Northwest Environmental Advocates

Response attached.

United States Department of the Interior

FISH AND WILDLIFE SERVICE
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June 28, 2001

Gregg Cooke
Regional Administrator
U.S. Environmental Protection Agency
1445 Ross Ave., Suite 1200
Dallas, Texas 75202-2733

Subject: EPA's noncompliance in Texas on National Pesticide Consultations

The purpose of this letter is to bring to your attention the concerns of the U.S. Fish and Wildlife Service's (Service) Austin Field Office in regard to the U.S. Environmental Protection Agency's (EPA) implementation of the: a) June 14, 1989 Biological Opinion on the National Pesticide Consultation, and b) the March 2, 1993 Biological Opinion on the Effects of 16 Vertebrate Control Agents on Threatened and Endangered Species. Biological opinions on impacts to Federally-listed species and their habitats from pesticides addressed from earlier consultations with EPA (notably the "cluster" opinions) were essentially superseded by these two biological opinions and will not be addressed by this letter.

The Service has five main concerns in regard to the two biological opinions as they relate to Federally-listed species, their habitats, and designated critical habitats in Texas:

1. EPA is not in compliance with reasonable and prudent alternatives (RPAs) and reasonable and prudent measures (RPMs) in the two biological opinions with respect to listed species in Texas as required by Section 7 of the Endangered Species Act of 1973, as amended, 16 U.S.C. 1531 et seq. (ESA). RPAs and RPMs are nondiscretionary actions that must be implemented by EPA and any applicant to ensure compliance with the ESA. Actions specified by RPAs and RPMs in the biological opinions include (but are not limited to) the following: specific buffer zone sizes for pesticide applications, limitations on pesticide application rates and types of application, prohibition of certain pesticides from occupied habitat, and establishment of a pesticide user education program. EPA has chosen to address these nondiscretionary measures as pesticide use limitations in county pesticide bulletins under its Endangered Species Protection Program as detailed in the notice of proposed program in the *Federal Register* (July 3, 1989, pp. 27894-28008). EPA has also chosen to require only voluntary compliance with county pesticide bulletins by pesticide applicators. The Service believes EPA's efforts to date for implementing RPAs and RPMs in Texas through county pesticide bulletins and through labelling of Section 18 and 24(c) pesticides is inadequate to meet the intent of the Service's biological opinions. To date, approximately 12 bulletins have been developed over the last decade

for Texas which has a total of 254 counties. The existing county bulletins do not address all threatened and endangered species and pesticides associated with an individual county nor do the bulletins provide all pertinent protection measures. The bulletins are not distributed throughout their respective county as promulgated by the 1989 *Federal Register* notice. In addition, coordination does not exist in referring pesticide users to county bulletins from FIFRA labels, i.e., Section 3, Section 18 emergency exemptions, and Section 24(c) local registrations. The Service believes that pesticide labels (including labels for FIFRA Section 18s and 24(c)s) should: a) refer applicators to the county bulletins, b) state that compliance with protection measures in county bulletins is a requirement for pesticide applications, and c) provide protection measures for pesticide applications involving listed species and critical habitat when this information is unavailable in a county bulletin. The Service also believes that EPA must use its regulatory authorities to enforce protection measures in county pesticide bulletins such as performing spot checks of applicators in the field and setting up pesticide monitoring networks for areas with a high potential for pesticide impacts on listed species and/or critical habitat.

2. There is not a comprehensive pesticide consultation for listed species, their habitats, and designated critical habitats in Texas. The Service believes that pesticide use in Texas adversely affects listed species as well as critical habitat since: a) a majority of pesticides used in Texas have had inadequate or no consultation, b) critical habitat has been insufficiently addressed, and c) no current mechanism exists for updating pesticide protection measures for recently listed species, critical habitat, or listed species that previously have undergone consultation. Currently in Texas there are 82 listed species, Federally-designated critical habitat for ten listed species, and over 300 pesticide active ingredients in use. The two aforementioned biological opinions are outdated since they represent consultations for only 19 Texas species and 125 pesticide active ingredients. Incidental take provisions and RPAs in biological opinions apply only to those listed species addressed in the opinions and cannot be used for additional species without concurrence of the Service. A number of the 125 pesticides originally consulted upon in the two biological opinions have had their registration canceled and/or are no longer used. Finally, new information in the spray drift/runoff literature indicates that buffer zones and other protection measures provided in the 1989 Biological Opinion should be revised.

The criteria by which a "may affect" determination is made by EPA for pesticide impacts on listed species in Texas during the risk assessment process are not appropriate. It is the Service's understanding that the risk assessment process as currently used by EPA for listed species is stipulated by the 1989 *Federal Register* notice. The risk assessment process relies in part on use of mathematical modeling through a quotient model which has a built-in safety coefficient factor for listed species. Recent information in the scientific literature (e.g., Tiebout and Brugger, 1995-see enclosure) indicates that assumptions underlying the quotient model approach do not adequately address interactions of listed species with pesticides. Other articles in the literature (e.g., Schauber et al. 1994-see enclosure) have shown that use of the quotient model has proven to be problematical when applied in field experiments. In particular, the quotient model and other mathematical modeling approaches in ecological risk assessment (including probabilistic models) cannot currently account for sublethal effects by pesticides on listed species such as endocrine disruption, abnormal behavioral changes, olfactory interference in anadromous fish spp., etc. Such sublethal effects from pesticide applications "may affect" listed species and therefore constitute harm as part of take as defined in the ESA. Since pesticide protection measures contained in the 1989 Biological Opinion have been based in part on use of the quotient model, the Service believes that: a) the biological opinion must be revised to provide more accurate protection measures for listed species, and b) the current process used by EPA for reaching "may affect" determinations for listed species must be re-evaluated including the role of mathematical models. To limit the uncertainty associated with mathematical modeling in ecological risk assessments, the Service also believes that an interagency memorandum should be instituted by EPA and the Service reflecting the following priorities (in descending order) for pesticide applications involving listed species:

a) Application of pesticides and their residues must avoid directly or indirectly impacting listed species or their habitat.

b) When pesticide applications interface with listed species or their habitat (e.g., species habitat bordering cropland), protection measures such as buffer zones must be required. Applicable protection measures for listed species should be evaluated according to: (1) the best available science, and (2) the relative hazard potential for individual pesticides assuming worst case scenarios.

c) When application of a particular pesticide directly involves a listed species or its habitat (e.g., a listed species in cropland or mosquito control areas), an ecological risk assessment must be made before application of the pesticide can be allowed. The assessments must be specific and as accurate for the species and/or habitat as possible and must include indirect effects such as loss of the prey base.

4. There are discrepancies between databases of EPA and the Service on locations of listed species on a county by county basis. It is the Service's understanding that EPA has primarily based its database on the National Heritage program for species. In Texas, this database was substantially underfunded in its early efforts to locate species and reflected efforts by individuals to cover local species of interest rather than provide comprehensive coverage for all species. Many of the records found in the original Texas Heritage program (now operated by the Texas Parks and Wildlife Department on a limited basis) are outdated or unconfirmed. Comparison of EPA's county map of species generated for the recent Endangered Species program meeting in Albuquerque in May, 2001 with that of the Service's database shows significant differences. For example, EPA lists no species in Kendall County of Texas, but the Service database lists black-capped vireo (*Vireo atricapillus*), golden-cheeked warbler (*Dendroica chrysoparia*), and bald eagle (*Haliaeetus leucocephalus*). These differences must be reconciled to adequately provide

protection for listed species with regulatory processes involving FIFRA or the ESA.

5. Application of pesticides in potential habitat of listed species "may affect" these species without prior determination of whether the species are present. It is the Service's understanding that as standard practice the EPA requires protection measures only in cases where habitat is known to be occupied by listed species and does not require surveys of potential habitat before application of potentially harmful pesticides. This issue came to the attention of the Service two years ago in a FIFRA Section 24(c) action by EPA involving use of the pesticide Arsenal® (active ingredient: imazapayr) for control of saltcedar (*Tamarix spp.*) in the Pecos River watershed of Texas. The Section 24(c) action had the potential for impacting the threatened Pecos sunflower *(Helianthus paradoxus)* by not requiring a survey in potential habitat within the watershed outside of the sunflower's known habitat in a Pecos River tributary. Local populations of listed species in Texas have not been completely documented by the Service or other agencies such as the Texas Parks and Wildlife Department since approximately 95 percent of the land ownership in Texas is private and access is ordinarily restricted. The Service believes that application of pesticides in potential habitat of listed species without appropriately authorized surveys does not comply with RPMs in the 1993 Biological Opinion and exceeds the level of take anticipated by both biological opinions.

Because of the aforementioned concerns, the Service believes that: a) Reasonable and Prudent Alternatives as well as terms and conditions of incidental take statements in the1989 Biological Opinion on the National Pesticide Consultation and the1993 Biological Opinion on the Effects of 16 Vertebrate Control Agents have been insufficiently addressed for Texas, and b) the protective coverage of the ESA's section 7(o)(2) as given to EPA has lapsed for Texas. The Service also believes that actions by EPA in implementing the two biological opinions constitute agency actions that have modified nondiscretionary stipulations in a manner inconsistent with the opinions. Therefore, EPA needs to reinitiate consultation with the Service on pesticides involving listed species and critical habitat in Texas. Listed species and critical habitat not currently covered in the two opinions need to be addressed in the reinitiated consultation.

We appreciate your attention to this matter. If you have further questions, please call myself or Allen White of my staff at 512-490-0057.

Sincerely,

/s/ David C. Frederick

David C. Frederick
Supervisor

Enclosures

cc: Field Supervisor, Arlington ESFO
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In Reply Refer to:
1-1-98-F-21

March 24, 2000

Ms. Felicia Marcus, Administrator
U.S. Environmental Protection Agency, Region 9
75 Hawthorne Street
San Francisco, California 94105

Dear Ms. Marcus:

This responds to your December 16, 1999, request to conclude formal consultation with the Fish and Wildlife Service (Service) and the National Marine Fisheries Service (NMFS), herein collectively referred to as the Services, on the Environmental Protection Agency's (EPA) "Final Rule for the Promulgation of Water Quality Standards: Establishment of Numeric Criteria for Priority Toxic Pollutants for the State of California" (CTR). This document represents the Services' final biological opinion on the effects of the final promulgation of the CTR on listed species and critical habitats in California in accordance with section 7 of the Endangered Species Act of 1973, as amended (16 USC 1531 et seq.; Act). A list of the species and critical habitats considered in this biological opinion is included as Table 1. Your request to conclude formal consultation on the CTR was received in the Sacramento Fish and Wildlife Service Office on December 30, 1999. Your initial October 27, 1997, request for formal consultation was received on October 30, 1997.

This document also includes a conference opinion, prepared pursuant to 50 **CFR** § 402.10, that addresses the effects of the final CTR on the following proposed threatened (PT) and proposed endangered (PE) species: Northern California ESU (Evolutionarily Significant Unit) of the steelhead trout(PT), Santa Ana sucker (*Catostomus santaanae*) (PT), the Southern California Distinct Population Segment of the Mountain Yellow-legged Frog (*Rana muscosa*)(PE), and the Santa Barbara County Distinct Population Segment of the California Tiger Salamander (*Ambystoma californiense*)(PE) . Critical habitat has been proposed the Tidewater goby. If any of these species or critical habitats become listed, this conference opinion can be converted to a biological opinion for those species/critical habitats, provided EPA formally requests such a conversion and the reinitiation criteria at 50 **CFR** § 402.16 do not apply.

The Services have reviewed EPA's biological evaluation for the proposed CTR and the effects of that action on the endangered salt marsh harvest mouse (*Reithrodontomys raviventris*), endangered least Bell's vireo (*Vireo belli pusillus*) and its critical habitat, endangered southwestern willow flycatcher (*Empidonax trailii extimus*) and its critical habitat, and the endangered San Joaquin kit fox (*Vulpes macrotis mutica*). The Services concur with EPA's determination that the CTR is not likely to adversely affect these species and critical habitats. Species the Services considered not likely to be adversely affected by the final CTR are listed in

Table 2. Therefore, unless new information reveals effects of the proposed action that may affect listed species in a manner or to an extent not considered, or a new species or critical habitat is designated that may be affected by the proposed action, no further action pursuant to the Act is necessary for the species listed above.

This biological and conference opinion is based on information provided in EPA's October 27, 1997, biological evaluation, the proposed CTR, correspondence that has occurred since the issuance of the Services' April 10, 1998, draft jeopardy biological opinion, supporting information contained within the Services' files, a review of the relevant published literature, discussions with specialists familiar with species ecology and toxicological issues presented in the CTR, numerous meetings and telephone conversations between our staffs, and EPA's December 16, 1999, proposed modifications to the CTR. The Services have prepared this biological and conference opinion in the absence of site-specific information on where numeric criteria will be applicable (areas not superseded by the promulgation of the proposed rule), and the lack of site-specific data on elements such as pH, water hardness, water effects ratios, and conversion factors. In the absence of these data we have used the ecologically most conservative estimate of effects for species and critical habitats considered in this opinion. Species and critical habitats the Services have determined likely to be adversely affected by the final CTR are listed in Table 3. A complete administrative record of this consultation is on file at the Service's Sacramento Fish and Wildlife Office.

CONSULTATION HISTORY

Informal consultation with EPA began on February 9, 1994, when the Service received EPA's request for a species list and a brief description of the draft CTR. On April 6 and 21, 1994, the Service and NMFS met with staff from EPA to discuss the CTR and begin informal discussions on the effects of the proposed numeric criteria on listed species and their critical habitats.

On May 31, 1994, the Service transmitted a species list to EPA for their consideration in the preparation of their biological evaluation. On June 26, 1997, the Service sent EPA an electronic update of the species list for the State of California.

On February 9, 1995, the Service participated in a teleconference call with EPA to discuss and categorize issues that were identified during internal strategy meetings between the Service and EPA. A list of issues was developed and categorized based on EPA's December 11, 1996, matrix of effects of the proposed criteria on listed species or their closely related surrogates. In addition, the Service provided EPA with a list of issues and concerns regarding the matrix and how to best address the effects of the proposed rule. During this meeting, the Service and EPA worked together to develop a table of issues and to identify the level to which these issues could be resolved.

On March 20, 1997, the Service and EPA met at EPA's request to re-initiate informal consultation. During this meeting, Service staff provided EPA with updated information on newly listed species and discussed key issues identified in previous meetings.

On June 19, 1997, the Service met with EPA to discuss outstanding issues regarding the proposed criteria for mercury, selenium, pentachlorophenol, the formula-based criteria for metals, and EPA's progress toward publishing a proposed rule. During this meeting, EPA indicated that the proposed CTR would likely be published, as drafted, in July of 1997, and would acknowledge the outstanding issues between the Service and EPA. During this meeting, the Service and EPA also discussed each of the following six issues: (1) the use of formula-based metals criteria; (2) the effects of copper on fish eggs, embryos, and non-gill breathing organisms; (3) the lack of analysis of the effects of pentachlorophenol on early life stages of fish species; (4) the lack of an aquatic criteria for Acrolein; (5) the threat of bioaccumulation to listed species by the promulgation of solely aquatic life criteria; and (6) the proposed selenium standard and its effects on listed species and aquatic ecosystems. At this time the Service indicated that it would prefer to resolve the disparity between the effects of proposed criteria and published scientific literature prior to publication of the proposed rule. Staff from EPA indicated that the Service would have numerous opportunities to resolve outstanding issues in the State's adoption of the CTR, and EPA's subsequent approval of the adoption and forthcoming basin plans. Time lines for completion of the draft CTR were discussed.

On July 25, 1997, the Service and EPA participated in a conference call to discuss the Service's concerns with the effects of the action on non-aquatic species, the proposed criteria for pentachlorophenol, and the formula-based metals criteria. Specifically, the Service discussed with EPA the draft biological evaluation and the lack of consideration of the bioaccumulative and interactive effects of the proposed criteria necessary to adequately assess the effects of the action on listed semi-aquatic and terrestrial wildlife species and their habitats. At this time the Service informed EPA that it could not concur with a "not likely to adversely affect" determination on the draft proposed rule and unless these issues were resolved, formal consultation under the Act would be necessary. Further, Service staff detailed the findings of published information which indicated that the proposed numeric criteria would have adverse effects on early life stages of salmonids at concentrations below the proposed numeric criteria for pentachlorophenol. Service staff also presented information regarding the use of formula-based criteria for metals considered in the CTR, and the potential for aquatic organisms to be adversely affected by the particulate fraction metals that would, in effect, be unregulated if EPA used the proposed formulae. No resolution of these issues was reached during this meeting; EPA provided the Service with an updated time line on the publication of the proposed rule.

On August 5, 1997, EPA published the proposed rule for the CTR (62 **FR** 42159).

On August 13, 1997, EPA and Service staff participated in a teleconference call to discuss the Service's ongoing concerns regarding the proposed promulgation of formula-based metals criteria. At this time staff from EPA suggested that the Service, in the absence of site-specific information necessary to calculate the criteria for each of eleven metals (Arsenic, Cadmium, Chromium (+3&+6), Copper, Lead, Silver, Selenium (+4&+6), Mercury, Nickel and Zinc), use a standard number for water hardness of 40. Service staff countered that hardness alone does not provide sufficient information to calculate a criterion (a conversion factor and water effect ratio are necessary in order to calculate criteria that are site-specific), and therefore, does not provide the Service with adequate information to consider the effects of the proposed formulae on listed species and critical habitat.

On September 25, 1997, Service staff provided written comments on the proposed CTR, reminding the EPA of their responsibilities to conserve listed species pursuant to sections 7(a)(1) and 7(a)(2) of the Act, and requested that EPA prepare a biological assessment on the effects of the proposed rule on listed species and critical habitats.

On October 30, 1997, the Service received EPA's biological evaluation for the CTR requesting concurrence with a finding that the proposed CTR was not likely to adversely affect listed species. On November 28, 1997, the Service issued a letter of non-concurrence, and acknowledged EPA's request to initiate formal consultation.

On December 10, 1997, the Service received a letter from EPA asking the Service to dispose of all previous drafts (including all drafts of the CTR issued between 1994 and August 1997) of the proposed numeric criteria in the CTR.

On January 8, 1998, staff from EPA, and the Services met to discuss the outstanding issues in the CTR, and the Service's progress on the biological opinion. At this time the Services presented their findings on the deficiency of the numeric criteria for mercury, selenium, pentachlorophenol, and dissolved metals. No agreements were made between the agencies on any changes to the proposed numeric criteria. This meeting's primary objective was to review the issues and the Services concerns regarding the proposed criteria, the apparent data gaps in the CTR, and the promulgation of the numeric criteria. The Services agreed to provide EPA with written documentation on the information they had reviewed on the proposed criteria and their failure to protect listed species. On January 29, 1998, the Services sent EPA a letter documenting their review of available information on the toxicity of pentachlorophenol on salmonids.

On April 10, 1998, the Services issued a draft jeopardy biological opinion (draft opinion) on the proposed CTR. In that opinion the Services concluded the CTR as proposed on August 5, 1997, was likely to jeopardize the continued existence of 25 listed species, and result in the adverse modification of 11 critical habitat units (see table 4). Since that time, staff from EPA Region IX and the Services have been discussing reasonable and prudent alternatives issued in the draft opinion. Those discussions have resulted in modifications to the proposed action by EPA and the Services subsequent revision of the April 10, 1998, and April 9, 1999, biological opinions.

For the purposes of our April 10, 1998, draft biological opinion and this opinion, findings of "no effect" were made for species which are not at any point in their development or foraging ecology dependent on the aquatic ecosystem. An example of a species that would not be affected by the proposed CTR is the desert slender salamander which is not dependent at any life stage on the aquatic ecosystem.

Findings of "not likely to adversely affect" were made for those species that may utilize the aquatic ecosystem, but whose foraging ecology or range results in a low likelihood of being exposed to problematic concentrations at or below proposed criteria concentrations. Examples of species not likely to be adversely affected are the Warner sucker, with a range that includes California but whose watershed boundaries are primarily outside of the State; and the least Bell's vireo, which is dependent on the aquatic/riparian ecosystem but its foraging ecology is not primarily dependent on the aquatic ecosystem.

The Services define jeopardy as an action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species. The Services concluded that a determination of "may affect, not likely to jeopardize the continued existence of the species" was appropriate when the potential exists for toxic effects to occur at or below the proposed numeric criteria concentrations of a pollutant considered in the CTR and one or more of the following conditions or combination of conditions were met: (1) the existing environmental conditions are currently not near or not likely to approach the proposed criteria concentrations; (2) the species is widely distributed, either within the State or within multiple states and proposed numeric criteria are likely to impact few individuals or an insignificant number of individuals within a population; (3) the foraging ecology of the species is not primarily dependent on the aquatic ecosystem, and dietary habits offer dilution by terrestrial food resources, significantly reducing adverse impacts associated with elevated levels of contaminants acquired while foraging in aquatic ecosystems; and (4) the species is migratory, and/or prolonged exposures to elevated concentrations of contaminants is not likely (dietary diversity).

Previously in the Services' April 10, 1998, and April 9, 1999, revised draft opinions we concluded that a determination of "may affect, likely to jeopardize the continued existence of a species" was appropriate when the species is primarily dependent upon the aquatic ecosystem for its foraging ecology, reproduction and survival, toxicity occurs at or below proposed criteria concentrations in water, and water concentrations within the habitat occupied by the species has a high probability of approaching or reaching a problematic concentration at or below criteria concentrations proposed in the CTR. Additional factors considered for a species or their critical habitat unit were: (1) whether the species is non-migratory and thus vulnerable to local contamination; (2) whether exposure to toxic concentrations at or below the proposed numeric criteria is likely to occur during the breeding season, a sensitive life stage, or during its entire life cycle; (3) whether exposures to toxic concentrations results in significant interactions with other stressors affecting the species such as susceptibility to disease, avoidance of introduced predators, etc.; and (4) the proposed numeric criteria are likely to significantly impair one or more primary constituent elements of a species' critical habitat. However, since EPA has modified the proposed action as presented in the "Description of the Proposed Action" section of this document, the Services are able to conclude that the action as modified is not likely to jeopardize the continued existence of these species, nor result in the adverse modification of their critical habitat. Species for which the Services previously concluded were likely to be jeopardized or their critical habitats adversely modified are presented in Table 4.

On April 27, 1998, the Services met with EPA staff to discuss the draft and EPA's concerns regarding the precedence of a jeopardy biological opinion on threatened and endangered species on their water quality criteria rule making process and their capacity to respond to the reasonable and prudent alternatives presented in the draft opinion.

On October 29, 1998, EPA Region IX staff, in cooperation with the Office of Science and Technology in Washington D.C., submitted a proposal to the Services to modify the CTR as proposed. Included in this proposal were draft agreements to change the scope of the CTR for criteria for mercury, selenium, and pentachlorophenol. As proposed these commitments made significant progress towards ameliorating the effects of the CTR. However, only the Administrator of EPA has the authority to make modifications to proposed rule making. Therefore, proposed modifications have yet to be completed.

Between October 1998 and March 17, 1999, EPA and Services' staff worked together to resolve issues and develop agreeable timelines and procedures to amend the proposed action as proposed in the August 5, 1997, version of the proposed CTR. On April 7, 1999, EPA sent the Services a letter documenting the proposed modifications. Services' staff utilized these draft agreements to formulate revised reasonable and prudent alternatives that were presented to EPA in a revised draft jeopardy biological opinion, informally transmitted to EPA on April 9, 1999.

Between April and August 2, 1999, and after review of the revised reasonable and prudent alternatives, EPA and the Services met on August 2, 1999, to discuss what further modifications to the proposed action were necessary to remove the jeopardizing effects of the CTR. On September 14, 1999, EPA transmitted a draft facsimile copy of their proposed modifications to the CTR for Services review.

Between August and December 16, 1999, EPA and Services' staff continued to refine the proposed modifications to the CTR. After numerous discussions between EPA and Services' staff regarding these modifications, EPA re-submitted their final proposed modifications on December 16, 1999. The Services have based this final opinion on those modifications. The final modifications to the proposed action are incorporated herein by reference in the following "Description of the Proposed Action", and "Conclusions" sections of this biological opinion.

DESCRIPTION OF THE PROPOSED ACTION

EPA is issuing a final rule on the CTR. This rule will promulgate legally enforceable water quality criteria for the state of California for inland surface waters, enclosed bays and estuaries, for all programs and purposes under the CWA. When completed these criteria are available to the State for immediate adoption and subsequent use by the State and Regional Water Quality

Control Boards (RWQCBs) for their use in permit writing and identification of impaired waters. The Final CTR will also Total Maximum Daily Loads (TMDL), Interim Permit Limits, Mixing Zones, and Variances

On August 5, 1997, EPA published a proposed rule on the CTR based on the Administrator's determination that criteria were needed in the State of California to meet the requirements of section 303(c)(2)(B) of the Clean Water Act of 1987, as amended (33 USC 1251 et seq.; CWA). This section of the CWA requires States to adopt numeric water quality criteria for priority toxic pollutants for which EPA has issued CWA section 304(a) criteria guidance and whose presence or discharge could be reasonably expected to interfere with designated beneficial uses. Priority toxic pollutants are identified in 40 **CFR** Part 131.36; currently, 126 constituents are classified as priority toxic pollutants.

The CTR is important for several environmental, programmatic and legal reasons. Control of toxic pollutants in surface waters is necessary to achieve the CWA's goals and objectives. Many of California's monitored river miles, lake acres, and estuarine waters have elevated levels of toxic pollutants. Recent studies on California water bodies indicate that elevated levels of toxic pollutants exist in fish tissue; this has resulted in the issuance of fishing advisories or bans. These toxic pollutants can be attributed to, among other sources, industrial and municipal discharges. Toxic pollutants for which fish advisories exist include mercury and selenium, two priority pollutants addressed in the CTR.

Water quality standards for toxic pollutants are important to State and EPA efforts to address water quality problems. Clearly established water quality goals enhance the effectiveness of many of the State's and EPA's water programs including permitting, coastal water quality improvement, fish tissue quality protection, non-point source controls, drinking water quality protection, and ecological protection. Numeric criteria for toxic pollutants allow the State and EPA to evaluate the adequacy of existing and potential control measures to protect aquatic ecosystems and human health. Numeric criteria also provide a more precise basis for deriving water quality-based effluent limitations in National Pollutant Discharge Elimination System (NPDES) permits to control toxic pollutant discharges.

EPA, through the CTR, establishes water quality criteria for toxic pollutants for inland surface waters, enclosed bays, and estuaries in the State of California. These numeric water quality criteria for priority toxic pollutants are necessary to fulfill the requirements of section 303(c)(2)(B) of the CWA. The CTR also authorizes a compliance schedule provision in the preamble allowing the RWCQB's to give existing dischargers up to five years after their first permit renewal following the final CTR to come into compliance. The maximum time that the CTR allows for a compliance schedule is ten years after the adoption of the final rule, regardless of how many years after the final rule the first permit renewal occurred.

EPA's publication of the final CTR will fill a gap in California water quality standards. This gap is the result of litigation by several dischargers who sued the California State Water Resources

Control Board (SWRCB) over whether the SWRCB adopted its statewide water quality control plans for inland surface waters, enclosed bays and estuaries in compliance with State law. The SWRCB's water quality control plans contained water quality criteria for many priority toxic pollutants. The California Superior Court for the County of Sacramento issued its final decision in favor of the plaintiffs in March 1994. In July 1994, the Court ordered the SWRCB to rescind the two water quality control plans, and the SWRCB formally did so in September of 1994. The State of California is currently without numeric water quality criteria for these priority toxic pollutants as required by the CWA, necessitating this action by EPA. The State of California is also in the process of readopting its statewide water quality control plans. When California completes its readoption process, and EPA approves the State plans, the Federal standards will no longer be needed.

In the interim, when these proposed Federal criteria take effect they will create legally applicable water quality criteria in California for inland surface waters, enclosed bays and estuaries, for all programs and purposes under the CWA. This proposed rule does not change or supersede any criteria that were previously promulgated for the State of California including those promulgated in the National Toxics Rule (NTR), as amended (Water Quality Standards; Establishment of Numeric Criteria for Priority Toxic Pollutants, 57 **FR** 60848, December 22, 1992; and the NTR as amended by the Administrative Stay of Federal Water Quality Criteria for Metals and Interim Final Rule, Water Quality Standards; Establishment of Numeric Criteria for Priority Toxic Pollutants; States Compliance Revision of Metals Criteria, 60 **FR** 22228, May 4, 1995 (referred to as the "NTR, as amended"). These criteria are footnoted in the table in the final CTR, so that readers may see the criteria previously promulgated in the NTR, as amended, together with the new proposed criteria. The CTR when finalized will not change or supersede federally approved, state-adopted, site-specific objectives.

Water Quality Criteria Overview

Section 303 of the CWA mandates that States adopt water quality standards to restore and maintain the chemical, physical, and biological integrity of the Nation's waters. Water quality standards consist of beneficial uses designated for specific water bodies and water quality criteria necessary to protect uses. Water quality criteria may be numeric, for example 9 µg/L of copper, or narrative, such as "no toxics in toxic amounts."

In order to avoid confusion, it must be recognized that the CWA uses the term "criteria" in two separate ways. In section 303 of the CWA, the term "criteria" is part of the definition of a water quality standard. "Criteria" refers to the ambient component of the water quality standard contained in state or Federal law. However, section 304(a) of the CWA directs EPA to publish water quality "criteria" guidance which encompass scientific assessments of the health and ecological effects of various pollutants listed pursuant to section 307(a) of the CWA and which are used to support development of ambient criteria as part of the water quality standards. CWA section 304(a) criteria guidance are developed using Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and their Uses (National

Guidelines) and are based on the results of toxicity tests conducted with organisms that are sensitive to specific toxicants. These section 304(a) criteria are intended as guidance only and have no binding effect. In contrast, the ambient criteria adopted by EPA pursuant to section 303 of the CWA are legally enforceable.

These legally enforceable criteria adopted pursuant to section 303 are based on: (1) the 304(a) criteria guidance; (2) 304(a) criteria guidance modified to reflect site-specific conditions; or (3) other scientifically defensible methods. EPA guidance as described in the Water Quality Standards Handbook, allows states to establish water quality criteria/objectives on a site-specific basis to reflect local conditions. EPA requires that a scientifically justifiable method be employed in deriving site-specific criteria. The method must be consistent with the assumptions, rationale, and spirit of the National Guidelines.

Modifications to the Final CTR

Based on the Services' April 9,1999, revised draft biological opinion EPA submitted the following proposed modifications to the CTR in their December 16, 1999, letter to the Services. These modifications will be incorporated by reference into section M of the preamble of EPA's final promulgation of the CTR. They are recorded here to reflect EPA's agreed-upon modifications to the proposed CTR.

I. EPA Modifications Addressing the Services' April 9, 1999 draft Reasonable and Prudent Alternatives for Selenium:

A. EPA will reserve (not promulgate) the proposed acute aquatic life criterion for selenium in the final CTR.

B. EPA will revise its recommended 304(a) acute and chronic aquatic life criteria for selenium by January 2002. EPA will propose revised acute and chronic aquatic life criteria for selenium in California by January of 2003. EPA will work in close cooperation with the Services to evaluate the degree of protection afforded to listed species by the revisions to these criteria. EPA will solicit public comment on the proposed criteria as part of its rulemaking process, and will take into account all available information, including the information contained in the Services' Opinion, to ensure that the revised criteria will adequately protect federally listed species. If the revised criteria are less stringent than those proposed by the Services in the Opinion, EPA will provide the Services with a biological evaluation/assessment on the revised criteria by the time of the proposal to allow the Services to complete a biological opinion on the proposed selenium criteria before promulgating final criteria. EPA will provide the Services with updates regarding the status of EPA's revision of the criterion and any draft biological evaluation/assessment associated with the revision. EPA will promulgate final criteria as soon as possible, but no later than 18 months, after proposal. EPA will continue to consult, under section 7 of ESA, with the Services on revisions to water quality standards contained in Basin Plans, submitted to EPA under CWA section 303, and affecting waters of California containing federally listed species and/or their habitats. EPA will annually submit to the Services a list of NPDES permits due for review to allow the Services to identify any potential for adverse effects on listed species and/or their habitats. EPA will coordinate with the Services on any permits that the Services identify as having potential for adverse effects on listed species and/or their habitat in accordance with procedures agreed to by the Agencies in the draft MOA published in the Federal Register at 64 FR 2755 (January 15, 1999) or any modifications to those procedures agreed to in a finalized MOA.

C. EPA will utilize existing information to identify water bodies impaired by selenium in the State of California. Impaired is defined as water bodies for which fish or waterfowl consumption advisories exist or where water quality criteria necessary to protect federally listed species are not met. Pursuant to Section 303(d) of the CWA, EPA will work, in cooperation with the Services, and the State of California to promote and develop strategies to identify sources of selenium contamination to the impaired water bodies where federally listed species exist, and use existing authorities and resources to identify, promote, and implement measures to reduce selenium loading into their habitat. (See also “Other Actions B.” below.)

II. <u>EPA Modifications Addressing the Services’ April 9, 1999 draft Reasonable and Prudent Alternatives for Mercury</u>:

A. EPA will reserve (not promulgate) the proposed freshwater and saltwater acute and chronic aquatic life criteria for mercury in the final CTR.

B. EPA will promulgate a human health criterion of 50 ng/l or 51 ng/l as designated within the final CTR for mercury <u>only</u> where no more restrictive federally-approved water quality criteria are now in place (e.g., the promulgation will not affect portions of San Francisco Bay).

C. EPA will revise its recommended 304(a) human health criteria for mercury by January 2002. EPA will propose revised human health criteria for mercury in California by January 2003. These criteria should be sufficient to protect federally listed aquatic and aquatic-dependent wildlife species. EPA will work in close cooperation with the Services to evaluate the degree of protection afforded to federally listed species by the revised criteria. EPA will solicit public comment on the proposed criteria as part of its rulemaking process, and will take into account all available information, including the information contained in the Services’ Opinion, to ensure that the revised criteria will adequately protect federally listed species. If the revised criteria are less stringent than those proposed by the Services in the Opinion, EPA will provide the Services with a biological evaluation/assessment on the revised criteria by the time of the proposal to allow the Services to complete a biological opinion on the proposed mercury criteria before promulgating final criteria. EPA will provide the Services with updates regarding the status of EPA's revision of the criterion and any draft biological evaluation/assessment associated with the revision. EPA will promulgate final criteria as soon as possible, but no later than 18 months, after proposal. EPA will continue to consult, under section 7 of ESA, with the Services on revisions to water quality standards contained in Basin Plans, submitted to EPA under CWA section 303, and affecting waters of California containing federally listed species and/or their habitats. EPA will annually submit to the Services a list of NPDES permits due for review to allow the Services to identify any potential for adverse effects on listed species and/or their habitats. EPA will coordinate with the Services on any permits that the Services identify as having potential for adverse effects on listed species and/or their habitat in accordance with procedures agreed to by the Agencies in the draft MOA published in the Federal Register at 64 FR 2755 (January 15, 1999) or any modifications to those procedures agreed to in a finalized MOA.

D. EPA will utilize existing information to identify water bodies impaired by mercury in the State of California. Impaired is defined as water bodies for which fish or waterfowl consumption advisories exist or where water quality criteria necessary to protect federally listed species are not met. Pursuant to Section 303(d) of the CWA, EPA will work, in cooperation with the Services, and the State of California to promote and develop strategies to identify sources of mercury contamination to the impaired water bodies where federally listed species exist, and use existing authorities and resources to identify, promote, and implement measures to reduce mercury loading into their habitat. (See also "Other Actions B." below.)

E. EPA promulgated a new more sensitive analytical method for measuring mercury (see 40 CFR Part 136).

III. <u>EPA Modifications Addressing the Services' April 9, 1999 draft Reasonable and Prudent Alternatives for Pentachlorophenol (PCP)</u>:

A. By March of 2001, EPA will review, and if necessary, revise its recommended 304(a) chronic aquatic life criterion for PCP sufficient to protect federally listed species and/or their critical habitats. In reviewing this criterion, EPA will generate new information on chronic sub-lethal toxicity of commercial grade PCP, and the interaction of temperature and dissolved oxygen, to protect early life-stage salmonids. If EPA, revises its recommended 304(a) criterion, EPA will then propose the revised PCP criterion in California by March 2002. If the proposed criterion is less protective than proposed by the Services in their Opinion or if EPA determines that a proposed criterion is not necessary, EPA will provide the Services with a biological evaluation/assessment by March 2002 and will reinitiate consultation. EPA will keep the Services informed regarding the status of EPA's review of the criterion and any draft biological evaluation/assessment associated with the review. If EPA proposes a revised PCP criterion by March 2002, EPA will promulgate a final criterion as soon as possible, but no later than 18 months, after proposal.

B. EPA will continue to use existing NPDES permit information to identify water bodies which contain permitted PCP discharges and Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) and Resource Conservation and Reclamation Act (RCRA) sites that potentially contribute PCP to surface waters. EPA, in cooperation with the Services, will review these discharges and associated monitoring data and permit limits, to determine the potential for the discharge to impact federally listed species and/or critical habitats. If discharges are identified that have the potential to adversely affect federally listed species and/or critical habitat, EPA will work with the Services and the State of California to address the potential effects to the species. EPA will give priority to review data for fresh water bodies within the range of federally listed salmonids that currently lack a MUN designation as specified in the Regional Water Quality Control Boards' Basin Plans.

IV. <u>EPA Modifications Addressing the Services' April 9, 1999 draft Reasonable and Prudent Alternatives for Cadmium</u>:

EPA will develop a revision to its recommended 304 (a) chronic aquatic life criterion for cadmium by January 2001 to ensure the protection of federally listed species and/or critical habitats and will propose the revised criterion in California by January 2002. However, if EPA utilizes the revised metals criteria model referred to below (see V.C.), EPA will develop a revision to its recommended 304(a) criterion by January 2002 and will propose the revised criterion in California by January 2003. EPA will solicit public comment on the proposed criteria as part of its rulemaking process, and will take into account all available information, including the information contained in the Services' Opinion, to ensure that the revised criterion will adequately protect federally listed species. If the revised criterion is less stringent than that proposed by the Services in the Opinion, EPA will provide the Services with a biological evaluation/assessment on the revised criterion by the time of the proposal to allow the Services to complete a biological opinion on the proposed cadmium criterion before promulgating final criteria. EPA will provide the Services with updates regarding the status of EPA's revision of the criterion and any draft biological evaluation/assessment associated with the revision. EPA will promulgate final criteria as soon as possible, but no later than 18 months, after proposal. EPA will continue to consult, under section 7 of ESA, with the Services on revisions to water quality standards contained in Basin Plans, submitted to EPA under CWA section 303, and affecting waters of California containing federally listed species and/or their habitats. EPA will annually submit to the Services a list of NPDES permits due for review to allow the Services to identify any potential for adverse effects on listed species and/or their habitats. EPA will coordinate with the Services on any permits that the Services identify as having potential for adverse effects on listed species and/or their habitat in accordance with procedures agreed to by the Agencies in the draft MOA published in the Federal Register at 64 FR 2755 (January 15, 1999) or any modifications to those procedures agreed to in a finalized MOA.

V. EPA Modifications to Address the Services' April 9, 1999 draft Reasonable and Prudent Alternatives for Dissolved Metals:

A. By December of 2000, EPA, in cooperation with the Services, will develop sediment criteria guidelines for cadmium, copper, lead, nickel and zinc, and by December of 2002, for chromium and silver. When the above guidance for cadmium, copper, lead, nickel and zinc is completed, Region 9, in cooperation with the Services, will draft implementation guidelines for the State of California to protect federally listed threatened and endangered species and critical habitat in California.

B. EPA, in cooperation with the Services, will issue a clarification to the *Interim Guidance on the Determination and Use of Water-Effect Ratios for Metals* (EPA 1994) concerning the use of calcium-to-magnesium ratios in laboratory water, which can result in inaccurate and under-protective criteria values for federally listed species considered in the Services' opinion. EPA, in cooperation with the Services, will also issue a clarification to the *Interim Guidance* addressing the proper acclimation of test organisms prior to testing in applying water-effect ratios (WERs).

C. By June of 2003, EPA, in cooperation with the Services, will develop a revised criteria calculation model based on best available science for deriving aquatic life criteria on the basis of hardness (calcium and magnesium), pH, alkalinity, and dissolved organic carbon (DOC) for metals. This will be done in conjunction with "Other Actions A." below.

D. In certain instances, the State of California may develop site-specific translators, using EPA or equivalent state/tribe guidance, to translate dissolved metals criteria into total recoverable permit limits. A translator is the ratio of dissolved metal to total recoverable metal in the receiving water downstream, from a discharge. A site-specific translator is determined on site-specific effluent and ambient data.

Whenever a threatened or endangered species or critical habitat is present within the geographic range downstream from a discharge where a State developed translator will be used and the conditions listed below exist, EPA will work, in cooperation with the Services and the State of California, to use available ecological safeguards to ensure protection of federally listed species and/or critical habitat. Ecological safeguards include: (1) sediment guidelines; (2) biocriteria; (3) bioassessment; (4) effluent and ambient toxicity testing; or (5) residue-based criteria in shellfish.

Conditions for use of ecosystem safeguards:

1. A water body is listed as impaired on the CWA section 303(d) list due to elevated metal concentrations in sediment, fish, shellfish or wildlife; or,

2. A water body receives mine drainage; or,

3. Where particulate metals compose a 50% or greater component of the total metal measured in a downstream water body in which a permitted discharge (subject to translator method selection) is proposed and the dissolved fraction is equal to or within 75% of the water quality criteria.

Whenever a threatened or endangered species is present downstream from a discharge where a State developed translator will be used, EPA will work with the permitting authority to ensure that appropriate information, which may be needed to calculate the translator in accordance with the applicable guidance, will be obtained and used. Appropriate information includes:

1. Ambient and effluent acute and chronic toxicity data;
2. Bioassessment data; and/or
3. An analysis of the potential effects of the metals using sediment guidelines, biocriteria and residue-based criteria for shellfish to the extent such guidelines and criteria exist and are applicable to the receiving water body.

EPA, in cooperation with the Services, will review these discharges and associated monitoring data and permit limits, to determine the potential for the discharge to impact federally listed species and/or critical habitats. If discharges are identified that have the potential to adversely affect federally listed species and/or critical habitat, EPA will work with the Services and the State of California in accordance with procedures agreed to by the Agencies in the draft MOA published in the Federal Register at 64 FR 2755 (January 15, 1999) or any modifications to those procedures agreed to in a finalized MOA.

Other Actions

A. EPA will initiate a process to develop a national methodology to derive site-specific criteria to protect federally listed threatened and endangered species, including wildlife, in accordance with the draft MOA between EPA and the Services concerning section 7 consultations.

B. EPA will use existing information to identify water bodies impaired by mercury and selenium in the State of California. "Impaired" is defined as water bodies for which fish or waterfowl consumption advisories exist or where water quality criteria necessary to protect the above species are not met. Pursuant to Section 303(d) of the CWA, EPA will work with the State of California to promote and develop strategies to identify sources of selenium and mercury contamination to the impaired water bodies where federally listed species exist, and use existing authorities and resources to identify, promote, and implement measures to reduce selenium and/or mercury loading into their habitat (e.g., San Joaquin River, Salton Sea, Cache Creek, Lake Nacimiento, Sacramento - San Joaquin Delta etc.). EPA will work closely with the Services on individual TMDLs to avoid delays associated with approvals of these actions. (See also Selenium C. and

Mercury D., above.)

The Services in our finalization of this biological opinion have formalized and refined the preceding agreements into non-discretionary terms and conditions presented in the "Incidental Take Statement" section of this document. The Services where necessary have included additional language in some areas of these agreements to ensure that these agreements/measures are enforceable.

Implementation of the CTR

In the CTR, EPA proposes numeric water quality criteria which, when combined with the designated uses for water bodies selected by the State, create water quality standards. These standards are applied to dischargers through implementation procedures adopted by the State. Subsections included in the implementation schedule of the CTR include the development of Total Maximum Daily Loads (TMDL), Interim Permit Limits, Mixing Zones, and Variances. The promulgation of the CTR is a Federal action and therefore all aspects of its implementation are subject to consultation requirements pursuant to section 7 of the Act. The State's adoption and implementation of the CTR must be approved by EPA and are therefore also subject to section 7 consultation requirements as part of EPA approval.

Wet Weather Flows

A wet weather point source means any discernible confined and discrete conveyance from which pollutants are, or may be, discharged as the result of a wet weather event. For the purposes of the CTR these discharges include only: discharges of storm water from a municipal separate storm sewer as defined at 40 **CFR** § 122.26(b)(8); storm water discharge associated with industrial activity as defined at 40 **CFR** § 122.26(b)(14); discharges of storm water and sanitary wastewater (domestic, commercial, and industrial) from a combined sewer overflow; or any storm water discharge for which a permit is required under § 402(p) of the CWA. NPDES permits for wet weather point source discharges must include limits necessary to implement applicable water quality standards, through application of water quality-based effluent limits (WQBELs). When the CTR rulemaking process is complete, these criteria will be used to determine water quality standards in California and will therefore be the basis for WQBELs in NPDES permits for wet weather point sources. Where it is infeasible to express WQBELs as numeric limits for wet weather discharges, best management practices (BMPs) may be used as WQBELs. It is anticipated that WQBELs, including those necessary to meet the criteria set forth in the CTR, will be expressed as BMPs in wet weather discharge NPDES permits when the permitting authority determines that it is infeasible to express WQBEL as numeric limits.

Schedules of Compliance

The CTR provides that compliance schedules may take up to five years to meet new or more stringent effluent limitations, and in cases where EPA has recently approved site-specific criteria, the criteria contained within the CTR may not be reached for up to 10 years. All site-specific criteria must be approved by the EPA and are therefore subject to consultation pursuant to section 7 of the Act.

DESCRIPTION OF THE ACTION AREA

The CTR covers surface waters in California, which are waters of the United States, and which have been designated as inland surface waters or enclosed bays and estuaries. These include all watersheds with their rivers, streams, channels, lakes, ponds, enclosed bays and estuaries in California. Ocean water is not covered by the CTR, because the State of California already has a valid statewide plan to control ocean water quality. This proposed rule does not change or supersede any criteria previously promulgated for the State of California in the NTR, as amended. This proposed rule is not intended to apply to waters within Indian Country (*sic*).

The CTR is a statewide rulemaking process promulgating water quality criteria for all parts of California, with limited exceptions, where water quality criteria have been adopted for specific water bodies. For instance, the selenium criteria for the San Francisco Bay have already been promulgated under the NTR. For a complete list of such exceptions see footnotes "o" through "t" to the table listing all priority toxic pollutants in the CTR itself.

Water quality criteria previously promulgated within the NTR (but not previously consulted on) are considered in this opinion for adequacy of protection of listed species. EPA has not provided the Services with a list of waters for which the CTR does not apply and therefore, the Services have considered all waters within the State equally.

SPECIES DESCRIPTIONS

Aleutian Canada Goose (*Branta canadensis leucoparia*)

Species Description and Life History: The Aleutian Canada goose was listed as threatened on December 12, 1990 (55 **FR** 51112). This subspecies was originally classified as endangered on March 11, 1967.

The Aleutian Canada goose can be distinguished from most other subspecies of Canada geese by their small size (only cackling Canada geese are smaller) and a ring of white feathers at the base of the black neck in birds older than 8 months. Lakes, reservoirs, ponds, large marshes, and flooded fields are used for roosting and loafing (Grinnell and Miller 1944, USDI-FWS 1982a).

Foraging Ecology: Aleutian Canada geese forage in harvested corn fields, newly planted or grazed pastures, or other agricultural fields (e.g., rice stubble and green barley).

Historic and Current Distribution: Historically, Aleutian Canada geese wintered from British

Columbia to California and northwestern Mexico (Delacour 1954). Although they occurred throughout California, the greatest concentrations were found in the Sacramento and San Joaquin Valleys (Grinnell and Miller 1944).

The subspecies nested throughout the Aleutian Islands and into Russia (Springer 1977). Predation by introduced arctic foxes eliminated most breeding colonies of the Aleutian Canada goose, and by 1962 the subspecies was nearly extinct, with only one breeding colony remaining on the tiny island of Buldir. This island was one of the few to escape the introduction of arctic foxes (USDI-FWS 1982a). In 1982, a new or remnant breeding population of Aleutian Canada geese of unknown size was discovered on Chagulak Island in the Islands of the Four Mountains (USDI-FWS 1982a).

The present population of Aleutian Canada geese migrates along the northern California coast and winters in the Central Valley near Colusa, and on scattered feeding and roosting sites along the San Joaquin River from Modesto to Los Banos (Nelson *et al.* 1984). Fall migration usually begins in late August or early September, with birds arriving in the Central Valley between October and early November. Spring migration usually occurs from mid-February to early March.

In California, the Aleutian goose occurs on agricultural lands along the north coast, and throughout the Sacramento and San Joaquin Valleys. Major migration and wintering areas include agricultural lands north of Crescent City in Del Norte County, around the Sutter Buttes in the Sacramento Valley, near El Sobrante in Contra Costa County, and along the San Joaquin River between Modesto and Los Banos.

<u>Reasons for Decline and Threats to Survival</u>: Predation by introduced arctic foxes on the breeding islands is the primary reason for the population decline. Avian cholera is currently a major threat to the concentrations of Aleutian Canada geese in the Central Valley. In 1991, 58 geese died during an outbreak of avian cholera in the San Joaquin Valley (USDI-FWS 1991). This subspecies is particularly vulnerable to cholera outbreaks because most of the population overwinters in a small geographical area. Sport hunting on its wintering grounds in California and by natives on the nesting grounds also contributed to the species' decline (USDI-FWS 1982a). At one time, recreational and subsistence take of this subspecies in the Pacific Flyway may have been a significant factor preventing the remnant breeding segments from recovering.

Changing land use practices in the wintering range, including the conversion of cropland and pastures to housing and other urban development, adversely affect Aleutian geese (USDI-FWS 1991). The lack of adequately protected migration and winter habitat for Aleutian geese is the greatest obstacle to full recovery of this species (USDI-FWS 1991). Habitat quality has also likely declined due to the concentrated effects of pollution, human disturbance, and disease (USDI-FWS 1991).

Bald Eagle (*Haliaeetus leucocephalus*)

Species Description and Life History: The bald eagle was federally listed as endangered on February 14, 1978 (43 **FR** 6233) in all of the coterminous United States except Minnesota, Wisconsin, Michigan, Oregon, and Washington, where it was classified as threatened. On August 15, 1995 (60 **FR** 36010), the bald eagle was down-listed to threatened throughout its range. Critical habitat has not been designated for the bald eagle. On July 6, 1999, the Service published a proposed rule to remove the bald eagle from the federal list of threatened and endangered species (64 **FR** 36454). The recovery plan for the Pacific population of the bald eagle describes the species biology, reasons for decline, and the actions needed for recovery (USDI-FWS 1986b).

The Pacific Recovery Region for the bald eagle includes the States of California, Oregon, Washington, Idaho, Montana, Wyoming, and Nevada. Other recovery plans exist for bald eagle populations in the Southeast, Southwest, Northern States, and Chesapeake Bay. Delisting/reclassification of the bald eagle in the Pacific Recovery Region is not dependent on the status of bald eagle populations covered by these other plans (USDI-FWS 1986b). For this reason, the Pacific Recovery Region for the bald eagle will be viewed as a recovery unit for purposes of this consultation.

Foraging Ecology: The bald eagle is a generalized predator/scavenger primarily adapted to edges of aquatic habitats. Typically fish comprise up to 70% of the nesting eagle diet with mammals, birds, and some amphibians and reptiles providing the balance of the diet. Wintering eagles forage fish, waterfowl, mammals, and a variety of carrion. Bald eagles can maneuver skillfully and frequently hunt from perches. They are also known to hunt by coursing low over the ground or water.

Historic and Current Distribution: The bald eagle is the only North American representative of the fish or sea eagles, and is endemic to North America. The breeding range of the bald eagle includes most of the continent, but they now nest mainly in Alaska, Canada, the Pacific Northwest states, the Great Lake states, Florida, and Chesapeake Bay. The winter range includes most of the breeding range, but extends primarily from southern Alaska and southern Canada, southward.

As of 1996, about 5,068 occupied bald eagle territories were estimated within its range. Of these, 1,274 (25 %) were estimated to occur within the Pacific Recovery Region. Within the 7-State Pacific Recovery Region, 105 occupied territories occurred in California, 90 in Idaho, 165 in Montana, 0 in Nevada, 266 in Oregon, 582 in Washington, and 66 in Wyoming (Jody Millar, Bald Eagle Recovery Coordinator, FWS, pers. comm.). The most recent estimates for Washington are 589 occupied territories (Jim Michaels, FWS, pers. comm.), 308 in Oregon (Diana Wang, FWS, pers. comm.), and 117 occupied territories in California (Maria Boroja, FWS, pers. comm.).

The California bald eagle nesting population has increased in recent years from 40 occupied territories in 1977 to 116 occupied territories in 1995 (Jurek 1995, CDFG data), approximately

800 individuals are known to winter in California in a given year. The majority of nesting eagles occur in the northern one-third of the state, primarily on public lands. Seventy percent of nests surveyed in 1979 were located near reservoirs (Lehman 1979), and this trend has continued, with population increases occurring at several reservoirs since the time of that study. In southern California, nesting eagles occur at Big Bear Lake, Cachuma Lake, Lake Mathews, Nacimiento Reservoir, and San Antonio Reservoir (Zeiner et. al., 1990). The Klamath Basin in northern California and southern Oregon supports the largest wintering population of eagles in the lower 48 states, where up to 400 birds may congregate at one time. Scattered smaller groups of wintering eagles occur throughout the State near reservoirs, and typically in close proximity to large concentrations of overwintering migratory waterfowl. Clear Lake, Lake County, may support up to 60 wintering eagles and is a mercury-impaired water body. San Antonio Reservoir has become an important wintering area for bald eagles. An estimate of 50+ eagles regularly winter there. Lake Nacimiento also supports as many as 14 wintering eagles, and is an identified mercury-impaired water of the State. Women are precautioned against consuming any large mouth bass and no one should eat more than 24 ounces of large mouth bass per month from this lake (Cal EPA public health warnings). The observed increase in populations is believed to be the result of a number of protective measures enacted throughout the range of the species since the early 1970s. These measures included the banning of the pesticide DDT, stringent protection of nest sites, and protection from shooting.

<u>Reasons for Decline and Threats to Survival</u>: The species has suffered population declines throughout most of its range, including California, due primarily to habitat loss, shooting, and environmental pollution (Snow 1973, Detrich 1986, Stalmaster 1987). The use of DDT and its accumulation caused thin shelled eggs in many predatory birds. After the ban of DDT and other organochlorine compounds, the bald eagle populations started to rebound (USDI-FWS 1986a).

Other environmental contaminants represent potentially significant threats to bald eagles. Dioxin, endrin, heptachlor epoxide, mercury, and polychlorinated biphenyls (PCB’s) still occur in eagle food supplies; however, their overall effects on eagle populations are poorly understood (USDI-FWS, 1986a).

Bald eagles are sensitive to human disturbances such as recreational activities, home sites, campgrounds, mines, and timber harvest (Thelander 1973, Stalmaster 1976) when roosting, foraging, and nesting areas are located near these sites. The bald eagle is protected under the Migratory Bird Treaty Act of 1918, as amended (16 U.S.C. §§ 703-712) and the Bald Eagle Protection Act of 1940, as amended (16 USC §§ 668-668d).

Olendorff and Lehman (1986) collected reports of bald eagles colliding with transmission lines from around the world and covering the period from 1965-1985. The reported mortality rate for bald eagles was 87%. Olendorff and Lehman (1986) suggest that the heavy weight of eagles could be a factor in the higher mortalities for eagles than for other smaller buteos. Olendorff *et al*. (1986) observed eagle flight patterns in wintering areas in the vicinity of proposed transmission line routes in California. Eagles were observed flying through drainages, canyons and saddles, across low ridges, over valleys, and were concentrated above high ridges. Eagles usually flew above 100 feet from the ground (Olendorff *et al*. 1986).

California Brown Pelican (*Pelecanus occidentalis californicus*)

Species Description and Life History: The brown pelican was federally listed as endangered in 1970 (35 **FR** 16047). The recovery plan describes the biology, reasons for decline, and the actions needed for recovery of the California brown pelican (USDI-FWS 1983).

The brown pelican is a large bird recognized by the long, pouched bill. Brown pelicans nest in colonies on small coastal islands that are free of mammalian predators and human disturbance, and are associated with an adequate and consistent food supply. During the non-breeding season brown pelicans roost communally, generally in areas that are near adequate food supplies, have some type of physical barrier to predation and disturbance, and provide some protection from environmental stresses such as wind and high surf.

Foraging Ecology: The brown pelican uses its pouched bill to catch surface schooling fishes by plunge-diving into the water. The brown pelican feeds exclusively on small schooling animals found in the marine environment. Species that occur in Salton Sea that may serve as pelican prey are *Tilapia* sp., juvenile orange mouth corvina (*Cynoscionxanthalus sp*), sailfin mollies (*Poecilia latipinna*), red shiner (*Notropis umbratilis*), and mosquito fish (*Gambusia sp.*).

Historic and Current Distribution: Nesting colonies range from the Channel Islands in the Southern California Bight to the islands off Nayarit, Mexico. Prior to 1959, intermittent nesting was observed as far north as Point Lobos in Monterey County, California. Dispersal between breeding seasons ranges from British Columbia, Canada, to southern Mexico and possibly to Central America. Variable numbers of brown pelicans also occur at the Salton Sea, Imperial County, California, with maximum numbers present in late July and August (Small 1994). Limited numbers of brown pelicans are known to occasionally winter there (Small 1994). Breeding at the Salton Sea has been recorded only once (16 nests in 1996) at this inland location (Gress, pers. comm. 1996). During the non-breeding season California brown pelicans roost communally, generally near areas with adequate food supplies, physical barriers that offer protection from predation, human disturbance, and environmental stressors such as high surf, and high winds.

Reasons for Decline and Threats to Survival: Brown pelicans experienced widespread reproductive failures in the 1960s and early 1970s. Much of the failure was attributed to eggshell thinning caused by high concentrations of DDE, a metabolite of DDT. Since the listing of the species the EPA has banned the use of DDT in the United States (37 **FR** 13369). Restrictions that banned use of aldrin and dieldrin were imposed in the United States (39 **FR** 37246). Following this ban, the production of California brown pelicans increased and was correlated with an increase in eggshell thickness (Anderson *et al*., 1975). Decline of DDE residues in California brown pelicans began leveling off in 1972, and the improvement reproductive success began stabilizing in 1974 (Anderson *et al.*, 1977). Other factors implicated in the decline of this subspecies include human disturbance at nesting colonies and food shortages. Brown pelicans have nested sporadically on Bird Island, north of the Channel Islands, since the subspecies' decline in the late 1950s and early 1960s. Oil spills pose a threat to both breeding and wintering birds.

Large die offs, such as those that have occurred at the Salton Sea may have a direct impact on populations of pelicans that nest in the Gulf of California. Long term effects of large die-offs have the potential to effect numbers of pelicans available for dispersal and ultimate recruitment to the Southern California Bight breeding populations.

California Clapper Rail (*Rallus longirostris obsoletus*)

Species Description and Life History: The California clapper rail was federally listed as endangered in 1970 (35 **FR** 1604). A detailed account of the taxonomy, ecology, and biology of the California clapper rail is presented in the approved Recovery Plan for this species (USDI-FWS 1984b). Supplemental information is provided below. Clapper rails are non-migratory and are year-round residents of San Francisco Bay tidal marshes. Evans and Page (1983) concluded from research in a north San Francisco Bay marsh that the clapper rail breeding season, including pair bonding and nest construction, may begin as early as February. Field observations in south San Francisco Bay marshes suggest that pair formation also occurs in February in some areas (J. Takekawa, pers. comm.). The clapper rail breeding season has two nesting peaks, one between mid-April and early-May and another between late-June and early-July. Harvey (1988) and Foerster *et al.* (1990) reported mean clutch sizes of 7.27 and 7.47 for clapper rails, respectively. The end of the breeding season is typically defined as the end of August, which corresponds with the time when eggs laid during renesting attempts have hatched and young are mobile.

Foraging Ecology: California clapper rails forage primarily on benthic invertebrates (J. Albertson, pers. comm.; Eddleman and Conway 1994; Varoujean 1972; Test and Test 1942; Moffitt 1941; Applegarth 1938; Williams 1929). The non-migratory nature of the California clapper rail makes them extremely vulnerable to local contamination. A significant portion of the reported prey include algal and detrital foragers, and filter feeders, including bivalves (i.e. *Macoma balthica*, *Ischadium demissum*), crabs (i.e. *Pachygrapsus crassipes*), amphipods, and polychaetes (i.e. *Nereis vexillosa*).

Historic and Current Distribution: Of the 193,800 acres of tidal marsh that bordered San Francisco Bay in 1850, about 30,100 acres currently remain (Dedrick 1993). This represents an 84 percent reduction from historical conditions. Furthermore, a number of factors influencing remaining tidal marshes limit their habitat values for clapper rails. Much of the east San Francisco Bay shoreline from San Leandro to Calaveras Point is rapidly eroding, and many marshes along this shoreline could lose their clapper rail populations in the future, if they have not already. In addition, an estimated 600 acres of former salt marsh along Coyote Creek, Alviso Slough, and Guadalupe Slough, has been converted to fresh- and brackish-water vegetation due to freshwater discharge from south San Francisco Bay wastewater facilities and is of lower quality for clapper rails. This conversion has at least temporarily stabilized as a result of the drought since the early 1990s.

The suitability of many marshes for clapper rails is further limited, and in some cases precluded, by their small size, fragmentation, and lack of tidal channel systems and other micro-habitat features. These limitations render much of the remaining tidal marsh acreage unsuitable or of low value for the species. In addition, tidal amplitudes are much greater in the south Bay than in San Pablo or Suisun bays (Atwater *et al.* 1979). Consequently, many tidal marshes are completely submerged during high tides and lack sufficient escape habitat, likely resulting in nesting failures and high rates of predation. The reductions in carrying capacity in existing marshes necessitate the restoration of larger tracts of habitat to maintain stable populations.

The clapper rail population is estimated to be approximately 500 to 600 individuals in the southern portion of San Francisco Bay, while a conservative estimate of the north San Francisco Bay population, including Suisun Bay, is 195 to 282 pairs. Historic populations at Humboldt Bay, Elkhorn Slough, and Morro Bay are now extinct; therefore, 30,100 acres of tidal marsh remaining in San Francisco Bay represent the current distribution of this subspecies.

Reasons for Decline and Threats to Survival: As described above, the clapper rail's initial decline resulted from habitat loss and degradation, and reduction in range. Throughout San Francisco Bay, the remaining clapper rail population is besieged by a suite of mammalian and avian predators. At least 12 native and 3 non-native predator species are known to prey on various life stages of the clapper rail (Albertson 1995). Artificially high local populations of native predators, especially raccoons, result as development occurs in the habitat of these predators around the Bay margins (J. Takekawa, pers. comm.). Encroaching development not only displaces lower order predators from their natural habitat, but also adversely affects higher order predators, such as coyotes, which would normally limit population levels of lower order native and non-native predators, especially red foxes (Albertson 1995).

Hunting intensity and efficiency by raptors on clapper rails also is increased by electric power transmission lines, which criss-cross tidal marshes and provide otherwise-limited hunting perches (J. Takekawa, pers. comm.). Non-native Norway rats (*Rattus norvegicus*) long have been known to be effective predators of clapper rail nests (DeGroot 1927, Harvey 1988, Foerster *et al.* 1990). Placement of shoreline riprap favors rat populations, which results in greater predation pressure on clapper rails in certain marshes. These predation impacts are exacerbated by a reduction in high marsh and natural high tide cover in marshes.

The proliferation of non-native red foxes into tidal marshes of the south San Francisco Bay since 1986 has had a profound effect on clapper rail populations. As a result of the rapid decline and almost complete elimination of rail populations in certain marshes, the San Francisco Bay National Wildlife Refuge implemented a predator management plan in 1991 (Foerster and Takekawa 1991) with an ultimate goal of increasing rail population levels and nesting success through management of red fox predation. This program has proven successful in increasing the overall south San Francisco Bay populations from an all-time low (see below); however, it has been difficult to effectively conduct predator management over such a large area as the south San Francisco Bay, especially with the many constraints associated with conducting the work in urban environments (J. Takekawa, pers. comm.).

Predator management for clapper rails is not being regularly practiced in the north San Francisco Bay, and rail populations in this area remain susceptible to red fox predation. Red fox activity has been documented west of the Petaluma River and along Dutchman Slough at Cullinan Ranch (J. Collins, pers. comm.). Along Wildcat Creek near Richmond, where recent red fox activity has been observed, the rail population level in one tidal marsh area has declined considerably since 1987 (J. Evens, pers. comm.), even though limited red fox management was performed in 1992 and 1993 (J. Takekawa, pers. comm.).

California Least Tern (*Sterna antillarum browni*)

Species Description and Life History: The California least tern (least tern) was listed as endangered on October 13, 1970 (35 **FR** 16047). A detailed account of the taxonomy, ecology, and biology of the least tern is presented in the approved Recovery Plan for this species (USDI-FWS 1980). The Service is currently developing an updated recovery plan, which incorporates information gathered since the publication of the first Recovery Plan (USDI-FWS 1980). Supplemental or updated information is provided below.

California least terns are migratory. They arrive in California in April to breed and depart to wintering areas in Central and South America by the end of September. Little is known about least tern wintering areas. While in California, least tern adults court, mate, and select nest sites; lay, incubate, and hatch eggs; and raise young to fledging prior to departing from the breeding site.

After their eggs hatch, breeding adults catch and deliver small fish to the flightless young. The adults shift their foraging strategy when chicks hatch in order to obtain the very small sized fish for nestlings (Collins *et al.* 1979, Massey 1988). The young begin to fly at about 20 days of age, but continue to be fed and are taught how to feed by their parents for some time after fledging. Reproductive success is, therefore, closely related to the availability of undisturbed nest sites and nearby waters with adequate supplies of appropriately sized fishes
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Terns typically employ a shallow plunge dive technique to capture fish immediately below the water's surface. Adults usually dive from a hover but occasionally dive directly from flight. Most foraging activity is conducted within a couple miles of the colony (Atwood and Minsky 1983).

California Least Terns are opportunistic in their foraging strategy and are known to take many different species of fish. However, they seem to select fish based on certain morphological characteristics. Massey and Atwood (1981) conclude that prey items are generally less than 9 cm in length and have a body depth of less than 1.5 cm.

Once their eggs hatch, the adult terns must feed their young as well as themselves. The adults shift their foraging strategy when chicks hatch in order to obtain the small fish for nestlings (Collins *et al.* 1979, Massey 1986). The adult terns begin foraging nearer the colony and in water with an abundance of small prey fish.

The adult tern does not dismember larger fish in order to feed its small chick. The adult captures a fish and disables it by shaking, and delivers it whole to the chick. A small, newly hatched least tern chick cannot swallow a fish that is too large or relatively deepbodied. The chicks can only eat small, elongated fish. Despite an abundance of larger fish that may be preferred food for an adult Least Tern, an inadequate supply of smaller fish will reduce chick survival.

After fledging, the young terns do not become fully proficient at capturing fish until after they migrate from the breeding grounds. Consequently, parents continue to feed their young even after they are strong fliers.

Foraging Ecology: Least terns feed exclusively on small fishes captured in shallow, nearshore waters, particularly at or near estuaries and river mouths (Massey 1974, Collins *et al.* 1979, Massey and Atwood 1981a, 1984, Atwood and Minsky 1983, Atwood and Kelly 1984, Minsky 1984, Bailey 1984). While in California during the breeding season, least terns forage for fish in nearshore waters which are generally productive foraging habitat areas. Collins (1995) summarized least tern prey selection studies conducted at Naval Air Station (NAS) Alameda from 1981 through 1995. Researchers counted fish, by species, dropped by least terns flying between foraging and nesting areas. Although studies of dropped fish do not provide direct evidence of prey consumed, they do provide a good indication of least tern diets. Least terns dropped larvae and juveniles of nearly 30 species; however, northern anchovy (*Engraulis mordax*) and silversides (*Atherinidae spp.*) comprised 25% and 60% of all dropped fish, respectively. Silversides included topsmelt (*Atherinops affinis*) and jacksmelt (*Atherinopsis californiensis*). Shiner surf-perch (*Cymatogaster agregata*) comprised approximately 5% of the tern's diet.

Thirty-seven different species of fish dropped by the least tern while breeding at the Venice Beach nesting site, next to the Ballona Creek Channel, Marina del Rey marina in Santa Monica Bay, were recorded by Massey and Atwood (1981). At Venice Beach and Huntington Beach in Orange County next to the Santa Ana River mouth, in 1978-81, northern anchovy (*Engraulis mordax*) and silversides including topsmelt (*Antherinops affinis*), jacksmelt (*Atherinopsis californiensis*), and California grunion (*Leuresthes tenuis*) composed most of the samples of fish found dropped in the nesting areas as well as most of the actually documented food items (Atwood and Kelly 1984). Very small or soft scaled species such as gobies (especially *Clevelandia ios, Quietula y-cauda*, and *Ilypnus gilberti*) are under represented in dropped fish surveys.

The larval and yearling sizes of anchovies and silversides fall well within the size range of fish taken by least terns. Northern anchovy are a planktivorous, schooling fish that broadcast-spawn in the Bay. Larvae begin schooling at 1.1-1.2 cm in length, and larvae and juveniles form tightly packed schools in nearshore areas. Topsmelt are a schooling fish that have a prolonged spawning period from April through October, with a peak in May and June. Moyle (1976a) described topsmelt as bottom feeding omnivores, based upon the organisms, detritus, and sand grains found in their stomachs. Stomach content analyses describe topsmelt diets as consisting of diatoms and filamentous algae (50% by volume), detritus (29%), chironomid midge larvae (10%), and amphipods (10%). Jacksmelt are omnivorous, schooling fish that spawn in late winter and early spring. Large schools of juveniles remain in the Bay through the summer, emigrating to coastal waters in the fall. Juvenile jacksmelt foraging behavior, described by Bane and Bane (1971), is similar to that of topsmelt. Jacksmelt juveniles are bottom feeding omnivores, primarily feeding on algae, detritus, small crustaceans, and amphipods. California least terns can therefore be considered exclusive consumers of trophic level 3 fish.

<u>Historic and Current Distribution</u>: The California least tern continues to occupy nesting sites distributed throughout its historic range. The historic breeding range extended along the Pacific Coast from Moss Landing, Monterey County, California, to San Jose del Cabo, southern Baja California, Mexico (A.O.U 1957, Dawson 1924, Grinnell 1928, Grinnell and Miller 1944). However, least terns were nesting several miles north of Moss Landing at the mouth of the Pajaro River, Santa Cruz County, California, at least from 1939 (W.E. Unglish, Western Foundation of Vertebrate Zoology egg collection) to 1954 (Pray 1954); and although nesting at San Francisco Bay was not confirmed until 1967 (Chandik and Baldridge 1967), numerous spring and summer records for the area suggest nesting may have occurred previously (Allen 1934, Chase and Paxton 1965, Grinnel and Wythe 1927, Sibley 1952). Since 1970, nesting sites have been documented in California from San Francisco Bay to the Tijuana River at the Mexican Border; and in Baja California from Ensenada to San Jose del Cabo at the tip of the peninsula.

There are no reliable estimates describing the historic numbers of California least terns along the Pacific Coast (USDI-FWS 1980). Early accounts describe the existence of substantial colonies along the southern and central California coast (Grinnell 1898; McCormick 1899, as cited in Bent 1921), including a colony of about 600 breeding pairs along a 3-mile stretch of beach in San Diego County (Shepardson 1909). At the time of its Federal listing as endangered in 1970, the U.S. population of the California least tern was estimated to be 600 breeding pairs (Fancher 1992). The dramatic decline in breeding least terns has been attributed to the degradation and loss of breeding sites, colonies, and foraging areas, which resulted from human development and disturbance, and pollution (USDI-FWS 1980).

Since its listing, the statewide population of the least tern has recovered to an estimated 4,009 breeding pairs in 1997 (Ron Jurek, pers. comm). Despite this dramatic increase in breeding pairs, statewide monitoring has revealed threats to the least tern which emphasizes the importance of demography to the least tern's survival and recovery. In 1983, for example, the presence of predators caused most of the NAS Alameda colony to attempt to breed at the Oakland Airport site, where 61 nesting pairs produced only 8 fledglings. This event and other stuff at other colony/nest sites has highlighted the importance of multiple nesting sites available to a colony. The effects of El Nino years on southern CA colonies has highlighted the significance of multiple clusters, distributed along the coast.

The current U.S. population of the California least tern is grouped into 5 geographically discrete clusters, which support multiple active and historic breeding sites. These clusters include: (1) San Diego County, (2) Los Angeles/Orange Counties, (3) Ventura County, (4) San Luis Obispo/Santa Barbara Counties, and (5) San Francisco Bay area. The maintenance of multiple viable clusters and multiple breeding sites within them is important to the least tern's survival and recovery.

San Diego County The San Diego County cluster includes 24 active nest sites and supports the majority of the U.S. population of the California least tern. The active nest sites and number of pairs recorded in 1997 (in parentheses) include White Beach (17), three sites at the Santa Margarita River mouth (728, 41, and 39), five sites in Batiquitos Lagoon (83, 59, 25, 0, and 104), San Elijo Lagoon (9), three sites in Mission Bay (20, 268, and 76), nine sites in San Diego Bay (0, 102, 22, 310, 15, 85, 0, 38, and 36), and the Tijuana River Estuary (211). Least tern foraging has been studied at Mission Bay (ERC 1989, SWRI 1994). Least tern foraging studies or observations in San Diego Bay indicate a very significant reliance upon the Bay’s tidal waters (Baird 1993, 1995, Manning 1995). While virtually every coastal area of southern California is vulnerable to exposure to toxic or environmentally contaminating discharges from the intense industrializing/urbanizing influences, San Diego Bay has been particularly developed as a commercial port, major U.S. Navy homeport, and industrial area.

Los Angeles/Orange Counties The Los Angeles County/Orange County cluster includes active nest sites at Venice Beach, Pier 300 (Terminal Island), Pier 400 and TC2 (new harbor sites), Seal Beach National Wildlife Refuge, Bolsa Chica, Huntington Beach, and Upper Newport Bay. In 1997, these sites supported 375, 4, 76, 178, 141, 373, and 82 nests, respectively. Atwood and Minsky (1983) studied the foraging patterns of breeding least terns at Huntington Beach and Venice Beach nesting colonies. Drainage channels from highly urbanized areas discharge near or directly into the least tern foraging areas. San Pedro Bay has been the focus of foraging studies of least terns nesting at the Terminal Island colony (MEC 1988, Keane 1997). The least tern relies upon fish captured in the nearshore zone, and in tidal sloughs and relatively shallow bodies of water that support large numbers of small fish. In highly urban LA and Orange Counties, these are water bodies under the influence of a very wide variety of industrial discharges, particularly San Pedro Bay which is also a commercial port and highly industrialized area.

San Luis Obispo/Santa Barbara Counties The San Luis Obispo County/Santa Barbara County cluster includes active least tern nest sites at Oceano (Pismo) Dunes State Vehicular Recreational Area, Mussel Rock (Guadalupe) Dunes, and Beach 2 and Purisima Point at Vandenberg Air Force Base. In 1997, these sites supported 6, 30, 3, and 25 nesting pairs, respectively. In this portion of their range California least terns are known to forage in the Santa Ynez and Santa

Maria River lagoons in the Pacific Ocean. Least terns also stage at area lagoons prior to post-breeding dispersal.

Ventura County The Ventura County cluster includes seven nest sites at three locations: Point Magu Naval Air Station, Ormond Beach, and McGrath State Beach at the Santa Clara River mouth. In 1997, these three locations supported approximately 74, 63, and 43 nesting pairs, respectively. In this portion of their range California least terns are known to forage in the Ormond, Ventura, and Santa Clara River Lagoons, Mugu Lagoon, Revolon Slough, and in the slough near the Mandalay Generating Station. Least terns also stage at area lagoons prior to post-breeding dispersal.

San Francisco Bay In the San Francisco Bay, least terns have nested at 6 sites in Contra Costa, Alameda, and San Mateo Counties. Most sites in the San Francisco Bay have not been used by breeding least terns in recent years. Presently, only NAS Alameda supports significant numbers of nesting pairs. There are two other minor least tern breeding sites that remain in the San Francisco Bay area, but the Oakland Airport site has not been used in years and the PG&E Pittsburg site supports only 1 to 4 pairs each year, including 4 pairs in 1997. Therefore, the NAS Alameda site currently represents the entire San Francisco Bay area population, and is the most northern of least tern breeding colonies by about 178 miles. Because of its northern location, the NAS Alameda site is relatively unaffected during El Nino years when many southern California sites experience pronounced breeding failure resulting from limited food availability. In the most recent El Nino year, 1992, the NAS Alameda site supported 6 percent of the statewide number of breeding pairs, but produced 16 percent of the total statewide number of fledglings.

According to Caffrey (1995), the least tern breeding site at NAS Alameda has played a significant role in recent increases in the number of least terns throughout California. The NAS Alameda site is consistently one of the most successful sites in California. Between 1987 and 1994, the NAS Alameda site supported 5 to 6 percent of the statewide breeding population out of 35 to 40 sites each year, but produced an average of 10.6 percent of the total number of fledglings produced statewide in each of those years. In 1997, an estimated 244 pairs of least terns nested at the colony out of a total population of over 4,000 nesting pairs at 37 breeding sites along the California and Baja California coasts. In 1997, an estimated 316 young fledged successfully at NAS Alameda; this represented 10.1 percent of the total number of fledglings produced throughout California that year. By consistently producing large numbers of fledglings each year, the colony has added large numbers of potential new breeding birds to the statewide population. Therefore, this site is considered to be one of the most important "source" populations in California serving to balance out losses at many "sink" locations throughout the state.

In San Francisco Bay, post-breeding adults and fledglings move to South San Francisco Bay salt ponds where they may remain for several weeks prior to migrating south (Feeney and Collins 1988, Collins 1989).

Reasons for Decline and Threats to Survival: California least terns were once common along the central and southern California coast. The decline of the California least tern is attributed to prolonged and widespread destruction and degradation of nesting and foraging habitats, and increasing human disturbance to breeding colonies. Conflicting uses of southern and central California beaches during the California least tern nesting season have led to isolated colony sites that are extremely vulnerable to predation from native, feral and exotic species, overwash by high tides, and vandalism and harassment by beach users. Since its classification as a Federal and State endangered species, considerable effort has been expended on annual population surveys, protection and enhancement of existing nesting colonies, and the establishment of new nesting locations. Control of predators constitutes one of the most crucial management responsibilities at California least tern nesting sites.

An important aspect of recovery is the protection of coastal feeding grounds of colonies by maintaining high water quality and preventing tideland fill and drainage projects. Protection of non-nesting, feeding, and roosting habitats from detrimental land or water use changes in San Diego and Los Angeles County is also important for recovery (USDI-FWS 1980).

Light-footed Clapper Rail (*Rallus longirostris levipe*)

Species Description and Life History: The light-footed clapper rail was listed as an endangered species on October 13, 1970 (35 **FR** 16047). A recovery plan for the species was issued in 1979 and revised in 1985 (USDI-FWS 1985a). This recovery plan describes the biology, reasons for decline, and the actions needed for recovery of light-footed clapper rails populations in California (USDI-FWS 1985a). The light-footed clapper rail's coloration blends with the dense stands of cordgrass (*Spartina foliosa)* dominating its preferred habitat in coastal salt or brackish water marshes. Male rails are approximately 12 inches in length and are slightly larger and more colorful than females. The birds are tawny-breasted with gray-brown backs, vertical white bars on the flanks and show whitish coloration under the short tail, on the chin, and over the eye. The rails' bills are mostly orange and the birds' legs and feet are largely brownish.

Rails breed from mid-March to mid-August, usually selecting dense stands of cordgrass (*Spartina foliosa*) as a nest site, although nest are occasionally observed in pickleweed (*Salicornia virginica*) or other marsh type vegetation. In addition to a brood nest, pairs usually build a number of nests, secured in to surrounding vegetation, to serve as refuges from high tides. Males and females usually share the responsibility for incubation of 4-10 eggs, which hatch in 18-27 days. Hatchling rails are covered in black down and are able to follow along after the adults in the marsh within a few hours of hatching. The young rails are dependent upon the adults for several weeks and are still being fed occasionally up to at least 6 weeks of age (Zembal 1989). Light-footed rails spend much of their time in lower salt marsh habitat, particularly in cordgrass. Although this plant species provides preferred nesting substrates, nest are also built in common pickleweed and other upper marsh plants on hummocks of high ground surrounded by low marsh (Massey *et al.* 1984).

Limited evidence exists for intermarsh movements by rails; this bird is resident in its home marsh except under unusual circumstances. Within-marsh movements are also confined and generally of no greater spread than 400 meters. Minimum home range sizes for 9 rails that were radio-harnessed for telemetry at Upper Newport Bay varied from approximately 0.8 to 4.1 acres. The larger areas and daily movements were by first-year birds attempting to claim their first breeding territories (Zembal 1989).

Foraging Ecology: The rail is an opportunistic omnivore. A wide variety of mostly animal foods is consumed using many different foraging strategies including gleaning, probing, crab hunting, fishing, and scavenging. Over 90% of the observed foraging has been of rails executing hundreds of gleans and usually shallow probes over the marsh substrate per hour and consuming hundreds of prey items. However, crabs are important in the diet, too, along with snails, insects, and invertebrates. Plant foods are uncommon (Zembal 1989).

Historic and Current Distribution: The light-footed clapper rail is a resident of coastal marshes, ranging historically from Carpinteria Marsh in Santa Barbara County, California south to San Quintin, Baja California, Mexico. The current distribution of the light-footed clapper rail is limited to Upper Newport Bay, Anaheim Bay, Tijuana Slough National Wildlife Refuge, and Mugu lagoon. The spring counts in 1997 revealed 307 pairs of rails in 16 marshes in California. Of this total, 48.5 percent of the rails were in Upper Newport Bay, Orange County, California (Zembal unpublished data, 1997).

Reasons for Decline and Threats to Survival: The destruction and degradation of habitat led to small, isolated subpopulations and prompted the listing of this species. The United States population has been censused annually over the past decade and the downward trend has continued. The spring counts in 1989 revealed only 163 pairs of rails in 8 marshes in California. Of this total, 116 pairs or 71.2 percent of were in Upper Newport Bay, Orange County, California (Zembal 1990). The one hundred thirty-six pairs detected in Upper Newport Bay in 1992 (Zembal 1993) may closely approach the maximum number of pairs that can be accommodated at this locale (Richard Zembal, personal communication, 1993).

Marbled Murrelet (*Brachyramphus marmoratus*)

Species Description and Life History: The marbled murrelet was federally listed as a threatened species in Washington, Oregon and California on September 28, 1992 (57 **FR** 45328), primarily due to loss of nesting habitat. The final recovery plan was released in 1997 (USDI-FWS 1997b). Critical habitat was designated in 1996 to include 32 critical habitat units (CHU's) in Washington, Oregon, and California, primarily on Federal lands. Primary constituent elements of the CHU's include 1) individual trees with potential nesting platforms, and 2) forested areas within 0.8 kilometers (0.5 miles) of individual trees with potential nesting platforms and a canopy height of at least one-half the site-potential tree height.

The Recovery Plan for the Marbled Murrelet (USDI-FWS, 1997) establishes six conservation zones for the species throughout its range in Washington, Oregon, and California. Conservation zones 4-6 are located in California. Narratives for each of these zones are included in the recovery plan. Conservation zone four, the Siskiyou Coast Range Zone, extends from North Bend, Oregon to the southern end of Humboldt Bay, California. Conservation zone five, Mendocino Zone, extends from the southern end of Humboldt Bay to the mouth of San Francisco Bay. Zone six, the Santa Cruz Mountains Zone, extends from the mouth of San Francisco Bay to Point Sur, Monterey County. Each of these zones include all nearshore waters, as previously defined, within 1.2 miles of the Pacific shoreline. Waters impacted by the CTR include all freshwater, and estuarine ecosystems coincidental with these conservation zones, including Humboldt, San Francisco, Tomales, Bodega, Half Moon, and Monterey Bays.

The marbled murrelet is a small diving seabird that breeds along the Pacific coast of North America from the Aleutian Archipelago and southern Alaska south to central California (USDI-FWS 1997b). The marbled murrelet is the only member of the Alcidae family known to nest in trees. Preferred nesting habitat for the species is characteristically old-growth, coniferous forests within 50 miles of the coast. Nesting stand characteristics include large, old trees, generally greater than 32 inches diameter at breast height (dbh), with large limbs which provide nest platforms. Nest are typically located near the bole of the tree and are simple depressions sometimes located in clumps of moss and lichens.

Marbled murrelets nest in old-growth forests, generally characterized by large trees ($\geq$ 32 inches dbh), multiple canopy layers, and moderate to high canopy closure. As of April 2, 1996, at least 95 active or previously used tree nests were located in North America: 9 in Washington, 41 in Oregon, and 12 in California (K. Nelson, pers. Comm. 1996; Binford *et al.* 1975; Varoujean *et al.* 1989; Quinlan and Hughes 1990; Hamer and Cummins 1990, 1991; Kuletz 1991; Singer *et al.* 1991, 1992; Hamer and Nelson 1995). All nests in Washington, Oregon, and California were located in old-growth trees that were greater than 32 inches dbh. Most nests were located on large or deformed, moss covered branches; however, a few nests were located on smaller branches, and some nests were situated on duff platforms composed of conifer needles or sticks rather than moss. Such locations allow easy access to the exterior of the forest and provide shelter from potential predators. Nest sites in California were located in stands containing old-growth redwood (*Sequoia sempervirens*) and Douglas-fir. Nest sites in Oregon and Washington were located in stands dominated by Douglas-fir, western hemlock (*Tsuga heterophylla*), and Sitka spruce (*Picea sitchensis*). Suitable marbled murrelet habitat is defined as forest stands with conditions that will support nesting marbled murrelets.

Marbled murrelets appear to be solitary in their nesting and feeding habits, but interact in groups over the forest and at sea (Sealy and Carter 1984, Carter and Sealy 1990, Nelson and Hamer 1995a). They lay on one egg on the limb of a large coniferous tree. Incubation lasts 30 days and fledging takes 28 days. Both sexes incubate the egg (Nelson and Hammer 1995a, Nelson and Peck 1995, Simons 1980, Singer *et al.* 1991, 1992).

Foraging Ecology: The marbled murrelet forages almost exclusively in the nearshore environment, including bays, estuaries, and island groups. Adult marbled murrelets forage on a variety of aquatic organisms including: Pacific sand lance (*Ammodytes hexapterus*), Pacific herring (*Clupea harengus*), northern anchovy (*Engraulis mordax*), capelin (*Clupea* spp.), and smelts (family *Osmeridae*), as well as invertebrates such as *Euphausia pacifica* and *Thysanoessa spinfera*. In the early 1900's, Pacific sardines (*Sardinops sagax*) were also documents as prey in California. Adults, subadults, and hatching year birds feed primarily on larval and juvenile fish, whereas nestlings are most commonly fed larger second year fish. The sand lance is the most common food of the marbled murrelet across its range, comprising up to 52% of the observed prey items, anchovy and herring comprised roughly 29% of observed prey items, and Osmerids comprised the remaining 24% of prey item observations (Burkett 1995). The species is an opportunistic forager, relying on numerous species of fish taken in the nearshore environment. This strategy is believed to have sustained the species after declines in historic prey species (Ralph et al 1995, USDI-FWS 1997b). Marbled murrelets will also forage in fresh water lakes on salmonid fry, fingerlings, and yearlings (Carter and Sealy 1986).

During the breeding season, the marbled murrelet tends to forage in well-defined areas along the coast in relatively shallow marine waters, including enclosed bays and estuaries.

Historic and Current Distribution: The historic distribution of the marbled murrelet within the listed range was continuous in nearshore waters and in coniferous forests near the coast from the Canadian border south to Point Sur, Monterey County, California. Current breeding populations are discontinuous and concentrated at sea in areas adjacent to remaining late-successional, coastal, coniferous forests. Off the California coast, marbled murrelets are concentrated in two areas at sea, corresponding to the three largest remaining blocks of older, coastal forest. These blocks of older forest are separated by areas of little or no habitat, which correspond to locations at sea where few marbled murrelets are found. A large gap (about 300 miles) occurs in the southern portion of the marbled murrelet’s breeding range, from San Mateo and Santa Cruz counties north to Humboldt and Del Norte counties, California. Marbled murrelets likely occurred in the gap prior to extensive logging of redwood forests (Paton and Ralph 1988).

Estimates of the marbled murrelet population size in California are based on research over the past 15 years. In 1979-1980, the breeding population was estimated to be about 2,000 birds, based on data collected while conducting surveys of other seabird colonies (Sowls *et al.* 1980). Utilizing Sowls’ data and similar information collected in 1989, Carter and Erickson (1992) and Carter *et al.* (1990) estimated the breeding population at 1,650 to 1,821 birds. Ralph and Miller (1995) conducted more intensive at-sea surveys in small portions of the murrelet’s range in northern California from 1989 to 1993. These multi-year surveys, specifically designed to estimate population size in California, used different methods and assumptions and estimate a total State population size of approximately 6,000 breeding and non-breeding birds. Ralph and Miller, however, extrapolated results from small areas to estimate numbers of murrelets over much larger areas; the result may be an overestimation of murrelet population size, given the non-uniform distribution of murrelets at sea.

Marbled murrelet populations in California, Oregon, and Washington apparently are declining rapidly. Current estimates of nesting success and recruitment are well below levels required to sustain populations in the Pacific Northwest (USDI-FWS 1997b). A population model which analyzed likely ranges of fecundity and survivorship estimated that murrelets population sizes in Washington, Oregon, and California are most likely declining at a rate between 4 and 6 percent per year (Beissinger 1995).

The distribution of the marbled murrelet in California is limited to three separate areas, primarily associated with remaining contiguous old growth forest habitat (Carter and Erickson 1992), Historically the species was plentiful during the winter months from Monterey county north to the Oregon border. Today the remaining populations of murrelets are disjunct and separated by great distances, largely the result of a lack of suitable breeding habitat. For further information on the status, distribution, and biology of the marbled murrelet refer to the *Ecology and Conservation of the Marbled Murrelet* (Ralph *et al.* 1995), Marshall 1988, and Carter and Morrison 1992.

Reasons for Decline and Threats to Survival: Suitable habitat has declined throughout the range of the marbled murrelet as a result of commercial timber harvest, with some loss attributable to natural disturbance such as fire and windthrow. Timber harvest has eliminated most suitable habitat on private lands within the three state area (Norse 1988, Thomas *et al.* 1990). A total of approximately 2,552,200 acres of suitable marbled murrelet habitat occur on Federal lands in California, Oregon, and Washington.

Marbled murrelet reproductive success may be adversely affected by forest fragmentation and associated effects from excessive amounts of edge. Fragmented forests can have higher numbers of predators that can adapt to the changing environment, leading to increased predation on murrelet nests that may be easier for a predator to locate in a fragmented forest. Relatively high observed predation rates are of great concern and have led the Service to conclude that maintenance and development of suitable habitat in relatively large contiguous blocks will contribute to the recovery of the murrelet (USDI-FWS 1997b).

Spills of oil and other pollutants along the coast of California, Oregon, and Washington can also do local harm to populations. The central California population of marbled murrelets is especially vulnerable to oil spill events. Changes in prey abundance from over-harvest, El Nino events, or pollution related deaths can also cause reproductive failure (USDI-FWS 1997b).

Industrial discharges from the population centers of San Francisco Bay, California, Puget Sound, Washington, and Vancouver, British Columbia, have contaminated estuarine sediments with heavy metals, petroleum hydrocarbons, and PCB. The major rivers with historic pollutant discharges in the murrelet range include the Sacramento-San Joaquin River System (Fry 1995).

Protection of the foraging areas is a critical component to a successful recovery strategy. The main threats to marbled murrelets identified in their marine habitat result in the loss of individuals through death or injury. Marbled murrelets are adversely affected by spills of oil and other pollutants. Given the essential role of the marine environment, protecting the quality of the marine environment and reducing adult and juvenile mortality in the marine environment are integral parts of the recovery effort. Important near-shore environments in California include Cape Mendocino to the Oregon border (including Humboldt and Arcata Bays, and river mouths of Smith, Eel, and Klamath Rivers and Redwood Creek), and central California from San Pedro Point south to the mouth of the Pajaro River, including the mouths of Pescadero and Waddell Creeks, as well as other creeks. Protection of areas where prey may concentrate should extend 2 km offshore and include estuaries, the mouths of bays, and eddies in the vicinity of headlands. Additionally prey breeding areas such as near-shore kelp beds, sand or gravel beaches, and sand banks should be protected (USDI-FWS 1997b).

Pacific Coast Population of the Western Snowy Plover (*Charadrius alexandrinus nivosus*)

Species Distribution and Life History: The Pacific coast population of the western snowy plover (plover) was federally listed as threatened on March 5, 1993 (50 **FR** 12864). A designation of critical habitat for the plover was federally proposed on March 2, 1995 (60 **FR** 11763), final critical habitat for the species was designated on January 6, 2000 (64 **FR** 68508).

The western snowy plover is a small shorebird that forages on invertebrates in areas such as intertidal zones, the wrack line, dry sandy areas above high tide line, salt pans, and the edge of salt marshes. The plover breeds primarily on coastal beaches from southern Washington to southern Baja California, Mexico. Other less common nesting habitat includes salt pans, coastal dredged spoil disposal sites, dry salt ponds, salt pond levees (Widrig 1980, Wilson 1980, Page and Stenzel 1981), and riverine gravel bars (Gary Lester, pers. comm.). Sand spits, dune-backed beaches, unvegetated beach strands, open areas around estuaries, and beaches at river mouths are the preferred coastal habitats for nesting (Stenzel *et al.* 1981, Wilson 1980).

Snowy plovers breed in colonies with the number of adults at coastal breeding sites ranging from 2 to 318 (Page and Stenzel 1981; Oregon Department of Fish and Wildlife 1994; Eric Cummins, pers. comm.). The breeding distribution is skewed towards the southern portion of the western snowy plover's range with the majority of breeding activity occurring in Ventura, Santa Barbara, San Luis Obispo, and Monterey counties (Ray Bransfield pers. comm. 1998). Nest sites typically occur in flat, open areas with sandy or saline substrates; vegetation and driftwood are usually sparse or absent (Widrig 1980, Wilson 1980, Stenzel *et al.* 1981). The majority of snowy plovers are site-faithful, returning to the same breeding site in subsequent breeding seasons (Warriner *et al.* 1986).

The breeding season of the coastal population of the western snowy plover extends from early March through late September. Nest initiation and egg laying occurs from mid March through mid July (Wilson 1980, Warriner *et al.* 1986). The usual clutch size is three eggs. Both sexes participate in incubation, which averages 27 days (Warriner *et al.* 1986). Plover chicks are precocious, leaving the nest within hours after hatching to search for food. Fledging (reaching flying age) requires an average of 31 days (Warriner *et al.* 1986). Broods rarely remain in the nesting territory until fledging (Warriner *et al.* 1986, Stern *et al.* 1990).

Snowy plovers will renest after loss of clutch or brood (Wilson 1980, Warriner *et al.* 1986). Double brooding and polygamy (i.e., the female successfully hatches more than one brood in a nesting season with different mates) have been observed in coastal California (Warriner *et al.* 1986) and also may occur in Oregon (Jacobs 1986). After loss of a clutch or brood or successful hatching of a nest, plovers may renest in the same site or move, sometimes up to several hundred miles, to other colony sites to nest (Gary Page, pers. comm.; Warriner *et al.* 1986).

Foraging Ecology: Snowy plovers forage on invertebrates in the wet sand and amongst surf cast kelp within the intertidal zone; in dry, sandy areas above the high tide; on salt pans; spoil sites; on mudflats; and along the edges of salt marshes and salt ponds. In San Francisco Bay, breeding plovers forage on invertebrates around salt ponds, and on nearby mudflats of tidal creeks and the Bay. Only anecdotal information exists on plover food habits. Page, *et al.* (1995) and Reeder (1951) listed known prey items of plovers on Pacific coast beaches and tidal flats: mole crabs (Emerita analoga), crabs (*Pachygrapsus crassipes*), polychaetes (Neridae, *Lumbrineris zonata*, *Polydora socialis*, *Scoloplos acmaceps*), amphipods (*Corophium* spp., *Ampithoe* spp., *Allorchestes angustus*, and sand hoppers [Orchestoidea]), tanadacians (*leptochelia dubia*, flies (Ephydridae, Dolichopodidae), beetles (Carabidae, Buprestidae, Tenebrionidae), clams (*Transenella* sp.), and ostracods. Feeney (1991) described plover prey items in salt evaporation ponds in South San Francisco Bay: flies (*Ephydra cinerea*), beetles (*Tanarthrus occidentalis*, *Bembidion* sp.), moths (*Perizoma custodiata*) and lepedopteran caterpillars.

Historic and Current Distribution: Snowy plovers occur along coastal beaches and estuaries from Washington to Baja California, Mexico. Based on the most recent surveys, a total of 28 snowy plover breeding sites or areas currently occur on the Pacific Coast of the United States. Two sites occur in southern Washington--one at Leadbetter Point, in Willapa Bay (Widrig 1980), and the other at Damon Point, in Grays Harbor (Anthony 1985). In Oregon, nesting birds were recorded in 6 locations in 1990 with 3 sites (Bayocean Spit, North Spit Coos Bay and spoils, and Bandon State Park-Floras Lake) supporting 81 percent of the total coastal nesting population (Oregon Department of Fish and Wildlife, unpubl. data, 1991). A total of 20 plover breeding areas currently occur in coastal California (Page *et al.* 1991). Eight areas support 78 percent of the California coastal breeding population: San Francisco Bay, Monterey Bay, Morro Bay, the Callendar-Mussel Rock Dunes area, the Point Sal to Point Conception area, the Oxnard lowland, Santa Rosa Island, and San Nicolas Island (Page *et al.* 1991).

The coastal population of the western snowy plover consists of both resident and migratory birds. Some birds winter in the same areas used for breeding (Warriner *et al.* 1986, Wilson-Jacobs, pers. comm. in Page *et al.* 1986). Other birds migrate either north or south to wintering areas (Warriner *et al.* 1986). Plovers occasionally winter in southern coastal Washington (Brittell *et al.* 1976), and about 70 plovers may winter in Oregon (Oregon Department of Fish and Wildlife 1994). The majority of birds, however, winter south of Bodega Bay, California (Page *et al.* 1986), and substantial numbers occur in the San Francisco Bay (Bay). Wintering coastal populations are augmented by individuals of the interior population that breed west of the Rocky Mountains (Page *et al.* 1986, Stern *et al.* 1988).

Reasons for Decline and Threats to Survival: Poor reproductive success, resulting from human disturbance, predation, and inclement weather, combined with permanent or long-term loss of nesting habitat to encroachment of introduced European beachgrass (*Ammophila arenaria*) and urban development has led to a decline in active nesting colonies, as well as an overall decline in the breeding and wintering population of the western snowy plover along the Pacific coast of the United States. Of the 87 historic breeding areas, only 28 remain (Page and Stenzel 1981; Charles Bruce, pers. comm.; E. Cummins, pers. comm.). The nesting population in the three states is estimated to be around 1,500 adults (Page *et al.*, 1991). Page and Stenzel (1981) estimated that the South Bay supports 10% of California's breeding snowy plovers, of which 90% can be found nesting in Alameda County salt pond systems.

Yuma Clapper Rail (*Rallus longirostris yumaensis*)

Species Description and Life History: The Yuma clapper rail was listed as endangered on March 11, 1967 (32 **FR** 4001). The Yuma clapper rail is a chicken-sized bird that is grayish-brown with a tawny breast and barred flanks. They prefer habitat that is densely vegetated with either cattails (*Typha* sp.) or giant bulrush (*Scirpus californicus*). Territories are generally in areas with a transition from standing water to saturated soils, but the presence of pond openings and flowing water are also important for foraging. Yuma clapper rails occur in fresh water marshes (e.g. cattail, alkali bulrush, and reed), within the vicinity of the Salton Sea and the Colorado River. This species is known to occur within agricultural drains which contain suitable habitat. Moreover, this species has been found to use extremely small patches of habitat within agricultural drains, patches which barely provide enough cover for concealment. Further information is found in Bennett and Ohmart 1978, Todd 1986, and Conway *et al.* 1993.

Foraging Ecology: The Yuma clapper rail has been documented to feed on a wide variety of invertebrates and some vegetation. Included in its diet are crayfish, fresh water prawns, weevils, isopods, clams, water beetles, leeches, damselfly nymphs, small fish, tadpoles, seeds and twigs. Based on the available information, crayfish appear to make up the majority of its food intake.

Historic and Current Distribution: The largest single breeding population of Yuma clapper rails in the United States is located in the Wister Unit of the California Department of Fish and Game's Imperial Wildlife Area. In the 1994 census, 309 individuals were located in the ponds of the Wister Unit (Steve Montgomery, SJM Biological Consultants, pers. comm.). In that same year, surveys of the Salton Sea National Wildlife Refuge and adjacent drainages located 95 individuals, most of which were breeding pairs (Ken Sturm, Salton Sea National Wildlife Refuge, pers. comm.). Large populations of this species occur in the Imperial and Palo Verde Valleys.

Additional Yuma clapper rails can be found along the Colorado River during the breeding season. Rails use the Lower Colorado River from the US border north to Topock Marsh. In the last complete census of the Lower Colorado River in 1994, the estimated total population was 1,145. Based on census data from 1990 to 1995, the Yuma clapper rail population along the Colorado River appears to be stable at this time.

Reasons for Decline and Threats to Survival: Significant habitat losses are believed to have occurred in the lower Colorado River and the delta with the construction of large water reclamation projects along the Colorado River. Recent studies of the Yuma clapper rail indicate that this species may be at risk of selenium-induced reproductive impacts (Rusk 1991, Roberts 1996). While census information has not indicated a decline, selenium concentrations in the rail eggs and tissues analyzed are at levels that could result in slight reductions in reproductive success.

Bonytail Chub (*Gila elegans*)

Species Description and Life History: The bonytail chub was first proposed for listing under the ESA on April 24, 1978, as an endangered species. The bonytail chub was listed as an endangered species on April 23, 1980 (45 **FR** 27713), with an effective date of the rule of May 23, 1980. In the final rule, the Service determined that at that time there were no known areas with the necessary requirements to be determined critical habitat. Critical habitat was designated in 1994. Critical habitat for the bonytail chub includes portions of the Colorado, Green, and Yampa Rivers. Critical habitat includes the Colorado River at Lake Havasu to its full pool elevation (USDI-FWS 1993a).

The bonytail chub is one of three closely related members of the genus *Gila* found in the Colorado River. Confusion about the proper taxonomy and the degree of hybridization between the bonytail chub, the humpback chub, (*Gila cypha*), and roundtail chub, (*G. robusta*), has complicated examinations of the status of these fish. The bonytail chub is a highly streamlined fish with a very thin, pencil-like, caudal peduncle and large, falcate fins (Allan and Roden 1978). A nuchal hump may be present behind the head. Maximum length is about 600 millimeters (mm), with 300-350 mm more common (USDI-FWS 1990). Weights are generally less than one kilogram (kg) (Vanicek and Kramer 1969). Bonytail chub are long-lived fish; some have reached at least 49 years of age (Minckley 1985).

With their streamlined bodies, bonytail chub appear to be adapted to the Colorado River and large tributary streams. Even with these adaptations, this species does not select areas of high velocity currents and use of pools and eddies by the fish is significant (Vanicek 1967, Vanicek and Kramer 1969).

Spawning takes place in the late spring to early summer (Jonez and Sumner 1954, Wagner 1955) in water temperatures about 18 degrees C (Vanicek and Kramer 1969). Riverine spawning of the bonytail chub has not been documented; however in reservoirs, gravel bars or shelves are used (Jonez and Sumner 1954).

The bonytail chub is adapted to the widely fluctuating physical environment of the historical Colorado River. Adults can live 45-50 years, and apparently produce viable gametes even when quite old. The ability to spawn in a variety of habitats is also a survival adaption. Fecundity measurements taken on adult females in the hatchery ranged from 1,015 to 10,384 eggs per fish with a mean of 4,677 (USDI-FWS 1990). With the fecundity of the species, it would be possible to quickly repopulate after a catastrophic loss of adults.

Foraging Ecology: Bonytail chub feed mostly on insects, algae, and plant debris.

Historic and Current Distribution: Occupied habitat as of 1993 is approximately 344 miles (15% of the historic range). Populations are generally small and composed of aging individuals. Recovery efforts under the Recovery Implementation Program in the Upper Basin have begun, but significant recovery results have not been seen for this species. In the Lower Basin, augmentation efforts along the Lower Colorado River propose to replace the aging populations in Lakes Havasu and Mohave with young fish from protected-rearing site programs. This may prevent the imminent extinction of the species in the wild, but appears less capable of ensuring long term survival or recovery of the bonytail chub. Overall, the status of the bonytail chub in the wild continues to be precarious.

Reasons for Decline and Threats to Survival: Severe reductions in both population numbers and individual bonytail chub numbers can be traced largely to impounding the lower Colorado River and introducing non-native fish into the modified environment. The bonytail chub was listed as an endangered species due to massive declines in or extirpation of all populations throughout the range of the species. The causes of these declines are changes to biological and physical features of the habitat. The effects of these changes have been most noticeable by the almost complete lack of natural recruitment to any population in the historic range of the species.

Chinook Salmon (Including Central Valley Spring-Run, California Coastal and Sacramento Winter-Run ESUs) (*Oncorhynchus tshawytscha*)

Species Description and Life History: Based on the best available scientific and commercial information, NMFS has identified 17 ESUs of chinook salmon from Washington, Oregon, Idaho, and California, including 11 new ESUs, and one re-defined ESU. Further detailed information on these ESUs is available in the NMFS "Status Review of Chinook Salmon from Washington, Idaho, Oregon, and California" (Myers *et al.*, 1998) and the NMFS proposed rule for listing chinook (63 FR 11482). Four of these are within the action area in California. The Sacramento River Winter-Run ESU was listed as endangered on January 4, 1994 (59 FR 440); critical habitat was designated in an earlier listing of the ESU as threatened (June 16, 1993; 58 FR 33212). On September 16, 1999, NMFS listed (64 FR 50394) the Central Valley Spring-Run ESU as threatened; redefined the Southern Oregon and California Coastal ESU, creating a distinct California Coastal ESU extending from the Russian River, Sonoma County, north to Redwood Creek, Humboldt County, and listed this new ESU as threatened. In the same rulemaking, NMFS also determined that the Central Valley Fall/Late Fall ESU and the Southern

Oregon / Northern California Coastal ESU (including those populations now considered separate from the California Coastal ESU) are not warranted for listing at this time.

Critical Habitat: On February 16, 2000, NMFS designated critical habitat for all ESUs of chinook salmon (except Sacramento River Winter-Run)(65 FR 7764). In evaluating the habitat requirements of listed chinook NMFS decided to designate only the current range of the listed ESUs as critical habitat. The current range encompasses a wide range of habitats, including small tributary reaches as well as mainstem, off-channel, and estuarine areas. Areas excluded from this proposed designation include historically occupied areas above impassible dams and headwater areas above impassable natural barriers (e.g., long-standing, natural waterfalls). NMFS has concluded that at the time of this designation, currently inhabited areas within the range of West Coast chinook salmon are the minimum habitat necessary to ensure conservation and recovery of the species. Critical habitat consists of the water, substrate, and adjacent riparian zone of accessible estuarine and riverine reaches for the following areas for chinook salmon located in California:

1) Central Valley Spring-Run chinook salmon geographic boundaries: Critical habitat is designated to include all river reaches accessible to chinook salmon in the Sacramento River and its tributaries in California. Also included are river reaches and estuarine areas of the Sacramento-San Joaquin Delta, all waters from Chipps Island westward to Carquinez Bridge, including Honker Bay, Grizzly Bay, Suisun Bay, and Carquinez Strait, all waters of San Pablo Bay westward of the Carquinez Bridge, and all waters of San Francisco Bay (north of the San Francisco/Oakland Bay Bridge) from San Pablo Bay to the Golden Gate Bridge. Excluded are areas above specific dams or above longstanding, naturally impassable barriers (i.e., natural waterfalls in existence for at least several hundred years).

2) California Coastal chinook salmon geographic boundaries: Critical habitat is designated to include all river reaches and estuarine areas accessible to chinook salmon along the California coast from the Russian River, in Sonoma County, north to Redwood Creek, Humboldt County. Also excluded are areas above specific dams or above longstanding, naturally impassable barriers (i.e., natural waterfalls in existence for at least several hundred years).

3) Sacramento River Winter-Run chinook geographic boundaries: Critical habitat is designated to include the Sacramento River from Keswick Dam (Shasta County) to Chipps Island at the westward margin of the Sacramento-San Joaquin Delta; all waters from Chipps Island westward to Carquinez Bridge, including Honker Bay, Grizzly Bay, Suisun Bay, and Carquinez Strait; all waters of San Pablo Bay westward of the Carquinez Bridge, and all waters of San Francisco Bay (north of the San Francisco/Oakland Bay Bridge) from San Pablo Bay to the Golden Gate Bridge. In addition, the critical habitat designation identifies those physical and biological features of the habitat that are essential to the conservation of the species and that may require special management considerations or protection. These features include (1) access from the Pacific Ocean to appropriate spawning areas in the upper Sacramento River, (2) the availability of clean gravel for spawning substrate, (3) adequate river flows for successful spawning,

incubation of eggs, fry development and emergence, and downstream transport of juveniles, (4) water temperatures between 42.5 and 57.5 degrees Fahrenheit for successful spawning, egg incubation and fry development, (5) habitat areas and adequate prey that are not contaminated, (6) riparian habitat that provides for successful juvenile development and survival, and (7) access downstream so that juveniles can migrate from spawning areas to San Francisco Bay and the Pacific Ocean.

Migration and Spawning (Coastal chinook ESUs): Chinook salmon are easily distinguished from other *Oncorhynchus* species by their large size. Adults weighing over 120 pounds have been caught in North American waters. Chinook salmon are very similar to coho salmon (*O. kisutch*) in appearance while at sea (blue-green back with silver flanks), except for their large size, small black spots on both lobes of the tail, and black pigment along the base of the teeth. Chinook salmon are anadromous and semelparous. This means that as adults they migrate from a marine environment into the fresh water streams and rivers of their birth (anadromous) where they spawn and die (semelparous). Adult female chinook will prepare a spawning bed, called a redd, in a stream area with suitable gravel composition, water depth and velocity. Redds will vary widely in size and in location within the stream or river. The adult female chinook may deposit eggs in 4 to 5 nesting pockets within a single redd. After laying eggs in a redd, adult chinook will guard the redd from 4 to 25 days before dying. Chinook salmon eggs will hatch, depending upon water temperatures, between 90 to 150 days after deposition. Stream flow, gravel quality, and silt load all significantly influence the survival of developing chinook salmon eggs. Juvenile chinook may spend from 3 months to 2 years in freshwater after emergence and before migrating to estuarine areas as smolts, and then into the ocean to feed and mature. Historically, chinook salmon ranged as far south as the Ventura River, California, and their northern extent reaches Alaska and the Russian Far East.

Among chinook salmon, two distinct races have evolved. One race, described as a stream-type chinook, is found most commonly in headwater streams. Stream-type chinook salmon have a longer freshwater residency, and perform extensive offshore migrations before returning to their natal streams in the spring or summer months. The second race is called the ocean-type chinook, which are commonly found in coastal streams or the mainstem portions of larger rivers draining inland basins in North America. Ocean-type chinook typically migrate to sea within the first three months of emergence, but they may spend up to a year in freshwater prior to emigration. They also spend their ocean life in coastal waters. Ocean-type chinook salmon return to their natal streams or rivers as spring, winter, fall, summer, and late-fall runs, but summer and fall runs predominate (Healey 1991). The difference between these life history types is also physical, with both genetic and morphological foundations. Juvenile stream- and ocean-type chinook salmon have adapted to different ecological niches. Ocean-type chinook salmon tend to utilize estuaries and coastal areas more extensively for juvenile rearing. The brackish water areas in estuaries also moderate physiological stress during parr-smolt transition. The development of the ocean-type life history strategy may have been a response to the limited carrying capacity of smaller stream systems and glacially scoured, unproductive, watersheds, or a means of avoiding the impact of seasonal floods in the lower portion of many watersheds (Miller and Brannon

1982). Stream-type juveniles are much more dependent on freshwater stream ecosystems because of their extended residence in these areas. A stream-type life history may be adapted to those watersheds, or parts of watersheds, that are more consistently productive and less susceptible to dramatic changes in water flow, or which have environmental conditions that would severely limit the success of subyearling smolts (Miller and Brannon 1982; Healey 1991). At the time of saltwater entry, stream-type (yearling) smolts are much larger, averaging 73-134 mm depending on the river system, than their ocean- type (subyearling) counterparts and are therefore able to move offshore relatively quickly (Healey 1991).

Coast wide, chinook salmon remain at sea for 1 to 6 years (more commonly 2 to 4 years), with the exception of a small proportion of yearling males (called jack salmon) which mature in freshwater or return after 2 or 3 months in salt water (Rutter 1904; Gilbert 1912; Rich 1920; Mullan *et al.* 1992). Ocean- and stream-type chinook salmon are recovered differentially in coastal and mid-ocean fisheries, indicating divergent migratory routes (Healey 1983 and 1991). Ocean- type chinook salmon tend to migrate along the coast, while stream-type chinook salmon are found far from the coast in the central North Pacific (Healey 1983 and 1991; Myers *et al.* 1984). Differences in the ocean distribution of specific stocks may be indicative of resource partitioning and may be important to the success of the species as a whole.

Migration and Spawning (Sacramento River Winter-Run chinook ESU): The first winter-run chinook upstream migrants appear in the Sacramento-San Joaquin Delta during the early winter months (Skinner 1972). On the upper Sacramento River, the first upstream migrants appear during December (Vogel and Marine 1991). The upstream migration of winter-run chinook typically peaks during the month of March, but may vary with river flow, water-year type, and operation of Red Bluff Diversion Dam. Keswick Dam completely blocks any further upstream migration, forcing adults to migrate to and hold in deep pools downstream, before initiating spawning activities.

Since the construction of Shasta and Keswick Dam, winter-run chinook spawning has primarily occurred between Red Bluff Diversion Dam and Keswick Dam. The spawning period of winter-run chinook generally extends from mid-April to mid-August with peak activity occurring in June (Vogel and Marine 1991). Winter-run chinook may also spawn below Red Bluff in some years. In 1988, for example, winter-run chinook redds were observed as far downstream as Woodson Bridge. Winter-run chinook eggs hatch after an incubation period of about 40-60 days depending on ambient water temperatures. Maximum survival of incubating eggs and pre-emergent fry occurs at water temperatures between 42 degrees F and 56 degrees F with a preferred temperature of 52 degrees F. Mortality of eggs and pre-emergent fry commences at 57.5 degrees F and reaches 100 percent at 62 degrees F (Boles 1988).

The pre-emergent fry remain in the redd and absorb the yolk stored in their yolk-sac as they grow into fry. This period of larval incubation lasts approximately 6 to 8 weeks depending on water temperatures. Emergence of the fry from the gravel begins during late June and continues through September. The fry seek out shallow, nearshore areas with slow current and good cover, and begin feeding on small terrestrial and aquatic insects and aquatic crustaceans. As they grow to 50 to 75 mm in length, the juvenile salmon move out into deeper, swifter water, but continue to use available cover to minimize the risk of predation and reduce energy expenditure.

The emigration of juvenile winter-run chinook from the upper Sacramento River is highly dependent on streamflow conditions and water year type. Peak outmigration from the Delta typically occurs from late January through April. Optimal water temperatures for the growth of juvenile chinook salmon in an estuary are 54 to 57 degrees F (Brett 1952). High river flows in the winter and early spring assist juvenile fish migrating downstream to the estuary, while positive outflow from the Delta improves juvenile survival and migration to the ocean.

Available information on winter-run chinook salmon ocean distribution indicates that marked winter-run chinook salmon are caught between Monterey Bay and Fort Bragg, California. However, this data may be biased towards areas where commercial and recreational fisheries occur.

Migration and Spawning (Central Valley Spring-Run chinook ESU): Impassable dams block access to most of the historical headwater spawning and rearing habitat of Central Valley spring-run chinook salmon. In addition, much of the remaining, accessible spawning and rearing habitat is severely degraded by elevated water temperatures, agricultural and municipal water diversions, unscreened and poorly screen water intakes, restricted and regulated streamflows, levee and bank stabilization, and poor quality and quantity of riparian and shaded riverine aquatic (SRA) cover.

Natural spawning populations of Central Valley spring-run chinook salmon are currently restricted to accessible reaches in 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 (DFG 1998; FWS, unpublished data). With the exception of Butte Creek and the Feather River, these populations are relatively small ranging from a few fish to several hundred. Butte Creek returns in 1998 and 1999 numbered approximately 20,000 and 3,600, respectively (DFG unpublished data). On the Feather River, significant numbers of spring-run chinook, as identified by run timing, return to the Feather River Hatchery. However, coded-wire-tag information from these hatchery returns indicates substantial introgression has occurred between fall-run and spring-run chinook populations in the Feather River due to hatchery practices. Over time, the spring-run within the Feather River may become homogeneous with Feather River fall-run fish unless current hatchery practices are changed.

Spring-run chinook salmon adults are estimated to leave the ocean and enter the Sacramento River from March to July (Myers et al. 1998). This run timing is well adapted for gaining access to the upper reaches of river systems, 1,500 to 5,200 feet in elevation, prior to the onset of high water temperatures and low flows that would inhibit access to these areas during the fall. Throughout this upstream migration phase, adults require streamflows sufficient to provide olfactory and other orientation cues used to locate their natal streams. Adequate streamflows are also necessary to allow adult passage to upstream holding habitat in natal tributary streams. The preferred temperature range for spring-run chinook salmon completing their upstream migration is 38° F to 56° F (Bell 1991; DFG 1998).

When they enter freshwater, spring-run chinook salmon are immature and they must stage for several months before spawning. Their gonads mature during their summer holding period in freshwater. Over-summering adults require cold-water refuges such as deep pools to conserve energy for gamete production, redd construction, spawning, and redd guarding. The upper limit of the optimal temperature range for adults holding while eggs are maturing is 59° F to 60° F (Hinz 1959). Unusual stream temperatures during spawning migration and adult holding periods can alter or delay migration timing, accelerate or retard mutations, and increase fish susceptibility to diseases. Sustained water temperatures above 80.6° F are lethal to adults (Cramer and Hammack 1952; DFG 1998).

Adults prefer to hold in deep pools with moderate water velocities and bedrock substrate and avoid cobble, gravel, sand, and especially silt substrate in pools (Sato and Moyle 1989). Optimal water velocities for adult chinook salmon holding pools range between 0.5-1.3 feet-per-second and depths are at least three to ten feet (G. Sato unpublished data, Marcotte 1984). The pools typically have a large bubble curtain at the head, underwater rocky ledges, and shade cover throughout the day (Ekman 1987).

Spawning typically occurs between late-August and early October with a peak in September. Once spawning is completed, adult spring-run chinook salmon die. Spawning typically occurs in gravel beds that are located at the tails of holding pools (USFWS 1995a). Spring-run adults have been observed spawning in water depths of 0.8 feet or more, and water velocities from 1.2-3.5 feet-per-second (Puckett and Hinton 1974). Eggs are deposited within the gravel where incubation, hatching, and subsequent emergence takes place. Optimum substrate for embryos is a mixture of gravel and cobble with a mean diameter of one to four inches with less than 5% fines, which are less than or equal to 0.3 inches in diameter (Platts et al. 1979, Reiser and Bjornn 1979). The upper preferred water temperature for spawning adult chinook salmon is 55° F (Chambers 1956) to 57° F (Reiser and Bjornn 1979).

Length of time required for eggs to develop and hatch is dependant on water temperature and is quite variable, however, hatching generally occurs within 40 to 60 days of fertilization (Vogel and Marine 1991). In Deer and Mill creeks, embryos hatch following a 3-5 month incubation period (USFWS 1995). The optimum temperature range for chinook salmon egg incubation is 44° F to 54° F (Rich 1997). Incubating eggs show reduced egg viability and increased mortality at temperatures greater than 58° F and show 100% mortality for temperatures greater than 63° F (Velson 1987). Velson (1987) and Beacham and Murray (1990) found that developing chinook salmon embryos exposed to water temperatures of 35° F or less before the eyed stage experienced 100% mortality (DFG 1998).

After hatching, pre-emergent fry remain in the gravel living on yolk-sac reserves for another two to four weeks until emergence. Timing of emergence within different drainages is strongly influenced by water temperature. Emergence of spring-run chinook typically occurs from November through January in Butte and Big Chico Creeks and from January through March in Mill and Deer Creeks (DFG 1998).

Post-emergent fry seek out shallow, nearshore areas with slow current and good cover, and begin feeding on small terrestrial and aquatic insects and aquatic crustaceans. As they grow to 50 to 75 mm in length, the juvenile salmon move out into deeper, swifter water, but continue to use available cover to minimize the risk of predation and reduce energy expenditure. The optimum temperature range for rearing chinook salmon fry is 50° F to 55° F (Boles et al. 1988, Rich 1997, Seymour 1956) and for fingerlings is 55° F to 60° F (Rich 1997).

In Deer and Mill creeks, juvenile spring-run chinook, during most years, spend 9-10 months in the streams, although some may spend as long as 18 months in freshwater. Most of these "yearling" spring-run chinook move downstream in the first high flows of the winter from November through January (USFWS 1995, DFG 1998). In Butte and Big Chico creeks, spring-run chinook juveniles typically exit their natal tributaries soon after emergence during December and January, while some remain throughout the summer and exit the following fall as yearlings. In the Sacramento River and other tributaries, juveniles may begin migrating downstream almost immediately following emergence from the gravel with emigration occurring from December through March (Moyle, et al. 1989, Vogel and Marine 1991). Fry and parr may spend time rearing within riverine and/or estuarine habitats including natal tributaries, the Sacramento River, non-natal tributaries to the Sacramento River, and the Delta. In general, emigrating juveniles that are younger (smaller) reside longer in estuaries such as the Delta (Kjelson et al. 1982, Levy and Northcote 1982, Healey 1991). The brackish water areas in estuaries moderate the physiological stress that occurs during parr-smolt transitions. Although fry and fingerlings can enter the Delta as early as January and as late as June, their length of residency within the Delta is unknown but probably lessens as the season progresses into the late spring months (DFG 1998).

Foraging Ecology: In an estuarine environment such as the Delta, juvenile chinook salmon forage in intertidal and shallow subtidal areas, such as marshes, mudflats, channels, and sloughs. These habitats provide protective cover and a rich food supply (McDonald 1960; Dunford 1975). The distribution of the juvenile fish appears to change tidally in an estuarine environment. Large fry and smolts tend to congregate in the surface waters of main and subsidiary sloughs and channels, moving into shallow subtidal areas only to feed (Allen and Hassler 1986).

Genetics: There is a significant genetic influence to the freshwater component of the returning adult migratory process. A number of studies show that chinook salmon return to their natal streams with a high degree of fidelity (Rich and Holmes 1928; Quinn and Fresh 1984; McIsaac and Quinn 1988). Salmon may have evolved this trait as a method of ensuring an adequate incubation and rearing habitat. It also provides a mechanism for reproductive isolation and local adaptation. Conversely, returning to a stream other than that of one's origin is important in colonizing new areas and responding to unfavorable or perturbed conditions at the natal stream (Quinn 1993).

Chinook salmon stocks exhibit considerable variability in size and age of maturation, and at least some portion of this variation is genetically determined. The relationship between size and length of migration may also reflect the earlier timing of river entry and the cessation of feeding for chinook salmon stocks that migrate to the upper reaches of river systems. Body size, which is correlated with age, may be an important factor in migration and redd construction success. Roni and Quinn (1995) reported that under high density conditions on the spawning ground, natural selection may produce stocks with exceptionally large-sized returning adults.

Artificial propagation and other human activities such as harvest and habitat modification can genetically change natural populations so much that they no longer represent an evolutionarily significant component of the biological species (Waples 1991). Artificial propagation is a common practice to supplement chinook salmon stocks for commercial and recreational fisheries. However, in many areas, a significant portion of the naturally spawning population consists of hatchery-produced chinook salmon. In several of the chinook salmon ESUs, over 50 percent of the naturally spawning fish are from hatcheries. Many of these hatchery- produced fish are derived from a few stocks which may or may not have originated from the geographic area where they are released. However, in several of the ESUs analyzed, insufficient or uncertain information exists regarding the interactions between hatchery and natural fish, and the relative abundance of hatchery and natural stocks. See the proposed rule for more information on the effects of artificial propagation on chinook salmon.

Among basins supporting only ocean-type chinook salmon, the Sacramento River system is somewhat unusual in that its large size and ecological diversity historically allowed for substantial spatial as well as temporal separation of different runs. Genetic and life history data both suggest that considerable differentiation among the runs has occurred in this basin. The Klamath River Basin, as well as chinook salmon in Puget Sound, share some features of coastal rivers but historically also provided an opportunity for substantial spatial separation of different temporal runs. As discussed below, the diversity in run timing made identifying ESUs difficult in the Klamath and Sacramento River Basins.

No allozyme data are available for naturally spawning Sacramento River spring chinook salmon. A sample from Feather River Hatchery spring-run fish, which may have undergone substantial hybridization with fall chinook salmon, shows modest (but statistically significant) differences from fall-run hatchery populations. DNA data show moderate genetic differences between the spring and fall/late-fall runs in the Sacramento River; however, these data are difficult to interpret because comparable data are not available for other geographic regions.

<u>Historic and Current Distribution</u>: NMFS considers differences in life history traits as a possible indicator of adaptation to different environmental regimes and resource partitioning within those regimes. The relevance of the ecologic and genetic basis for specific chinook salmon life-history traits as they pertain to each ESU is discussed in the brief summary that follows. NMFS calculated trends from the most recent 10 years using data collected after 1984 for series having at least 7 observations since 1984. No attempt was made to account for the influence of

hatchery-produced fish on these estimates, so the estimated trends include the progeny of naturally spawning hatchery fish. After evaluating patterns of abundance drawn on these quantitative and qualitative assessments, and evaluating other risk factors for chinook salmon from these ESUs, NMFS reached the conclusions summarized below.

Central Valley Spring-Run ESU (Threatened): Existing populations in this ESU spawn in the Sacramento River and its tributaries. Historically, spring chinook salmon were the dominant run in the Sacramento and San Joaquin River Basins (Clark 1929), but native populations in the San Joaquin River have apparently all been extirpated (Campbell and Moyle 1990). This ESU includes chinook salmon entering the Sacramento River from March to July and spawning from late August through early October, with a peak in September. Spring-run fish in the Sacramento River exhibit an ocean-type life history, emigrating as fry, subyearlings, and yearlings. Recoveries of hatchery chinook salmon implanted with coded-wire-tags (CWT) are primarily from ocean fisheries off the California and Oregon coast. There were minimal differences in the ocean distribution of fall- and spring-run fish from the Feather River Hatchery (as determined by CWT analysis); however, due to hybridization that may have occurred in the hatchery between these two runs, this similarity in ocean migration may not be representative of wild runs. Substantial ecological differences in the historical spawning habitat for spring-run versus fall- and late-fall-run fish have been recognized. Spring chinook salmon run timing was suited to gaining access to the upper reaches of river systems (up to 1,500 m elevation) prior to the onset of prohibitively high water temperatures and low flows that inhibit access to these areas during the fall. Differences in adult size, fecundity, and smolt size also occur between spring- and fall/late fall-run chinook salmon in the Sacramento River.

Native spring chinook salmon have been extirpated from all tributaries in the San Joaquin River Basin, which represents a large portion of the historic range and abundance of the ESU as a whole. The only streams considered to have wild spring-run chinook salmon are Mill and Deer Creeks, and possibly Butte Creek (tributaries to the Sacramento River), and these are relatively small populations with sharply declining trends. Demographic and genetic risks due to small population sizes are thus considered to be high. Current spawning is restricted to the mainstem and a few river tributaries in the Sacramento River. Most of the fish in this ESU are hatchery produced.

California Coastal ESU (Threatened): This ESU includes all naturally spawned coastal spring and fall chinook salmon spawning from the Russian River, in Sonoma County north to Redwood Creek in Humboldt County. Chinook salmon from the Central Valley and Klamath River Basin upstream form the Trinity River confluence are genetically and ecologically distinguishable from those in this ESU. Chinook salmon in this ESU exhibit an ocean-type life-history; ocean distribution (based on marine CWT recoveries) is predominantly off of the California and Oregon coasts. Life-history information on smaller populations, especially in the southern portion of the ESU, is extremely limited. Additionally, only anecdotal or incomplete information exists on abundance of several spring-run populations including, the Chetco, Winchuck, Smith, Mad, and Eel Rivers. Allozyme data indicate that this ESU is genetically distinguishable from the

Oregon Coast, Upper Klamath and Trinity River, and Central Valley ESUs. Life history differences also exist between spring- and fall-run fish in this ESU, but not to the same extent as is observed in larger inland basins. Ecologically, the majority of the river systems in this ESU are relatively small and heavily influenced by a maritime climate. Low summer flows and high temperatures in many rivers result in seasonal physical and thermal barrier bars that block movement by anadromous fish.

This ESU contains chinook salmon from the Russian River in Sonoma County, north to Redwood Creek in Humboldt County. Chinook salmon spawning abundance in this ESU is highly variable among populations. There is a general pattern of downward trends in abundance in most populations for which data are available, with declines being especially pronounced in spring-run populations. The extremely depressed status of almost all coastal populations south of the Klamath River is an important source of risk to the ESU. NMFS has a general concern that no current information is available for many river systems in the southern portion of this ESU, which historically maintained numerous large populations.

Sacramento River Winter-Run ESU (Endangered): The Sacramento River winter-run chinook salmon is a unique population of chinook salmon in the Sacramento River. It is distinguishable from the other three Sacramento River chinook runs by the timing of its upstream migration and spawning season.

Prior to construction of Shasta and Keswick dams in 1945 and 1950, respectively, winter-run chinook were reported to spawn in the upper reaches of the Little Sacramento, McCloud, and lower Pit rivers (Moyle *et al.* 1989). Specific data relative to the historic run sizes of winter-run chinook prior to 1967 are sparse and anecdotal. Numerous fishery researchers have cited Slater (1963) to indicate that the winter-run chinook population may have been fairly small and limited to the spring-fed areas of the McCloud River before the construction of Shasta Dam. However, recent CDFG research in California State Archives has cited several fisheries chronicles that indicate the winter-run chinook population may have been much larger than previously thought. According to these qualitative and anecdotal accounts, winter-run chinook reproduced in the McCloud, Pit and Little Sacramento rivers and may have numbered over 200,000 (Rectenwald 1989).

Completion of the Red Bluff Diversion Dam in 1966 enabled accurate estimates of all salmon runs to the upper Sacramento River based on fish counts at the fish ladders. These annual fish counts document the dramatic decline of the winter-run chinook population. The estimated number of winter-run chinook passing the dam from 1967 to 1969 averaged 86,509. During 1990, 1991, 1992, 1993, 1994, 1995, 1996, and 1997 the spawning escapement of winter-run chinook past the dam was estimated at 441, 191, 1180, 341, 189, 1361, 940, and 841 adults (including jacks), respectively.

<u>Reasons for Decline and Threats to Survival</u>: *Central Valley Spring-Run ESU*: Habitat problems are the most important source of ongoing risk to the Central Valley spring-run ESU. Spring-run

fish cannot access most of their historical spawning and rearing habitat in the Sacramento and San Joaquin River Basins (which is now above impassable dams). The remaining spawning habitat accessible to fish is severely degraded. Collectively, these habitat problems greatly reduce the resiliency of this ESU to respond to additional stresses in the future. The general degradation of conditions in the Sacramento River Basin (including elevated water temperatures, agricultural and municipal diversions and returns, restricted and regulated flows, entrainment of migrating fish into unscreened or poorly screened diversions, and the poor quality and quantity of remaining habitat) has severely impacted important juvenile rearing habitat and migration corridors. There appears to be serious concern for threats to genetic integrity posed by hatchery programs in the Central Valley. Most of the spring-run chinook salmon production in the Central Valley is of hatchery origin, and naturally spawning populations may be interbreeding with both fall/late fall- and spring-run hatchery fish. Related harvest regimes may not be allowing recovery of this at-risk population.

California Coastal ESU: Habitat loss and/or degradation is widespread throughout the range of the California Coastal ESU. The California Advisory Committee on Salmon and Steelhead Trout (CACSST) reported habitat blockages and fragmentation, logging and agricultural activities, urbanization, and water withdrawals as the most predominant problems for anadromous salmonids in California's coastal basins (CACSST 1988). They identified associated habitat problems for each major river system in California. CDFG (1965, Vol. III, Part B) reported that the most vital habitat factor for coastal California streams was “degradation due to improper logging followed by massive siltation, log jams, etc.” They cited road building as another cause of siltation in some areas. They identified a variety of specific critical habitat problems in individual basins, including extremes of natural flows (Redwood Creek and Eel River), logging practices (Mad, Eel, Mattole, Ten Mile, Noyo, Big, Navarro, Garcia, and Gualala Rivers), and dams with no passage facilities (Eel and Russian Rivers), and water diversions (Eel and Russian Rivers). Recent major flood events (February 1996 and January 1997) have probably affected habitat quality and survival of juveniles within this ESU. Artificial propagation programs in the California Coastal ESU are less extensive than those in Klamath/Trinity or Central Valley ESUs. The Rogue, Chetco and Eel River Basins and Redwood Creek have received considerable releases, derived primarily from local sources. Current hatchery contribution to overall abundance is relatively low except for the Rogue River spring run.

Sacramento River Winter-Run ESU: The main cause of decline of the winter-run chinook salmon was the damming of rivers that prevented instream migration. Associated factors contributing to the decline and threat of survival for winter-run chinook salmon include forestry, agriculture, mining, and urbanization that have degraded, simplified, and fragmented habitat significantly throughout the range of the species. Potential sources of mortality during the incubation period include redd dewatering, insufficient oxygenation, physical disturbance, and water-borne contaminants.

Infectious disease is one of the many factors that can influence adult and juvenile survival.

Chinook salmon are exposed to numerous bacterial, protozoan, viral, and parasitic organisms in spawning and rearing areas, hatcheries, migratory routes, and the marine environment, poor water quality within these habitats increase steelhead vulnerability to disease and predation.

Overall Threats to Survival for all ESU's: Chinook salmon on the west coast of the United States have experienced declines in abundance in the past several decades as a result of loss, damage or change to their natural environment. Water diversions for agriculture, flood control, domestic, and hydropower purposes (especially in the Columbia River and Sacramento-San Joaquin Basins) have greatly reduced or eliminated historically accessible habitat and degraded remaining habitat. Forestry, agriculture, mining, and urbanization have degraded, simplified, and fragmented habitat. Studies indicate that in most western states, about 80 to 90 percent of the historic riparian habitat has been eliminated (Botkin *et al.*, 1995; Norse, 1990; Kellogg, 1992; California State Lands Commission, 1993). Washington and Oregon wetlands are estimated to have diminished by one-third, while California has experienced a 91 percent loss of its wetland habitat. Loss of habitat complexity and habitat fragmentation have also contributed to the decline of chinook salmon. For example, in national forests within the range of the northern spotted owl in western and eastern Washington, there has been a 58 percent reduction in large, deep pools due to sedimentation and loss of pool-forming structures such as boulders and large wood (Forest Ecosystem Management Assessment Team (FEMAT) 1993). Similar or even an elevated level of effects are likely in California.

Introductions of non-native species and habitat modifications have resulted in increased predator populations in numerous rivers. Predation by marine mammals is also of concern in areas experiencing dwindling chinook salmon run sizes. However, salmonids appear to be a minor component of the diet of marine mammals (Scheffer and Sperry 1931; Jameson and Kenyon 1977; Graybill 1981; Brown and Mate 1983; Roffe and Mate 1984; Hanson 1993). Principal food sources are small pelagic schooling fish, juvenile rockfish, lampreys (Jameson and Kenyon 1977; Roffe and Mate 1984), benthic and epibenthic species (Brown and Mate 1983) and flatfish (Scheffer and Sperry 1931; Graybill 1981). Predation may significantly influence salmonid abundance in some local populations when other prey are absent and physical conditions lead to the concentration of adults and juveniles (Cooper and Johnson 1992).

Infectious disease is one of many factors that can influence adult and juvenile chinook salmon survival. Chinook salmon are exposed to numerous bacterial, protozoan, viral, and parasitic organisms in spawning and rearing areas, hatcheries, migratory routes, and the marine environment. Very little current or historical information exists to quantify changes in infection levels and mortality rates attributable to these diseases for chinook salmon. However, studies have shown that naturally spawned fish tend to be less susceptible to pathogens than hatchery-reared fish (Buchanon *et al.* 1983; Sanders *et al.* 1992).

Competition, genetic introgression, and disease transmission resulting from hatchery introductions may significantly reduce the production and survival of native, naturally-reproducing chinook salmon. Collection of native chinook salmon for hatchery brood stock purposes often harms small or dwindling natural populations. Artificial propagation may play an important role in chinook salmon recovery, and some hatchery populations of chinook salmon may be deemed essential for the recovery of threatened or endangered chinook salmon ESUs. While some limits have been placed on hatchery production of anadromous salmonids, more careful management of current programs and scrutiny of proposed programs is necessary in order to minimize impacts on listed species.

The CWA, enforced in part by the EPA, is intended to protect beneficial uses, including fishery resources. To date, implementation has not been effective in adequately protecting fishery resources, particularly with respect to non-point sources of pollution. In addition, section 404 of the CWA does not adequately address the cumulative and additive effects of loss of habitat through continued development of waterfront, riverine, coastal, and wetland properties that also contribute to the degradation and loss of important aquatic ecosystem components necessary to maintain the functional integrity of these habitat features.

Sections 303 (d) (1) (C) and (D) of the CWA require states to prepare Total Maximum Daily Loads (TMDLs) for all water bodies that do not meet State water quality standards. Development of TMDLs is a method for quantitative assessment of environmental problems in a watershed and identification of pollution reductions needed to protect drinking water, aquatic life, recreation, and other uses of rivers, lakes, and streams. Appropriately protective aquatic life criteria are critical to the TMDL process for affecting the recovery of salmon populations, as the criteria exceedance will determine which waterbodies will engage in the TMDL process and criteria compliance goals are the impetus for developing mass loading strategies. The ability of these TMDLs to protect chinook salmon should be significant in the long term; however, it will be difficult to develop them quickly in the short term, and their efficacy in protecting chinook salmon habitat will be unknown for years to come.

Coho Salmon (Including Central California Coast and Southern Oregon/Northern California Coast ESUs) (*Oncorhynchus kisutch*)

Species Description and Life History: General life history information for coho salmon is summarized below, followed by information on population trends for each coho salmon ESU. Further detailed information on these coho salmon ESUs is available in the NMFS Status Review of coho salmon from Washington, Oregon, and California (Weitkamp *et al.* 1995), the NMFS proposed rule for listing coho (60 **FR** 38011), and the NMFS final listings for the Central California Coast coho ESU (61 **FR** 56138) and the Southern Oregon/Northern California Coast coho ESU (62 **FR** 24588). On May 5, 1999, NMFS designated critical habitat for the Central California Coast and the Southern Oregon/Northern California Coast coho salmon ESUs (64 **FR** 24049). The designation includes all accessible reaches of rivers between the Elk River in Oregon and the San Lorenzo River in Santa Cruz County, California. This designation also includes two rivers entering the San Francisco Bay: Mill Valley Creek and Corte Madera Creek. For both ESUs, critical habitat includes the water, substrate, and adjacent riparian zones.

Critical Habitat: Central California Coast ESU coho geographic boundaries encompass accessible reaches of all rivers (including estuarine areas and tributaries) between Punta Gorda (near the Mattole River, Mendocino County) and the San Lorenzo River (Santa Cruz County), inclusive, and including two streams that enter San Francisco Bay: Arroyo Corte Madera Del Presidio and Corte Madera Creeks.

Southern Oregon/Northern California Coast ESU coho geographic boundaries encompass accessible reaches of all rivers (including estuarine areas and tributaries) between the Mattole River (Mendocino County) and the Elk River in Oregon, inclusive.

Migration and Spawning: Most coho salmon adults are 3-year-olds, having spent approximately 18 months in freshwater and 18 months in salt water (Gilbert 1912; Pritchard 1940; Briggs 1953; Shapovalov and Taft 1954; Loeffel and Wendler 1968). The primary exception to this pattern are 'jacks', which are sexually mature males that return to freshwater to spawn after only 5-7 months in the ocean.

Most west coast coho salmon enter rivers in October and spawn from November to December and occasionally into January. However, both run and spawn-timing of Central California coho salmon are very late (peaking in January) with little time spent in freshwater between river entry and spawning. This compressed adult freshwater residency appears to coincide with the single, brief peak of river flow characteristic of this area. Many small California systems have sandbars which block their mouths for most of the year except during winter. In these systems, coho salmon and other salmon species are unable to enter the rivers until sufficiently strong freshets break the sandbars (Sandercock 1991).

While central California coho spend little time between river entry and spawning, northern stocks may spend 1 or 2 months in fresh water before spawning (Flint and Zillges 1980; Fraser *et al.* 1983). In larger river systems like the Klamath River, coho salmon have a broad period of freshwater entry spanning from August until December (Leidy and Leidy 1984). In general, earlier migrating fish spawn farther upstream within a basin than later migrating fish, which enter rivers in a more advanced state of sexual maturity (Sandercock 1991). Adult coho salmon normally migrate when water temperatures are 44.96 to 60.08 degrees F, minimum water depth is seven inches and streamflow velocity does not exceed 2.44 m/s (Reiser and Bjornn 1979). If the conditions are not right, coho will wait at the mouth of the river or stream for the correct conditions. Most coho stocks migrate upstream during daylight hours. Generally, the coho build their redds at the head of riffles where there is good intra-gravel flow and oxygenation. Gribanov (1948) found that spawning coho appear to favor areas where the stream velocity is 0.30 to 0.55 m/s. Water quality can be clear or heavily silted with varying substrate of fine gravel to coarse rubble. California coho spawn in water temps of 42.08 to 55.94 degrees F (Briggs 1953).

Coho salmon eggs hatch in approximately 38 days at 51.26 degrees F, but, this duration depends on ambient water temperatures (Shopovalov and Taft 1954). Young fry hide in gravel and under large rocks during daylight hours. After several days growth, they move closer to the banks seeking out quiet backwaters, side channels and small creeks, especially those with overhanging riparian vegetation (Gribanov (1948). As they grow, they move into areas with less cover and higher velocity flows (Lester and Genoe 1970). Most fry move out of the system with winter and early spring freshets; however, some level of emigration may occur all year long. Brett (1952) found that coho salmon juveniles had an upper lethal temperature of 77 degrees F with a preferred rearing and emigration range of 53.6 to 57.2 degrees F. Taking advantage of cooler ambient temperatures and the afforded protection from predators, the bulk of seaward migration occurs at night.

Peak outmigration timing generally occurs in May, about a year after they emerge from the gravel. In California, smolts migrate to the ocean somewhat earlier, from mid-April to mid-May. Most smolts measure 90-115 mm, although Klamath River Basin smolts tend to be larger, but this is possibly due to influences of off-station hatchery plants. After entering the ocean, immature coho salmon initially remain in near-shore waters close to the parent stream. In general, coded-wire tag (CWT) recoveries indicate that coho salmon remain closer to their river of origin than do chinook salmon, but coho may nevertheless travel several hundred miles (Hassler 1987).

Foraging Ecology: Coho salmon fry usually emerge from the gravel at night from March to May. Coho salmon fry begin feeding as soon as they emerge from the gravel, and grow rapidly. In California, fry move into deep pools in July and August, where feeding is reduced and growth rate decreased (Shapovalov and Taft 1954). Between December and February winter rains result in increased stream flows and by March, following peak flows, fish feed heavily again on insects and crustaceans and grow rapidly.

<u>Historic and Current Distribution</u>: *Southern Oregon/Northern California Coast ESU (Threatened):* Recently, most coho salmon production in the Oregon portion of this ESU has been in the Rogue River. Recent run-size estimates (1979-1986) have ranged from about 800 to 19,800 naturally-produced adults, and from 500 to 8,300 hatchery-produced adults (Cramer 1994). Average annual run sizes for this period were 4,900 natural and 3,900 hatchery fish, with the total run averaging 45 percent hatchery fish. Adult passage counts at Gold Ray dam provide a long-term view of coho salmon abundance in the upper Rogue River (Cramer *et al.* 1985). In the 1940s, passage counts averaged about 2,000 adults per year. Numbers declined and fluctuated during the 1950s and early 1960s, then stabilized at an average of fewer than 200 adults during the late 1960s and early 1970s. In the late 1970s, the run increased with returning fish produced at Cole Rivers Hatchery. The remaining data is angler catch, which has ranged from less than 50 during the late 1970s to a peak of about 800 in 1991. Average annual catch over the least 10 years has been about 500 fish.

In the northern California region of this ESU, CDFG reported that coho salmon including hatchery stocks could be less than 6 percent of their abundance during the 1940s and have experienced at least a 70 percent decline in numbers since the 1960s (CDFG 1994). The Klamath River Basin (including the Trinity River) historically supported abundant coho salmon runs. In both systems, runs have greatly diminished and are now composed largely of hatchery fish, although small wild runs may remain in some tributaries (CDFG 1994).

Of 396 streams within the range of this ESU identified as once having coho salmon runs, recent survey information is available for 115 streams (30 percent) (Brown *et al.* 1994). Of these 117 streams, 73 (62 percent) still support coho salmon runs while 42 (36 percent) have lost their coho salmon runs. The rivers and tributaries in the California portion of this ESU were estimated to have average recent runs of 7,080 natural spawners and 17,156 hatchery returns, with 4,480 identified as native fish occurring in tributaries having little history of supplementation with non-native fish. Combining recent run-size estimates for the California portion of this ESU with the Rogue River estimates provides a run-size estimate for the entire ESU of about 12,000 natural fish and 21,000 hatchery fish.

Central California Coast ESU (Threatened): Statewide (including areas outside this ESU) coho salmon spawning escapement in California apparently ranged between 200,000 to 500,000 adults per year in the 1940s (Brown *et al.* 1994). By the mid-1960s, statewide spawning escapement was estimated to have fallen to about 100,000 fish per year (CDFG 1965; California Advisory Committee on Salmon and Steelhead Trout 1988), followed by a further decline to about 30,000 fish in the mid-1980s (Wahle and Pearson 1987; Brown *et al.* 1994). From 1987 to 1991, spawning escapement averaged about 31,000 with hatchery populations composing 57% of this total (Brown *et al.* 1994). Brown *et al.* (1994) estimated that there are probably less than 5,000 naturally-spawning coho salmon spawning in California each year, and many of these fish are in populations that contain less than 100 individuals.

Estimated average coho salmon spawning escapement in the Central California ESU for the period from the early 1980s through 1991 was 6,160 naturally spawning coho salmon and 332 hatchery spawned coho salmon (Brown *et al.*1994). Of the naturally-spawning coho salmon, 3,880 were from the tributaries in which supplementation occurs (the Noyo River and coastal streams south of San Francisco). Only 160 fish in the range of this ESU (all in the Ten Mile River) were identified as “native” fish lacking a history of supplementation with the non-native hatchery stocks. Based on redd counts, the estimated run of coho salmon in the Ten Mile River was 14 to 42 fish during the 1991-1992 spawning season (Maahs and Gilleard 1994).

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

Reasons for Decline and Threats to Survival: The factors threatening naturally reproducing coho salmon throughout its range are varied and numerous. For coho populations in the Central California coast ESU, the present depressed condition is the result of several long-standing, human induced factors (e.g., habitat degradation, timber harvest, water diversions, and artificial propagation).

Among other factors contributing to the decline and threat of survival for west coast coho, forestry, agriculture, mining, and urbanization have degraded, simplified, and fragmented habitat significantly throughout the range of the species. Water diversions for agriculture, flood control, domestic, and hydropower purposes have greatly reduced or eliminated historically accessible habitat. Studies estimate that during the last 200 years, the lower 48 states have lost approximately 53% of all wetlands and the majority of the rest are severely degraded (Dahl, 1990; Tiner, 1991). California has experienced a 91 percent loss of its wetland habitat (Dahl, 1990; Jensen *et al.*,1990; Barbour *et al.*, 1991; Reynolds *et al.*, 1993).

Infectious disease is one of the many factors that can influence adult and juvenile survival. Coho are exposed to numerous bacterial, protozoan, viral, and parasitic organisms in spawning and rearing areas, hatcheries, migratory routes, and the marine environment, poor water quality within these habitats increase coho vulnerability to disease and predation.

Implementation of existing regulatory mechanisms, specifically sections 303 (d) (1) (C) and (D) of the CWA, designed to protect beneficial resources including fisheries resources have not been effective in protecting fisheries resources or the aquatic ecosystem on which they depend, particularly with respect to non-point sources of pollution.. In addition, section 404 of the CWA does not adequately address the cumulative and additive effects of loss of habitat through continued development of waterfront, riverine, coastal, and wetland properties that also contribute to the degradation and loss of important aquatic ecosystem components necessary to maintain the functional integrity of these habitat features.

Delta Smelt (*Hypomesus transpacificus*)

Species Description and Life History: The delta smelt was federally listed as a threatened species on March 5, 1993 (58 **FR** 12854). On December 19, 1994, a final rule designating critical habitat for the delta smelt was published in the Federal Register (59 **FR** 65256). Critical habitat for delta smelt was originally proposed in the lower Sacramento-San Joaquin Delta and Suisun and Honker bays. However, after considerable debate, critical habitat was reproposed and is now contained within Contra Costa, Sacramento, San Joaquin, Solano, and Yolo counties.

The delta smelt is a slender-bodied fish with a steely blue sheen on the sides, and appears almost translucent (Moyle 1976a). They have an average length of 60 to 70 mm (about two to 3 inches). The delta smelt is a euryhaline species (tolerant of a wide salinity range) that spawns in fresh water and has been collected from estuarine waters up to 14 parts per thousand (ppt) salinity (Moyle *et al.* 1992). For a large part of its annual life span, this species is associated with the freshwater edge of the mixing zone (a saltwater-freshwater interface; also called X2), where the salinity is approximately two ppt (Ganssle 1966; Moyle *et al.* 1992; Sweetnam and Stevens 1993).

The delta smelt is adapted to living in the highly productive San Francisco Bay/Delta Estuary (Estuary) where salinity varies spatially and temporally according to tidal cycles and the amount of freshwater inflow. Despite this tremendously variable environment, the historical Estuary probably offered relatively constant suitable habitat conditions for the delta smelt because it could move upstream or downstream with the mixing zone (Moyle, pers. comm., 1993).

Feeding ecology: Delta smelt feed primarily on planktonic copepods, cladocerans (small crustaceans), amphipods, and to a lesser extent, insect larvae. Larger fish may also feed on the opossum shrimp (*Neomysis mercedis*). The most important food item for all age classes is the euryhaline copepod (*Eurytemora affinis*). Delta smelt are a pelagic fish and their food source is within the water column.

Spawning behavior: Shortly before spawning, adult delta smelt migrate upstream from the brackish-water habitat associated with the mixing zone to disperse widely into river channels and tidally-influenced backwater sloughs (Radtke 1966; Moyle 1976a; Wang 1991). Migrating adults with nearly mature eggs were taken at the Central Valley Project's (CVP) Tracy Pumping Plant from late December 1990 to April 1991 (Wang 1991). Spawning locations appear to vary widely from year to year (DWR and USDI 1993). Sampling of larval delta smelt in the Delta suggests spawning has occurred in the Sacramento River, Barker, Lindsey, Cache, Georgiana, Prospect, Beaver, Hog, and Sycamore sloughs, in the San Joaquin River off Bradford Island including Fisherman's Cut, False River along the shore zone between Frank's and Webb tracts, and possibly other areas (Dale Sweetnam, Calif. Dept. Of Fish and Game, pers. comm.; Wang 1991). Delta smelt also may spawn north of Suisun Bay in Montezuma and Suisun sloughs and their tributaries (Sweetnam, Calif. Dept. Of Fish and Game, pers. comm.).

Delta smelt spawn in shallow, fresh, or slightly brackish water upstream of the mixing zone (Wang 1991). Most spawning occurs in tidally-influenced backwater sloughs and channel edgewaters (Moyle 1976a; Wang 1986, 1991; Moyle *et al.* 1992). Although delta smelt spawning behavior has not been observed in the wild (Moyle *et al.* 1992), the adhesive, demersal eggs are thought to attach to substrates such as cattails, tules, tree roots, and submerged branches (Moyle 1976a; Wang 1991).

The spawning season varies from year to year, and may occur from late winter (December) to early summer (July). Moyle (1976a) collected gravid adults from December to April, although ripe delta smelt were most common in February and March. In 1989 and 1990, Wang (1991) estimated that spawning had taken place from mid-February to late June or early July, with peak spawning occurring in late April and early May. A recent study of delta smelt eggs and larvae (Wang and Brown 1994 as cited in DWR & USDI 1994) confirmed that spawning may occur from February through June, with a peak in April and May. Spawning has been reported to occur at water temperatures of about 7° to 15° C. Results from a University of California at Davis (UCD) study (Swanson and Cech 1995) indicate that although delta smelt tolerate a wide range of temperatures (<8° C to >25° C), warmer water temperatures restrict their distribution more than colder water temperatures.

Laboratory observations indicate that delta smelt are broadcast spawners that spawn in a current, usually at night, distributing their eggs over a local area (Lindberg 1992 and Mager 1993 as cited in DWR & USDI 1994). The eggs form an adhesive foot that appears to stick to most surfaces. Eggs attach singly to the substrate, and few eggs were found on vertical plants or the sides of a culture tank (Lindberg 1993 as cited in DWR & USDI 1994).

Delta smelt eggs hatched in nine to 14 days at water temperatures ranging from 13° to 16° C during laboratory observations in 1992 (Mager 1992 as cited in Sweetnam and Stevens 1993). In this study, larvae began feeding on phytoplankton on day four, rotifers on day six, and *Artemia nauplii* at day 14. In laboratory studies, yolk-sac fry were found to be positively phototaxic, swimming to the lightest corner of the incubator, and negatively buoyant, actively swimming to the surface. The post-yolk-sac fry were more evenly distributed throughout the water column (Lindberg 1992 as cited in DWR & USDI 1994). After hatching, larvae and juveniles move downstream toward the mixing zone where they are retained by the vertical circulation of fresh and salt waters (Stevens *et al.* 1990). The pelagic larvae and juveniles feed on zooplankton. When the mixing zone is located in Suisun Bay where there is extensive shallow water habitat within the euphotic zone (depths less than four meters), high densities of phytoplankton and zooplankton may accumulate (Arthur and Ball 1978, 1979, 1980). In general, estuaries are among the most productive ecosystems in the world (Goldman and Horne 1993). Estuarine environments produce an abundance of fish and zooplankton as a result of plentiful food and shallow, productive habitat.

Swimming behavior. Observations of delta smelt swimming in the swimming flume and in a large tank show that these fish are unsteady, intermittent, slow-speed swimmers (Swanson and Cech 1995). At low velocities in the swimming flume (<three body lengths per second), and during spontaneous, unrestricted swimming in a 1-meter tank, delta smelt consistently swam with a "stroke and glide" behavior. This type of swimming is very efficient; Weihs (1974) predicted energy savings of about 50 percent for "stroke and glide" swimming compared to steady swimming. However, the maximum speed delta smelt are able to achieve using this preferred mode of swimming, or gait, is less than three body lengths per second, and the fish did not readily or spontaneously swim at this or higher speeds (Swanson and Cech 1995). Juvenile delta smelt proved stronger swimmers than adults. Forced swimming at these speeds in a swimming flume was apparently stressful; the fish were prone to swimming failure and extremely vulnerable to impingement. Unlike fish for which these types of measurements have been made in the past, delta smelt swimming performance was limited by behavioral rather than physiological or metabolic constraints *(e.g.,* metabolic scope for activity; Brett 1976). Please refer to the Service (USDI-FWS 1994a, 1996a) and Department of Water Resources and United States Department of Interior - Bureau of Reclamation (DWR & USDI 1994) for additional information on the biology and ecology of this species.

Primary Constituent Elements of Critical Habitat: In designating critical habitat for the delta smelt, the Service identified the following primary constituent elements essential to the conservation of the species: physical habitat, water, river flow, and salinity concentrations required to maintain delta smelt habitat for spawning, larval and juvenile transport, rearing, and adult migration.

Spawning habitat. Specific areas that have been identified as important delta smelt spawning habitat include Barker, Lindsey, Cache, Prospect, Georgiana, Beaver, Hog, and Sycamore sloughs and the Sacramento River in the Delta, and tributaries of northern Suisun Bay.

Larval and juvenile transport. Adequate river flow is necessary to transport larvae from upstream spawning areas to rearing habitat in Suisun Bay and to ensure that rearing habitat is maintained in Suisun Bay. To ensure this, X2 must be located westward of the confluence of the Sacramento-San Joaquin Rivers, located near Collinsville (Confluence), during the period when larvae or juveniles are being transported, according to historical salinity conditions. X2 is important because the "entrapment zone" or zone where particles, nutrients, and plankton are "trapped", leading to an area of high productivity, is associated with its location. Habitat conditions suitable for transport of larvae and juveniles may be needed by the species as early as February 1 and as late as August 31, because the spawning season varies from year to year and may start as early as December and extend until July.

Rearing habitat. An area extending eastward from Carquinez Strait, including Suisun, Grizzly, and Honker bays, Montezuma Slough and its tributary sloughs, up the Sacramento River to its confluence with Three Mile Slough, and south along the San Joaquin River including Big Break, defines the specific geographic area critical to the maintenance of suitable rearing habitat. Three Mile Slough represents the approximate location of the most upstream extent of historical tidal incursion. Rearing habitat is vulnerable to impacts of export pumping and salinity intrusion from the beginning of February to the end of August.

Adult migration. Adequate flow and suitable water quality are needed to attract migrating adults in the Sacramento and San Joaquin river channels and their associated tributaries, including Cache and Montezuma sloughs and their tributaries. These areas are vulnerable to physical disturbance and flow disruption during migratory periods.

<u>Historic and Current Distribution</u>: The delta smelt is endemic to Suisun Bay upstream of San Francisco Bay through the Delta in Contra Costa, Sacramento, San Joaquin, Solano and Yolo counties, California. Historically, the delta smelt is thought to have occurred from Suisun Bay upstream to at least the city of Sacramento on the Sacramento River, and Mossdale on the San Joaquin River (Moyle *et al.* 1992; Sweetnam and Stevens 1993).

<u>Reasons for Decline and Threats to Survival</u>: The delta smelt is adapted to living in the highly productive Estuary where salinity varies spatially and temporally according to tidal cycles and the amount of freshwater inflow. Despite this tremendously variable environment, the historical Estuary probably offered relatively consistent spring transport flows that moved delta smelt juveniles and larvae downstream to the mixing zone (P. Moyle, pers. comm.). Since the 1850's, however, the amount and extent of suitable habitat for the delta smelt has declined dramatically. The advent in 1853 of hydraulic mining in the Sacramento and San Joaquin rivers led to

increased siltation and alteration of the circulation patterns of the Estuary (Nichols *et al.* 1986; Monroe and Kelly 1992). The reclamation of Merritt Island for agricultural purposes, in the same year, marked the beginning of the present-day cumulative loss of 94 percent of the Estuary's tidal marshes (Nichols *et al.* 1986; Monroe and Kelly 1992).

In addition to the degradation and loss of estuarine habitat, the delta smelt has been increasingly subject to entrainment, upstream or reverse flows of waters in the Delta and San Joaquin River, and constriction of low salinity habitat to deep-water river channels of the interior Delta (Moyle *et al.* 1992). These adverse conditions are primarily a result of drought and the steadily increasing proportion of river flow being diverted from the Delta by the CVP and State Water Project (SWP) (Monroe and Kelly 1992). The relationship between the portion of the delta smelt population west of the Delta as sampled in the summer townet survey and the natural logarithm of Delta outflow from 1959 to 1988 (Department and Reclamation 1994) indicates that the summer townet index increased dramatically when outflow was between 34,000 and 48,000 cfs which placed X2 between Chipps and Roe islands. Placement of X2 downstream of the Confluence, Chipps and Roe islands provides delta smelt with low salinity and protection from entrainment, allowing for productive rearing habitat that increases both smelt abundance and distribution.

Delta smelt critical habitat has been affected by activities that destroy spawning and refugial areas and change hydrology patterns in Delta waterways. Critical habitat also has been affected by diversions that have shifted the position of X2 upstream of the confluence of the Sacramento and San Joaquin rivers. This shift has caused a decreased abundance of delta smelt. Existing baseline conditions and implementation of the Service's 1994 and 1995 biological opinions concerning the operation of the CVP and SWP, provide a substantial part of the necessary positive riverine flows and estuarine outflows to transport delta smelt larvae downstream to suitable rearing habitat in Suisun Bay outside the influence of marinas, agricultural diversions, and Federal and State pumping plants.

The Service's 1994 and 1995 biological opinions provided for adequate larval and juvenile transport flows, rearing habitat, and protection from entrainment for upstream migrating adults (USDI-FWS 1994a). Please refer to 59 **FR** 65255 for additional information on delta smelt critical habitat.

Desert Pupfish (*Cyprinodon macularius*)

<u>Species Description and Life History</u>: On March 31, 1986 (51 **FR** 10850), the Service determined the desert pupfish to be an endangered species and critical habitat was designated for this species in Imperial County, California and Pima County, Arizona.

The desert pupfish is a small laterally compressed fish with a smoothly rounded body shape. Adult fish rarely grow larger than 75 millimeters (3 inches) in total length. Males are larger than females and during the reproductive season become brightly colored with blue on the dorsal portion of the head and sides and yellow on the caudal fin and the posterior part of the caudal peduncle. Females and juveniles typically have tan to olive backs and silvery sides. Most adults have narrow, vertical, dark bars on their sides, which are often interrupted to give the impression of a disjunct, lateral band. They are adapted to harsh desert environments and are capable of surviving extreme environmental conditions (Moyle 1976a; and Lowe *et al.* 1967). Although desert pupfish are extremely hardy in many respects, they cannot tolerate competition or predation and are thus readily displaced by exotic fishes.

Desert pupfish mature rapidly and may produce up to three generations per year. Spawning males typically defend a small spawning and feeding territory in shallow water. The eggs are usually laid and fertilized on a flocculent substrate and hatch within a few days. After a few hours, the young begin to feed on small plants and animals. Spawning occurs throughout the spring and summer months. Individuals typically survive for about a year. Desert pupfish forage on a variety of insects, other invertebrates, algae, and detritus.

Foraging Ecology: Desert pupfish typically occur in shallow water and forage on a variety of insects, other invertebrates, algae, and detritus.

<u>Historic and Current</u> Distribution: The desert pupfish was once common in the desert springs, marshes, and tributary streams of the lower Gila and Colorado River drainages in Arizona, California, and Mexico (Minckly 1973 & 1980; Miller and Fuiman 1987; USDI-FWS 1993b). It also formerly occurred in the slow-moving reaches of some large rivers, including the Colorado, Gila, San Pedro, and Santa Cruz. In California, this species is currently known from only a few historic locations. It still exists in two Salton Sea tributaries (San Felipe Creek system and its associated wetland San Sebastian Marsh, Imperial County, and Salt Creek, Riverside County) and a few shoreline pools and irrigation drains along the Salton Sea in Imperial and Riverside Counties (Nichol *et al.* 1991; USDI-FWS 1993b).

<u>Reasons for Decline and Threats to Survival</u>: There are many reasons for declines of desert pupfish populations. They include habitat loss (dewatering of springs, some headwaters, and lower portions of major streams and marshlands), habitat modification (stream impoundment, channelization, diversion, and regulation of discharge, plus domestic livestock grazing and other watershed uses such as mining, and road construction), pollution, and interactions with non-native species (competition for food and space, and predation) (Matsui 1981; Minckley 1985; Miller and Fuiman 1987; USDI-FWS 1993b).

Many historic pupfish localities have been dried by groundwater pumping, channel erosion or arroyo formation, and water impoundment and diversion (Hastings and Turner 1965, Fradkin 1981, Rea 1983, Hendrickson and Minckley 1985). Impoundment also creates upstream habitat unsuitable for pupfish because of increased depth which, because of its lentic character, is more conducive to occupation by non-native fishes. Grazing by domestic livestock may reduce terrestrial vegetative cover, enhance watershed erosion, exacerbate problems of arroyo cutting, and increase sediment loads and turbidity in receiving waters. Habitats may be further impacted by trampling where cattle feed or drink in or adjacent to water. Contamination of the habitat of desert pupfish may have contributed to its decline.

Non-native fishes pose the greatest threat to extant desert pupfish populations (Minckley and Deacon 1968, Deacon and Minckley 1974, Schoenherr 1981 & 1988, Meffe 1985, Miller and Fuiman 1987). Non-native fishes that occupy habitats also used by pupfish include mosquitofish (*Gambusia affinis*), sailfin molly (*Poecilia latipinna*), largemouth bass (*Micropterus salmoides*), and juvenile cichlids (*Oreochromis ssp.* and *Tilapia ssp.*). Primary mechanisms of replacement include predation and aggression (mosquitofish and largemouth bass) and behavioral activities that interfere with reproduction (mollies and cichlids) (Matsui 1981, Schoenherr 1988).

As part of the National Irrigation Water Quality Program, the Service conducted a study to determine body burdens of contaminants in a surrogate species, sailfin mollies (*Poecilia latipinna*) for the endangered desert pupfish. Sailfin mollies were trapped in 13 agricultural drains. At one drain sampling site both mollies and desert pupfish were collected and submitted for analysis; contaminant levels between the two species were generally in agreement, especially for selenium. Mollies collected from 10 of 13 drains and pupfish contained 3 to 6 ppm dry weight selenium, above the levels of concern for warmwater fishes (CAST, 1994; Gober, 1994; Ohlendorf, 1996). Mollies in two other drains contained 6.4 and 10.2 ppm, dry weight selenium, above thresholds for toxicity for warmwater fish reproductive hazards (Lemly 1993a). Lemly (1993a), concluded that 4 ppm dry weight whole body selenium should be considered the toxic effect threshold for the overall health of and reproductive vigor for freshwater fish. These findings indicate that the desert pupfish is likely at risk to reduced reproductive vigor and condition as a result of elevated levels of selenium in its environment.

Lahontan Cutthroat Trout (*Oncorhynchus clarki henshawi*)

Species Description and Life History: The Lahontan cutthroat trout is an inland subspecies of cutthroat trout endemic to the physiographic Lahontan basin of northern Nevada, eastern California, and southern Oregon. It was listed as endangered by the Service in 1970 (35 **FR** 13520) and subsequently reclassified as threatened in 1975 (40 **FR** 229864). No critical habitat has been designated for this species.

The Lahontan cutthroat trout can be distinguished from other subspecies of cutthroat trout by three characteristics identified by Behnke (1979, 1992). These characteristics include: (1) the pattern of medium-large rounded spots, somewhat evenly distributed over the sides of the body, on the head, and often on the abdomen; (2) the highest number of gill rakers found in any trout, 21 to 28, with mean values ranging from 23 to 26; and (3) a high number of pyloric caeca, 40 to 75 or more, with mean values of more than 50.

Lahontan cutthroat trout inhabit both lakes and streams, but are obligatory stream spawners. Intermittent tributary streams are frequently used as spawning sites (Coffin 1981; Trotter 1987). Spawning generally occurs from April through July, depending on stream flow, elevation, and water temperature (Calhoun 1942; La Rivers 1962; McAfee 1966; Lea 1968; Moyle 1976a). Eggs are deposited in 0.25 to 0.5 inch gravels within riffles, pocket water, or pool crests. Spawning beds must be well oxygenated and relatively silt free for good egg survival. Optimum Lahontan cutthroat trout habitat is characterized by 1:1 pool-riffle ratios, well vegetated stable stream banks, over 25 percent cover, and a relatively silt free rocky substrate (Hickman and Raleigh 1982). They can tolerate much higher alkalinities than other trout and seem to survive daily temperature fluctuations of 14-20 degrees C (57-68 degrees F). They do best in waters with average maximum temperatures of 13 degrees C (55 degrees F).

Foraging Ecology: Lahontan cutthroat trout are opportunistic feeders; in streams they feed on the most common terrestrial and aquatic insects which get caught in the drift (Coffin 1983).

Historic and Current Distribution: Lahontan cutthroat trout historically occupied a wide variety of cold water habitats, including large terminal alkaline lakes, oligotrophic alpine lakes, meandering low-gradient rivers, montane rivers, and small headwater tributary streams. Prior to this century, there were 11 lake populations and an estimated 300 to 600 river populations in more than 3,600 miles of streams (USDI-FWS 1995). The western Lahontan Basin population segment includes the Truckee, Carson, and Walker River basins in California.

Lahontan cutthroat trout currently occupy between 155 and 160 streams as well as six lakes and reservoirs in California, Nevada, Oregon, and Utah. Self-sustaining populations occur in 10.7 percent of fluvial and 0.4 percent of lacustrine historical habitat (USDI-FWS 1995). The species has been introduced outside of its native range, primarily for recreational angling purposes. Three distinct vertebrate population segments have been identified by the Service based on geographical, ecological, behavioral, and genetic factors (USDI-FWS 1995).

Lahontan cutthroat trout were introduced into the Upper Truckee River watershed in 1990 and 1991 as part of the species' recovery program. The Upper Truckee River is within a watershed that historically contained Lahontan cutthroat trout. During the summer and fall of 1990, 5,000 fingerlings and 200 adults were planted. In 1991, 2,000 fingerlings and 110 adults were planted into the Upper Truckee River watershed. Before Lahontan cutthroat trout were introduced into these waters, the streams and lakes were treated by CDFG to remove non-native salmonids. The LTBMU has conducted ocular surveys annually since the introduction. In 1995, just under 250 fish were observed, mostly adults. This is down from the 1994 survey of approximately 360 Lahontan cutthroat trout.

Reasons for Decline and Threats to Survival: Major impacts to Lahontan cutthroat trout habitat and abundance include 1) reduction and alteration of stream discharge; 2) alteration of stream channels and morphology; 3) degradation of water quality ; 4) reduction of lake levels and concentrated chemical components in natural lakes; and 5) introduction of non-native fish species. There alterations are usually associated with agricultural use, livestock and feral horse grazing, mining, and urban development. Alteration and degradation of trout habitat have also resulted from logging, highway and road construction, dam building, and the discharge of effluent form wastewater treatment facilities. All these factors reduce the suitability of habitat for the trout (USDI-FWS 1995).

Little Kern Golden Trout (*Oncorhynchus aquabonita whitei*)

Species Description and Life History: The Little Kern golden trout was federally listed as threatened and critical habitat was designated concurrently on April 13, 1978 (43 **FR** 15427). Critical habitat was defined to include all streams and tributaries in the Little Kern River drainage above a barrier falls on the Little Kern River located one mile below the mouth of Trout Meadows Creek. The CDFG has prepared a management plan that has been accepted by the Service as the official recovery plan for Little Kern golden trout. The fishery objectives for conditions within the proposed project boundaries are restoration of pure strain Little Kern golden trout to its critical habitat, protection of critical habitat, and protection and/or restoration of the native Sacramento sucker (*Catostomus occidentalis*).

The Little Kern golden trout requires diverse habitat composed of pools for refugia, instream cover, shade from bankside vegetation to regulate temperature, and gravel substrates for spawning (USDA-FS 1993). Desired habitat includes deep, narrow channels within low gradient meadow environments. Low width to depth ratios and a large percentage of undercut banks are considered indicators of desirable meadow habitat conditions. Desirable habitat outside meadows contains good cover from cobble and boulders (USDA-FS 1993). Little Kern golden trout reach sexual maturity at three years, although some younger fish do exhibit courtship behavior (Smith 1977). Spawning occurs during the spring. Males establish spawning sites on the downstream edge of pools over gravel substrates. Spawning occurs at a water depth of 5 to 15 cm (Smith 1977).

Foraging Ecology: Little Kern golden trout forage on a variety of invertebrates, eating whatever is most abundant in the water column. Diet includes larval and adult insects and planktonic crustaceans (Moyle 1976a).

Historic and Current Distribution: The historical distribution of Little Kern golden trout was restricted to the Little Kern River drainage down to a barrier falls that isolated Little Kern golden trout from Kern River rainbow trout in the Kern River. Approximately 40 of the estimated 100 miles of suitable trout habitat in the Little Kern River drainage are thought to have supported Little Kern golden trout prior to human influence (USDA-FS 1993). Early activities of settlers in the area included transplanting Little Kern golden trout into many nearby waters (Schreck 1969). After human influence, nearly 90 miles of streams and several lakes contained Little Kern golden trout (USDA-FS 1993). Between 1900 and 1950, rainbow trout and brook trout were also transplanted into the Little Kern River watershed. The Little Kern golden trout does not compete well with other species and also hybridizes with rainbow trout. By 1970, only 10.2 miles of streams in the Little Kern River system contained pure Little Kern golden trout (USDA-FS 1993).

The CDFG has been involved in an intensive program to eradicate the non-native fish species within the Little Kern River system. Over the last 20 years, treatment with antimycin or rotenone (fish toxicants) have been used to treat many of the streams, lakes, and a portion of the Little Kern River. Populations of pure strain Little Kern golden trout are now inhabiting many of the treated sections of streams and lakes. Treatments were completed in 1995, with delisting of the species the future goal once studies determine that the fish are pure and at adequate population levels according to the Revised Plan.

Reasons for Decline and Threats to Survival: Little Kern golden trout do not compete well with other species. Hybridization and interspecific competition result in reduced genetic purity and lower population numbers (USDA-FS 1993).

Lost River Sucker (*Deltistes luxatus*)

Species Description and Life History: The Lost River sucker was described by Cope (1879) from specimens he collected from Upper Klamath Lake. A complete discussion of the taxonomy of the species can be found in the Service's Lost River and Shortnose Sucker Recovery Plan (USDI-FWS 1993c). The Lost River sucker was federally listed as endangered species on July 18, 1988 (53 **FR** 27134). The Clear Lake watershed is considered Unit 1 of the proposed designation of six Critical Habitat Units (CHUs) for Lost River and shortnose suckers. Primary constituent elements include water of sufficient quantity and quality to provide conditions required for the particular life stage of the species; physical habitat inhabited or potentially habitable by shortnose suckers for use as refugia, spawning, nursery, feeding, or rearing areas, or as corridors between these areas; and food supply and a natural scheme of predation, parasitism, and competition in the biological environment.

Scoppettone (1988) found shortnose suckers up to 33 years of age in Copco Reservoir and Lost River suckers to 43 years of age in upper Klamath Lake. In the Clear Lake drainage, Scoppettone (1988) found shortnose suckers from one to 23 years of age, and Lost River suckers from one to 27 years old. Lost River suckers can achieve lengths approaching one meter. Sexual maturity is achieved in approximately nine years for Lost River suckers (Scoppettone, pers. comm., cited in USDI-FWS 1994c).

The role upstream populations of Lost River suckers play in the maintenance and viability of downstream populations is poorly understood at this time.

Foraging Ecology: The diet of Lost River suckers includes detritus, zooplankton, algae, and aquatic insects (Buettner and Scoppetone 1990).

Historic and Current Distribution:. The Lost River sucker (along with the shortnose sucker) is endemic to the upper Klamath Basin, Oregon and California, and were once quite abundant. Cope (1884) noted that Upper Klamath Lake sustained "a great population of fishes" and was "more prolific in animal life" than any body of water known to him at that time. Gilbert (1898) noted that the Lost River sucker was "the most important food-fish of the Klamath Lake region." At that time, spring sucker runs "in incredible numbers" (Gilbert 1898) were relied upon as a food source by the Klamath and Modoc Indians and were taken by local settlers for both human consumption and livestock feed (Cope 1879; Coots 1965; Howe 1968). Sucker runs were so numerous, that a cannery was established on the Lost River (Howe 1968) and several other commercial operations processed "enormous amounts" of suckers into oil, dried fish, and other products (Andreasen 1975).

The Lost River sucker was historically found in Upper Klamath Lake and its tributaries, including the Williamson, Sprague, and Wood rivers (Williams *et al.* 1985), Crooked, Seven Mile, Four Mile, Odessa, and Crystal creeks (Stine 1982). It was also found in the Lost River system, Tule Lake, Lower Klamath Lake, and Sheepy Lake (Moyle 1976a).

In a distributional survey of the Clear Lake watershed conducted in the summers of 1989 and 1990, Lost River suckers were collected in lower Willow Creek and Boles Creek upstream to Avanzino Reservoir (Buettner and Scoppettone 1991). Under higher flow conditions, such as the spring of 1993, the range probably extended upstream in all of the creeks in the Clear Lake watershed (M. Buettner, pers. comm., cited in USDI-FWS 1994c). Lost River and shortnose suckers have been captured in the Lost River below Clear Lake and were taken to Malone Reservoir in 1992 during Reclamation's salvage operation at Clear Lake. Buettner (pers. comm. 1995) believes it is unlikely that many suckers remain in Malone Reservoir. The reservoir is drained each fall to a small pool and most of the fish were likely washed down stream into the Lost River.

<u>Reasons for Decline and Threats to Survival</u>: The factors believed to be responsible for the decline of the Lost River suckers include the damming of rivers, dredging and draining of marshes, instream flow diversions, a shift toward hyper eutrophication in Upper Klamath Lake, and other traditional land use practices. A recent analysis of the population genetics of the shortnose and Lost River suckers (Moyle and Berg 1991) suggested that "if populations continue to decline, these species may cross below the minimum viable population threshold and be lost". Entire stocks may have already been lost [e.g., Harriman Springs (Andreasen 1975)].

Suckers appear to be strongly influenced by poor water quality induced by high water temperatures, nutrient enrichment, algal blooms and die-offs, low dissolved oxygen, high pH, and possibly high ammonia (Kann and Smith 1993; Perkins 1997). Higher recruitment success occurs during above-average water quality years; in contrast, large-scale fish kills of adult suckers in the Upper Klamath Lake and Williamson Rivers appear related to poor water quality (Perkins 1997). Although fish kills have occurred sporadically in the 1900s, they appear to have increased in size, duration, and areal extent in recent years and may be adversely affecting current recovery efforts (Perkins 1997). A 1996 August-September fish kill, consisting almost exclusively of the endangered suckers, had the documented deaths of more than 6049 individuals, with many thousands of additional fish estimated to have been killed (Perkins 1997). Another subsequent kill in the Lake in 1997 involved primarily tui chubs, but more than 1400 endangered suckers deaths were also documented (Mark Buettner, Reclamation, pers. comm.). Although the ultimate causes of these fish kills was identified as the bacterial infections of the skin and gills by *Flavobacterium columnare*, degenerative changes in the intestines, livers and kidneys of many of the fish were also observed in the 1996 fish. Lesions of the kidneys were indicative of toxic tubular necrosis, typically caused by heavy metals, pesticides, and other poisons (Foote 1996). Foote suggested that a likely source of toxins in the Upper Klamath Lake system was *Microcystis*, a cyanobacterium producing the toxin microcystin. This bacterium was in bloom during the 1996 fish kill and its toxin was detected in 3 of 9 dead suckers from the 1996 fish kill (Klamath Falls Fish and Wildlife Office, U.S. Fish and Wildlife Service, unpublished data).

In addition, to fish kills, suckers in the Klamath Basin suffer from abnormally high rates of parasitism and physical deformities (Biological Research Division, U.S. Geological Survey, unpublished) that may be related to water quality, nutritional deficiencies, or contaminant exposures. Fish in the Tule Lake area also suffer very high rates of parasitism and deformities (Littleton 1993), although sucker health has not specifically been documented. Overharvest and chemical contamination may have also contributed to the decline. Reduction and degradation of lake and stream habitats in the upper Klamath Basin is considered to be the most important factor in the decline of the endangered suckers (USDI-FWS 1993c). Very low numbers of benthic organisms in many locations and an overall reduction in numbers of aquatic reptiles in the habitat of the sucker may have been caused by pollution of organochlorine pesticides and other pollutants (USDI-FWS 1993c).

Modoc Sucker (*Catostomus microps*)

Species Description and Life History: The Modoc sucker is a dwarf catostomid. The species was federally listed as endangered, with critical habitat designated on June 11, 1985 (50 **FR** 42530). Critical habitat was described to include the following reaches: Johnson Creek from the confluence with Rush Creek upstream approximately four river miles including two tributaries in Higgins Flat and Rice Flat; Rush Creek from the gaging station on highway 299 upstream to the Upper Rush Creek campground; Turner Creek from its confluence with the Pit River upstream about 4.5 river miles; Washington Creek from its confluence with Turner Creek upstream approximately four river miles, including 1.5 miles of Coffee Mill Creek; and approximately 3.5 miles of Hulbert Creek from its confluence with Turner Creek, including 1.5 miles of Cedar Creek. The Modoc sucker also exists in Coffee Mill, Willow, Ash, and Rush creeks (Studinski 1993) for a total of 25 miles (Gina Sato, BLM, pers. comm. 1991). Previously, the California Department of Fish and Game had classified the Modoc sucker as “rare” in 1973 and “endangered” in 1980.

The Modoc sucker was first described in 1908 by C. Rutter from three paratypes collected from Rush Creek in 1898. Unlike many other native fish species, the Modoc sucker’s nomenclature has never been questioned. *Catostomus* refers to the inferior position of the mouth (Moyle 1976a), and *microps* means “small eye” (Mills 1980). The species can be distinguished from other catostomids by the number of dorsal rays (n = 10-12), the number of scales in the lateral line (n = 79-89), and their small body size (<160 mm) (Mills 1980).

Life history studies (Moyle and Marciochi 1975) indicate Modoc suckers are most successful in small, relatively undisturbed, pool-dominated streams where they are isolated from Sacramento sucker (*Catostomus occidentalis*), with which they can hybridize. Modoc sucker habitat is typified by extreme water flows (Studinski 1993). Flows are very high in winter and spring months, but by mid-summer, large reaches of habitat dry up. During these times, fish populations are confined to relatively small, permanent pools. Adults (>70 - 85 mm TL) prefer pools from one foot to over four feet deep during summer. Smaller fish have been observed in riffles and shallow pools in large schools (Studinski 1993). Moyle and Marciochi (1975) found that Modoc suckers were most abundant in areas with low flows, large shallow pools with muddy bottoms or gravel to cobble substrate, partial shade, and moderately clear water. Studinski (1993) found Modoc sucker in pools with maximum water temperature of less than 21°C with a daily temperature variation of less than 2°C. Little is known about Modoc sucker winter habitat requirements.

Moyle and Marciochi (1975) collected ripe males and females from mid-April to late May. They did not observe actual spawning behavior. Modoc suckers were observed spawning during a 1978 study. Boccone and Mills (1979) observed spawning occurring from mid-April through the first week of June. They reported that spawning behavior of Modoc sucker closely resembled that of the Tahoe sucker, a close relative. Spawning took place over coarse to fine gravel in the lower end of pools. Pools were located in meadow areas with abundant cover. Boccone and Mills (1979) also noted spawning coloration and tubercle development on mature male Modoc suckers, but they further noted that ripe females did not express these characteristics. Water temperature and photoperiod were thought to be factors controlling timing of spawning. Spawning was observed from midmorning to late afternoon with water temperature from 13.3°C to 16.1°C (Boccone and Mills 1979).

Foraging Ecology: The diet of the Modoc sucker consists mostly of detritus and algae, with insects and crustaceans making up 25% of the diet.

<u>Historic and Current Distribution</u>: The Modoc sucker is endemic to small streams tributary to the upper Pit River drainage in Modoc and Lassen counties, California. Its current range is restricted to the Turner and Ash Creek subsystems in Modoc County.

Past habitat and populations surveys gave different estimates to Modoc sucker population size. Moyle (1974) estimated the population of Modoc suckers to be less than 5,000 individuals, with an effective population of 200. Ford (1977) found 2,605 suckers, and estimated the effective population to be 104, based on length-frequency analyses. Mills (1980) estimated that only 1,300 genetically pure Modoc sucker remained. During recent habitat and population surveys for six of the nine known Modoc sucker streams, Scoppettone *et al.* (1994) estimated the population to be 3,000 suckers. Biologists on this research project did not differentiate between Modoc sucker and Sacramento sucker during their visual surveys.

Approximately 50 percent of Modoc sucker habitat lies on Modoc National Forest. Modoc sucker populations are generally considered to be stable to improving. Exclosures protect much of the species habitat. Most recovery actions, as outlined in the Modoc sucker recovery action plan (USDA-FS 1989) have been completed. During a recent drought, Modoc suckers were found in deep perennial pools.

Reasons for Decline and Threats to Survival: Main threats are habitat loss from overgrazing, siltation, channelization, and hybridization with a closely related *Catostomid*. Past and present grazing and channelization on both private and public lands have caused severe erosion and siltation, dramatically degrading the species' habitat. In some streams, erosional cutting of stream banks exposed as much as 10 vertical feet of earth. These habitat changes limited the distribution and abundance of the sucker to a point where, at the time the species was listed, only 1,300 genetically pure individuals were thought to remain (Mills 1980). Besides these changes in the habitat, the extreme erosion and channelization also removed natural barriers separating the Modoc sucker from the Sacramento sucker. Hybridization between these two species has occurred.

Mohave Tui Chub (*Gila bicolor mohavensis*)

Species Description and Life History: The Mohave tui chub was listed as endangered on October 13, 1970, without critical habitat (35 **FR** 16047). This account is based on Moyle 1976a and Moyle *et al.* 1989.

The Mohave tui chub, a member of the minnow family, can reach over 10 inches in length. The Mohave tui chub is the only fish native to the Mohave River basin in California. This species was thought to inhabit the deep pools and slough-like areas of the Mohave River. Mohave tui chubs are adapted to the Mohave River's alkaline, hard water. Mohave tui chubs have survived in habitats where dissolved oxygen was less than one microgram per liter; they also have some tolerance for high salinity and high water temperatures. Mohave tui chubs use aquatic vegetation to attach their eggs and for cover and thermal refuges.

Foraging Ecology: Mohave tui chubs are morphologically adapted for feeding on plankton. However, they readily consume food, such as bread and lunch meat, provided by visitors to their refugia.

Historic and Current Distribution: The Mohave tui chub is native to the Mohave River basin. Currently, the only known genetically pure Mohave tui chub populations are found in three artificial ponds, one natural spring, and a series of constructed drainage channels in San Bernardino County. The pond at the Desert Studies Center at Soda Dry Lake is maintained by groundwater pumping; MC Spring is a natural spring also located at the Desert Studies Center. The water supplying both of these habitats is likely from the underflow of the Mohave River.

The two ponds at Camp Cady receive water pumped from the underflow of the Mohave River. The remaining population at the Naval Air Weapons Station, China Lake, California resides in drainage channels which carry percolating water from a system of sewage ponds. The estimated population at China Lake is between 10,000 and 20,000 fish.

Reasons for Decline and Threats to Survival: The primary causes for the decline of the Mohave tui chub were the introduction of arroyo chubs and other exotic species into the Mohave River system and habitat alteration. The construction of headwater reservoirs altered natural flow regimes and provided favorable habitat for exotic species. Water diversions and pollution have decreased habitat suitability in other locations. Increases in permissible levels of environmental contaminants to the species' restricted habitat may have a deleterious effect on the species. The Mohave tui chub is native to the Mohave River basin, which has been identified as an impaired water body.

Owens Pupfish (*Cyprinodon radiosus*)

Species Description and Life History: The Owens pupfish was listed as endangered on March 11, 1967 (32 **FR** 4001). Population declines attributed to competition and predation by non-native species and habitat modification caused by water diversions from the Owens River and its tributaries were identified as the principal causes of the declines. The following information is summarized from the draft recovery plan for the wetland and aquatic species of the Owens Basin (USDI-FWS 1996a).

The Owens pupfish rarely exceeds 2.5 inches in length. Males can easily be distinguished from females by coloration; males are bright blue, particularly during the breeding season, while females are a dusky olive green.

Owens pupfish occupy habitat where water is relatively warm and food is plentiful. Spawning occurs over soft substrates. Eggs are laid singly and hatch in approximately 6 days when temperatures are from 24 to 27 degrees C. They reach maturity in three to four months and rarely live longer than one year.

Foraging Ecology: The Owens pupfish is an opportunistic omnivore. Their diet changes seasonally to include the most abundant organisms in their habitat. They forage in schools, mostly on insects such as chironomid larvae. They were probably the main predator on mosquito larvae when they were abundant (Moyle 1976a).

Historic and Current Distribution: Owens pupfish were reported as common in habitats throughout the Owens Valley in Inyo and Mono counties from Fish Slough, approximately 12 miles north of Bishop, south to Lone Pine. They were most abundant near the margins of marshes, from shallow sloughs bordering the Owens River, and from springs. They are currently known from four sites, all of which are managed to protect Owens pupfish from non-native fish: Warms Springs and the White Mountain Research Station in Inyo County, and BLM Spring and

Owens Valley Native Fish Sanctuary in Mono County. This species was thought to be extinct in 1942; all of the remaining fish have been propagated from a remnant population found in Fish Slough in 1964.

Reasons for Decline and Threats to Survival: The transfer of Owens River water to the Los Angeles Aqueduct and the subsequent loss of habitat almost caused the extinction of the Owens pupfish. Because all of the remaining Owens pupfish are descendants of one population, this species may lack the genetic variability found in other species of pupfish. This factor, along with the relatively brief life span, should be considered in any analysis of the effects of toxic substances on the Owens pupfish. The Owens River, the primary water course through the valley floor where this species occurs, has been declared an impaired water body.

Owens pupfish are extremely limited in distribution. The recovery plan for the Owen's pupfish determined that a population would be determined to be secure when 1) exotic species are controlled or eliminated, 2) emergent vegetation is controlled, and 3) sufficient water quality is guaranteed (USDI-FWS 1984a).

Owens Tui Chub (*Gila bicolor snyderi*)

Species Description and Life History: The Owens tui chub was listed as endangered on August 5, 1985 (50 **FR** 31592). The introduction of non-native fish that affect the Owens tui chub through competition, predation, and hybridization and diversion of water for agricultural and municipal use were the principal reasons for the listing. Critical habitat was designated for this species along eight miles of the Owens River in the Owens Gorge and at two springs at Hot Creek Fish Hatchery. Both of these locations are in Mono County. The following information is summarized from the draft recovery plan for the wetland and aquatic species of the Owens Basin (USDI-FWS 1996a).

The Owens tui chub may reach a length of 12 inches. Its dorsal coloration ranges from bronze to dusky green; its belly is silver or white. Reproductive information is not well-known for the Owens tui chub; however, information derived from other subspecies of tui chub may be applicable. They prefer pool habitats that provide adequate cover and dense aquatic vegetation. Spawning occurs over aquatic vegetation or gravel. Females can produce large numbers of eggs; an eleven-inch long female from Lake Tahoe contained 11,200 eggs. They reach sexual maturity in 2 years and may live more than 30 years.

Foraging Ecology: Owens tui chubs prey primarily on aquatic insects, although they also consume detritus and aquatic vegetation.

Historic and Current Distribution: Owens tui chubs were reported as common from Long Valley in Mono County south to Owens Lake in Inyo County. Although tui chubs remain common in this area, the only non-introgressed populations of the Owens tui chub occur in the headsprings at the Hot Creek Fish Hatchery, the Owens River downstream from Crowley Lake, ponds at Cabin

Bar Ranch in Olancha, and at Mule Spring near Big Pine in Inyo County.

Reasons for Decline and Threats to Survival: The Owens tui chub declined due to Owens River water diversions and introduction of predatory fishes. Hybridization with other tui chub also threatens the genetic purity of the Owens tui chub. The Owens River, the primary water course through the valley floor where this species occurs, has been declared an impaired water body. The Town of Mammoth Lakes deposits sewage effluent in a percolation pond several miles uphill from the headsprings; however, an influence of this water and a hydrologic connection between the pond and the head springs has not been demonstrated.

The draft recovery plan for the Owens tui chub identifies only one specific water quality issue in its discussions of the threats or recovery of this species. Whitmore Hot Springs currently discharges treated swimming pool water into an area identified in the draft recovery plan as a potential conservation area for the Owens tui chub. Chemicals used to treat the swimming pool could be harmful to Owens tui chubs. The draft recovery plan also calls for the maintenance of water quality in the other natural and artificial springs and ponds where the Owens tui chub currently occurs or could be re-introduced.

Paiute Cutthroat Trout (*Oncorhynchus clarki seleniris*)

Species Description and Life History: The Paiute cutthroat trout is an inland subspecies of cutthroat trout endemic to the Lahontan Basin of eastern California. The species was listed as endangered on October 13, 1970 (35 **FR** 16047) and subsequently reclassified as threatened on July 16, 1975 (40 **FR** 29863). The species is believed to have evolved from Lahontan cutthroat trout during the last 5,000 to 8,000 years (Behnke and Zarn 1976).

Paiute cutthroat trout are distinguished from other subspecies of cutthroat by the absence, or near absence, of body spots, the slender body form, relatively small scales, and vivid coloration (USDI-FWS 1985b). Paiute cutthroat trout life history and spawning requirements are similar to other stream-dwelling cutthroat trout. Paiute cutthroat trout reach sexual maturity at age two and peak spawning occurs in June and July (Wong 1975). To spawn successfully, they must have access to flowing waters with clean gravel substrates (USDI-FWS 1985b). Adults and juveniles favor pools, runs, and backwater pools where current velocities are quite low. Fry are most often found in backwaters and pools (USDA-FS 1994). Paiute cutthroat trout commonly select areas of low water velocities during spring, summer and fall. Their use of habitat in the winter is unknown.

Foraging Ecology: Paiute cutthroat trout are opportunistic, foraging on a variety of invertebrates that are abundant in the water column. Insects make up the bulk of their diet (Moyle 1976a).

Historic and Current Distribution: The Paiute cutthroat has a very limited historical range in the eastern Sierra Nevada river drainage of Silver King Creek, a tributary of the East Fork Carson River drainage. Within the Silver King Creek drainage, populations of Paiute cutthroat trout occur in Fly Valley, Fourmile Canyon, Coyote Valley, and Corral Valley Creeks. Transplanted populations occur in the Sierra and Inyo National Forests, in Stairway, Sharktooth, and Cottonwood Creeks. Populations thought to be introgressed occur at a few additional sites. All current populations are in relatively small tributary creeks that do not support large populations. However, these Paiute cutthroat trout populations appear to have normal age/class distributions (Russ Wickwire and Bill Somer pers comm).

Reasons for Decline and Threats to Survival: The principal threats to the species include habitat loss due to livestock grazing and recreational use, hybridization and competition with non-native trout, and over-exploitation by angling. A Recovery Plan for the species was prepared in 1985. Critical habitat has not been designated. Recovery Plan goals include establishing pure populations and secure habitat for Paiute cutthroat trout in Silver King Creek above Llewellyn Falls, in Cottonwood Creek, and in Stairway Creek.

Razorback Sucker (*Xyrauchen texanus*)

Species Description and Life History: The razorback sucker was first proposed for listing under the ESA on April 24, 1978, as a threatened species (56 **FR** 54967). The proposed rule was withdrawn on May 27, 1980, due to changes to the listing process included in the 1978 amendments to the ESA. In March, 1989, the Service was petitioned by a consortium of environmental groups to list the razorback sucker as an endangered species. The Service made a positive finding on the petition in June, 1989, that was published in the Federal Register on August 15, 1989. The proposed rule to list the species as endangered was published on May 22, 1990, and the final rule was published on October 23, 1991. Critical habitat was designated in 1994. Critical habitat for the razorback sucker includes the Colorado, Gila, Salt, and Verde Rivers in the Lower Basin, including the 100-year floodplain of the Colorado River from Parker Dam to Imperial Dam.

The razorback sucker is the only representative of the genus *Xyrauchen*. This native sucker is distinguished from all others by the sharp edged, bony keel that rises abruptly behind the head. The body is robust with a short and deep caudal peduncle (Bestgen 1990). The razorback sucker may reach lengths of one meter and weigh five to six kg (Minckley 1973). Adult fish in Lake Mohave reached about half this maximum size and weight (Minckley 1983). Razorback suckers are long-lived, reaching the age of at least 40 years (McCarthy and Minckley 1987).

Adult razorback suckers utilize most of the available riverine habitats, although there may be an avoidance of whitewater type habitats. Main channel habitats used tend to be low velocity ones such as pools, eddies, nearshore runs, and channels associated with sand or gravel bars (summarized in Bestgen 1990). Backwaters, oxbows, and sloughs adjacent to the main channel are well-used habitat areas ; flooded bottom lands are important in the spring and early summer (summarized in Bestgen 1990). Razorback suckers may be somewhat sedentary, however considerable movement over a year has been noted in several studies (USDI-FWS 1993a). Spawning migrations have been observed or inferred in several locales (Jordan 1891; Minckley

1973; Osmundson and Kaeding 1989; Bestgen 1990; Tyus and Karp 1990).

Spawning takes place in the late winter to early summer depending upon local water temperatures. In general, temperatures between 10° to 20° C are appropriate (summarized in Bestgen 1990). Spawning areas include gravel bars or rocky runs in the main channel (Tyus and Karp 1990), and flooded bottom lands (Osmundson and Kaeding 1989).

Habitat needs of larval razorback suckers are not well known. Warm, shallow water appears to be important. Shallow shorelines, backwaters, inundated bottom lands and similar areas have been identified (Sigler and Miller 1963; Marsh and Minckley 1989; Tyus and Karp 1989, 1990; Minckley *et al.* 1991). For the first period of life, larval razorbacks are nocturnal and hide during the day. Young fish grow fairly quickly with growth slowing once adult size is reached (McCarthy and Minckley 1987). Little is known of juvenile habitat preferences.

The razorback sucker is adapted to the widely fluctuating physical environment of the historical Colorado River. Adults can live 45-50 years and, once reaching maturity between two and seven years of age (Minckley 1983), apparently produce viable gametes even when quite old. The ability of razorback suckers to spawn in a variety of habitats, flows and over a long season are also survival adaptations. Average fecundity recorded in studies ranged from 10,800 to 46,740 eggs per female (Bestgen 1990). With a varying age of maturity and the fecundity of the species, it would be possible to quickly repopulate after a catastrophic loss of adults.

Foraging Ecology: Young fish eat mostly plankton (Marsh and Langhorst 1988, Papoulias 1988). Adults are bottom dwellers, foraging on a variety of algae, detritus, and invertebrates.

Historic and Current Distribution: Occupied habitat as of 1993 is approximately 1,824 river miles, of which 336 miles are reintroduction habitats (52% of historic range). Populations are generally small and composed of aging individuals. Augmentation efforts along the Lower Colorado River propose to replace the aging populations in Lakes Havasu and Mohave and below Parker Dam with young fish from protected-rearing site programs. This may prevent the imminent extinction of the species in the wild, but appears less capable of ensuring long term survival or recovery. Overall, the status of the razorback sucker in the wild continues to decline.

Reasons for Decline and Threats to Survival: The razorback sucker was listed as an endangered species due to declining or extirpated populations throughout the range of the species. The causes of these declines are changes to biological and physical features of the habitat, largely through impounding of the lower Colorado River and introduction of non-native fish species. The effects of these changes have been most clearly noted by the almost complete lack of natural recruitment to any population in the historic range of the species.

Sacramento Splittail (*Pogonichthys macrolepidotus*)

Species Description and Life History: On January 6, 1994, a proposed rule to list the Sacramento splittail (*Pogonichthys macrolepidotus*) as a threatened species was published in 59 **FR** 862. The final rule listing the Sacramento splittail as a threatened species was published on February 8, 1999, and became effective March 10, 1999 (64 **FR** 5963).

The Sacramento splittail is a large cyprinid that can reach greater than 12 inches in length (Moyle 1976a). Adults are characterized by an elongated body, distinct nuchal hump, and a small blunt head with barbels usually present at the corners of the slightly subterminal mouth. This species can be distinguished from other minnows in the Central Valley of California by the enlarged dorsal lobe of the caudal fin. Sacramento splittail are a dull, silvery-gold on the sides and olive-grey dorsally. During the spawning season, the pectoral, pelvic and caudal fins are tinged with an orange-red color. Males develop small white nuptial tubercles on the head.

Feeding Ecology: Sacramento splittail are benthic foragers that feed on opossum shrimp, although detrital material makes up a large percentage of their stomach contents (Daniels and Moyle 1983). Earthworms, clams, insect larvae, and other invertebrates are also found in the diet. Predators include striped bass and other piscivores. Sacramento splittail are sometimes used as bait for striped bass.

Spawning behavior: Sacramento splittail are long-lived, frequently reaching five to seven years of age. Generally, females are highly fecund, producing more than 100,000 eggs each year (Daniels and Moyle 1983). Populations fluctuate annually depending on spawning success. Spawning success is highly correlated with freshwater outflow and the availability of shallow-water habitat with submersed, aquatic vegetation (Daniels and Moyle 1983). Sacramento splittail usually reach sexual maturity by the end of their second year at which time they have attained a body length of 180 to 200 mm. There is some variability in the reproductive period because older fish reproduce before younger individuals (Caywood 1974). The largest recorded individuals of the Sacramento splittail have measured between 380 and 400 mm (Caywood 1974; Daniels and Moyle 1983). Adults migrate into fresh water in late fall and early winter prior to spawning. The onset of spawning is associated with rising water temperature, lengthening photoperiod, seasonal runoff, and possibly endogenous factors from the months of March through May, although there are records of spawning from late January to early July (Wang 1986). Spawning occurs in water temperatures from 9° to 20° C over flooded vegetation in tidal freshwater and euryhaline habitats of estuarine marshes and sloughs, and slow-moving reaches of large rivers. The eggs are adhesive or become adhesive soon after contacting water (Caywood 1974; Bailey, UCD, pers. comm., 1994, as cited in DWR & USDI 1994). Larvae remain in shallow, weedy areas close to spawning sites and move into deeper water as they mature (Wang 1986).

Sacramento splittail can tolerate salinities as high as 10 to 18 ppt (Moyle 1976a; Moyle and Yoshiyama 1992). Sacramento splittail are found throughout the Delta (Turner 1966), Suisun Bay, and the Suisun and Napa marshes. They migrate upstream from brackish areas to spawn in freshwater. Because they require flooded vegetation for spawning and rearing, Sacramento splittail are frequently found in areas subject to flooding. Please refer to the Service (USDI-FWS

1994c, 1996c), and Department of Water Resources and United States Department of Interior - Bureau of Reclamation (DWR & USDI 1994) for additional information on the biology and ecology of the Sacramento splittail.

Historic and Current Distribution: Sacramento splittail are endemic to California's Central Valley where they were once widely distributed in lakes and rivers (Moyle 1976a). Historically, Sacramento splittail were found as far north as Redding on the Sacramento River and as far south as the site of Friant Dam on the San Joaquin River (Rutter 1908). Rutter (1908) also found Sacramento splittail as far upstream as the current Oroville Dam site on the Feather River and Folsom Dam site on the American River. Anglers in Sacramento reported catches of 50 or more Sacramento splittail per day prior to damming of these rivers (Caywood 1974). Sacramento splittail were common in San Pablo Bay and Carquinez Strait following high winter flows up until about 1985 (Messersmith 1966; Moyle 1976a; and Wang 1986 as cited in DWR & USDI 1994).

In recent times, dams and diversions have increasingly prevented upstream access to large rivers and the species is restricted to a small portion of its former range (Moyle and Yoshiyama 1989). Sacramento splittail enter the lower reaches of the Feather (Jones and Stokes 1993) and American rivers on occasion, but the species is now largely confined to the Delta, Suisun Bay, and Suisun Marsh (USDI-FWS 1994c). Stream surveys in the San Joaquin Valley reported observations of Sacramento splittail in the San Joaquin River below the mouth of the Merced River and upstream of the confluence of the Tuolumne River (Saiki 1984 as cited in DWR & USDI 1994).

Reasons for Decline and Threats to Survival: The decline of the Sacramento splittail has been documented over the past 10 years using fall midwater trawl data. This decline is due to hydrologic changes in the Estuary and loss of shallow water habitat due to dredging and filling (Monroe and Kelly, 1992). These changes include increases in water diversions during the spawning period of January through July. Most of the factors that caused delta smelt to decline have also caused the decline of this species. Diversions, dams and reduced outflow, coupled with severe drought years, introduced aquatic species such as the Asiatic clam (Nichols *et al.* 1986), and loss of wetlands and shallow-water habitat apparently have perpetuated the species' decline.

Sources of selenium contamination into the habitat of Sacramento splittail include: subsurface agricultural drainwater from westside San Joaquin Valley agricultural lands, non-point source runoff from Coast Range ephemeral streams flowing into the westside San Joaquin Valley (exacerbated by overgrazing of livestock), oil refinery wastewater disposal in San Francisco Bay and west Delta, and concentrated animal feeding operations (where feedlots supplement animal food with selenium) upstream of the Delta.

Santa Ana Sucker (*Catostomus santaanae*)

Species Description and Life History: The Santa Ana sucker was originally described by Snyder

(1908) from specimens collected in the Santa Ana River, hence its name. The Santa Ana sucker, a small, short-lived sucker, was proposed for threatened status by the Service on January 26, 1999 (64 FR 3915). Moyle (1976) described the Santa Ana sucker as less than 16 centimeters (cm) (6.3 inches (in)) in length. The Santa Ana sucker is silvery below, darker along the back with irregular blotches, and the membranes connecting the rays of the tail are pigmented (Moyle 1976).

The Santa Ana sucker inhabits streams that are generally small and shallow, with currents ranging from swift (in canyons) to sluggish (in the bottomlands). All the streams are subject to periodic severe flooding (Moyle 1976). Santa Ana suckers appear to be most abundant where the water is cool (less than 22° Celsius) (72° Fahrenheit), unpolluted and clear, although they can tolerate and survive in seasonally turbid water. Santa Ana suckers feed mostly on detritus, algae, and diatoms which they scrape off of rocks and other hard substrates, with aquatic insects making up a very small component of their diet. Larger fish generally feed more on insects than do smaller fish (Greenfield et al. 1970).

Santa Ana suckers usually live no more than 3 years (Greenfield et al. 1970). Spawning generally occurs from early April to early July, with a peak in late May and June (Greenfield et al. 1970, Moyle 1976). Spawning period may be variable and protracted, however. Recent field surveys on the East Fork of the San Gabriel River, found evidence of an extended spawning period. These surveys found small juveniles (<30 mm standard length (1.2 in)) in December 1998, and March of 1999 (U. S. Geological Survey (USGS) data in litt. 1999). This data indicates that spawning may be very protracted in this stream, and begin as early as November. Fecundity appears to be exceptionally high for a small sucker species (Moyle 1976). The combination of early sexual maturity, protracted spawning period, and high fecundity should allow the Santa Ana sucker to quickly repopulate streams following periodic flood events that can decimate populations (Moyle 1976).

Historic and Current Distribution: The Santa Ana sucker is one of seven native freshwater fishes that occurred historically in the Los Angeles Basin of California. Of these seven species, the Santa Ana sucker is the most common in the basin today. Four of the native Los Angeles Basin fishes are extinct within the basin, and two are very rare. Historically, the Santa Ana sucker occurred form near the Pacific Ocean to the headwaters of Los Angeles Basin streams. Urbanization and the associated anthropogenic impacts to habitats in the Los Angeles megalopolis have reduced the Santa Ana sucker's range to small reaches of Big Tujunga Creek (a tributary of the Los Angeles River), the headwaters of the San Gabriel River, and a lowland reach of the Santa Ana River, in Los Angeles, San Bernardino, Riverside and Orange counties (Swift et al. 1993).

A population also occurs throughout portions of the Santa Clara River drainage system, in Ventura and Los Angeles counties. The Santa Clara population is presumed to be an introduced population, although this presumption is based entirely on negative data (its absence from early collections), and not on a documented record of introduction (Bell 1978, Hubbs et al. 1943, Miller 1968, Moyle 1976). The Santa Clara River population was not included in the proposal to list the Santa Ana sucker as threatened because of its presumed introduced status (64 FR 3915).

Reasons for Decline and Threats to Survival: Moyle and Yoshiyama (1992) concluded that the native range of the Santa Ana sucker is largely coincident with the Los Angeles metropolitan area. Intensive urban development of the area has resulted in water diversions, extreme alteration of stream channels, changes in the watershed that result in erosion and debris torrents, pollution, and the establishment of introduced non-native fishes. Moyle and Yoshiyama (1992) stated, "[e]ven though Santa Ana suckers seem to be quite generalized in their habitat requirements, they are intolerant of polluted or highly modified streams." The impacts associated with urbanization are likely the primary cause of the extirpation of this species from lowland reaches of the Los Angeles, San Gabriel, and Santa Ana rivers.

As the Los Angeles urban area expanded, the rivers of the Los Angeles Basin, the Los Angeles, Santa Ana, and San Gabriel rivers, were highly modified, channelized, or moved in an effort to either capture water runoff or protect property. As Moyle (1976) stated, "[t]he lower Los Angeles River is now little more than a concrete storm drain." The same is true for the Santa Ana and San Gabriel rivers. These channelized rivers and canals with uniform and altered substrates are not suitable for sustaining Santa Ana sucker populations (Chadwick and Associates 1996). Past and continuing projects have resulted (or will result) in channelization and concrete lining of the Santa Ana River channel throughout most of the range of the Santa Ana sucker in Orange County. Urban development threatens the Santa Ana sucker in the Los Angeles and Santa Ana river basins. This urban development has resulted in changes in water quality and quantity, and the hydrologic regime of these rivers. The Santa Ana sucker is one of seven native freshwater fish species of the Los Angeles Basin. Four of these species, the steelhead (Oncorhynchus mykiss), Pacific lamprey (Lampetra tridentata), Pacific brook lamprey (Lampetra cf. pacifica), and the unarmored threespine stickleback (Gasterosteus aculeatus williamsoni) have been extinct within the Los Angeles Basin since the 1950's, and two others are very rare (Santa Ana speckled dace (Rhinichthys osculus ssp.) and arroyo chub (Gila orcutti)) presumably due to the same factors that have caused the decline of the Santa Ana sucker (Swift et al. 1993).

All three river systems within the historic range of the Santa Ana sucker have dams that isolate and fragment fish populations. Dams likely have resulted in some populations being excluded from suitable spawning and rearing tributaries. Reservoirs also provide areas where introduced predators and competitors can live and reproduce (Moyle and Light 1996). The newly completed Seven Oaks Dam, upstream from the present range of Santa Ana sucker in the Santa Ana River, will prevent future upstream movement of fish and further isolate the Santa Ana sucker populations from their native range in the headwaters of that system.

A recent study of environmental variables affecting Santa Ana sucker abundace found some evidence that deteriorating water quality (electrical conductivity and turbidity) negatively impacts Santa Ana suckers. Results from this study also indicated that the presence of non-native introduced fish species was more strongly correlated with the absence of Santa Ana suckers than any water quality variable. Strongly significant negative associations were found with common carp (Cyprinus carpio), largemouth bass (Micropterus salmoides), bluegill (Lepomis macrochirus), and fathead minnow (Pimephales promelas), indicating nonnative fishes may exclude Santa Ana suckers by competition, or eliminate via predation (Mike Saiki, U.S. Geological Survey, pers. com. 1999). Non-native introduced fishes have long been recognized as having far reaching negative impacts to native fishes in North America (Moyle et al. 1986). Accordingly, introduced predators and competitors likely threaten the continued existence of Santa Ana suckers throughout most of the range of the species.

Shortnose Sucker (*Chasmistes brevirostris*)

Species Description and Life History: The shortnose sucker was described by Cope (1879) from specimens he collected from Upper Klamath Lake. A complete discussion of the taxonomy of the species can be found in the Service's Lost River and Shortnose Sucker Recovery Plan (USDI-FWS 1993c). The shortnose sucker was federally listed as endangered species on July 18, 1988 (53 **FR** 27134). The Clear Lake watershed is considered Unit 1 of the proposed designation of six Critical Habitat Units (CHUs) for Lost River and shortnose suckers. Primary constituent elements include water of sufficient quantity and quality to provide conditions required for the particular life stage of the species; physical habitat inhabited or potentially habitable by shortnose suckers for use as refugia, spawning, nursery, feeding, or rearing areas, or as corridors between these areas; and food supply and a natural scheme of predation, parasitism, and competition in the biological environment.

Scoppettone (1988) found shortnose suckers up to 33 years of age in Copco Reservoir and Lost River suckers to 43 years of age in upper Klamath Lake. In the Clear Lake drainage, Scoppettone (1988) found shortnose suckers from one to 23 years old. Shortnose suckers are generally not larger than 50 centimeters (cm). Sexual maturity for shortnose suckers in Clear Lake appears to be five years (CDFG 1993). Buettner and Scoppettone (1990) found that most growth occurred in the first six to eight years of life for female shortnose suckers sampled from Upper Klamath Lake.

The majority of shortnose suckers spawning in the tributaries of Upper Klamath Lake have been observed in water depths ranging from 21 to 60 cm and in water velocities of 41 to 110 centimeters per second. Fecundity for shortnose suckers is reportedly between 18,000 to 46,000 eggs for suckers measuring about 360 millimeters (mm) to 445 mm in fork length (Buettner and Scoppettone 1990). Shortnose suckers have also been observed spawning in lacustrine habitats at Ouxy Springs and springs adjacent to Sucker Springs (L. Dunsmoor, pers. comm., cited in USDI-FWS 1994b), although little is known about the suitability of this habitat for incubation.

Foraging Ecology: The diet of shortnose suckers includes detritus, zooplankton, algae, and aquatic insects (Buettner and Scoppetone 1990).

Historic and Current Distribution: The shortnose sucker is endemic to the upper Klamath Basin, Oregon and California, and were once quite abundant. Cope (1884) noted that Upper Klamath Lake sustained "a great population of fishes" and was "more prolific in animal life" than any body of water known to him at that time.

The historical distribution of the shortnose sucker was Upper Klamath Lake and its tributaries (Miller and Smith 1981; Williams *et al.* 1985), Lake of the Woods (Moyle 1976a), and possibly the Lost River drainage. This species is now found throughout the Upper Klamath Basin, including the Lost River, Clear Lake Reservoir, Gerber Reservoir, and Tule Lake. Shortnose suckers have also been collected on the Upper Klamath River from Copco Reservoir to the Link River Dam. Those found in Gerber Reservoir and Clear Lake show some morphological differences from those in Upper Klamath Lake (Buettner and Scoppettone 1991). The taxonomic status of various shortnose sucker populations is yet to be resolved. Genetic evaluations are in progress by Dr. Don Buth at the University of California, Los Angeles (UCLA). Andreason (1975) included Clear Lake as the upstream limit of the sucker in the Lost River system.

The largest population of shortnose suckers occurs in Upper Klamath Lake and Clear Lake (Scoppettone, pers. comm., cited in USDI-FWS 1994b). Under higher flow conditions, such as the spring of 1993, the range probably extended upstream in all of the creeks in the Clear Lake watershed (M. Buettner, pers. comm., cited in USDI-FWS 1994b). Shortnose suckers have been captured in the Lost River below Clear Lake and were taken to Malone Reservoir in 1992 during Reclamation's salvage operation at Clear Lake. Buettner (pers. comm. 1995) believes it is unlikely that many suckers remain in Malone Reservoir. The reservoir is drained each fall to a small pool and most of the fish were likely washed down stream into the Lost River.

Reasons for Decline and Threats to Survival: The factors believed to be responsible for the decline of the shortnose sucker include the damming of rivers, dredging and draining of marshes, instream flow diversions, a shift toward hyper eutrophication in Upper Klamath Lake, and other traditional land use practices. A recent analysis of the population genetics of the shortnose and Lost River suckers (Moyle and Berg 1991) suggested that "if populations continue to decline, these species may cross below the minimum viable population threshold and be lost". Entire stocks may have already been lost [e.g., Harriman Springs (Andreasen 1975)].

Suckers appear to be strongly influenced by poor water quality induced by high water temperatures, nutrient enrichment, algal blooms and die-offs, low dissolved oxygen, high pH, and possibly high ammonia (Kann and Smith 1993; Perkins 1997). Higher recruitment success occurs during above-average water quality years; in contrast, large-scale fish kills of adult suckers in the Upper Klamath Lake and Williamson Rivers appear related to poor water quality (Perkins 1997). As indicated above, fish kills appear to have increased in size, duration, and areal extent in recent years and may be adversely affecting current recovery efforts (Perkins 1997).

In addition, to fish kills, suckers in the Klamath Basin suffer from abnormally high rates of parasitism and physical deformities (Biological Research Division, U.S. Geological Survey, unpublished) that may be related to water quality, nutritional deficiencies, or contaminant exposures. Fish in the Tule Lake area also suffer very high rates of parasitism and deformities (Littleton 1993), although sucker health has not specifically been documented. Overharvest and chemical contamination may have also contributed to the decline. Reduction and degradation of lake and stream habitats in the upper Klamath Basin is considered to be the most important factor in the decline of the endangered suckers (USDI-FWS 1993a). Very low numbers of benthic organisms in many locations and an overall reduction in numbers of aquatic reptiles in the habitat of the sucker may have been caused by pollution of organochlorine pesticides and other pollutants (USDI-FWS 1993a).

Steelhead Trout(Including all California ESUs) (*Oncorhynchus mykiss*)

Species Description and Life History: General life history information for steelhead is summarized below, followed by more detailed information on each steelhead ESU, including any unique life history traits as well as their population trends. Further detailed information on these steelhead ESUs is available in the NMFS Status Review of west coast steelhead from Washington, Idaho Oregon, and California (Busby *et al.* 1996); the NMFS proposed rule for listing steelhead (61 FR 41541); the NMFS Status Review for Klamath Mountains Province Steelhead (Busby *et al.* 1994), and the NMFS final rule listing the Southern California steelhead ESU as endangered and the South-Central California Coast and the Central California Coast steelhead ESUs as threatened (62 FR 43937). On March 19, 1998, the Central Valley ESU of steelhead was listed as threatened, and the Klamath Mountains Province and Northern California ESUs were deferred for listing (63 FR 13347). The listing decision for the Northern California steelhead ESU was revisited, and on February 11, 2000, this ESU was proposed for listing as threatened (65 FR 6960).

Critical Habitat: Critical habitat was designated on February 16, 2000 (65 FR 7764) for Central Valley, Central California Coast, South-Central California Coast, and Southern California steelhead ESUs. Critical habitat has not been proposed for the Northern California and Klamath Mountain Province steelhead ESUs. Critical habitat has been designated to include all river reaches accessible to listed steelhead within the range of the ESUs listed, except for reaches on Indian lands within Indian Reservations. Critical habitat consists of the water, substrate, and adjacent riparian zone of estuarine and riverine reaches for all of the steelhead ESUs. Accessible reaches are those within the historical range of the ESUs that can still be occupied by any life stage of steelhead. Inaccessible reaches are those above longstanding, naturally impassable barriers (i.e., natural waterfalls in existence for at least several hundred years) and specific dams within the historical range of each ESU identified in Tables 16 through 19 of the final critical habitat designation.

1. Central California Coast steelhead geographic boundaries. Critical habitat is designated to include all river reaches and estuarine areas accessible to listed steelhead in coastal river basins from the Russian River to Aptos Creek, California (inclusive), and the drainages of San

Francisco and San Pablo Bays. Also included are all waters of San Pablo Bay westward of the Carquinez Bridge and all waters of San Francisco Bay from San Pablo Bay to the Golden Gate Bridge. Excluded is the Sacramento-San Joaquin River Basin of the California Central Valley as well as areas above specific dams or above longstanding, naturally impassable barriers (i.e., natural waterfalls in existence for at least several hundred years).

2. South-Central California Coast steelhead geographic boundaries. Critical habitat is designated to include all river reaches and estuarine areas accessible to listed steelhead in coastal river basins from the Pajaro River (inclusive) to, but not including, the Santa Maria River, California. Excluded are areas above specific dams or above longstanding, naturally impassable barriers (i.e., natural waterfalls in existence for at least several hundred years).

3. Southern California steelhead geographic boundaries. Critical habitat is designated to include all river reaches and estuarine areas accessible to listed steelhead in coastal river basins from the Santa Maria River to Malibu Creek, California (inclusive). Excluded are areas above specific dams or above longstanding, naturally impassable barriers (i.e., natural waterfalls in existence for at least several hundred years).

4. Central Valley steelhead geographic boundaries. Critical habitat is designated to include all river reaches accessible to listed steelhead in the Sacramento and San Joaquin Rivers and their tributaries in California. Also included are river reaches and estuarine areas of the Sacramento-San Joaquin Delta, all waters from Chipps Island westward to Carquinez Bridge, including Honker Bay, Grizzly Bay, Suisun Bay, and Carquinez Strait, all waters of San Pablo Bay westward of the Carquinez Bridge, and all waters of San Francisco Bay (north of the San Francisco/Oakland Bay Bridge) from San Pablo Bay to the Golden Gate Bridge. Excluded are areas of the San Joaquin River upstream of the Merced River confluence and areas above specific dams or above longstanding, naturally impassable barriers (i.e., natural waterfalls in existence for at least several hundred years).

Proposed ESUs: The geographic boundaries of the Northern California ESU, proposed as threatened, include the coastal river basins from Redwood Creek, Humboldt County, to the Gualala River, in Mendocino County, California, inclusive.

Migration and Spawning: The most widespread run type of steelhead is the winter (ocean-maturing) steelhead, while summer (stream-maturing) steelhead (including spring and fall steelhead in southern Oregon and northern California) are less common. The stream-maturing type enters fresh water in a sexually immature condition and requires several months in freshwater to mature and spawn. The ocean-maturing type enters fresh water with well-developed gonads and spawns shortly thereafter (Barnhart 1986). There is a high degree of overlap in spawn timing between populations, regardless of run-type. California steelhead generally spawn earlier than steelhead in northern areas. Both summer and winter steelhead in California generally begin spawning in December, whereas most populations in Washington begin spawning in February or March. Among inland steelhead populations, Columbia River populations from tributaries upstream of the Yakima River spawn later than most downstream populations.

Steelhead spawn in cool, clear streams featuring suitable gravel size, water depth, and current velocity. The timing of upstream migration is correlated with higher flow events, such as freshets or sand bar breaches, and associated lower water temperatures. Unusual stream temperatures during spawning migration periods can alter or delay migration timing, accelerate or retard mutations, and increase fish susceptibility to diseases. The minimum stream depth necessary for successful upstream migration is 18 cm (Thompson 1972). Reiser and Bjornn (1979) indicated that steelhead preferred a depth of 24 cm or more. The preferred water velocity for upstream migration is in the range of 40-90 cm/second, with a maximum velocity, beyond which upstream migration is not likely to occur, of 2.4 m/second (Thompson 1972, Smith 1973). Intermittent streams may be used for spawning (Barnhart 1986; Everest 1973). Steelhead may spawn more than once before dying, in contrast to other species of the *Oncorhynchus* genus. It is relatively uncommon for steelhead populations north of Oregon to have repeat spawning, and more than two spawning migrations is rare. In Oregon and California, the frequency of two spawning migrations is higher, but more than two is unusual. The number of days required for steelhead eggs to hatch varies from about 19 days at an average temperature of 60 degrees F to about 80 days at an average of 42 degrees F. Fry typically emerge from the gravel two to three weeks after hatching (Barnhart 1986).

After emergence, steelhead fry usually inhabit shallow water along perennial stream banks. Older fry establish territories which they defend. Stream side vegetation and cover are essential. Steelhead juveniles are usually associated with the bottom of the stream. In winter, they become inactive and hide in any available cover, including gravel or woody debris. Juvenile steelhead live in freshwater between one and four years and then become smolts and migrate to the sea from November through May with peaks in March, April, and May. The smolts can range from 14 to 21 cm in length. Steelhead spend between one and four years in the ocean (usually two years in the Pacific Southwest) (Barnhart 1986). Water temperatures influence the growth rate, population density, swimming ability, ability to capture and metabolize food, and ability to withstand disease of these rearing juveniles.

Reiser and Bjornn (1979) recommended that dissolved oxygen concentrations remain at or near saturation levels with temporary reductions to not less than 5.0 mg/L for successful rearing of juvenile steelhead. Low dissolved oxygen levels decrease the rate of metabolism, swimming speed, growth rate, food consumption rate, efficiency of food utilization, behavior, and ultimately the survival of the juveniles.

North American steelhead typically spend two years in the ocean before entering freshwater to spawn. The distribution of steelhead in the ocean is not well known. Coded wire tag recoveries indicate that most steelhead tend to migrate north and south along the Continental Shelf (Barnhart 1986). Steelhead stocks from the Klamath and Rogue rivers probably mix together in a nearshore ocean staging area along the northern California before they migrate upriver (Everest

1973).

All Central Valley steelhead are currently considered winter steelhead, although three distinct runs, including summer steelhead, may have occurred as recently as 1947 (CDFG 1995; McEwan and Jackson 1996). Steelhead within this ESU have the longest freshwater migration of any population of winter steelhead. There is essentially a single continuous run of steelhead in the upper Sacramento river. River entry ranges from July through May, with peaks in September and February; spawning begins in late December and can extend into April (McEwan and Jackson 1996).

There are two recognized forms of native *O. mykiss* within the Sacramento River Basin: coastal steelhead/rainbow trout (*O. m. irideus*, Behnke 1992) and Sacramento redband trout (*O. m. stonei*, Behnke 1992). It is not clear how the coastal and Sacramento forms of *O. mykiss* interacted in the Sacramento River prior to construction of Shasta Dam in the 1940s which blocked anadromous fish passage. Behnke (1992) reported that coastal and resident redband trout were spawned together at the McCloud River egg-taking station (1879-1888). Therefore, it appears the two forms co-occurred historically at spawning time, but may have maintained reproductive isolation. In addition, the relationship between anadromous and non-anadromous forms of coastal *O. mykiss,* including possible residualized fish upstream from dams, is unclear.

Migration and life history patterns of southern California steelhead depend more strongly on rainfall and streamflow than is the case for steelhead populations farther north (Moore 1980; Titus *et al.* in press). Average rainfall is substantially lower and more variable in southern California than in regions to the north, resulting in increased duration of sand berms across the mouths of streams and rivers and, in some cases, complete dewatering of the lower reaches of these streams from late spring through fall. Environmental conditions in marginal habitats may be extreme (e.g., elevated water temperatures, droughts, floods, and fires) and presumably impose selective pressures on steelhead populations. Their utilization of southern California streams and rivers with elevated temperatures (in some cases much higher than the preferred range for steelhead) suggests that steelhead within this ESU are able to withstand higher temperatures than populations to the north. The relatively warm and productive waters of the Ventura River have resulted in more rapid growth of juvenile steelhead than occurs in more northerly populations (Moore 1980; Titus *et al.* in press; McEwan and Jackson 1996). However, we have relatively little life history information for steelhead from this ESU.

Large rivers, such as the Klamath and Rogue rivers, may have adult steelhead migrating throughout the year (Shapovalov and Taft 1954; Rivers 1957; Barnhart 1986). For example, summer steelhead in the Rogue River were historically divided into spring and fall steelhead (Rivers 1963). More recently, some researchers contend spring and fall steelhead of the Rogue, Klamath, Mad and Eel rivers are summer steelhead (Everest 1973; Roelofs 1983), while others classify fall steelhead separately (Heubach 1992) or as winter steelhead.

Foraging Ecology: Juvenile steelhead feed on a wide variety of aquatic and terrestrial insects, and emerging fry are sometimes preyed upon by older juveniles.

<u>Historic and Current Distribution</u>: *Central Valley ESU (Threatened)* (63 FR 13347): Historical abundance estimates are available for some stocks within this ESU, but no overall estimates are available prior to 1961. In the Sacramento River including San Francisco Bay, the total run-size of steelhead was estimated at 40,000 in 1961 (Hallock *et al.* 1961). In the mid-1960s, steelhead spawning populations in this ESU were estimated at 27,000 fish (CDFG 1965). The present total run size for this ESU is probably less than 10,000 fish based on dam counts, hatchery returns and past spawning surveys.

At the Red Bluff Diversion Dam, counts have averaged 1,400 fish over the last 5 years, compared with runs in excess of 10,000 in the late 1960s. In the American River, estimates of hatchery produced fish average less than 1,000 fish, compared to 12,000 to 19,000 in the early 1970s (McEwan and Jackson 1996). Data to estimate population trends at the Red Bluff Diversion Dam show a significant decline of 9 percent per year from 1966 to 1992.

The majority of native, natural steelhead production in this ESU occurs in the upper Sacramento tributaries (Antelope, Deer, Mill, and other creeks), but these populations are nearly extirpated. The American, Feather, and Yuba rivers (and possibly the upper Sacramento and Mokelumne rivers) also have naturally-spawning populations (CDFG 1995). However, these rivers have also had substantial hatchery influence, and their ancestry is unknown. In the San Joaquin River Basin, there are reports of: (1) a small remnant steelhead run in the Stanislaus River (McEwan and Jackson 1996); (2) observations of steelhead in the Tuolumne River; and (3) large rainbow trout (possibly steelhead) at the Merced River hatchery.

Southern California ESU (Endangered) (62 FR 43937): The Southern California ESU of steelhead trout occupies rivers from the Santa Maria River to the southern extent of the species range. Historically, *O. mykiss* occurred at least as far south as Rio del Presidio in Mexico (Behnke 1992, Burgner *et al.* 1992). Spawning populations of steelhead did not occur that far south but may have extended to the Santo Domingo River in Mexico (Barnhart 1986); however, some reports state that steelhead may not have existed south of the U.S.-Mexico border (Behnke 1992; Burgner *et al.* 1992). The present southernmost stream used by steelhead for spawning is generally thought to be Malibu Creek, California (Behnke 1992; Burgner *et al.* 1992); however, in years of substantial rainfall, spawning steelhead can be found as far south as the Santa Margarita River, San Diego County (Barnhart 1986; Higgins 1991).

Previous assessments within this ESU have identified several stocks as being at risk or of special concern. Nehlsen *et al.* (1991) identified 11 stocks as extinct and 4 as at high risk. Titus *et al.* (in press) provided a more detailed analysis of these stocks and identified stocks within 14 drainages in this ESU as extinct, at risk, or of concern. They identified only two stocks, those in Arroyo Sequit and Topanga Creek, as showing no significant change in production from historical levels.

Historically, steelhead may have occurred naturally as far south as Baja California. Estimates of historical (pre-1960s) abundance are available for several rivers in this ESU: Santa Ynez River, before 1950, 20,000-30,000; Ventura River, pre-1960, 4,000-6,000; Santa Clara River, pre-1960, 7,000-9,000; Malibu Creek, pre-1960, 1,000. In the mid-1960s, CDFG (1965) estimated steelhead spawning populations for smaller tributaries in San Luis Obispo County as 20,000, but they provided no estimates for streams farther south.

The present total run sizes for 6 streams in this ESU were summarized by Titus *et al.* (in press); all were less than 200 adults. Titus *et al.* (in press) concluded that populations have been extirpated from all streams south of Ventura County, with the exception of Malibu Creek in Los Angeles County. However, steelhead are still occasionally reported in streams where stocks were identified by these authors as extirpated.

Of the populations south of San Francisco Bay (including part of the Central California Coast ESU) for which past and recent information was available, they concluded that 20% had no discernible change, 45% had declined, and 35% were extinct.

Central California Coast ESU (Threatened) (62 FR 43937): Only two estimates of historical (pre-1960s) abundance specific to this ESU are available: an average of about 500 adults in Waddell Creek in the 1930s and early 1940s (Shapovalov and Taft 1954), and 20,000 steelhead in the San Lorenzo River before 1965 (Johnson 1964). In the mid-1960s, 94,000 steelhead adults were estimated to spawn in the rivers of this ESU, including 50,000 and 19,000 fish in the Russian and San Lorenzo rivers, respectively (CDFG 1965). Recent estimates indicate an abundance of about 7,000 fish in the Russian River (including hatchery steelhead) and about 500 fish in the San Lorenzo River. These estimates suggest that recent total abundance of steelhead in these two rivers is less than 15 percent of their abundance 30 years ago. Recent estimates for several other streams (Lagunitas Creek, Waddell Creek, Scott Creek, San Vincente Creek, Soquel Creek, and Aptos Creek) indicate individual run sizes of 500 fish or less. Steelhead in most tributaries to San Francisco and San Pablo bays have been extirpated (McEwan and Jackson 1996). Fair to good runs of steelhead still apparently occur in coastal Marin County tributaries.

Little information is available regarding the contribution of hatchery fish to natural spawning, and little information on present run sizes or trends for this ESU exists. However, given the substantial rates of declines for stocks where data do exist, the majority of natural production in this ESU is likely not self-sustaining.

South-Central California Coast ESU (Threatened) (62 FR 43937): In the mid-1960s, total spawning populations of steelhead in the rivers in this ESU were estimated as 27,750 (CDFG 1965). Recent estimates for those rivers show a substantial decline during the past 30 years. Other estimates of steelhead include 1,000 to 2,000 in the Pajaro River in the early 1960s (McEwan and Jackson 1996), and about 3,200 steelhead for the Carmel River for the 1964-1975 period (Snider 1983). No recent estimates for total run size exist for this ESU. However, recent run-size estimates are available for five streams (Pajaro River, Salinas River, Carmel River, Little Sur River, and Big Sur River). The total of these estimates is less than 500 fish, compared with a total of 4,750 fish for the same streams in 1965.

Adequate adult escapement information was available to compute a trend for only one stock within this ESU (Carmel River above San Clemente Dam). This data series shows a significant decline of 22 percent per year from 1963 to 1993, with a recent 5-year average count of only 16 adult steelhead at the dam. In 1996, however, 700 adults were reported to have passed the ladder at San Clemente Dam.

Little information exists regarding the actual contribution of hatchery fish to natural spawning, and little information on present total run sizes or trends are available for this ESU. However, given the substantial reductions from historical abundance or recent negative trends in the stocks for which data exist, it is likely that the majority of natural production in this ESU is not self-sustaining.

Northern California ESU (Proposed Threatened) (65 FR7764): Population abundance has been determined to be very low relative to historical estimates (1930's dam counts), and recent trends are downward in stocks for which data were available, with the exception of two summer steelhead stocks. Summer steelhead abundance in particular is very low in this ESU. The most complete data set available in this ESU is a time series of winter steelhead counts on the Eel River at Cape Horn Dam. The updated abundance data (through 1997) showed moderately declining long-term and short-term trends in abundance, and the vast majority of these fish were believed to be of hatchery origin. These data show a strong decline in abundance prior to 1970, but no significant trend thereafter. Additional winter steelhead data are available for Sweasy Dam on the Mad River which show a significant decline, but that data set ends in 1963. For the seven populations where recent trend data were available, the only runs showing recent increases in abundance in the ESU were the relatively small populations of summer steelhead in the Mad River, which has had high hatchery production, and winter steelhead in Prairie Creek where the increase may be due to increased monitoring or mitigation efforts.

<u>Reasons for Decline and Threats to Survival</u>: (*All ESUs*) Steelhead on the West Coast have experienced declines in abundance in the past several decades as a result of natural and human factors. Forestry, agriculture, mining, and urbanization have degraded, simplified, and fragmented habitat. Water diversions for agriculture, flood control, domestic, and hydropower purposes have greatly reduced or eliminated historically accessible habitat. Among other factors, NMFS specifically identified timber harvest, agriculture, mining, habitat blockages, and water diversions as important factors for the decline of steelhead.

The status reviews and listing notices have cited extensive loss of steelhead habitat due to water development, including impassable dams and dewatering of portions of rivers, as principal threats to the steelhead. They also reported that of 32 tributaries for the southern California ESU, 21 have blockages due to dams, and 29 have impaired mainstem passage. Habitat problems in these ESUs relate primarily to water development resulting in inadequate flows, flow fluctuations, blockages, and entrainment into diversions (McEwan and Jackson 1996, Titus *et al.* in press).

Other problems related to land use practices and urbanization also certainly contribute to depressed stock conditions. Habitat fragmentation and population declines have also resulted in small, isolated populations that may face genetic risk from inbreeding, loss of rare alleles, and genetic drift.

During rearing, suspended and deposited fine sediments can directly affect salmonids by abrading and clogging gills, and indirectly cause reduced feeding, avoidance reactions, destruction of food supplies, reduced egg and alevin survival, and changed rearing habitat (Reiser and Bjornn 1979). See also Reasons for Decline and Threats to Survival for chinook and coho salmon sections of this biological opinion for further information on factors affecting steelhead trout.

Tidewater Goby (*Eucyclogobius newberryi*)

Species Description and Life History: The tidewater goby was listed by the Service as endangered on March 7, 1994 (59 **FR** 10584). A recovery plan has not been published, and critical habitat has not been proposed. On June 24, 1999, the Service published a proposed rule to remove northern populations of the tidewater goby from the federal list of threatened and endangered species (64 **FR** 33816). This proposed rule identifies a distinct population segment (DPS) of tidewater goby known from six locations in Orange and San Diego counties, and would remove protection for all populations of tidewater goby north of these locations. On August 3, 1999, the Service published a proposed rule to designate critical habitat for this DPS (64 **FR** 42250). Detailed information regarding the biology of the tidewater goby can be found in Wang (1982), Irwin and Soltz (1984), Swift *et al.* (1989), Worcester (1992), and Swenson (1995).

The tidewater goby rarely exceeds 50 millimeters standard length. The species, which is endemic to California, is found primarily in waters of coastal lagoons, estuaries, and marshes. Its habitat is characterized by brackish shallow lagoons and lower stream reaches where the water is fairly still but not stagnant (Miller and Lea 1972; Moyle 1976a; Swift 1980; Wang 1982; Irwin and Soltz 1984). Tidewater gobies have been documented in waters with salinity levels from 0 to 42 parts per thousand, temperature levels from 8 to 25° Celsius, and water depths from 25 to 200 centimeters (Irwin and Soltz 1984; Swift *et al.* 1989; Worcester 1992; Swenson 1994; Lafferty 1997; Smith 1998). The species can withstand very low dissolved oxygen levels, and is regularly collected in waters with levels below 1 mg/l (Worcester 1992; Swift *et al.* 1997).

The tidewater goby appears to spend all life stages in lagoons. It may enter the marine environment only when flushed out of the lagoon by normal breaching of the sandbars following storm events. These events are important in the normal metapopulation dynamics and distribution of the species (Swift *et al.* 1989; Lafferty *et al.* 1997; Swift *et al.* 1997; Lafferty *et al.* in review). The tidewater goby seems to be an annual species although some variation has been observed (Swift 1980; Wang 1982; Irwin and Soltz 1984). Reproduction can occur year-round although distinct peaks in spawning, often in late spring and late summer or early fall, do occur. Both males and females can breed more than once in a season, with a lifetime reproductive potential of 3 - 12 spawning events. Females deposit an average of 400 eggs (range 100 - 1000) per spawning effort (Swenson 1995, in press). When breeding, males dig vertical burrows for females to deposit eggs. Within nine to ten days larvae emerge and are approximately five to seven mm in length. The larvae live in vegetated areas within the lagoon until they are 15 to 18 mm long (Wang 1982; Swift *et al.* 1989; Swenson 1994).

Historic and Current Distribution: The tidewater goby historically occurred in at least 110 California coastal lagoons (USDI-FWS in prep.) from the Smith River, Del Norte County, to Agua Hedionda Lagoon in San Diego County. The southern extent of its distribution has been reduced by approximately 13 kilometers (8 miles), and the species is currently known to occur in about 85 locations. Exact numbers of sites fluctuate with normal climatic conditions.

Reasons for Decline and Threats to Survival: The decline of the tidewater goby can be attributed primarily to urban, agricultural and industrial development in and surrounding the coastal wetlands and alteration of habitats from seasonally closed lagoons to tidal bays and harbors. The extent and magnitude of these threats has diminished since the promulgation of protective environmental legislation. Some extirpations are believed to be related to pollution, upstream water diversions, and the introduction of exotic fish species. These threats continue to affect remaining populations of tidewater gobies. Tidewater gobies have been extirpated from several impaired water bodies (e.g., Mugu Lagoon, Ventura County), but still occur in others (e.g., Santa Clara River, Ventura County). Lagoons where the goby resides receive municipal and industrial contaminated run-off from coastal streams. The short life-cycle of the species leaves it vulnerable to stochastic events. A single pulse of a contaminant may inhibit growth, survival, and reproduction of an entire cohort.

Unarmored Threespine Stickleback (*Gasterosteus aculeatus williamsoni*)

Species Description and Life History: The unarmored threespine stickleback was listed as endangered in 1970 (35 **FR** 16047). The following information is summarized from the recovery plan for the unarmored threespine stickleback (USDI-FWS 1985d). Two reaches of the Santa Clara River, and a single reach of both San Francisquito Creek and San Antonio Creeks were proposed as critical habitat in 1980 (45 **FR** 76012). However, critical habitat has not been designated.

Unarmored threespine sticklebacks are small fish (up to 6 centimeters) inhabiting slow moving reaches or quiet water microhabitats of streams and rivers. Favorable habitats usually are shaded by dense and abundant vegetation but in more open reaches algal mats or barriers may provide refuge for the species. Unarmored threespine sticklebacks reproduce throughout the year with a minimum of breeding activity occurring from October to January. Unarmored threespine sticklebacks are believed to live for only one year (USDI-FWS 1985d).

Foraging Ecology: Unarmored threespine sticklebacks feed on insects, small crustaceans, and snails, and to a lesser degree, on flat worms and nematodes.

Historic and Current Distribution: Unarmored threespine sticklebacks historically were distributed throughout southern California but are now restricted to the upper Santa Clara River and its tributaries in Los Angeles and Ventura counties, San Antonio and Canada Honda creeks on Vandenberg Air Force Base, Shay Creek in San Bernardino County, and San Felipe Creek in San Diego County. The population in Canada Honda Creek on Vandenberg Air Force Base is a transplanted population, as is the population that may persist in San Felipe Creek.

Reasons for Decline and Threats to Survival: Competition with non-native fish, introgression with other subspecies of sticklebacks, and loss of habitat to urbanization were contributing factors that led to the decline of the unarmored threespine stickleback. The greatest risk of continued urbanization of the Santa Clara River watershed is the degradation of water quality (USDI-FWS 1977). In the Santa Clara River, populations of unarmored threespine sticklebacks are affected by effluent from the Saugus and Valencia water reclamation plants, operated by the County Sanitation Districts of Los Angeles County. Pending modifications to the Valencia Water Reclamation Plant would improve the quality of effluent waters by removing ammonia. Effluent from this plant currently contains concentrations of ammonia that approach the toxic level for some aquatic species. Recovery plan objectives for this species include the regulation, maintenance, and restoration of water quality and quantity to ensure the survival and recovery of the species (USDI-FWS 1977).

Potential for Exposure and Adverse Effects: Contaminants associated with effluent discharges may have contributed to the decline of the unarmored threespine stickleback and may preclude recovery.

Arroyo Toad (*Bufo microscaphus californicus*)

Species Description and Life History: The arroyo toad was listed as endangered on December 16, 1994 (59 **FR** 64589). A draft recovery plan is in preparation, but has not yet been published. Critical habitat has not been proposed. Information regarding the biology of the arroyo toad can be found in Sweet (1992) and Campbell *et al.* (1996). The arroyo toad is a small (adults: snout-urostyle length (SUL) (2.2 to 2.9 inches), light-olive green or gray to tan, dark-spotted toad with a distinctive light-colored, V-shaped stripe across the head and the eyelids.

Arroyo toads are restricted to perennial and intermittent rivers and streams that have shallow, sandy to gravelly pools adjacent to sand or fine gravel terraces. Breeding occurs from March until mid-June (Sweet 1992). Eggs are deposited and larvae develop in shallow pools with minimal current, little or no emergent vegetation, and sand or pea gravel substrate. After metamorphosis from June to August, juveniles remain on the bordering gravel bars until the pool no longer persists (Sweet 1992). Juveniles spend more time exposed on these terraces during the daytime than do adults, and are thus vulnerable to diurnal predators. Adults excavate shallow burrows which are used for shelter during the day when the surface is damp or during longer intervals in the dry season (Sweet 1992). Sexual maturity is reached in one to two years, and toads may live for as few as five years (Sweet 1993). Little is known about movements or other behavior in the non-breeding season.

Foraging Ecology: Juveniles and adults forage for insects, especially ants and small beetles, on sandy stream terraces. Subadults and adults move into surrounding riparian and upland areas to forage.

Historic and Current Distribution: Arroyo toads historically were known to occur in coastal drainages in southern California from San Luis Obispo County to San Diego County and in Baja California, Mexico. In Orange and San Diego Counties, it occurred from the estuaries to the headwaters. The species also was reported from fewer than half a dozen desert slope drainages (USDI in preparation). In 1996, arroyo toads were discovered on Fort Hunter Liggett, Monterey County. This discovery constituted a northern range expansion for the species. Arroyo toads now survive primarily in the headwaters of coastal streams as small isolated populations (Sweet 1992), having been extirpated from much of their historic habitat.

Reasons for Decline and Threats to Survival: Urbanization, agriculture, dam construction, water manipulation, mining, livestock grazing and recreational activities in riparian areas have caused extensive habitat degradation leading to the decline and isolation of the remaining populations of arroyo toads. The introduction of bullfrogs and exotic fish may have severe impacts on toad populations due to predation. Exotic plant species degrade arroyo toad habitat, making it unsuitable, and may cause changes in the invertebrate fauna upon which the toad feeds. Changes in hydrologic regimes and loss of overwintering habitat as streamside areas are developed are probably the most important factors in the decline of arroyo toads.

California Red-Legged Frog (*Rana aurora draytonii*)

Species Description and Life History: The California red-legged frog was federally listed as threatened on May 23, 1996, (61 **FR** 25813). Critical habitat has not been proposed for the species. The Service is currently developing a recovery plan for the species. This species is the largest native frog in the western United States (Wright and Wright 1949), ranging from 4 to 13 centimeters (1.5 to 5.1 inches) in length (Stebbins 1985). The abdomen and hind legs of adults are largely red; the back is characterized by small black flecks and larger irregular dark blotches with indistinct outlines on a brown, gray, olive, or reddish background color. Dorsal spots usually have light centers (Stebbins 1985), and dorsolateral folds are prominent on the back. Larvae (*i.e.,* tadpoles) range from 14 to 80 millimeters (mm) (0.6 to 3.1 inches) in length, and the background color of the body is dark brown and yellow with darker spots (Storer 1925).

California red-legged frogs have paired vocal sacs and vocalize in air (Hayes and Krempels 1986). Female frogs deposit egg masses on emergent vegetation so that the egg mass floats on the surface of the water (Hayes and Miyamoto 1984). California red-legged frogs breed from November through March with earlier breeding records occurring in southern localities (Storer 1925). California red-legged frogs found in coastal drainages are active year-round (Jennings *et al.* 1992), whereas those found in interior sites may be more seasonally inactive.

California red-legged frogs spend most of their lives in and near sheltered backwaters of ponds, marshes, springs, streams, and reservoirs. The largest densities of California red-legged frogs currently are associated with deep pools with dense stands of overhanging willows (*Salix spp.*) and an intermixed fringe of cattails (*Typha latifolia*) (Hayes and Jennings 1988, Jennings 1988). This is considered optimal habitat. California red-legged frog eggs, larvae, transformed juveniles, and adults also have been found in ephemeral creeks and drainages and in ponds that do not have riparian vegetation. Accessability to sheltering habitat is essential for the survival of California red-legged frogs within a watershed, and can be a factor limiting frog population numbers and survival. Sheltering habitat includes mammal burrows, damp leaf litter, downed wood and other cover objects, both natural and manmade, and dense shrubbery up to several hundred meters distant from aquatic sites. California red-legged frogs may shelter in such places for weeks at a time in the wet season. During winter rain events, juvenile and adult California red-legged frogs are known to wander perhaps up to 1-2 km from summer aquatic sites (Rathbun and Holland, unpublished data, cited in Rathbun *et al.* 1991).

Egg masses contain about 2,000 to 5,000 moderate-sized (2.0 to 2.8 mm [0.08 to 0.11 inches] in diameter), dark reddish brown eggs and are typically attached to vertical emergent vegetation, such as bulrushes (*Scirpus* spp.) or cattail (Jennings *et al.* 1992). California red-legged frogs are often prolific breeders, laying their eggs during or shortly after large rainfall events in late winter and early spring (Hayes and Miyamoto 1984). Eggs hatch in 6 to 14 days (Jennings 1988). In coastal lagoons, the most significant mortality factor in the pre-hatching stage is water salinity (Jennings *et al.* 1992). One hundred percent mortality occurs in eggs exposed to salinity levels greater than 4.5 parts per thousand (Jennings and Hayes 1990). Increased siltation that occurs during the breeding season can cause asphyxiation of eggs and small larvae. Larvae undergo metamorphosis 3.5 to 7 months after hatching (Storer 1925, Wright and Wright 1949, Jennings and Hayes 1990). Of the various life stages, larvae probably experience the highest mortality rates, with less than 1 percent of eggs laid reaching metamorphosis (Jennings *et al.* 1992). Sexual maturity normally is reached at 3 to 4 years of age (Storer 1925, Jennings and Hayes 1985). California red-legged frogs may live 8 to 10 years (Jennings *et al.* 1992).

Foraging Ecology: The diet of California red-legged frogs is highly variable. Hayes and Tennant (1985) found invertebrates to be the most common food items. Vertebrates, such as Pacific tree frogs (*Pseudacris* (= *Pseudacris (= Hyla) regilla*) and California mice (*Peromyscus californicus*), represented over half of the prey mass eaten by larger frogs (Hayes and Tennant 1985). Hayes and Tennant (1985) found juvenile frogs to be active diurnally and nocturnally, whereas adult frogs were largely nocturnal. Feeding activity probably occurs along the shoreline and on the surface of the water (Hayes and Tennant 1985). Larvae likely eat algae (Jennings *et al.* 1992).

<u>Historic and Current Distribution</u>: The California red-legged frog has been extirpated or nearly extirpated from 70 percent of its former range. Historically, this species was found throughout the Central Valley and Sierra Nevada foothills. At present, California red-legged frogs are known to occur in 243 streams or drainages from 22 counties, primarily in central coastal

California. The most secure aggregations of California red-legged frogs are found in aquatic sites that support substantial riparian and aquatic vegetation and lack non-native predators [*e.g.*, bullfrogs (*Rana catesbeiana*), bass (*Micropterus spp.*), and sunfish (*Lepomis spp.*)].

Reasons for Decline and Threats to Survival: Over-harvesting, habitat loss, non-native species introduction, and urban encroachment are the primary factors that have negatively affected the California red-legged frog throughout its range (Jennings and Hayes 1985, Hayes and Jennings 1988). Ongoing causes of decline include direct habitat loss due to stream alteration and disturbance to wetland areas, indirect effects of expanding urbanization, and competition or predation from non-native species.

Giant Garter Snake (*Thamnophis gigas*)

Species Description and Life History: The Service published a proposal to list the giant garter snake as an endangered species on December 27, 1991 (56 **FR** 67046). The Service reevaluated the status of the giant garter snake before adopting the final rule. The giant garter snake was listed as a threatened species October 20, 1993 (58 **FR** 54053).

The giant garter snake is one of the largest garter snakes, reaching a total length of at least 64 inches (160 centimeters). Females tend to be slightly longer and proportionately heavier than males. The weight of adult female giant garter snakes is typically 1.1-1.5 pounds (500-700 grams). Dorsal background coloration varies from brownish to olive with a checkered pattern of black spots, separated by a yellow dorsal stripe and two light colored lateral stripes. Background coloration and prominence of the black checkered pattern and the three yellow stripes are geographically and individually variable (Hansen 1980). The ventral surface is cream to olive or brown and sometimes infused with orange, especially in northern populations.

Endemic to wetlands in the Sacramento and San Joaquin valleys, the giant garter snake inhabits marshes, sloughs, ponds, small lakes, low gradient streams, and other waterways and agricultural wetlands, such as irrigation and drainage canals and rice fields, and the adjacent uplands. Giant garter snakes feed on small fishes, tadpoles, and frogs (Fitch 1941, Hansen 1980, Hansen 1988). Essential habitat components consist of: (1) adequate water during the snake's active season (early-spring through mid-fall) to provide food and cover; (2) emergent, herbaceous wetland vegetation, such as cattails and bulrushes, for escape cover and foraging habitat during the active season; (3) upland habitat with grassy banks and openings in waterside vegetation for basking; and (4) higher elevation uplands for cover and refuge from flood waters during the snake's dormant season in the winter (Hansen 1980). Giant garter snakes are typically absent from larger rivers and other water bodies that support introduced populations of large, predatory fish, and from wetlands with sand, gravel, or rock substrates (Hansen 1980, Rossman and Stewart 1987, Brode 1988, Hansen 1988). Riparian woodlands do not typically provide suitable habitat because of excessive shade, lack of basking sites, and absence of prey populations (Hansen 1980).

Foraging ecology - Giant garter snakes are extremely aquatic, are rarely found away from water, forage in the water for food, and will retreat to water to escape predators and disturbance. This species occupies a niche similar to some eastern water snakes (*Nerodia* spp.). Giant garter snakes are active foragers, feeding primarily on aquatic prey such as fish and amphibians. Historically, prey likely consisted of Sacramento blackfish (*Orthodon microlepidotus*), thick-tailed chub (*Gila crassicauda*), and red-legged frog (*Rana aurora*). Because these species are no longer available (the thick-tailed chub is extinct, the red-legged frog is extirpated from the Central Valley, and the blackfish is declining/in low numbers), the predominant food items are now introduced species such as carp (*Cyprinus carpio*), mosquito-fish (*Gambusia affinis*), bullfrogs (*Rana catesbiana*), and Pacific treefrogs (*Pseudacris regilla*) (Fitch 1941, Rossman et al, 1996).

The breeding season extends through March and April, and females give birth to live young from late July through early September (Hansen and Hansen 1990). Brood size is variable, ranging from 10 to 46 young, with a mean of 23 (Hansen and Hansen 1990). At birth young average about 20.6 cm snout-vent length and 3-5 g. Young immediately scatter into dense cover and absorb their yolk sacs, after which they begin feeding on their own. Although growth rates are variable, young typically more than double in size by one year of age (G. Hansen, pers. comm.). Sexual maturity averages three years in males and five years for females (G. Hansen, pers. comm.).

The giant garter snake inhabits small mammal burrows and other soil crevices above prevailing flood elevations throughout its winter dormancy period (i.e., November to mid-March). Giant garter snakes typically select burrows with sunny exposure along south and west facing slopes. Giant garter snakes also use burrows as refuge from extreme heat during their active period. The Biological Resources Division (BRD) of the USGS (Wylie *et al.* 1997) has documented giant garter snakes using burrows in the summer as much as 165 feet (50 meters) away from the marsh edge. Overwintering snakes have been documented using burrows as far as 820 feet (250 meters) from the edge of marsh habitat.

During radio-telemetry studies conducted by the BRD giant garter snakes typically moved little from day to day. However, total activity varied widely between individuals. Snakes have been documented moving up to 5 miles (8 kilometers) over the period of a few days (Wylie *et al.* 1997). In agricultural areas, giant garter snakes were documented using rice fields 19-20% of the observations, marsh habitat 20-23% of observations, and canal and agricultural waterway habitats 50-56% of the observations (Wylie *et al.* 1997). Within canal and agricultural waterway habitats, giant garter snakes are likely to prefer drainage rather than delivery canals, because drainage canals are often less heavily maintained and are allowed to become vegetated.

<u>Historic and Current Distribution</u>: Fitch (1940) described the historical range of the species as extending from the vicinity of Sacramento and Contra Costa Counties southward to Buena Vista Lake, near Bakersfield, in Kern County. Prior to 1970, the giant garter snake was recorded historically from 17 localities (Hansen and Brode 1980). Five of these localities were clustered in and around Los Banos, Merced County, and the paucity of information makes it difficult to

determine precisely the species' former range. Nonetheless, these records coincide with the historical distribution of large flood basins, fresh water marshes, and tributary streams. Reclamation of wetlands for agriculture and other purposes apparently extirpated the species from the southern one-third of its range by the 1940's-1950's, including the former Buena Vista Lake and Kern Lake in Kern County, and the historic Tulare Lake and other wetlands in Kings and Tulare Counties (Hansen and Brode 1980, Hansen 1980). Surveys over the last two decades have located the giant garter snake as far north as the Butte Basin in the Sacramento Valley.

As recently as the 1970s, the range of the giant garter snake extended from near Burrel, Fresno County (Hansen and Brode 1980), northward to the vicinity of Chico, Butte County (Rossman and Stewart 1987). California Department of Fish and Game (CDFG) studies (Hansen 1988) indicate that giant garter snake populations currently are distributed in portions of the rice production zones of Sacramento, Sutter, Butte, Colusa, and Glenn Counties; along the western border of the Yolo Bypass in Yolo County; and along the eastern fringes of the Sacramento-San Joaquin River delta from the Laguna Creek-Elk Grove region of central Sacramento County southward to the Stockton area of San Joaquin County. This distribution largely corresponds with agricultural land uses throughout the Central Valley.

Surveys over the last two decades have located the giant garter snake as far north as the Butte Basin in the Sacramento Valley. Currently, the Service recognizes 13 separate populations of giant garter snakes, with each population representing a cluster of discrete locality records (58 **FR** 54053). The 13 extant populational clusters largely coincide with historical riverine flood basins and tributary streams throughout the Central Valley (Hansen 1980, Brode and Hansen 1992): (1) Butte Basin, (2) Colusa Basin, (3) Sutter Basin, (4) American Basin, (5) Yolo Basin--Willow Slough, (6) Yolo Basin--Liberty Farms, (7) Sacramento Basin, (8) Badger Creek--Willow Creek, (9) Caldoni Marsh, (10) East Stockton--Diverting Canal and Duck Creek, (11) North and South Grasslands, (12) Mendota, and (13) Burrel/Lanare. These populations span the Central Valley from just southwest of Fresno (i.e., Burrel-Lanare) north to Chico (i.e., Hamilton Slough). The 11 counties where the giant garter snake is still presumed to occur are: Butte, Colusa, Glenn, Fresno, Merced, Sacramento, San Joaquin, Solano, Stanislaus, Sutter and Yolo.

In 1994, the BRD (formerly the National Biological Survey [NBS]) began a study of the life history and habitat requirements of the giant garter snake in response to an interagency submittal for consideration as an NBS Ecosystem Initiative. Since April of 1995, the BRD has further documented occurrences of giant garter snakes within some of the 13 populations identified in the final rule. The BRD has studied populations of giant garter snakes at the Sacramento and Colusa National Wildlife Refuges within the Colusa Basin, at Gilsizer Slough within the Sutter Basin, and at the Badger Creek area of the Cosumnes River Preserve within the Badger Creek-Willow Creek area (Wylie et al, 1997). These populations, along with the American Basin population of giant garter snakes represent the largest extant populations. With the exception of the American Basin, these populations are largely protected from many of the threats to the species. Outside of these protected areas, giant garter snakes in these population clusters are still subject to all threats identified in the final rule. The remaining nine population clusters identified in the final rule are distributed discontinuously in small isolated patches and are vulnerable to extirpation by stochastic environmental, demographic, and genetic processes. All 13 population clusters are isolated from each other with no protected dispersal corridors. Opportunities for recolonization of small populations which may become extirpated are unlikely given the isolation from larger populations and lack of dispersal corridors between them.

Further descriptions of the status of the thirteen subpopulations are given in Table 4 and in Appendix A.

Reasons for Decline and Threats to Survival: The current distribution and abundance of the giant garter snake is much reduced from former times. Agricultural and flood control activities have extirpated the giant garter snake from the southern one third of its range in former wetlands associated with the historic Buena Vista, Tulare, and Kern lakebeds. These lakebeds once supported vast expanses of ideal giant garter snake habitat, consisting of cattail and bulrush dominated marshes. Vast expanses of bulrush and cattail floodplain habitat also typified much of the Sacramento Valley historically (Hinds 1952). Prior to reclamation activities beginning in the mid to late 1800's, about 60 percent of the Sacramento Valley was subject to seasonal overflow flooding in broad, shallow flood basins that provided expansive areas of giant garter snake habitat (ibid.). All natural habitats have been lost and an unquantifiable small percentage of semi-natural wetlands remain extant. Only a small percentage of extant wetlands currently provide habitat suitable for the giant garter snake. Valley floor wetlands are also subject to the cumulative effects of upstream watershed modifications, water storage and diversion projects, as well as urban and agricultural development. Although some giant garter snake populations have persisted at low levels in artificial wetlands associated with agricultural and flood control activities, many of these altered wetlands are now threatened with urban development. Cities within the current range of the giant garter snake that are rapidly expanding include: (1) Chico, (2) Yuba City, (3) Sacramento, (4) Galt, (5) Stockton, (6) Gustine, and (7) Los Banos.

A number of land use practices and other human activities currently threaten the survival of the giant garter snake throughout the remainder of its range. Ongoing maintenance of aquatic habitats for flood control and agricultural purposes eliminate or prevent the establishment of habitat characteristics required by giant garter snakes and can fragment and isolate available habitat, prevent dispersal of snakes among habitat units, and adversely affect the availability of the garter snake's food items (Hansen 1988, Brode and Hansen 1992). Livestock grazing along the edges of water sources degrades habitat quality in a number of ways: (1) eating and trampling aquatic and riparian vegetation needed for cover from predators, (2) changes in plant species composition, (3) trampling snakes, (4) water pollution, (5) and reducing or eliminating fish and amphibian prey populations. Overall, grazing has contributed to the elimination and reduction of the quality of available habitat at four known locations (Hansen 1982, 1986).

In many areas, the restriction of suitable habitat to water canals bordered by roadways and levee tops renders giant garter snakes vulnerable to vehicular mortality. Fluctuation in rice and agricultural production affects stability and availability of habitat. Recreational activities, such as fishing, may disturb snakes and disrupt basking and foraging activities. Non-native predators, including introduced predatory gamefish, bullfrogs, and domestic cats also threaten giant garter snake populations. While large areas of seemingly suitable giant garter snake habitat exist in the form of duck clubs and waterfowl management areas, water management of these areas typically does not provide summer water needed by giant garter snakes. Although giant garter snakes on NWRs are relatively protected from many of the threats to the species, water quality continues to be a threat to the species both on and off NWRs.

Documented declines due to selenium contamination - San Joaquin Valley subpopulations of giant garter snakes have suffered severe declines and possible extirpations over the last two decades. Prior to 1980, several areas within the San Joaquin Valley supported populations of giant garter snakes. Until recently, there were no post-1980 sightings from Stockton, San Joaquin County, southward, despite several survey efforts (G. Hansen, 1988). Surveys during 1986 of prior localities did not detect any giant garter snakes. During 1995 surveys of prior locality records and adjacent waterways, one road killed giant garter snake was found, and three presumed giant garter snakes were observed but not captured (G. Hansen, 1996). Two sightings occurred at Mendota Wildlife Area, and two occurred several miles south of the town of Los Banos. These data indicate that giant garter snakes are still extant in two localities within the San Joaquin, but in extremely low to undetectable numbers.

Although habitat has been lost or degraded throughout the Central Valley, there have been many recent sightings of giant garter snakes in the Sacramento Valley while there have been very few recent sightings within the San Joaquin Valley. The 1995 report on the status of giant garter snakes in the San Joaquin Valley (G. Hansen, 1996) indicates that Central San Joaquin Valley giant garter snake numbers appear to have declined even more dramatically than has apparently suitable habitat. Factors in addition to habitat loss may be contributing to the decline. These are factors which affect giant garter snakes within suitable habitat and include interrupted water supply, poor water quality, and contaminants (G. Hansen, 1996).

Selenium contamination and impaired water quality have been identified in the final rule listing the giant garter snake as a threat to the species and a contributing factor in the decline of giant garter snake populations, particularly for the North and South Grasslands subpopulation (i.e., Kesterson NWR area). The bioaccumulative food chain threat of selenium contamination on fish, frogs, and fish-eating birds has been well documented. Though there is little data specifically addressing toxicity of selenium, Hg, or metals to reptiles, it is expected that reptiles would have toxicity thresholds similar to those of fish and birds. (58 **FR** 54053 under Factor E - Contaminants)

Threats due to contaminants and impaired water quality - The range of the giant garter snake occurs entirely within the Central Valley of California, putting giant garter snakes at risk of exposure to numerous contaminants from agricultural, urban, and industrial/mining runoff. Current water sources and supplies to areas supporting giant garter snakes indicate that the species is at risk of exposure to both mercury and selenium. Many areas supporting populations of giant garter snake receive water from agricultural drainage, which may contain elevated levels of selenium and other contaminants. Selenium contamination of drainwater has been identified in the San Joaquin Valley giant garter snake subpopulations (58 **FR** 54053 and references therein). However, refuges in the Sacramento Valley which currently support giant garter snakes also receive agricultural return flows as part of their water supplies. These include Gray Lodge Wildlife Area, Sacramento NWR, Delevan NWR, Colusa NWR, and Sutter NWR (USDI 1997). In addition, streams draining the coastal ranges may contribute selenium to aquatic systems within the Central Valley.

Mercury also is present in numerous drainages in the Central Valley due to past mercury and gold mining activity. Sacramento Valley refuges and other areas supporting giant garter snake populations also receive water from drainages which may contribute mercury to the aquatic systems. These drainages include the Sacramento, Feather, American, and Cosumnes Rivers, and Laguna, Morrison, Stony, Auburn Ravine, Putah, and Cache Creeks.

Table 4 describes known giant garter snake locations within the thirteen giant garter snake subpopulations, the status of the subpopulations, the potential for exposure to selenium and mercury, and the potential for synergistic effects of selenium and mercury. Appendix A further describes the status of the thirteen subpopulations, and also describes some water supply sources to refuges and other areas that support giant garter snakes. Although giant garter snake populations on refuges may be protected from many of the threats to the species, they are not protected from exposure to poor water quality and contaminants introduced from water supply sources.

Water quality impairment of aquatic habitat that supports giant garter snakes could reduce the prey base, contribute to bioaccumulation, impair essential behaviors, and reduce reproductive success. Appendix A lists existing impaired water bodies (from California Impaired Waterbodies list) that either currently support giant garter snakes or supply water to areas that support giant garter snakes. Although the level of impairment and specific contaminants were not listed, this information identifies that significant water quality impairment already exists. The list of water bodies that may support or supply giant garter snake populations indicates that the species is currently challenged with poor water quality. Unprotective water quality standards proposed in the CTR could further impair water quality within these giant garter snake subpopulations and represent the potential for cumulative and synergistic effects of contaminants and poor water quality.

Summary of contaminants threats to giant garter snakes - The giant garter snake has a restricted distribution and is entirely dependent on its aquatic ecosystem. The thirteen population clusters identified in the final rule are distributed discontinuously in small isolated patches and are vulnerable to extirpation by stochastic environmental, demographic, and genetic processes. It is probable that elevated selenium levels in the San Joaquin Valley contributed to the severe decline and possible extirpation of the giant garter snake from the majority of this area. The remaining giant garter snake populations are exposed to impaired waterbodies and existing or potential sources of selenium and mercury. As top predators, giant garter snakes are at risk of exposure to elevated levels of contaminants such as mercury and selenium. Over the life of the giant garter snake it is possible to accumulate contaminants that can impact the growth, survival, and reproduction of individuals, leading to declines in distribution.

Mountain Yellow-legged Frog: Southern California Distinct Population Segment (*Rana muscosa*)

Species Description and Life History: The mountain yellow-legged frog is a true frog in the family Ranidae. Mountain yellow-legged frogs were originally described by Camp in 1917 (as cited by Zweifel 1955) as a subspecies of Rana boylii. Zweifel (1955) demonstrated that frogs from the high Sierra and the mountains of southern California were somewhat similar to each other yet were distinct from the rest of the R. boylii (= boylei) group. Since that time, most authors have followed Zweifel, treating the mountain yellow-legged frog as a full species, Rana muscosa.

Mountain yellow-legged frogs are moderately sized, about 40 to 80 millimeters (mm) (1.5 to 3 inches (in)) from snout to urostyle (the pointed bone at the base of the backbone) (Jennings and Hayes 1994; Zweifel 1955). The pattern is variable, ranging from discrete dark spots that can be few and large, to smaller and more numerous spots with a mixture of sizes and shapes, to irregular lichen-like patches or a poorly defined network (Zweifel 1955). The body color is also variable, usually a mix of brown and yellow, but often with gray, red, or green-brown. Some individuals may be dark brown with little pattern (Jennings and Hayes 1994). The back half of the upper lip is pale. Folds are present on each side of the back, but usually they are not prominent (Stebbins 1985). The throat is white or yellow, sometimes with mottling of dark pigment (Zweifel 1955). The belly and undersurface of the high limbs are yellow, which ranges in hue from pale lemon yellow to an intense sun yellow. The iris is gold with a horizontal, black counter shading stripe (Jennings and Hayes 1994).

In the Sierra Nevada Mountains of California, the mountain yellow-legged frog ranges from southern Plumas County to southern Tulare County (Jennings and Hayes 1994), at elevations mostly above 1,820 meters (m) (6,000 feet (ft)). The frogs of the Sierra Nevada are isolated from the frogs of the mountains of southern California by the Tehachapi Mountains and a distance of about 225 kilometers (km) (140 miles (mi)). The southern California frogs now occupy portions of the San Gabriel, San Bernardino, and San Jacinto Mountains. Zweifel (1955) noted the presence of an isolated southern population on Mt. Palomar in northern San Diego County, but this population appears to be extinct (Jennings and Hayes 1994). In southern California, the elevation range reported by Stebbins (1985) is 182 m (600 ft) to 2,273 m (7,500 ft). Representative localities, including some that are no longer occupied, which demonstrate the wide elevation range that mountain yellow-legged frogs inhabited in southern California, include Eaton Canyon, Los Angeles County (370 m (1,220 ft)) and Bluff Lake, San Bernardino County (2,290 m (7,560 ft)). The southern California locations now occupied by mountain yellow-legged frogs range from City Creek, in the San Bernardino Mountains (760 m (2,500 ft)), to Dark

Canyon in the San Jacinto Mountains (1,820 m (6,000 ft)).

Southern California mountain yellow-legged frogs are diurnal, highly aquatic frogs, occupying rocky and shaded streams with cool waters originating from springs and snowmelt. In these areas, juveniles and adults feed on small, streamside arthropods (Jennings and Hayes 1994). They do not occur in the smallest creeks. The coldest winter months are spent in hibernation, probably under water or in crevices in the bank. Mountain yellow-legged frogs emerge from overwintering sites in early spring, and breeding soon follows. Eggs are deposited in shallow water where the egg mass is attached to vegetation or the substrate. In the Sierra Nevada, larvae select warm microhabitats (Bradford 1984 cited in Jennings and Hayes 1994), and the time to develop from fertilization to metamorphosis reportedly varies from 1 to 2.5 years (Jennings and Hayes 1994).

Prior to the late 1960s, mountain yellow-legged frogs were abundant in many southern California streams (G. Stewart, in litt. 1995), but they now appear to be absent from most places in which they previously occurred. Jennings and Hayes (1994) believe that mountain yellow-legged frogs are now absent from more than 99 percent of their previous range in southern California. This decline is part of a well-known larger pattern of declines among native ranid frogs in the western United States (Hayes and Jennings 1986; Drost and Fellers 1996). Some of the western ranid frog species experiencing noticeable declines are the California red-legged frog (Rana aurora draytonii) (61 FR 25813), the spotted frog (R. pretiosa and R. luteventris), the Cascades frog (R. cascadae), and the Chiricahua leopard frog (R. chiricauhensis) (62 FR 49398). Nowhere have the declines been any more pronounced than in southern California, where, besides declines in mountain yellow-legged frogs, the California red-legged frog has been reduced to a few small remnants (61 FR 25813), and the foothill yellow-legged frog (R. boylii) may be extinct (Jennings and Hayes 1994.)

Distinct Vertebrate Population Segment: We analyzed the mountain yellow-legged frog according to the joint Service and National Marine Fisheries Service Policy Regarding the Recognition of Distinct Vertebrate Populations, published in the Federal Register on February 7, 1996 (61 FR 4722). We consider three elements in determining whether a vertebrate population segment could be treated as threatened or endangered under the Act: discreteness, significance, and conservation status in relation to the standards for listing. Discreteness refers to the isolation of a population from other members of the species and is based on two criteria: (1) Marked separation from other populations of the same taxon resulting from physical, physiological, ecological, or behavioral factors, including genetic discontinuity, or (2) populations delimited by international boundaries. We determine significance either by the importance or contribution, or both, of a discrete population to the species throughout its range. Our policy lists four examples of factors that may be used to determine significance: (1) Persistence of the discrete population segment in an ecological setting unusual or unique for the taxon; (2) evidence that loss of the discrete population segment would result in a significant gap in the range of the taxon; (3) evidence that the discrete population segment represents the only surviving natural occurrence of the taxon that may be more abundant elsewhere as an introduced population outside its historic range; and (4) evidence that the discrete population segment differs markedly from other populations of the taxon in its genetic characteristics. If we determine that a population segment is discrete and significant, we evaluate it for endangered or threatened status based on the Act's standards.

Discreteness: The range of the mountain yellow-legged frog is divided by a natural geographic barrier, the Tehachapi Mountains, which isolate Sierran frogs from those in the mountains of southern California. The distance of the separation is about 225 km (140 mi), but the separation may not have been this great in the recent past because a frog collected in 1952 on Breckenridge Mountain in Kern County was identified by Jennings and Hayes (1994) as a mountain yellow-legged frog. The geographic separation of the Sierran and southern California frogs was recognized in the earliest description of the species by Camp (1917, cited in Zweifel 1955), who treated frogs from the two localities as separate subspecies within the R. boylii group. He designated the Sierran frogs R. b. sierrae and the southern California frogs R. b. muscosa, based on geography and subtle morphological differences. Zweifel (1955) reevaluated the morphological evidence and found it insufficient to warrant Camp's recognition of two subspecies, the chief difference between the two being hind-limb length.

More recently, Ziesmer (1997) analyzed the calls of Sierran (Alpine and Mariposa Counties) and southern California (San Jacinto Mountains and Riverside County) mountain yellow-legged frogs. He found that the calls of Sierran frogs differed from southern California frogs in pulse rate, harmonic structure, and dominant frequency. Based on a limited sample, Ziesmer concluded that the results supported the hypothesis that mountain yellow-legged frogs from the Sierra Nevada and southern California are separate species.

Allozyme (a form of an enzyme produced by a gene) variation throughout the range of the mountain yellow-legged frog has been examined, but the results are open to interpretation (Jennings and Hayes 1994 and references therein). In the work most applicable to the question of the distinctiveness of the Sierran and southern California frogs, David Green (pers. comm., 1998) analyzed allozyme variation in central Sierran mountain yellow-legged frogs (four individuals, Tuolumne County) and southern California mountain yellow-legged frogs (two individuals, Riverside County). He found fixed differences at 6 of 28 loci (sites on a chromosome occupied by specific genes). These limited, unpublished data suggest that Sierran and southern California mountain yellow-legged frogs are different at a level that could support the recognition of full species. However, because of the small number of individuals per sample and the limited number of samples, we view these results cautiously. It is possible that existing variation at those six loci may not have been detected with such a small number of individuals sampled. To better understand whether a genetic discontinuity significant enough to warrant full species rank exists between Sierran frogs and those from the mountains of southern California, samples of frogs from the southern Sierra Nevada, especially the Greenhorn Mountains, would be of particular interest.

Although Green's limited allozyme analysis may not be sufficient to support recognizing the Sierran and southern California populations as separate species, it does support the conclusion of significant geographic separation. This conclusion is also supported by earlier observations of morphological differences (Zweifel 1955, and references therein) and differences in vocalizations (Ziesmer 1997). Considered together, the evidence supports an interpretation of isolation between the two populations of frogs over a very long period. We find that the southern California frogs meet the criterion of "marked separation from other populations of the same taxon" and qualify as discrete according to the Policy Regarding the Recognition of Distinct Vertebrate Populations (61 FR 4722).

Significance. One of the most striking differences between Sierran and southern California mountain yellow-legged frogs is the habitats they occupy. Zweifel (1955) observed that the frogs in southern California are typically found in steep gradient streams in the chaparral belt, even though they may range up into small meadow streams at higher elevations. In contrast, Sierran frogs are most abundant in high elevation lakes and slow-moving portions of streams. Bradford's (1989) southern Sierra Nevada study site, for example, was in Sequoia and Kings Canyon National Parks at high elevations (between 2,910-3,430 m (9,600-11,319 ft)). The rugged canyons of the arid mountain ranges of southern California bear little resemblance to the alpine lakes of the Sierra Nevada. On the basis of habitat alone, one might easily conclude that these are two very different frogs.

The mountain yellow-legged frogs of southern California comprise the southern portion of the species' range. The extinction of this southern group would be significant because it would substantially reduce the overall range as it is currently understood, and what is now a gap in the distribution, the Tehachapi Mountains, would become the southern limit of the species' range.

In addition, evidence exists that the mountain yellow-legged frog is not simply a single species with a disjunct distribution (cited in Zweifel 1955; Stebbins 1985). As discussed above, vocal and genetic differences exist between Sierran and southern California mountain yellow-legged frogs. Although the data are limited and some important variation may have been missed, they are consistent with the earlier interpretation by Camp (1917 cited in Zweifel 1955) and numerous other authors prior to Zweifel (e.g., Stebbins 1954) who treated the two forms as taxonomically distinct. If the differences in vocalization described by Ziesmer (1997) and the allozyme variation described by Green (per. comm., 1998) accurately characterize differences between the two forms, then the Sierran and southern California frogs are quite different and have been isolated for a very long time.

Our conclusion that Sierran and southern California frogs are very different from each other, and may even merit recognition as separate subspecies or possibly even species, is based on the cumulative weight of the available evidence. We find that the mountain yellow-legged frogs inhabiting the mountains of southern California meet the significance criteria under our Policy Regarding the Recognition of Distinct Vertebrate Populations (61 FR 4722) on the basis of the geographical, ecological, vocal, and genetic discontinuities described above.

Reasons for Decline and Threats to Survival: The mechanisms causing the declines of western frogs are not well understood and are certain to vary somewhat among species, but the two most common and well-supported hypotheses for widespread declines of western ranid frogs are: (1) Past habitat destruction related to unregulated activities such as logging and mining and more recent habitat conversions for water development, irrigated agriculture, and commercial development (Hayes and Jennings 1986; 61 FR 25813); and (2) alien predators and competitors (Bradford 1989; Knapp 1996; Kupferberg 1997). Natural populations may be killed off directly by these factors operating alone or in combination, or these factors so severely disrupt the normal population dynamics that when local extinctions occur, regardless of the cause, natural recolonization is impossible. Other environmental factors that could have adverse effects over a wide geographic range include pesticides, certain pathogens, and ultraviolet-B (beyond the visible spectrum) radiation, but their role, if any, in amphibian declines is not well understood (Reaser 1996). These factors, acting singly or in combination, may be contributing to widespread, systematic declines of western ranid frogs. Determining their effects, however, is not an easy task (Reaser 1996; Wake 1998), and the Department of the Interior (USDOI) currently supports an initiative to fund research on the causes of amphibian declines (see examples in USDOI 1998).

Some of the same factors that are hypothesized to have caused declines of other western ranid frogs are likely to be responsible for the reduction of the mountain yellow-legged frog in southern California. Because the declines have been so precipitous, and have spared only a small number of frogs in a few localities, the factors, and their interactions, that caused the decline may never be fully understood. We believe that these factors are still operating, and unless reversed, a high probability exists that this frog may be extinct in southern California within a few decades. In the case of the mountain yellow-legged frog, the only factor listed above that we believe can be ruled out as a likely cause of decline is habitat destruction related to activities such as logging, mining, irrigated agriculture, and commercial development. The range of the mountain yellow-legged frog in southern California is mainly on public land administered by the U.S. Forest Service (FS). Most of the rugged canyons and surrounding mountainous terrain have been altered little and look much the same today as they did when earlier naturalists such as Lawrence Klauber collected mountain yellow-legged frogs there in the early decades of the 1900s.

<u>Historic and Current Distribution:</u> In southern California, mountain yellow-legged frogs can still be found in four small streams in the San Gabriel Mountains, the upper reaches of the San Jacinto River system in the San Jacinto Mountains, and at a single locality on City Creek, a tributary of the Santa Ana River, in the San Bernardino Mountains (Jennings and Hayes 1994; M. D. Wilcox <u>in</u> <u>litt</u>., 1998). These areas along with the numbers of frogs most recently observed in each area are described below.

<u>San Gabriel Mountains</u>: Surveys conducted from 1993 to 1997 revealed small isolated populations in the upper reaches of Prairie Creek/Vincent Gulch, Devil's Canyon, and Alder Creek/East Fork, on the East Fork of the San Gabriel River, and Little Rock Creek on the Mojave River (Jennings and Hayes 1994 and references therein; Jennings 1995; Jennings 1998). The surveys involved one to three field biologists and were conducted over 1-5 days per site.

Over the course of these field studies, 15 adults or fewer were observed at any 1 site, and, after the 1995 season, Jennings (1995) concluded that the actual population at each of the sites was only 10-20 adults.

San Jacinto Mountains: Small populations of mountain yellow-legged frogs also occur in four tributaries in the upper reaches of the North Fork, San Jacinto River on Mount San Jacinto: Dark Canyon, Hall Canyon, Fuller Mill Creek, and the main North Fork, San Jacinto River (Jennings and Hayes 1994; Jennings 1995; Jennings 1998). The number of frogs occupying these sites is not known, but fewer than 10 adult frogs per site per year have been observed in surveys from 1995 to the present.

San Bernardino Mountains: A few tadpoles and 26 recently transformed juveniles, but no adults, were rediscovered on a roughly 1-mile reach of the East Fork, City Creek during the summer of 1998 (M. D. Wilcox in litt., 1998). Previous to this finding, mountain yellow-legged frogs had not been observed in the San Bernardino Mountains since the 1970s (Jennings and Hayes 1994), even though surveys were conducted during the summer and fall of 1997 and 1998 (Holland 1997; Tierra Madre 1999).

When frogs were encountered during field surveys accomplished between 1988 and 1995, only a few individuals were observed. Jennings and Hayes (1994) and Jennings (1995) suggested that the entire population of mountain yellow-legged frogs in the San Gabriel and San Jacinto Mountains (8 more or less isolated sites) was probably fewer than 100 adult frogs. Their rough estimate is based on a compilation of the results of visual surveys generally conducted on a single day, not on formal population abundance estimation techniques. While the precise number of adult frogs may be greater than 100, we concur with Jennings and Hayes (1994) that, in the San Gabriel and San Jacinto Mountains, the available data indicate that this once widespread species is now found in only a small number of relatively isolated populations. We do not know the population size of adult frogs at the recently rediscovered site on the east fork of City Creek in the San Bernardino Mountains, but because no adults and only a few juveniles and tadpoles were encountered, the adult population is probably small. Thus, we conclude that each of the three mountain ranges (San Gabriel, San Jacinto, San Bernardino) contains a small number of small, relatively isolated populations.

San Francisco garter snake (*Thamophis sirtalis tetrataenia*)

Species Description and Life History: The San Francisco garter snake was listed as a Federal endangered species in March, 1967 (32 **FR** 4001). The San Francisco garter snake is an extremely colorful snake. It is identified by its burnt orange head, yellow to greenish-yellow dorsal stripe edged in black, and its red lateral stripe which may be continuous or broken with black blotches and edged in black. The belly color varies from greenish-blue to blue. Large adults can reach three feet in length.

The San Francisco garter snakes preferred habitat is a densely vegetated pond near an open hillside where it can sun itself, feed, and find cover in rodent burrows. The snakes are extremely shy, difficult to locate and capture, and quick to flee to water or cover when disturbed (Willy, pers. comm.). Adult snakes may estivate in rodent burrows during summer months when ponds may dry. On the coast snakes hibernate during the winter, but further inland, if the weather is suitable, snakes may be active year round.

San Francisco garter snakes breed in the spring or late fall (Larsen, pers. comm.) and bear live young from May through October (Stebbins 1985). The average litter size is 12-18 (Stebbins 1985). Many species of snakes, including garter snakes, breed adjacent to their hibernacula. Although highly vagile, adults spend considerable time after emergence in their hibernacula.

Foraging Ecology: Although primarily a diurnal species, captive snakes housed in an outside enclosure were observed foraging after dark on warm evenings (Larsen, pers. comm.). Adult snakes feed primarily on California red-legged frogs, and may also feed on juvenile bullfrogs (*Rana catesbeiana*). In laboratory studies, Larsen (1994) fed adult San Francisco garter snakes two year old bullfrog tadpoles and found that only the largest adults could eat and digest the tadpoles; smaller adults regurgitated partially digested tadpoles, apparently unable to fully digest them. Larsen (1994) also found that when these smaller adult snakes were fed bullfrogs and California red-legged frogs of comparable size, they were unable to hold and eat the bullfrogs although they had no trouble with the California red-legged frogs. Newborn and juvenile San Francisco garter snakes depend heavily upon Pacific treefrogs (*Hyla regilla*) as prey (Larsen 1994). If newly metamorphosed Pacific treefrogs are not available, the young snakes may not survive.

<u>Historic and Current Distribution</u>: Historically, San Francisco garter snakes occurred in scattered wetland areas on the San Francisco Peninsula from approximately the San Francisco County line south along the eastern and western bases of the Santa Cruz Mountains, at least to the Upper Crystal Springs Reservoir, and along the coast south to Año Nuevo Point, San Mateo County, and Waddell Creek, Santa Cruz County, California. Currently, the species has been reduced to only six populations in San Mateo County and the extreme northern Santa Cruz County. Sag ponds--small seasonal freshwater ponds formed along the San Andreas fault--historically supported this snake, but most of these former locations have been destroyed by urbanization.

The species has been extirpated from most of its historical distribution in the Skyline Boulevard area of San Mateo County. Fox (1951) reported typical populations of the snake on the coast around Sharp Park (Laguna Salada), and along Skyline Boulevard. Since then, the sag ponds along Skyline Boulevard were drained and filled for urban development and the Sharp Park area has been severely impacted. In 1987, the seawall at Sharp Park failed, allowing the intrusion of salt water into Laguna Salada. In 1989, abandoned quarry ponds adjacent to Calera Creek (over the ridge from Sharp Park) were found to support a small population of snakes. These snakes may have migrated from Laguna Salada after the failure of the sea wall. In August 1989, the quarry ponds were illegally drained and filled. The current population status at the quarry ponds and Sharp Park is unknown. In 1985, the population at Año Nuevo State Reserve was thought to be stable at fewer than 50 snakes, but in 1995 the population appeared to be declining (Paul Keel, pers. comm.). This decline may be caused by inadequate management for the San Francisco garter snake and the recent introduction of bullfrogs.

The Recovery Plan for the San Francisco garter snake (USDI-FWS 1985c) identified six significant populations. These were the Airport (west-of-Bayshore), San Francisco State Fish and Game Refuge (Refuge), Laguna Salada (Pacifica), Pescadero Marsh Natural Preserve (Pescadero) and Año Nuevo State Reserve (Año Nuevo) populations, and an isolated population fragment north of Half Moon Bay. Of the six populations known in 1985, the Pacifica population was heavily impacted in 1989 and is no longer considered significant, four have declined drastically (Airport, Refuge, Pescadero and Año Nuevo). The status of the Half Moon Bay population is unknown.

Reasons for Decline and Threats to Survival: Current threats to the San Francisco garter snakes' existence include reservoir construction and management, agricultural practices, poor management practices on lands where San Francisco garter snakes currently survive, and isolation of populations. Introduced predators such as predatory fish and bullfrogs impact not only the San Francisco garter snake, but also its principal prey species, the Pacific treefrog and the threatened California red-legged frog. Because there are so few remaining populations of the San Francisco garter snake extant populations are extremely vulnerable to local contamination. The San Francisco garter snake has a narrow foraging niche, if contamination of forage species occurs it is likely to significantly impact the species ability to survive. The San Francisco garter snake's beautiful coloration also makes it valuable to both amateur and professional illegal collectors. Extirpation of California red-legged frogs in San Francisco garter snake habitat is likely to cause a local extinction event for the snake.

California Tiger Salamander - Santa Barbara County Distinct Population Segment (*Ambystoma califórnniense*)

Species Description and Life History: The California tiger salamander is a large, stocky, terrestrial salamander with a broad, rounded snout. This distinct population segment (DPS)of the species was proposed as endangered on January 19, 2000 (65 **FR** 3110). California tiger salamanders are restricted to California, and their range does not overlap with any other species of tiger salamander (Stebbins 1985). Within California, the Santa Barbara County population is separated by the Coast Ranges, particularly the La Panza and Sierra Madre Ranges, and the Carrizo Plain from the closest other population, which extends into the Temblor Range in eastern San Luis Obispo and western Kern Counties (Shaffer, et al. 1993).

Adults may reach a total length of 207 millimeters (mm) (8.2 inches (in)), with males generally averaging about 200 mm (8 in) in total length and females averaging about 170 mm (6.8 in) in total length. For both sexes, the average snout–vent length is approximately 90 mm (3.6 in). The small eyes have black irises and protrude from the head. Coloration consists of white or pale yellow spots or bars on a black background on the back and sides. The belly varies from almost uniform white or pale yellow to a variegated pattern of white or pale yellow and black. Males can be distinguished from females, especially during the breeding season, by their swollen cloacae (a common chamber into which the intestinal, urinary, and reproductive canals discharge), more developed tail fins, and larger overall size (Stebbins 1962; Loredo and Van Vuren 1996).

Subadult and adult California tiger salamanders spend much of their lives in small mammal burrows found in the upland component of their habitat, particularly those of ground squirrels and pocket gophers (Loredo and Van Vuren 1996, Trenham 1998a). During estivation (a state of dormancy or inactivity in response to hot, dry weather), California tiger salamanders eat very little (Shaffer, et al. 1993). Once fall and winter rains begin, they emerge from these retreats on nights of high relative humidity and during rains to feed and to migrate to the breeding ponds (Stebbins 1985, 1989; Shaffer, et al. 1993). The salamanders breeding in and living around a pool or seasonal pond, or a local complex of pools or seasonal ponds, constitute a local subpopulation. The rate of natural movement of salamanders among subpopulations depends on the distance between the ponds or complexes and on the intervening habitat (e.g., salamanders may move more quickly through sparsely covered and more open grassland versus more densely vegetated scrublands).

Adults may migrate up to 2 kilometers (km) (1.2 miles (mi)) from summering to breeding sites. The distance from breeding sites may depend on local topography and vegetation, the distribution of ground squirrel or other rodent burrows, and climatic conditions (Stebbins 1989, Hunt 1998). In Santa Barbara County, juvenile California tiger salamanders have been trapped over 360 m (1,200 ft) while dispersing from their natal (birth) pond (Ted Mullen, Science Applications International Corporation (SAIC), personal communication, 1998), and adults have been found along roads over 2 km (1.2 mi) from breeding ponds (S. Sweet, in litt. 1998a). Migration is concentrated during a few rainy nights early in the winter, with males migrating before females (Twitty 1941; Shaffer, et al. 1993; Loredo and Van Vuren 1996; Trenham 1998b). Males usually remain in the ponds for an average of about 6 to 8 weeks, while females stay for approximately 1 to 2 weeks. In dry years, both sexes may stay for shorter periods (Loredo and Van Vuren 1996, Trenham 1998b). Although most marked salamanders have been recaptured at the pond where they were initially captured, in one study approximately 20 percent were recaptured at different ponds (Trenham 1998b). As with migration distances, the number of ponds used by an individual over its lifetime will be dependent on landscape features.

Female California tiger salamanders mate and lay their eggs singly or in small groups (Twitty 1941; Shaffer, et al. 1993). The number of eggs laid by a single female ranges from approximately 400 to 1,300 per breeding season (Trenham 1998b). The eggs typically are attached to vegetation near the edge of the breeding pond (Storer 1925, Twitty 1941), but in ponds with no or limited vegetation, they may be attached to objects (rocks, boards, etc.) on the bottom (Jennings and Hayes 1994). After breeding, adults leave the pond and typically return to small mammal burrows (Loredo et al. 1996; Trenham 1998a), although they may continue to come out nightly for approximately the next 2 weeks to feed (Shaffer, et al. 1993).

Eggs hatch in 10 to 14 days with newly hatched larvae ranging from 11.5 to 14.2 mm (0.45 to 0.56 in) in total length. Larvae feed on algae, small crustaceans, and mosquito larvae for about 6 weeks after hatching, when they switch to larger prey (P.R. Anderson 1968). Larger larvae have been known to consume smaller tadpoles of Pacific treefrogs (Hyla regilla) and California red-legged frogs (Rana aurora) as well as many aquatic insects and other aquatic invertebrates (J.D. Anderson 1968; P.R. Anderson 1968). Captive salamanders appear to locate food by vision and olfaction (smell) (J.D. Anderson 1968).

Amphibian larvae must grow to a critical minimum body size before they can metamorphose (change into a different physical form) to the terrestrial stage (Wilbur and Collins 1973). Feaver (1971) found that California tiger salamander larvae metamorphosed and left the breeding ponds 60 to 94 days after the eggs had been laid, with larvae developing faster in smaller, more rapidly drying ponds. The longer the ponding duration, the larger the larvae and metamorphosed juveniles are able to grow. The larger juvenile amphibians grow, the more likely they are to survive and reproduce (Semlitsch et al. 1988; Morey 1998).

In the late spring or early summer, before the ponds dry completely, metamorphosed juveniles leave the ponds and enter small mammal burrows after spending up to a few days in mud cracks or tunnels in moist soil near the water (Zeiner et al. 1988; Shaffer, et al. 1993; Loredo et al. 1996). Like the adults, juveniles may emerge from these retreats to feed during nights of high relative humidity (Storer 1925; Shaffer, et al. 1993) before settling in their selected estivation sites for the dry summer months.

Many of the pools California tiger salamanders lay eggs water is not retained water long enough to support successful metamorphosis. Generally, 10 weeks is required to allow sufficient time to metamorphose. The larvae will desiccate (dry out and perish) if a site dries before larvae complete metamorphosis (P.R. Anderson 1968, Feaver 1971). Pechmann et al. (1989) found a strong positive correlation with ponding duration and total number of metamorphosing juveniles in five salamander species. In one study, successful metamorphosis of California tiger salamanders occurred only in larger pools with longer ponding durations (Feaver 1971), which is typical range-wide (Jennings and Hayes 1994). Even though there is little difference in the number of pools used by salamanders between wet and dry years, pool duration is the most important factor to consider in relation to persistence and survival (Feaver 1971; Shaffer, et al. 1993; Seymour and Westphal 1994, 1995).

Lifetime reproductive success for California and other tiger salamanders is typically low, with fewer than 30 metamorphic juveniles per breeding female. While individuals may survive for more than 10 years, many may breed only once, and, in some populations, less than 5 percent of marked juveniles survive to become breeding adults (Trenham 1998b). With such low recruitment, isolated subpopulations can decline greatly from unusual, randomly occurring natural events as well as from human-caused factors that reduce breeding success and individual survival. Factors that repeatedly lower breeding success in isolated ponds that are too far from other ponds for migrating individuals to replenish the population can quickly drive a local population to extinction.

Historic and Current Distribution: The California tiger salamander inhabits low elevation, below 300 meters (m) (1000 feet (ft)), vernal pools and seasonal ponds and the associated coastal scrub, grassland, and oak savannah plant communities of the Santa Maria, Los Alamos, and Santa Rita Valleys in western Santa Barbara County (Shaffer, et al. 1993; Sam Sweet, University of California, Santa Barbara, in litt. 1993, 1998a). Although California tiger salamanders still exist across most of their historic range in Santa Barbara County, the habitat available to them has been reduced greatly. Ponds available to salamanders for breeding have been degraded and reduced in number. In addition, upland habitats inhabited by salamanders for most of their life cycle have been degraded and reduced in area through changes in agriculture practices, urbanization, building of roads and highways, chemical applications, and overgrazing (Gira et al. 1999; S. Sweet, in litt. 1993, 1998a,b).

Currently, California tiger salamanders in Santa Barbara County are found in four discrete regions (S. Sweet, in litt. 1998a). Collectively, salamanders in these regions constitute a single genetic population or DPS, reproductively separate from the rest of the California tiger salamanders (Jones 1993; Shaffer, et al. 1993; Shaffer and McKnight 1996). Ponds and associated uplands in southwestern (West Orcutt) and southeastern (Bradley-Dominion) Santa Maria Valley, Los Alamos Valley, and Santa Rita Valley constitute the four discrete regions or metapopulations where California tiger salamanders now exist in Santa Barbara County (S. Sweet, in litt. 1998a). For the purposes of this account, a metapopulation is defined as a group of subpopulations or "local populations" linked by genetic exchange. Of 14 known breeding sites or subpopulations within this DPS, 1 was destroyed in 1998, the upland habitat around 3 has been converted into more intensive agriculture practices (i.e. vineyards, gladiolus fields, and row crops, which may have eliminated the salamander subpopulations), 1 is surrounded by agriculture and urban development, 2 are affected by overgrazing, 4 are imminently threatened with conversion to vineyards or other intensive agriculture practices, and the remaining 3 are in areas rapidly undergoing conversion to vineyards and row crops (Sweet, et al. 1998; Sweet, in litt. 1998; Santa Barbara County Planning and Development 1998; Grace McLaughlin, Service, personal observations, 1998). Thus, only 6 or 7 of 13 existing ponds potentially provide breeding habitat for viable subpopulations of Santa Barbara County California tiger salamanders. Although other breeding ponds could exist within each of the four metapopulations noted above, searches around extant localities in the county, as well as in other areas with suitable habitat, have not identified additional subpopulations of the species (Paul Collins, Santa Barbara Museum of Natural History, in litt. 1998, pers. comm. 1999; S. Sweet, in litt. 1998a). Four possible breeding ponds or pond complexes (three in the Bradley-Dominion area, one in Santa Rita Valley) have been identified from aerial photography and by finding salamanders on roads in the vicinity (Sweet, et al. 1998) but have not been sampled. Most of the upland habitats around the ponds have been converted to vineyards or row crops within the last 6 years (Santa Barbara County Planning and Development 1998). All of the known and potential localities of the California tiger salamander in Santa Barbara County are on private lands, none are protected by conservation easements or agreements, and access is limited.

Reasons for Decline and Threats to Survival: The factors believed to responsible for the decline of the species are habitat loss due to conversion of natural habitat to intensive agriculture, urban development, habitat fragmentation, and agricultural contaminants.

Santa Cruz Long-Toed Salamander (*Ambystoma macrodactylum croceum*)

Species Description and Life History: The Santa Cruz long-toed salamander was listed on March 11, 1967 (32 **FR** 4001). At that time, only two breeding localities of the Santa Cruz long-toed salamander, Valencia Lagoon and Ellicott Slough, were known. A recovery plan was approved in 1977, and revised in 1985; currently the Service is working on another revision to the existing recovery plan.

The Santa Cruz long-toed salamander spends most of its life underground in small mammal burrows and along the root systems of plants in upland chaparral and woodland areas of coast live oak (*Quercus agrifolia)* or Monterey pine (*Pinus radiata*) as well as riparian strips of arroyo willows (*Salix lasiolepis*). These areas are desirable because they are protected from heat and the drying rays of the sun (Reed 1979, 1981). The breeding ponds are usually shallow, ephemeral, freshwater ponds. The breeding ponds at the Seascape, Larkin Valley, Calabasas, and Buena Vista sites are man-made. The extent of the upland habitat adjacent to the ponds varies from a ring of riparian vegetation on the perimeter of the pond to as far as a mile or more out from the pond (Ruth and Tollestrup 1973). However, examination of all currently available studies on the Santa Cruz long-toed salamander reveals that adult salamanders typically do not move more than 0.6 mile (straight line distance) from a breeding site.

Adult Santa Cruz long-toed salamanders leave their upland chaparral and woodland summer retreats with the onset of the rainy season in mid- to late-November or December and begin their annual nocturnal migration to the breeding pond (Anderson 1960). Adult salamanders migrate primarily on nights of rain, mist, or heavy fog (Anderson 1960, 1967; Ruth and Tollestrup 1973; Reed 1979, 1981). They arrive at the breeding pond from November through March, with most arriving in January and February (Anderson 1967, Reed 1979, Ruth 1988b). Peak breeding occurs during January and February because earlier rains are usually insufficient to fill the breeding ponds (Anderson 1967). Adult salamanders may skip breeding for one or more seasons if no surface water is present during drier years (Russell and Anderson 1956). Female Santa Cruz long-toed salamanders have specialized and selective egg-laying habits. Eggs are laid singly on submerged stalks of spike rush (*Eleocharis* sp.) or other vegetation about one inch apart (Anderson 1960, 1967). Free floating, unattached, and clustered eggs have also been observed (Reed 1981). Each female lays about 300 (range 215 to 411) eggs per year (Anderson 1967). After courtship and egg laying, most adult salamanders leave the pond in March or April and return to the same general areas where they spent the previous summer. Some adults may remain in the vicinity of the breeding site for a year or more before returning to more distant terrestrial retreats (Ruth 1988b). The eggs and the subsequent larvae are left unattended by the adults.

According to Reed (1979, 1981) and Ruth (1988a), eggs usually hatch after 15 to 30 days and enter the aquatic larval stage. The exact amount of time for development depends on water temperature (Anderson 1972). Larvae may metamorphose in a relatively short period of time if the pond environment becomes unsuitable (i.e., dries up, limited food source) for continued larval growth. However, a complex of factors determines the timing of metamorphosis in ambystomatid salamanders (Werner 1986, Wilbur and Collins 1973, Wilbur 1976, Smith-Gill and Berven 1979). Metamorphosis typically occurs from early May to mid-August (Anderson 1967, Reed 1979, 1981; Ruth 1988a). In closely related *A. talpoideum*, metamorphosis can be induced in the laboratory by starvation, pollution of the water, increased water temperatures, or drying of the aquatic habitat (Shoop 1960). If water is available to the larvae for a longer period of time, remaining in the pond may be advantageous for the juveniles. A larger body size at metamorphosis increases resistance to desiccation, makes the individual less vulnerable to predation, and increases the size range of food items that can be eaten (Werner 1986). As the pond begins to dry, the juvenile salamanders move at night and seek underground refuge at or near the pond (Reed 1979, 1981). During the next rainy seasons, these recently metamorphosed juveniles disperse farther away from the pond, not returning until they reach sexual maturity at two to three years (Ruth 1988a).

Adults of closely related *A. m. sigillatum* and *A. m. krausei* are known to have lived over six years in captivity (Snider and Bowler 1992) and ten years in the wild (Russell *et al.* 1995), respectively. An adult *A. m. croceum* confiscated by law enforcement officials was kept in captivity for eight years until its death (Stephen B. Ruth, Science Research and Consulting Services, Marina, California, *in litt.*). Thus, Santa Cruz long-toed salamanders are probably long-lived creatures, possibly living for a decade or more.

Santa Cruz long-toed salamanders are vulnerable to several predators including opossums (*Didelphis virginiana*), striped skunks (*Mephitis mephitis*), and ringneck snakes (*Diadophis punctatus*) (Reed 1979), raccoons (*Procyon lotor*), large California tiger salamanders *(A. californiense*), coast garter snakes (*Thamnophis atratus*), western terrestrial garter snakes (*T. elegans*), and common garter snakes (*T. sirtalis*). Larval *A. m. croceum* are parasitized by a digenetic trematode (Plagiorchiidae) which causes the creation of supernumerary limbs as well as other limb deformities (Sessions and Ruth 1990).

Foraging Ecology: The larvae of Santa Cruz long-toed salamanders subsist largely on aquatic invertebrates, other larval amphibians such as *Hyla regilla*, and conspecifics. Adults often forage for invertebrates, especially isopods (Anderson 1968), on the surface in and around breeding sites during the rainy season.

<u>Historic and Current Distribution</u>: Breeding of Santa Cruz long-toed salamanders have been documented at Valencia Lagoon, Ellicott pond, Seascape pond, Calabasas pond, Buena Vista pond, Green pond, and Rancho Road pond in Santa Cruz County and at McClusky Slough, Moro Cojo Slough, Bennett Slough, and Zmudowski pond in Monterey County. However, many of these sites have not been surveyed recently and may no longer support breeding populations.

Juvenile Santa Cruz long-toed salamanders have also been found at several other sites in Santa Cruz and Monterey counties (California Natural Diversity Data Base, unpubl. data). Whether any of these juveniles represent undiscovered breeding populations or merely wandering individuals from marginal or currently identified breeding habitats is unknown. Further discovery of new breeding sites is likely given the amount of privately owned habitat in the region that has not been surveyed for Santa Cruz long-toed salamanders.

Reasons for Decline and Threats to Survival: The very restricted and disjunct distribution of the Santa Cruz long-toed salamander has made the species particularly susceptible to population declines resulting from both human-associated and natural factors, including habitat loss and degradation, predation by introduced and native organisms, and weather conditions. Highway construction, urban and agricultural development, siltation, vehicles, exotic fish and vegetation, and saltwater intrusion are some of the perturbations affecting Santa Cruz long-toed salamander habitat. Runoff from adjacent agricultural and urban areas into many of the breeding ponds of the Santa Cruz long-toed salamander is a potential threat. Santa Cruz long-toed salamanders occur in several impaired water bodies.

California Freshwater Shrimp (*Syncaris pacifica*)

Species Description and Life History: The California freshwater shrimp was listed as endangered in 1988 (53 **FR** 43889). The California freshwater shrimp is a decapod crustacean of the family Atyidae. Females are generally larger and deeper bodied than males. Shrimp coloration is quite variable. Male shrimp are translucent to nearly transparent, with small surface and internal chromatophores (color-producing cells) clustered in a pattern to help disrupt their body outline and to maximize the illusion that they are submerged, decaying vegetation. Eng (1981) observed that the coloration of female range from a dark brown to a purple color. In some females, a broad tan dorsal band also may be present. Females may change rapidly from this very dark cryptic color to opaque with diffuse chromatophores, a distinctly different coloration. Undisturbed shrimp move slowly and are virtually invisible on submerged leaf and twig substrates, and among the fine, exposed, live roots of trees along undercut stream banks. Atyid shrimps can be separated from others based on the lengths of chelae (pincer-like claws) and presence of terminal setae (bristles) at the tips of the first and second chelae (Eng 1981, Pennak 1989). The presence of a short supraorbital (above the eye) spine on the carapace (body) and the angled articulation of the second chelae with the carpus (wrist) separate the California freshwater shrimp from other shrimp found in California.

Shrimp have been found only in low elevation (less than 16 meters) and low gradient (generally less than 1 percent) streams. With the exception of Yulupa Creek, shrimp have not been found in stream reaches with boulder and bedrock bottoms. In fact, high velocities and turbulent flows in such reaches may hinder upstream movement of shrimp. The California freshwater shrimp has evolved to survive a broad range of stream and water temperature conditions characteristic of small, perennial coastal streams. The shrimp appears to be able to tolerate warm water temperatures (greater than 23 degrees Celsius, 73 degrees Fahrenheit) and low flow conditions that are detrimental or fatal to native salmonids.

The shrimp are generally found in stream reaches where banks are structurally diverse with undercut banks, exposed roots, overhanging woody debris, or overhanging vegetation (Eng 1981, Serpa 1986 and 1991). Excellent habitat conditions for the shrimp involve streams 30 to 90 centimeters (cm) in depth with exposed live roots (e.g., alder and willow trees) along undercut banks (greater than 15 cm) with overhanging stream vegetation and vines (Serpa 1991). During the winter, the shrimp is found in undercut banks with exposed fine root systems or dense, overhanging vegetation. Such microhabitats may provide velocity refugia as well as some protection from high suspended sediment concentrations typically associated with high stream flows.

Habitat preferences apparently change during late-spring and summer months. Eng (1981) rarely found shrimp beneath undercut banks in the summer; submerged leafy branches were the preferred summer habitat. Highest concentrations of shrimp were in reaches with adjacent vegetation comprised of stinging nettles (*Urtica* sp.) grasses, vine maple (Serpa *in litt.* 1994 suspects periwinkle was misidentified as vine maple), and mint (*Mentha* sp.). None were caught from cattails (*Typha* sp.), cottonwood (*Populus fremontii*), or California laurel (*Umbellularia californica*). Serpa also noted that populations of shrimp were proportionately correlated with the quality of summer habitat provided by trailing terrestrial vegetation. However, during summer low flows, shrimp have been found in apparently poor habitat such as isolated pools with minimal cover. In such streams, opaque waters may allow shrimp to escape predation and persist in open pools despite the lack of cover (Serpa 1991).

Although largely absent from existing streams, large, complex organic debris dams may have been prevalent in streams supporting shrimp populations. These structures may have been important feeding and refugial sites for the shrimp. Such structures are known to collect detrital material (shrimp food) as well as leaf litter, which can be later broken down by microbial activity and invertebrates to finer, detrital material (Triska *et al.* 1982). In addition, debris dams may offer refugia during high flow events and reduce displacement of invertebrates (Covich *et al.* 1991).

Adult females produce relatively few eggs, generally, 50 to 120 (Hedgpeth 1968, Eng 1981). The eggs adhere to the pleopods (swimming legs on the abdomen) where they are protected and cared for during the winter incubation. The California freshwater shrimp is one of the few atyid species that breeds during the winter period.

California freshwater shrimp are preyed upon by fish, western pond turtles, salamanders, and newts, which are probably present throughout many of the streams. Invertebrate predators may include water scorpions, predaceous diving beetles, and dragonfly and damselfly nymphs.

Foraging Ecology: Atyid shrimps can be described as collectors feeding upon fine particulate organic matter. The food sources may range from fecal material produced by shredders (a functional group that feeds on coarse particulate organic matter), organic fines produced by physical abrasion and microbial maceration, senescent periphytic algae, planktonic algae, aquatic macrophyte plant fragments, zooplankton, and particles formed by the flocculation of dissolved organic matter. Shrimp observed on pool bottoms, submerged twigs, and vegetation seemed to feed on fine particulate matter (Eng 1981). Atyid shrimp use their claws to scrape and sweep detritus and small organisms from substrates. Much of the material ingested is probably indigestible cellulose. Shrimp may use visual, tactile, or chemical cues in foraging activities (USDI-FWS 1997a).

Historic and Current Distribution: Distribution of the shrimp is assumed, prior to human disturbances, to have been common in low elevation, perennial freshwater streams within Marin, Sonoma, and Napa counties. Today, the shrimp is found in 16 stream segments within these counties. The distribution of the shrimp can be separated into four general geographic regions: 1) tributary streams in the lower Russian River drainage which flows westward into the Pacific Ocean, 2) coastal streams flowing westward directly into the Pacific Ocean, 3) streams draining into a small coastal-embayment (Tomales Bay), and 4) streams flowing southward into northern San Pablo Bay. Many of these streams contain shrimp populations that are now isolated from each other. Distribution of shrimp populations within streams is not expected to be static because of habitat changes by natural or anthropogenic (man made) forces. Distribution within streams may expand and contact depending upon existing conditions. Gradual removal of unnatural barriers to shrimp dispersal and restoration of natural habitat conditions are expected to expand the distribution of shrimp beyond its existing occurrence.

Reasons for Decline and Threats to Survival: Existing populations of the California freshwater shrimp are threatened by introduced fish, deterioration or loss of habitat resulting from water diversion, impoundments, livestock and dairy activities, agricultural activities and developments, flood control activities, gravel mining, timber harvesting, migration barriers, and water pollution.

Fairy Shrimp (Including Conservancy, Longhorn, Riverside, San Diego, and Vernal Pool Fairy Shrimp)

Species Description and Life History: The Riverside fairy shrimp (*Streptocephalus woottoni*) was listed as endangered in 1993 (58 **FR** 41391). The vernal pool fairy shrimp (*Brachinecta lynchi*), conservancy fairy shrimp (*B. conservatio*), longhorn fairy shrimp (*B. longiatenna*), were listed as threatened (vernal pool) or endangered (all others) in 1994 (59 **FR** 48153). The San Diego fairy shrimp (*B. sandiegonensis*) was listed as endangered in 1997 (62 **FR** 4925). Further details on the life history and ecology of the fairy shrimp are provided by Eng *et al.* (1990) and Simovich *et al.* (1992)

Fairy shrimp have a delicate elongate body, large stalked compound eyes, no carapace, and 11 pairs of swimming legs. It swims or glides gracefully upside down by means of complex beating movements of the legs that pass in a wave-like anterior to posterior direction. The females carry the eggs in an oval or elongate ventral brood sac. The eggs are either dropped to the pool bottom or remain in the brood sac until the female dies and sinks. The "resting" or "summer" eggs are capable of withstanding heat, cold, and prolonged desiccation. When the pools fill in the same or subsequent seasons, some, but not all, of the eggs may hatch. The egg bank in the soil may consist of eggs from several years of breeding (Donald 1983). The eggs hatch when the vernal pools fill with rainwater. The early stages of the fairy shrimp develop rapidly into adults. These non-dormant populations often disappear early in the season long before the vernal pools dry up.

The primary historic dispersal method for the fairy shrimp likely was large scale flooding resulting from winter and spring rains which allowed the animals to colonize different individual vernal pools and other vernal pool complexes (J. King, pers. comm., 1995). This dispersal currently is non-functional due to the construction of dams, levees, and other flood control measures, and widespread urbanization within significant portions of the range of this species. Waterfowl and shorebirds likely are now the primary dispersal agents for fairy shrimp (Brusca, in. litt., 1992, King, in. litt., 1992, Simovich, in. litt., 1992). The eggs of these crustaceans are either ingested (Krapu 1974, Swanson *et al.* 1974, Driver 1981, Ahl 1991) and/or adhere to the legs and feathers where they are transported to new habitats.

Fairy shrimp are restricted to vernal pools/swales, an ephemeral freshwater habitat in California that forms in areas with Mediterranean climates where slight depressions become seasonally saturated or inundated following fall and winter rains. Due to local topography and geology, the pools are usually clustered into pool complexes (Holland and Jain 1988). In southern California, these pools/swales typically form on mesa tops or valley floors and are surrounded by very low hills, usually referred to as mima mounds (Zedler 1987). None of these listed branchiopods are known to occur in permanent bodies of water, riverine waters, or marine waters. Water remains in these pools/swales for a few months at a time, due to an impervious layer such as hardpan, claypan, or basalt beneath the soil surface.

The San Diego fairy shrimp is a habitat specialist found in small, shallow vernal pools, which range in depth from 5 to 30 centimeters (cm) (2 to 12 in.) and in water temperature from 10 to 20 degrees Celsius (C)(50 to 68 degrees Fahrenheit (F)) (Simovich and Fugate 1992, Hathaway and Simovich undated). Water chemistry is one of the most important factors in determining the distribution of fairy shrimp (Belk 1977, Branchiopod Research Group 1996). The San Diego fairy shrimp appears to be sensitive to high water temperatures (Branchiopod Research Group 1996). Hathaway and Simovich (undated) presented data indicating that pools located in the inland mountain and desert regions may be too cool (below 5 degrees C (41 degrees F)) or too warm (above 30 degrees C (86 degrees F)) for this species. Adult San Diego fairy shrimp are usually observed from January to March; however, in years with early or late rainfall, the hatching period may be extended.

The vernal pool fairy shrimp inhabits vernal pools with clear to tea-colored water, most commonly in grass or mud-bottomed swales, or basalt flow depression pools in unplowed grasslands, but one population occurs in sandstone rock outcrops and another population in alkaline vernal pools. The vernal pool fairy shrimp has been collected from early December to early May. It can mature quickly, allowing populations to persist in short-lived shallow pools (Simovich *et al.* 1992).

The genetic characteristics of these species, as well as ecological conditions, such as watershed continuity, indicate that populations of these animals are defined by pool complexes rather than by individual vernal pools (Fugate 1992; J. King, pers. comm., 1995). Therefore, the most accurate indication of the distribution and abundance of these species is the number of inhabited vernal pool complexes. Individual vernal pools occupied by these species are most appropriately referred to as subpopulations. The pools and, in some cases, pool complexes supporting these species are usually small.

Foraging Ecology: Fairy shrimp feed on algae, bacteria, protozoa, rotifers, and bits of detritus.

<u>Historic</u> <u>and</u> <u>Current</u> <u>Distribution</u>: These crustaceans are restricted to vernal pools and swales in California. Holland (1978) estimated that between 67 and 88 percent of the area within the Central Valley of California which once supported vernal pools had been destroyed by 1973. However, an analysis of this report by the Service revealed apparent arithmetic errors which resulted in a determination that a historic loss between 60 and 85 percent may be more accurate. Regardless, in the ensuing 23 years, threats to this habitat type have continued and resulted in a substantial amount of vernal pool habitat being converted for human uses in spite of Federal regulations implemented to protect wetlands. For example, the Corps' Sacramento District has authorized the filling of 189 hectares (467 acres) of wetlands between 1987 and 1992 pursuant to Nationwide Permit 26 (USDI-FWS 1992). The Service estimates that a majority of these wetland losses within the Central Valley involved vernal pools. Current rapid urbanization and agricultural conversion throughout the ranges of the species continue to pose the most severe threats to the continued existence of the fairy shrimp. The Corps' Sacramento District has several thousand vernal pools under its jurisdiction (Coe 1988), which includes most of the known populations of the vernal pool fairy shrimp. It is estimated that within 20 years 60 to 70 percent of these pools will be destroyed by human activities (Coe 1988).

Conservancy Fairy Shrimp (Endangered): The Conservancy fairy shrimp inhabits vernal pools with highly turbid water. The species is known from six disjunct populations: Vina Plains, north of Chico, Tehama County; south of Chico, Butte County; Jepson Prairie, Solano County; Sacramento National Wildlife Refuge, Glenn County; near Haystack Mountain northeast of Merced in Merced County; and the Lockwood Valley of northern Ventura County.

Longhorn Fairy Shrimp (Endangered): The longhorn fairy shrimp inhabits clear to turbid grass-bottomed vernal pools in grasslands and clear-water pools in sandstone depressions. This species is known only from four disjunct populations along the eastern margin of the central coast range from Concord, Contra Costa County south to Soda Lake in San Luis Obispo County: the Kellogg Creek watershed, the Altamont Pass area, the western and northern boundaries of Soda Lake on the Carrizo Plain, and Kesterson National Wildlife Refuge in the San Joaquin Valley.

Riverside Fairy Shrimp (Endangered): The Riverside fairy shrimp has a restricted distribution and is known only from vernal pools in the Santa Rosa Plateau, Skunk Hollow, and several small scattered pools in Riverside County; from El Toro Marine Cavalry Air Station and Saddleback Meadows in Orange County; from Otay Mesa, Camp Pendleton, and Miramar Naval Air Station in San Diego County; from the Moorpark area of Ventura County; and the Canyon Country/Santa Clarita area of Los Angeles County.

San Diego Fairy Shrimp (Endangered): The San Diego fairy shrimp belongs to the Family Branchinectidae. These fairy shrimp have a very restricted distribution and are only known from vernal pools in southwestern coastal California and extreme northwestern Baja California, Mexico. Less than 81 hectares (ha) (200 acres (ac)) of habitat likely remains.

No individuals have been found in riverine waters, marine waters, or other permanent bodies of water. All known localities are below 700 meters (m) (2,300 feet (ft)) and within 65 kilometers (km) (40 miles (mi)) of the Pacific Ocean, from Santa Barbara County south to northwestern Baja California. The majority of the vernal pools in this region, including many which likely served as habitat for the species, were destroyed prior to 1990. Between 1979 and 1986, approximately 68 percent of the privately owned vernal pools under the City of San Diego's jurisdiction were destroyed (Wier and Bauder 1991).

Vernal Pool Fairy Shrimp (Threatened): The vernal pool fairy shrimp inhabits vernal pools with clear to tea-colored water, most commonly in grass or mud bottomed swales, or basalt flow depression pools in unplowed grasslands. The vernal pool fairy shrimp has been collected from early December to early May. The vernal pool fairy shrimp is known from 34 populations extending from Stillwater Plain in Shasta County through most of the length of the Central Valley to Pixley in Tulare County, and along the central coast range from northern Solano County to Pinnacles in San Benito County (Eng *et al.* 1990, Fugate 1992, Sugnet and Associates 1993). In wet years, Fort Hunter Liggett, in southern Monterey County, supports hundreds of pools containing this species. Camp Roberts, which straddles the Monterey-San Luis Obispo county line, also contains pools with vernal pool fairy shrimp. Four additional, disjunct populations exist: one near Soda Lake in San Luis Obispo County; one in the mountain grasslands of northern Santa Barbara County; one on the Santa Rosa Plateau in Riverside County, and one near Rancho California in Riverside County. Three of these four isolated populations each contain only a single pool known to be occupied by the vernal pool fairy shrimp.
Conservancy Fairy Shrimp (Endangered): The Conservancy fairy shrimp inhabits vernal pools with highly turbid water. The species is known from six disjunct populations: Vina Plains, north of Chico, Tehama County; south of Chico, Butte County; Jepson Prairie, Solano County; Sacramento National Wildlife Refuge, Glenn County; near Haystack Mountain northeast of Merced in Merced County; and the Lockewood Valley of northern Ventura County.

<u>Reasons for Decline and Threats to Survival</u>: Fairy shrimp are imperiled by a variety of human-caused activities, primarily urban development, water supply/flood control projects, and land conversion for agricultural use. Habitat loss occurs from direct destruction and modification

of pools due to filling, grading, discing, leveling, and other activities, as well as modification of surrounding uplands which alters vernal pool watersheds. Other activities which adversely affect these species include off-road vehicle use, certain mosquito abatement measures, and pesticide/herbicide use, alterations of vernal pool hydrology, fertilizer and pesticide contamination, activity, invasions of aggressive non-native plants, gravel mining, and contaminated stormwater runoff.

In addition to direct habitat loss, the vernal pool habitat for the vernal pool fairy shrimp also has been and continues to be highly fragmented throughout their ranges due to conversion of natural habitat for urban and agricultural uses. This fragmentation results in small isolated vernal pool fairy shrimp populations. Ecological theory predicts that such populations will be highly susceptible to extirpation due to chance events, inbreeding depression, or additional environmental disturbance (Gilpin and Soule 1986, Goodman 1987a,b). Should an extirpation event occur in a population that has been fragmented, the opportunities for recolonization would be greatly reduced due to physical (geographical) isolation from other (source) populations.

Only a small proportion of the habitat of these species is protected from these threats. State and local laws and regulations have not been passed to protect these species, and other regulatory mechanisms necessary for the conservation of the habitat of these species have proven ineffective.

Shasta Crayfish (*Pacifastacus fortis*)

Species Description and Life History: The Shasta crayfish was federally listed as endangered in 1988 (53 **FR** 190). A detailed account of the taxonomy, ecology, and biology of the Shasta crayfish is presented in the Draft Recovery Plan for this species (USDI-FWS 1997). Supplemental information is provided below.

The Shasta crayfish occurs in cool, clear, spring-fed lakes, rivers and streams, usually at or near a spring inflow source, where waters show relatively little annual fluctuation in temperature and remain cool during the summer. Most Shasta crayfish are found in still and slowly to moderately flowing waters. Although Shasta crayfish have been observed in groups under large rocks situated on clean, firm sand or gravel substrates (Bouchard, 1978; Eng and Daniels, 1982), they also have been observed on a fine, probably organic, material 1-3 centimeters thick on the bottom of Crystal Lake. Shasta crayfish is most abundant where plants are absent. The most important habitat requirement appears to be the presence of adequate volcanic rock rubble to provide escape cover from predators.

Foraging Ecology: Although the food habits of the Shasta crayfish are not well known, the morphology of the mouthparts suggests that the species relies primarily on predation, browsing on encrusting organisms, and grazing on detritus to obtain food. Aquatic invertebrates and dead fish probably provide food for the Shasta crayfish. Feeding and mating takes place at night.

Historic and Current Distribution: The Shasta crayfish is found only in Shasta County, California, in the Pit River drainage and two tributary systems, Fall River and Hat Creek subdrainages. In the Fall River subdrainage, populations occur in the Tule and Fall Rivers, Big Lake, Spring, Squaw and Lava Creeks, and in Crystal and Rainbow Springs. An additional population occurs in Sucker Spring Creek, a tributary of the Pit River just downstream from Powerhouse I, which lies between the two subdrainages (Bouchard, 1978; Eng and Daniels, 1982). In the Hat Creek subdrainage, historically, populations have been found in Lost Creek, Crystal, Baum, and Rising River Lakes. The populations in Lake Britton, Burney, Clark, Kosk, Goose, Lost, and Rock Creeks were extirpated prior to 1974 (Bouchard, 1977). Since 1978 the Shasta crayfish has been extirpated from Crystal Lake, Baum Lake and Spring Creek near its confluence with the Pit River, Rising River and Sucker Spring Creek near Pit Powerhouse I (McGriff, personal communication, 1986).

Reasons for Decline and Threats to Survival: The invasion of non-native crayfish species, in particular the signal crayfish, is the single largest threat to the continued existence of the Shasta crayfish. Human activities (such as levee repairs) in the historic range of the Shasta crayfish caused increased siltation, covering the volcanic rubble and reducing the amount of suitable habitat for the species. Two entire populations have been extirpated since 1978.

Vernal Pool Tadpole Shrimp (*Lepidurus packardi*)

Species Description and Life History: The vernal pool tadpole shrimp was listed as endangered on September 19, 1994 (59 **FR** 48153). Further details on the life history and ecology of the fairy shrimp are provided by Eng *et al.* (1990) and Simovich *et al.* (1992).

The vernal pool tadpole shrimp has dorsal compound eyes, a large shield-like carapace that covers most of the body, and a pair of long cercopods at the end of the last abdominal segment (Linder 1952, Longhurst 1955, Pennak 1989). It is primarily a benthic animal that swims with its legs down. Tadpole shrimp climb or scramble over objects, as well as move along or in bottom sediments. The females deposit their eggs on vegetation and other objects on the pool bottom. Tadpole shrimp populations pass the dry summer months as diapaused eggs in pool sediments. Some of the eggs hatch as the vernal pools are filled with rainwater in the fall and winter of subsequent seasons.

The life history of the vernal pool tadpole shrimp is linked to the phenology of its vernal pool habitat. After winter rainwater fills the pools, the populations are reestablished from diapaused eggs which lie dormant in the dry pool sediments (Lanaway 1974, Ahl 1991). Ahl (1991) found that eggs in one pool hatched within three weeks of inundation and sexual maturation was reached in another three to four weeks. The eggs are sticky and readily adhere to plant matter and sediment particles (Simovich *et al.* 1992). A portion of the eggs hatch immediately and the rest enter diapause and remain in the soil to hatch during later rainy seasons (Ahl 1991). The vernal pool tadpole shrimp matures slowly and is a long-lived species (Ahl 1991). Adults are often present and reproductive until the pools dry up in the spring (Ahl 1991, Simovich *et al.* 1992).

The genetic characteristics of this species, as well as ecological conditions, such as watershed continuity, indicate that populations of these animals are defined by pool complexes rather than by individual vernal pools (Fugate 1992; J. King, pers. comm., 1995). Therefore, the most accurate indication of the distribution and abundance of the species is the number of inhabited vernal pool complexes. Individual vernal pools occupied by the species are most appropriately referred to as subpopulations. The pools and, in some cases, pool complexes supporting these species are usually small.

The primary historic dispersal method for the vernal pool tadpole shrimp and likely was large scale flooding resulting from winter and spring rains which allowed the animals to colonize different individual vernal pools and other vernal pool complexes (J. King, pers. comm., 1995). This dispersal currently is non-functional due to the construction of dams, levees, and other flood control measures, and widespread urbanization within significant portions of the range of this species. Waterfowl and shorebirds likely are now the primary dispersal agents for vernal pool tadpole shrimp (Brusca, in. litt., 1992, King, in. litt., 1992, Simovich, in. litt., 1992). The eggs of these crustaceans are either ingested (Krapu 1974, Swanson *et al.* 1974, Driver 1981, Ahl 1991) and/or adhere to the legs and feathers where they are transported to new habitats.

Vernal pool tadpole shrimp are restricted to vernal pools/swales, an ephemeral freshwater habitat in California that forms in areas with Mediterranean climates where slight depressions become seasonally saturated or inundated following fall and winter rains. Due to local topography and geology, the pools are usually clustered into pool complexes (Holland and Jain 1988). Tadpole shrimp are not known to occur in permanent bodies of water, riverine waters, or marine waters. Water remains in these pools/swales for a few months at a time, due to an impervious layer such as hardpan, claypan, or basalt beneath the soil surface.

Foraging Ecology: The diet of tadpole shrimp consists of organic detritus and living organisms, such as fairy shrimp and other invertebrates (Pennak 1989).

Historic and Current Distribution: Holland (1978) estimated that between 67 and 88 percent of the area within the Central Valley of California which once supported vernal pools had been destroyed by 1973. However, an analysis of this report by the Service revealed apparent arithmetic errors which resulted in a determination that a historic loss between 60 and 85 percent may be more accurate. Regardless, in the ensuing 23 years, threats to this habitat type have continued and resulted in a substantial amount of vernal pool habitat being converted for human uses in spite of Federal regulations implemented to protect wetlands. For example, the Corps' Sacramento District has authorized the filling of 189 hectares (467 acres) of wetlands between 1987 and 1992 pursuant to Nationwide Permit 26 (USDI-FWS 1992). The Service estimates that a majority of these wetland losses within the Central Valley involved vernal pools, the endemic habitat of the vernal pool tadpole shrimp and vernal pool fairy shrimp. Current rapid urbanization and agricultural conversion throughout the ranges of these two species continue to pose the most severe threats to the continued existence of the vernal pool tadpole shrimp and vernal pool fairy shrimp. The Corps' Sacramento District has several thousand vernal pools under its jurisdiction (Coe 1988), which includes most of the known populations of these listed species. It is estimated that within 20 years 60 to 70 percent of these pools will be destroyed by human activities (Coe 1988).

The vernal pool tadpole shrimp is known from 19 populations in the Central Valley, ranging from east of Redding in Shasta County south to Fresno County, and from a single vernal pool complex located on the San Francisco Bay National Wildlife Refuge in Alameda County. It inhabits vernal pools containing clear to highly turbid water, ranging in size from 5 square meters (54 square feet) in the Mather Air Force Base area of Sacramento County, to the 36-hectare (89-acre) Olcott Lake at Jepson Prairie in Solano County. Vernal pools at Jepson Prairie and Vina Plains (Tehama Co.) have a neutral pH, and very low conductivity, total dissolved solids, and alkalinity (Barclay and Knight 1984, Eng *et al.* 1990). These pools are located most commonly in grass-bottomed swales of grasslands in old alluvial soils underlain by hardpan or in mud-bottomed claypan pools containing highly turbid water.

<u>Reasons for Decline and Threats to Survival</u>: Fairy shrimp are imperiled by a variety of human-caused activities, primarily urban development, water supply/flood control projects, and land conversion for agricultural use. Habitat loss occurs from direct destruction and modification of pools due to filling, grading, discing, leveling, and other activities, as well as modification of surrounding uplands which alters vernal pool watersheds. Other activities which adversely affect these species include off-road vehicle use, certain mosquito abatement measures, and pesticide/herbicide use, alterations of vernal pool hydrology, fertilizer and pesticide contamination, activity, invasions of aggressive non-native plants, gravel mining, and contaminated stormwater runoff.

In addition to direct habitat loss, the vernal pool habitat for the vernal pool fairy shrimp also has been and continues to be highly fragmented throughout their ranges due to conversion of natural habitat for urban and agricultural uses. This fragmentation results in small isolated vernal pool fairy shrimp populations. Ecological theory predicts that such populations will be highly susceptible to extirpation due to chance events, inbreeding depression, or additional environmental disturbance (Gilpin and Soule 1986, Goodman 1987a,b). Should an extirpation event occur in a population that has been fragmented, the opportunities for recolonization would be greatly reduced due to physical (geographical) isolation from other (source) populations.

Only a small proportion of the habitat of these species is protected from these threats. State and local laws and regulations have not been passed to protect these species, and other regulatory mechanisms necessary for the conservation of the habitat of these species have proven ineffective.

Southern Sea Otter (*Enhydra lutris nereis*)

<u>Species Description and Life History</u>: The southern sea otter was listed as threatened in 1977 (42 **FR** 2968). Sea otters are one of the largest members of the family Mustelidae. Adult males are larger than adult females. Standard lengths of adult males and females average 51 inches and 47

inches, respectively, with males averaging 64 pounds and females averaging 44 pounds. Pups weigh between 3 to 5 pounds at birth. This account is based on information in Bonnell *et al.* 1983, and Costa & Kooyman 1980, 1982.

Unlike most other marine mammals, sea otters have very little subcutaneous fat, depending instead on their clean, dense, water-resistant fur for insulation against the cold. Contamination of the fur by oily substances can destroy the insulating properties of the fur and lead to hypothermia and death.

Although mating and pupping take place throughout the year, a peak period of pupping occurs from January to March. The general yearly reproductive pattern consists of a winter-spring pupping season and a summer-fall breeding season. Males may reach sexual maturity at about 5 years of age; however males probably do not establish territories or actively participate in breeding for some time after reaching puberty. Preliminary observations indicate that female southern sea otters may also reach sexual maturity between 4 and 5 years of age. Current estimates indicate that most adult females give birth to one pup each year, with a reproductive cycle ranging from 11-14 months in length. Gestation periods have been estimated at 4-6 months. Pup dependency periods in California range from 5-8 months. There appears to be a potential for considerable individual variation and plasticity with respect to the temporal phases of the reproductive cycle.

Foraging Ecology: Otters forage in both rocky and soft-sediment communities as well as in the kelp understory and canopy. Foraging occurs in both the intertidal and subtidal zones, but seldom deeper than 25 meters. The diet of sea otters is almost exclusively of a variety of nearshore macroinvertebrates. Prey items include abalones, rock crabs, sea urchins, kelp crabs, clams, turban snails, mussels, octopus, barnacles, scallops, sea stars, and chitons. Sea otter teeth are adapted for crushing hard-shelled macro-invertebrates.

<u>Historic and Current Distribution</u>: Southern sea otters inhabit a narrow zone of shallow, littoral waters along the counties of Santa Cruz, Monterey, San Luis Obispo, and Santa Barbara. A reintroduced colony is located on San Nicolas Island, Ventura county. The majority of otters remain within 1.2 miles of shore, inshore of the outer kelp bed edge, which generally corresponds to the 60-foot (10 fathom) depth curve. However, some individuals may be found further off shore to the 30 fathom depth curve. Foraging activity is generally restricted to water depth of 90 feet (15 fathoms) or less. Southern sea otters are primarily associated with subtidal habitats characterized by rocky, creviced substrate, although they are also found in sandy substrate areas. Sea otter density within most of the range (with the exception of the north and south population fronts) is related to substrate type; rocky bottom habitats support an average density of 13 otters per square mile whereas sandy bottom areas support an average of 2 otters per square mile.

The number of southern sea otters increased to 2,377 in 1995, but has since declined to 2,229 in 1997. The Service is currently assessing whether this lower count represents an actual decline or an artifact of survey technique and a redistribution of southern sea otters.

Reasons for Decline and Threats to Survival: Threats to the survival of the southern sea otter include reduced population size, increased tanker traffic, oil spills, drowning in commercial fishing nets, municipal pollution, and increased harassment caused by increased use of near-shore areas. Some evidence suggests that the decline in population growth rate is due to infectious disease.

Elevated levels of heavy metals, chlorinated hydrocarbons, PCB's, and petroleum hydrocarbons were found in sea otters in the past. Chemical contamination may also reduce suitable foraging areas (USDI-FWS 1981).

Elevated levels of mercury are known to occur in Elkhorn Slough, a tributary to Monterey Bay. Elkhorn Slough is impacted by upstream discharges of mercury. Livers collected from sea otters found dead at this location had a maximum mercury concentration of (60mg/kg) (Mark Stephenson pers comm 1998). Wren, 1986 suggested normal mercury concentrations in river otter livers were 4 mg/kg (ppm). O'Conner and Nielsen (1981) found that length of exposure was a better predictor of tissue residue level than dose in otters but higher doses produced an earlier onset of clinical signs. Acute mercury poisoning in mammals is primarily manifested in Central Nervous System damage, sensory and motor deficits, and behavioral impairment. Animals initially become anorexic and lethargic. A dose of 0.09 mg/kg body weight (2 ppm in diet) for 181 days was enough to produce anorexia and ataxia in two of three otters (Lutra candensis. Associated liver residues were 32.6 mg/kg (O'Conner and Nielsen 1981). Muscle ataxia, motor control deficits, and visual impairment develop as toxicity progresses with convulsions preceding death. River otters fed 8 ppm died within a mean time of 54 days. Associated liver concentrations were 32.3 mg/kg (ppm) (O'Conner and Nielsen 1981). Smaller carnivores are more sensitive to methylmercury toxicity than larger species as reflected in shorter times of onset of toxic signs and time to death.

DIRECT AND INDIRECT EFFECTS OF THE PROPOSED ACTION

For the purposes of this opinion the Services have conducted their effects analysis based on the potential for the numeric criteria to result in effects to the aquatic ecosystem and the species that are dependent on its function for their survival and recovery. While 126 priority pollutants are addressed within the CTR, the Services have focused upon the numeric criteria for selenium, mercury, pentachlorophenol, cadmium and formula based criteria for metals on a dissolved basis as the most problematic for listed species and critical habitat. The Services have prepared this analysis of criteria for priority pollutants based on: (1) the adequacy of the proposed aquatic life criteria, including the necessity of wildlife criteria where aquatic life criteria are not sufficiently protective of wildlife; (2) the toxic effects to listed species or surrogates which may occur at proposed criteria concentrations; (3) the bioaccumulative nature of the priority pollutants at issue; and (4) the potential for interactive effects of pollutants at the proposed criteria concentrations. In some cases, such as mercury, if the aquatic life criteria were not protective and the human health criteria were lower, the adequacy of the human health numeric criteria to protect aquatic life was also considered.

Our analysis of criteria assessed whether there was the potential for toxicity that would affect listed species to occur at concentrations at or below the proposed criteria concentrations in water. EPA has stipulated that the promulgation of the CTR is solely for the purpose of providing the State of California with criteria. Although the Services recognize that criteria are sometimes not met within some California waterbodies and that implementation and enforcement issues also determine the degree of protection, the analysis within this opinion assesses the degree of protection likely to be afforded to listed species by the CTR if concentrations of toxic pollutants allowable by the proposed CTR are achieved. While EPA has not specifically proposed any wildlife criteria as part of the CTR, the Services are required to evaluate the degree of protection afforded to listed wildlife species by the proposed criteria for all California waterbodies.

The Services have evaluated the effects of the proposed action based on the assumptions that: (1) the proposed numeric criteria will apply throughout the geographic distribution of the species; and (2) the ambient concentrations of constituents could rise to the concentrations allowed by the numeric criteria proposed by EPA. Included in these findings are the Services' analysis of the demonstrated potential for adverse effects to occur to species at or below the proposed criteria concentrations, the likelihood of these problematic concentrations being achieved within the range of the species, and the degree to which these adverse effects will impact the species' environmental baseline.

The Services in the development of this final biological opinion have used the same rationale for evaluating effect thresholds of criteria as previously presented in our April 10, 1998, and April 9, 1999, draft biological opinions. That rationale is presented in the "Consultation History" section of this document. The Services based the following effects section on EPA's August 5, 1997, proposed CTR. Since that time EPA has modified the proposed action as presented in EPA's December 16, 1999, letter to the Services, and memorialized in the "Description of the Proposed Action" section of this document. The subsequent conclusions contained in this document are contingent on EPA's implementation of these modifications.

Selenium

<u>Assessment of Adequacy of Proposed Selenium Criteria to protect listed species</u>

Chronic Aquatic Life Criterion for Selenium

The Services find that the chronic aquatic life criterion for selenium proposed in the CTR does not protect listed fish and wildlife dependent on the aquatic ecosystem for development and/or foraging. The Federal Register notice for the proposed rule (EPA 1997c) states that the chronic criterion of 5 µg/L for selenium (derived in 1987) continues to be scientifically valid and protective of aquatic life. However, nearly every major review of experimental and field data conducted over the past decade has concluded that a chronic criterion of 5 µg/L is not fully protective of fish and wildlife resources. The list of scientific reviews known to the Service that contradict the 5 µg/L chronic criterion includes: Lemly and Smith (1987), Davis *et al.* 1988, Lillebo *et al*. (1988), UC Committee (1988), DuBowy (1989), Johns 1989, Lemly 1989, U.S. Dept. of Interior and California Resources Agency (1990), Sorensen (1991), Environment Canada (1991), Pease *et al*. (1992), Peterson and Nebeker (1992), CH2M HILL *et al*. (1993), Emans et al. 1993, Lemly (1993a), Lemly (1993b), CAST (1994), Gober (1994), Maier and Knight (1994), New Mexico (1994), California Regional Water Board (1995), Lemly (1995), Seiler and Skorupa (1995), California Regional Water Board (1996), Lemly (1996a), Lemly (1996b), Ohlendorf (1996), Roux et al. (1996), Skorupa *et al*. (1996), Van Derveer and Canton (1997), Engberg *et al*. (1998), Skorupa (1998), Naftz and Jarman (1998), Stephens and Waddell (1998), Adams *et al*. (1998), Seiler and Skorupa (In Press), and Hamilton and Lemly, 1999. Each of these reviews, incorporates the findings from numerous individual studies, for example, Skorupa *et al*. (1996) cite results from about 200 individual studies. In aggregate, the weight of scientific evidence supporting a chronic criterion for selenium of $\leq$ 2 µg/L is now overwhelming.

As early as 1991, the evidence available in the scientific literature was sufficient for Canada to issue a national water quality guideline stipulating that the concentration of total selenium should not exceed 1 µg/L (Environment Canada 1991). Based on data collected by the U.S. Department of Interior's National Irrigation Water Quality Program (NIWQP) from 26 study areas in 14 western states (including 5 California study areas), a 5 µg/L chronic criterion for selenium is only 50-70 percent protective (Adams *et al*. 1998; Seiler and Skorupa, In Press), as opposed to the 95 percent level of protection that EPA's national water quality criteria are intended to achieve (Stephan *et al*. 1984). The Service believes the NIWQP data suggest that on a dissolved basis a criterion of 1 µg/L would be required to achieve 95 percent protection, which is approximately equivalent to a 2 µg/L criterion on a total recoverable basis (Peterson and Nebeker 1992).

Acute Aquatic Life Criterion for Selenium

The Services find that the speciation-weighted acute criterion for selenium proposed in the CTR does not protect listed fish and wildlife dependent on the aquatic ecosystem for development and/or foraging. The EPA proposed changing the acute criterion for selenium from 20 µg/L (total recoverable) to a speciation-weighted criterion based on the relative concentrations of selenite, selenate, and all other forms of selenium found in a particular water body. Depending on the specific water body in question, this proposed acute criterion for selenium could range from 12.8 µg/L (if 100 percent selenate were present) to 185.9 µg/L (if 100 percent selenite were present). A 20 µg/L (total recoverable) acute site-specific criterion was promulgated in the NTR (and would not be changed by the CTR) and applies to the following water bodies in California: San Francisco Bay upstream to and including Suisun Bay, Sacramento-San Joaquin Delta, Mud Slough, Salt Slough, San Joaquin River, and Sack Dam to the mouth of the Merced River. The Services believe that the promulgation of the proposed speciation weighted acute criterion for selenium in the CTR would not afford adequate protection to listed species because: (1) selenium bioaccumulates rapidly in aquatic organisms and a single pulse of selenium ($\geq$10 µg/L) into aquatic ecosystems could have lasting ramifications, including elevated selenium concentrations in aquatic food webs (Maier *et al*. (in press); Hansen's Biological Consulting et al. 1997a, 1997b, 1998; Hanson et al. 1996; Tulare Lake Drainage District 1996); (2) EPA's speciation-weighted criterion assumes that selenate is more toxic than selenite, which is the reverse of what has been found in most acute selenium toxicity studies; (3) and the site-specific criterion of 20 µg/L promulgated in the NTR may fail to adequately protect aquatic-dependent fish and wildlife (Lemly 1997; Maier *et al*. 1998; Hansen's Biological Consulting et al. 1997a, 1997b, 1998; Hanson et al. 1996; Tulare Lake Drainage District 1996). For example, in February 1995, the Tulare Lake Drainage District established a flow-thru compensation wetland. Although the water supplied to the wetland was generally managed to keep its selenium content at or below about 2-3 µg/L, a pulse of 23 µg/L was documented on March 29, 1995 (Tulare Lake Drainage District 1996; Hanson et al. 1996). Three months later (June 20, 1995), and without any additional selenium pulses, avian eggs sampled at the site contained up to 6.2 µg/g Se which exceeds the embryotoxic risk threshold reported in Skorupa (1998). In June 1995, 12% of sampled eggs exceeded 6 µg/g Se which very plausibly may have been linked to the late March pulse of 23 µg/L Se that passed through the system. Additional support for a "pulse-effect" hypothesis, is provided by monitoring data for 1996-1998. In each of those three years, water supplied to the wetland was never documented to exceed 2.8 to 4.2 µg/L Se, and in all three years, in the absence of a $\geq$ 10 µg/L Se pulse, none of the avian eggs collected at the site exceeded the embryotoxicity threshold of 6 µg/g Se (Hansen's Biological Consulting et al. 1997a, 1997b, 1998).

The Services believe the acute toxicity data that were reviewed and compiled in Maier *et al.* (1987), Lillebo *et al.* (1988), Moore *et al.* (1990), and Skorupa *et al.* (1996) should be incorporated by EPA into the database that is employed for deriving a speciation-weighted acute criterion. These sources, and field studies (cf. Skorupa 1998), unanimously indicate that a lower criterion is warranted for selenite-dominated waters than for selenate-dominated waters (the reverse of the currently proposed weighting formula). Canton (1996) suggested that EPA's erroneous acute toxicity weighting of selenate versus selenite is the result of the influence of unusual outlier data for one taxon, *Gammarus*, and the small data base for acute toxicity testing of selenate. This suggests that only strictly matched comparative data should be used to derive a speciation-weighted acute criterion for selenium.

<u>Hazards of Selenium</u>

Selenium Sources

Selenium, a semi-metallic trace element with biochemical properties very similar to sulfur, is widely distributed in the earth's crust, usually at trace concentrations (<1 µg/g, ppm; e.g., Wilber 1980; Eisler 1985). Some geologic formations, however, are particularly seleniferous (e.g., Presser and Ohlendorf 1987; Presser 1994; Presser et al. 1994; Piper and Medrano 1994; Seiler 1997; Presser and Piper 1998) and when disturbed by anthropogenic activity provide pathways for accelerated mobilization of selenium into aquatic ecosystems. Abnormally high mass-loading of selenium into aquatic environments is most typically associated with the use of fossil fuels, with intensive irrigation and over-grazing of arid lands, and with mining of sulfide ores (Skorupa 1998). Intensive confined livestock production facilities and municipal wastewater treatment plants may also contribute to accelerated mass-loading of selenium into surface water bodies.

The use of fossil fuels can result in accelerated mass-loading of selenium into aquatic environments via the leaching of coal-mining spoils and/or overburden, via disposal of process wastewater from oil refineries, via downwind drift and deposition from industrial-scale coal combustion, and via aquatic disposal and/or leaching of fly ash from coal-fired electric power plants (Lemly 1985; Skorupa 1998). Agricultural irrigation over large areas of the western United States causes accelerated leaching of selenium from soils into groundwater. Natural and anthropogenic discharge of subsurface agricultural drainage water to surface waters is a major pathway for the mass-loading of selenium into aquatic ecosystems (Presser et al. 1994; Presser 1994; Seiler 1997; Presser and Piper 1998; Skorupa 1998). Overgrazing of high-gradient watersheds can cause accelerated erosion of seleniferous soils and detrital litter into surface waters, but no case studies of this pathway have been systematically documented. Mining of sulfide ores (other than coal) such as uranium, copper, bentonite, and phosphoria is also a common source of artificially mobilized selenium. Selenium concentrations as high as 4,500 µg/g (ppm) have been reported in the overburden from uranium mining (USDI-BOR/FWS/GS/BIA 1998). Leachates from phosphoria overburden drains have been documented to contain > 2,000 µg/L (ppb) selenium and to have caused selenium toxicosis among livestock in downstream pastures where creeks contained 300 µg/L waterborne selenium (Talcott and Moller 1997).

The recent rapid expansion of high-density confined livestock production facilities pose yet another potential pathway for accelerated mobilization of selenium into aquatic ecosystems. Most commercial livestock feeding operations (and dairies) add supplemental selenium to the feeds and Oldfield (1994) reported that liquid manure pits beneath feed barns contained 50-150 µg/L of selenium. Unlike human wastes, animal wastes are often discharged to surface water bodies without any prior waste treatment. The biochemistry of selenium in liquid manure might be unique compared to other artificial mobilization pathways (CAST 1994), but this has not been confirmed. The environmental fate of "feed barn" selenium has not been systematically researched to date. Solid manure is also a common ingredient in commercial fertilizers and can reach surface waters via drift during fertilizer application , equipment cleansing, and downslope drainage of leachates. Although most municipal wastewater treatment systems process nonseleniferous wastewater (Westcot and Gonzalez 1988), on a regional and local basis mass-loading of selenium to surface waters from public wastewater treatment facilities can be ecologically significant (Pease *et al*. 1992; CRWQCB 1995). This may be of particular concern where constructed wetlands, that attract use by wildlife, are a component of the water treatment process.

Toxicity

For vertebrates, selenium is an essential nutrient (Wilber 1980). Inadequate dietary uptake (food and water) of selenium results in selenium deficiency syndromes such as reproductive impairment, poor body condition, and immune system dysfunction (Oldfield 1990; CAST 1994).

However, excessive dietary uptake of selenium results in toxicity syndromes that are similar to the deficiency syndromes (Koller and Exon 1986). Thus, selenium is a "hormetic" chemical, i.e., a chemical for which levels of safe dietary uptake are bounded on both sides by adverse-effects thresholds. Most essential nutrients are hormetic; what distinguishes selenium from other nutrients is the very narrow range between the deficiency threshold and the toxicity threshold (Wilber 1980; Sorensen 1991). Nutritionally adequate dietary uptake (from feed) is generally reported as 0.1 to 0.3 µg/g (ppm) on a dry feed basis, whereas, the toxicity threshold for sensitive vertebrate animals is generally reported as 2 µg/g (ppm). That dietary toxicity threshold is only one order-of-magnitude above nutritionally adequate exposure levels (see review in Skorupa *et al*. 1996; USDI-BOR/FWS/GS/BIA 1998).

Hormetic margin-of-safety data suggest that environmental regulatory standards for selenium should generally be placed no higher than one order of magnitude above normal background levels (unless there are species-specific and site-specific data to justify a variance from the general rule). For freshwater ecosystems that are negligibly influenced by agricultural or industrial mobilization of selenium, normal background concentrations of selenium have been estimated as 0.25 µg/L (ppb; Wilber 1980), 0.1-0.3 µg/L (ppb; Lemly 1985), 0.2 µg/L (ppb; Lillebo *et al*. 1988), and 0.1-0.4 µg/L (ppb; average <0.2, Maier and Knight 1994). These estimates suggest, based on a margin-of-safety line of reasoning, that the aquatic life chronic criterion for selenium should be *no higher* than 4 µg/L (= 10-times the upper boundary for normal background), and that a criterion of 2 µg/L would be most consistent with the central tendency value (0.2 µg/L) for normal background levels of waterborne selenium and a one order-of-magnitude margin of safety.

Direct Waterborne Contact Toxicity

Selenium occurs in natural waters primarily in two oxidation states, selenate (+6) and selenite (+4). Waters associated with various fossil-fuel extraction, refining, and waste disposal pathways contain selenium predominantly in the selenite (+4) oxidation state. Waters associated with irrigated agriculture in the western United States contain selenium predominantly in the selenate (+6) oxidation state. Based on traditional bioassay measures of toxicity (24- to 96-hour contact exposure to contaminated water *without* concomitant dietary exposure), selenite is more toxic than selenate to most aquatic taxa (e.g., see review in Moore *et al*. 1990).

Most aquatic organisms, however, are relatively insensitive to waterborne contact exposure to either dissolved selenate or dissolved selenite, with adverse-effects concentrations generally above 1,000 µg/L (ppb). By contrast, waterborne contact toxicity for selenium in the form of dissolved seleno-amino-acids (such as selenomethionine and selenocysteine) has been reported at concentrations as low as 3-4 µg/L for striped bass (*Morone saxitilis)* (ppb; Moore *et al*. 1990). It would be expected, however, that at a chronic standard of 5 µg/L (ppb) *total selenium* the concentration of dissolved seleno-amino-acids would be substantively below 3-4 µg/L (ppb) because seleno-amino-acids usually make up much less than 60-80 percent of *total dissolved selenium* in natural waters. For example, it was estimated that organoselenium made up only 4.5

percent of the total dissolved selenium in highly contaminated drainage water from the San Joaquin Valley (Besser *et al*. 1989). Under most circumstances, a 5 µg/L chronic criterion should be protective of aquatic life *with regard to direct contact toxicity*. Selenium, however, is bioaccumulative and therefore direct contact exposure is only a minor exposure pathway for aquatic organisms (e.g., see review by Lemly 1996a).

Bioaccumulative Dietary Toxicity

Although typical concentrations of different chemical forms of selenium would be unlikely to cause direct contact toxicity at an aquatic life chronic standard of 5 µg/L (ppb), as little as 0.1 µg/L of dissolved selenomethionine has been found sufficient, via bioaccumulation, to cause an average concentration of 14.9 µg/g (ppm, dry weight) selenium in zooplankton (Besser *et al*. 1993), a concentration that would cause dietary toxicity to most species of fish (Lemly 1996a). Based on Besser *et al.* (1993) bioaccumulation factors (BAFs) for low concentrations of selenomethionine, as little as 6 ng/L (ppt) of dissolved selenomethionine would be sufficient to cause foodchain bioaccumulation of selenium to concentrations exceeding toxic thresholds for dietary exposure of fish and wildlife. Thus, at a chronic aquatic life standard of 5 µg/L (ppb) as *total selenium*, if more than 0.1 percent of the total dissolved selenium were in the form of selenomethionine, foodchain accumulation of selenium to levels sufficient to cause dietary toxicity in sensitive species of fish and birds would occur. For highly contaminated water (100-300 µg/L selenium) in the San Joaquin Valley about 4.5 percent of all dissolved selenium was in the form of organoselenium (Besser *et al*. 1989). Unfortunately, relative concentrations of seleno-amino-acids have not been determined in the field in California for waters where total selenium is found in the critical 1-5 µg/L range. Further research is required to characterize typical proportions of seleno-amino-acids in waters containing 1-5 µg/L (ppb) *total selenium*.

Based on waters containing 1-5 µg/L (ppb) *total selenium*, composite bioaccumulation factors (defined as: the total bioaccumulation of selenium from exposure to a composite mixture of different selenium species measured only as *total selenium*) for aquatic foodchain items (algae, zooplankton, macroinvertebrates) are typically between 1,000 and 10,000 (on dry weight basis; Lillebo *et al*. 1988; Lemly 1996a). Therefore, based on risk from bioaccumulative dietary toxicity, a generic aquatic life chronic criterion in the range of 0.2 to 2 µg/L (ppb) would be justified (where generic is defined as: the absence of site-specific and species-specific toxicological data). In fact, based on an analysis of bioaccumulative dietary risk and a literature database, Lillebo *et al*. (1988) concluded that a chronic criterion of 0.9 µg/L (ppb) for *total selenium* is required to protect fish from adverse toxic effects. Furthermore, Peterson and Nebeker (1992) applied a bioaccumulative risk analysis to semi-aquatic wildlife taxa and concluded that a chronic standard of 1 µg/L (ppb) for *total selenium* was warranted. Most recently, Skorupa (1998) has compiled a summary of field data that includes multiple examples of fish and wildlife toxicity in nature at waterborne selenium concentrations below 5 µg/L (ppb), supporting the criteria recommendations of Lillebo *et al*. (1988) and Peterson and Nebeker (1992). Furthermore, a recently concluded regional survey of irrigation related selenium mobilization in the western United States, conducted jointly by several agencies of the U.S.

Department of the Interior over a ten-year period, found that at 5 µg/L total Se in surface waters about 60% of associated sets of avian eggs exceeded the toxic threshold for selenium, i.e., that 5 µg/L Se was only about 40% protective against excessive bioaccumulation of selenium into the eggs of waterbirds (Seiler and Skorupa, In Press).

Interaction Effects Enhancing Selenium Toxicity

Toxic thresholds for fish and wildlife dietary exposure to selenium have been identified primarily by means of controlled feeding experiments with captive animals (e.g., see reviews by NRC 1980, 1984, 1989; Heinz 1996; Lemly 1996a; Skorupa *et al*. 1996; USDI-BOR/FWS/GS/BIA 1998). Such experiments are carefully designed to isolate the toxic effects of selenium as a *solitary stressor*. Consequently, the toxic thresholds identified by such studies are prone to overestimating the levels of selenium exposure that can be tolerated, without adverse effects, in an environment with *multiple stressors* as is typical of the real ecosystems (Cech et al. 1998). There are at least three well-known multiple-stressor scenarios for selenium that dictate a very conservative approach to setting water quality criteria for aquatic life:

1. Winter Stress Syndrome - More than 60 years ago it was first discovered in experiments with poultry housed in outdoor pens that dietary toxicity thresholds were lower for experiments done in the winter than at other times of the year (Tully and Franke 1935). More recently this was confirmed for mallard ducks (*Anas platyrhynchos*) by Heinz and Fitzgerald (1993). Lemly (1993b), studying fish, conducted the first experimental research taking into account the interactive effects of winter stress syndrome and confirmed that such effects are highly relevant even for waters containing <5 µg/L (ppb) selenium. Consequently, Lemly (1996b) presents a general case for winter stress syndrome as a critical component of hazard assessments. It can be further generalized that any metabolic stressor (cold weather, migration, smoltification, pathogen challenge, etc.) would interact similarly to lower the toxic thresholds for dietary exposure to selenium. Based on a comparison of results from Heinz and Fitzgerald (1993) and Albers *et al.* (1996), the dietary toxicity threshold in the presence of winter stress was only 0.5-times the threshold level for selenium as a solitary stressor. Thus, it appears that criteria based on single-stressor data should be reduced by at least a factor of two. The proposed chronic criterion for selenium of 5 µg/L (ppb) is based, in part, on field data from Belews Lake (EPA 1987a), presumably including multiple stressors as typically encountered in nature. However, as recently noted in a presentation by Dr. Dennis Lemly to the EPA Peer Consultation Committee on selenium (EPA 1998:3-5), EPA's 5 µg/L (ppb) criterion was based on the erroneous presumption that the Hwy. 158-Arm of Belews Lake was "unaffected." Dr. Lemly argues that in fact multiple lines of evidence indicate adverse effects of selenium on fish in the Hwy. 158-Arm of Belews Lake at concentrations of 0.2-4 µg/L (ppb). Dr. Lemly concludes that the true (multiple stressor) ". . . threshold for detrimental impacts [at Belews Lake] is well below 5 µg/L."

2. Immune System Dysfunction - Also more than 60 years ago, it was first noted that chickens exposed to elevated levels of dietary selenium were differentially susceptible to pathogen challenges (Tully and Franke 1935). More recently this was confirmed for mallard ducks by

Whiteley and Yuill (1989). Numerous other studies have confirmed the physiological and histopathological basis for selenium-induced immune system dysfunctions in wildlife (Fairbrother and Fowles 1990; Schamber *et al*. 1995; Albers *et al*. 1996). Based on Whiteley and Yuill's (1989) results, *in ovo* exposure of mallard ducklings to as little as 3.9 µg/g (ppm dry weight basis) selenium was sufficient to significantly increase mortality when ducklings were challenged with a pathogen. The lowest confirmed *in ovo* toxicity threshold for selenium as a solitary stressor is 10 µg/g (ppm dry weight basis; Heinz 1996, reported as 3 µg/g wet weight basis and about 70% moisture). In this case the multiple-stressor toxicity threshold is only 0.39-times the threshold level for selenium as a solitary stressor. Based, in part, on the solitary stressor toxic threshold reported by Heinz (1996) for mallard eggs, Adams et al. (1998) concluded that 6.77 µg/L Se would be 90% protective against excessive bioaccumulation of selenium into avian eggs. Therefore based on a pathogen challenge multiple-stressor scenario a protective water quality criterion would be (0.39) X (6.77 µg/L) = 2.6µg/L (ppb). Again, the multiple-stressor threshold would appear to be well below the proposed chronic criterion of 5 µg/L (ppb).

3. Chemical Synergism - Multiple stressors can also consist of other contaminants. For example, Heinz and Hoffman (1998) recently reported very strong synergistic effects between dietary organo-selenium and organo-mercury with regard to reproductive impairment of mallard ducks. The experiment of Heinz and Hoffman (1998) did not include selenium treatments near or below the threshold for diet-mediated reproductive toxicity and therefore no ratio of single-stressor versus multiple-stressor threshold levels is available. A field study involving 12 lakes in Sweden, however, found that in the presence of threshold levels of mercury contamination, the waterborne threshold for selenium toxicity was about 2.6 µg/L (ppb; see review in Skorupa 1998; and review in USDI-BOR/FWS/GS/BIA 1998). The Swedish lakes' result is in agreement with multiple-stressor derived criteria suggested above for winter stress and for pathogen challenge as multiple stressors. Based on the Swedish lakes study, which encompassed 98 different lakes, Lindqvist et al. (1991) concluded, "It is important not to dose so that Se concentrations in water rise above about 1 to 2 µg Se/L." Likewise, Meili (1996) concluded that, "The results [of the Swedish Lakes studies] suggest that a selenium concentration of only 3 µg/L can seriously damage fish populations."

At least one field study of birds also provides circumstantial evidence of lowered toxicity thresholds for selenium-induced reproductive impairment in the presence of mercury contamination (Henny and Herron 1989).

Environmental Partitioning and Waterborne Toxicity Thresholds

Risk management via water concentration-based water quality criteria is an inherently flawed process for selenium (Pease *et al*. 1992; Taylor *et al*. 1992, 1993; Canton 1997). The process is flawed because the potential for toxic hazards to fish and wildlife is determined by the rate of mass loading of selenium into an aquatic ecosystem and the corresponding environmental partitioning of mass loads between the water column, sediments, and biota (food chain). However, a water column concentration of selenium can be an imperfect and uncertain measure of mass loading and foodchain bioaccumulation. For example, a low concentration of waterborne selenium can occur because mass loading into the system is low (= low potential for hazard to fish and wildlife) or because there has been rapid biotic uptake and/or sediment deposition from elevated mass loading (= high potential for hazard to fish and wildlife). Toxicity to fish and wildlife is ultimately determined by how much selenium is partitioned into the food chain. Therefore, water quality criteria are useful guides for risk management only to the extent that they protect aquatic food chains from excessive bioaccumulation of selenium. As evidenced by the literature cited above, a water quality chronic criterion of 2 µg/L will protect aquatic food chains from excessive bioaccumulation under most permutations of environmental and anthropogenic factors (i.e., the probability of adverse effects is sufficiently low). However, several examples of potentially hazardous foodchain bioaccumulation of selenium at waterborne selenium concentrations <2 µg/L are known from California (Maier and Knight 1991; Pease *et al.* 1992; Luoma and Linville 1997; San Francisco Estuary Institute [SFEI] 1997a; Setmire *et al.* 1990, 1993; Bennett 1997) and elsewhere (Birkner 1978; Lemly 1997; Hamilton 1998). To substantively decrease the regulatory uncertainty of water quality criteria for selenium, ultimately a criterion-setting protocol will have to be formulated that links risk management and regulatory goals directly to aquatic food chain contamination (for example, see Taylor *et al.* 1992, 1993).

<u>Selenium Summary</u>

A variety of conceptual bases for deriving a generally applicable chronic water quality criterion for selenium that is protective of fish and wildlife have been presented above with the following results:

Hormetic Margin of Safety Basis: 1-4 µg/L (ppb), with 2 µg/L (ppb) being most consistent with central tendency data.

Waterborne Exposure Only Basis (= Traditional Bioassay Testing): 3-4 µg/L (ppb) for selenium in the form of seleno-amino-acids (e.g., selenomethionine); current EPA chronic criterion of 5 µg/L (ppb) adequate for selenium as inorganic ions (e.g., selenite and selenate).

Bioaccumulative Dietary Exposure Basis (with Selenium as solitary stressor):
0.2-2.0 µg/L (ppb), with 0.9-1.0 µg/L (ppb) supported by the two most detailed reviews to date.

Winter Stress Syndrome Multiple Stressor Basis: ". . . well below . . . " 5 µg/L (ppb).

Pathogen Challenge Multiple Stressor Basis: 2.6 µg/L (ppb).

Mercury Synergism Multiple Stressor Basis: 2-3 µg/L (ppb).

Overwhelmingly, the available body of scientific evidence (the majority of which has been produced subsequently to EPA's 1987 criterion derivation for selenium) consistently supports a chronic criterion of 2 µg/L (ppb) for the protection of sensitive taxa of fish and wildlife. Even a criterion of 2 µg/L, however, can fail to be protective in specific cases where water column contamination with selenium fails to accurately reflect food chain contamination. There is a strong need for developing a method to link criteria directly to food chain contamination. In the absence of site-specific and species-specific data regarding the sensitivity of particular species and/or populations, a general criterion of at least 2 µg/L is required to assure adequate protection of threatened and endangered species of fish and wildlife. This is especially warranted considering the steep response curves for selenium (Hoffman *et al*. 1996; Lemly 1998; Skorupa 1998) and the well-demonstrated potential for selenium-facilitated pathogen susceptibility that can rapidly extirpate entire populations of fish and wildlife via epizootic events.

Summary of Effects of Selenium to Listed Species

Birds

The Services conclude that selenium poisoning of birds foraging in aquatic systems may occur at or below concentrations permissible under the aquatic life criteria proposed in the CTR. The effects of selenium poisoning on avian species include: gross embryo deformities, winter stress syndrome, depressed resistance to disease due to depressed immune system function, reduced juvenile growth and survival rates, mass wasting, loss of feathers (alopecia), embryo death, and altered hepatic enzyme function. In addition the interactive effects between mercury and selenium produce super-toxic effects greater than effects of each compound individually that may include embryo deformities, embryo death, reduced juvenile survival, behavioral abnormalities, depressed immune response, mass wasting, and mortality. It is the aggregation of these effects that the Service believes are likely to adversely affect the bald eagle, California clapper rail, California brown pelican, California least tern, light-footed clapper rail, marbled murrelet, and the Yuma clapper rail, based on the potential for these species to be impacted by elevated levels of selenium through their dietary habits, dependence on the aquatic ecosystem, and their limited distribution.

A species which the Service believes will not be adversely affected is the snowy plover. The coastal populations of the snowy plover have a significant terrestrial component to their diet which likely provides dietary dilution of aquatic system selenium exposures, and have been shown on a species-specific basis to be very tolerant to selenium exposure.

Aleutian Canada Goose: As herbivorous waterbirds, with a fairly unique ecological niche, all forms of Canada geese can be expected to be extremely sensitive to dietary exposure to selenium. The basis for this sensitivity was presented via energetic modeling by DuBowy (1989) for American coots (*Fulica americana*), another herbivorous species of waterbird. Herbivorous birds consume such a large bulk of vegetation to meet caloric requirements (compared to birds feeding on high caloric dense animal matter) that their mass dosing of selenium can be very high even though the diet contains a lower concentration of selenium than normally considered toxic for other species.

A field study of Canada geese (*Branta canadensis*) in Wyoming (See *et al.* 1992) reported widespread reproductive failure among geese with relatively low exposure to selenium (eggs averaging 5-10 µg/g Se). If selenium caused the observed reproductive failure in Wyoming as the authors of the report believed, but which was not well established (Skorupa 1998), and if as little as 5 µg/g Se in eggs of geese is reproductively hazardous, then a 5 µg/L water quality criterion for selenium would fail to protect geese (most avian species exhibit water to egg bioaccumulation factors of at least 1,000-fold; Ohlendorf *et al.* 1993, Skorupa *et al.* unpubl. data).

The Aleutian Canada goose would be most likely to encounter selenium-contaminated vegetation in wetlands. In contrast to breeding geese, which would be expected to feed in the wetlands used for nesting, wintering Aleutian Canada geese in California feed primarily in upland crops and fallow fields. Thus, it is expected that exposure to wetland vegetation would be rare for the Aleutian Canada goose while wintering in California and that selenium standards for such wetlands are not an important issue for the survival and recovery of this subspecies.

Bald Eagle: At least two citations in the selenium literature provide a basis for doubting that a chronic selenium standard of 5 µg/L (ppb) would be sufficiently protective of bald eagles. Lillebo *et al.* (1988) derived levels of selenium to protect various species of waterbirds. Based on an analysis of bioaccumulation dynamics and an estimated critical dietary threshold for toxicity of 3 µg/g, they concluded that piscivorous birds would be at substantially greater risk of toxic exposure than mallards (*Anas platyrhynchos*). The calculated water criterion to protect piscivorous birds was 1.4 µg/L (ppb) as opposed to 6.5 µg/L (ppb) for mallards. The proposed CTR criterion of 5 µg/L (ppb) is more than 3-times the calculated criterion for piscivorous birds. It should also be noted that the 6.5 µg/L (ppb) calculated criterion for mallards exceeds the actual threshold point for ducks in the wild which is somewhere below 4 µg/L (ppb) (Skorupa 1998). Thus, the 1.4 µg/L (ppb) calculated criterion for piscivorous birds may be biased high compared to the wild as well.

Applying an energetics modeling approach, modified from the Wisconsin Department of Natural Resources, Peterson and Nebeker (1992) calculated a chronic criterion specifically for Bald eagles. Peterson and Nebeker's estimate of a protective criterion is 1.9 µg/L (ppb). Again, the estimate is below the CTR proposed criterion of 5 µg/L (ppb). However, Peterson and Nebeker calculated a mallard criterion (2.1 µg/L; ppb) that was much closer to their Bald eagle criterion than Lillebo *et al.'s* results would suggest. Peterson and Nebeker's mallard criterion is consistent with real-world data (cf. Skorupa 1998) and therefore their bald eagle criterion may also be reliable.

Consequently, best available evidence suggests that widespread expansion of aquatic habitats containing > 1.9 µg/L (ppb) selenium, as could occur with a criterion of 5 µg/L (ppb), could put substantial numbers of California's bald eagles at risk of toxic effects of selenium.

California Brown Pelican: As a large-bodied piscivorous bird, much of the discussion provided above for the bald eagle regarding the inadequacy of the CTR-proposed selenium criteria may also apply to the California brown pelican. Consequently, until species-specific data are collected or species-specific modeling is conducted for the California brown pelican, a selenium criterion on the order of 1.4 µg/L (ppb) (generic piscivorous bird model; Lillebo *et al.* 1988) to 1.9 µg/L (ppb) (bald eagle model; Peterson and Nebeker 1992) must be viewed as the applicable guidance for protection of California brown pelicans from selenium poisoning. The CTR-proposed criterion of 5 µg/L (ppb) must therefore be viewed as unprotective of California brown pelicans foraging in the Salton Sea and enclosed bays and estuaries in the State of California.

In the 1990's there have been at least 4 major avian epizootic events at California's Salton Sea, including suspected algal toxin poisoning of more than 175,000 eared grebes (in two episodes), botulism poisoning of about 15,000 piscivorous birds (including more than 1,400 Brown Pelicans) and a Newcastle's disease outbreak in a cormorant colony (Bennett 1994; USGS 1996; USDI-FWS 1997c). Normal selenium nutrition is a well-documented requirement for the proper functioning of avian and fish immune systems (e.g., Larsen *et al.* 1997; Wang and Lovell 1997). Deficient and toxic levels of selenium equally cause immune system dysfunctions (e.g., Larsen *et al.* 1997) and for 60 years it has repeatedly been demonstrated clinically that birds and fish suffering from selenium-induced immune dysfunctions are hypersensitive to pathogen challenges (e.g., Tully and Franke 1935; Whiteley and Yuill 1989; Larsen *et al.* 1997; Wang *et al.* 1997).

In addition to weakening the immune defenses of listed species such as the brown pelican, excessive environmental selenium can also trigger pathogen and toxin challenges that would not otherwise have occurred. For example, a red tide flagellate (*Chattonella verruculosa)* which has caused the mortality of fish such as yellowtail, amberjack, red and black sea bream, has recently been discovered to require above-normal exposure to selenium (Imai *et al.* 1996). Only when selenium extracted from contaminated sediments is added to growth media can *C. verruculosa* sustain rapid growth (i.e., toxic blooms). The level of contamination required to sustain rapid growth is only about 2-times normal background. Clearly, the potential effects of selenium-mediated algal toxins must be considered when evaluating potential hazards associated with selenium criteria. The two episodes involving massive eared-grebe die-offs illustrate how quickly algal toxins can remove 10 percent or more of the entire continental population of a species. Selenium-mediated algal toxins should probably be viewed as a serious potential threat to any endangered species that could have major portions of its extant population exposed. The CTR-proposed criterion of 5 µg/L, which is more than 10-times the normal background concentration of waterborne selenium (e.g., Maier and Knight 1994), would almost always be associated with more than 2-times normal sediment selenium and therefore could facilitate toxic algal blooms.

The case of botulism that killed more than 1,400 brown pelicans at California's Salton Sea was a very unusual case of botulism that was mediated by a bacterial epizootic among fish (USDI-FWS 1997c). This bacterially-mediated pathway for an avian botulism epizootic had never been encountered before. Fish in the Salton Sea contain substantially elevated tissue selenium (e.g., Saiki 1990) which very plausibly leaves them immune impaired and hypersensitive to the *Vibrio* bacterial attacks that facilitated the botulism outbreak.

California Clapper Rail: The extant range of the California clapper rail is restricted to marshes of the San Francisco Bay Estuary, an aquatic system already receiving substantial selenium input from agricultural and industrial sources (Pease *et al.* 1992). California clapper rails feed almost exclusively on benthic invertebrates, a well-documented pathway for bioaccumulation of selenium (see review by Pease *et al.* 1992). Total inflows of water to the San Francisco Bay Estuary average less than 5 µg/L (ppb) selenium (e.g., inflows diverted to the Central Valley Project and State Water Project canals usually average about 1 µg/L (ppb) selenium). The Regional Monitoring Program for 1997 (SFEI, 1999) reported total selenium concentrations ranged from 0.03 µg/L (ppb) to 2.20 µg/L (ppb) with highest concentrations found in the south bay. Lonzarich *et al.* (1992) reported that eggs of California clapper rails collected from the north bay in 1987 contained up to 7.4 µg/g selenium. Water data from this time and location are not available. The *in ovo* threshold for selenium exposure that causes toxic effects on embryos of California clapper rails is unknown. For another benthic-foraging marsh bird, the black-necked stilt, the *in ovo* threshold for embryotoxicity is 6 µg/g selenium (Skorupa 1998). More recent investigations of fail to hatch California clapper rail eggs in the south bay in 1992 and the north bay in 1998 have not duplicated the higher selenium results of Lonzarich *et al.* and maximum egg selenium concentrations have not exceeded 3.2 µg/g (dw)(FWS unpublished data).

It has recently been demonstrated for mallard ducks that interactive effects of selenium and mercury can be super-toxic with regard to embryotoxic effects (Heinz and Hoffman 1998). Lonzarich *et al.* (1992) also reported potentially embryotoxic concentrations of mercury in eggs of California clapper rails. Abnormally high numbers of nonviable eggs, 13.7-22.9 percent, have also been reported for the California clapper rail (Schwarzbach 1994). Since the main avenue of impacts from selenium and mercury alone, and interactively, would be manifested as reproductive impairment (especially inviable eggs), it strongly appears that populations of the California clapper rail could not tolerate the increased selenium loading to the San Francisco Bay Estuary that would be allowable under a CTR-proposed criterion of 5 µg/L (ppb). Based, in part, on the data for California clapper rails, staff technical reports prepared for the San Francisco Bay Regional Water Quality Control Board recommend decreasing current selenium loading to the estuary by 50 percent or more (Taylor *et al.* 1992, 1993). By comparison, the CTR-proposed selenium criteria would possibly accommodate increases in selenium loading to the bay or locally elevated selenium in effluent dominated tributaries. If selenium concentrations or selenium loads were increased in San Francisco Bay, clapper rail egg selenium would be expected to increase. The rail is particularly vulnerable to any locally elevated effluent concentrations of selenium as the rail generally occupies small home ranges of only a few acres

California Least Tern: As a piscivorous bird, much of the discussion provided above for the bald eagle regarding the inadequacy of the CTR-proposed selenium criteria may also apply to the California least tern. Consequently, until species-specific data are collected or species-specific modeling is conducted for the California least tern, a selenium criterion on the order of 1.4 µg/L (ppb) (generic piscivorous bird model; Lillebo *et al.* 1988) to 1.9 µg/L (ppb) (Bald eagle model; Peterson and Nebeker 1992) must be viewed as the applicable guidance for protection of California least terns from selenium poisoning. The CTR-proposed criterion of 5 µg/L (ppb) must therefore be viewed as unprotective of California least terns.

Selenium analyses of least tern eggs collected from San Francisco Bay and San Diego Bay are reported by Hothem and Zador (1995). In San Francisco Bay the eggs contained up to 3.1 µg/g selenium and in San Diego Bay the eggs contained up to 2.9 µg/g selenium. Neither of those maximum values exceed currently recognized thresholds for avian embryotoxicity (for selenium as a solitary stressor). However, both sets of eggs also exhibited elevated concentrations of mercury which raises the possibility of super-toxic interaction effects as demonstrated for mallards by Heinz and Hoffman (1998). Waterborne concentrations of selenium in the San Francisco Bay Estuary are currently well below 5 µg/L (ppb) (e.g., <1 µg/L (ppb); Pease *et al*. 1992).

Eggs of the Interior least tern (*Sterna antillarum athalassos*) collected from the Missouri River system in the central United States have contained as much as 11-12 µg/g selenium (Ruelle 1993; Allen and Blackford 1997). Allen and Blackford (1997) reported that Least Tern nesting success from 1992-1994 at most locations in the study area was not sufficient to ensure survival of the studied populations. They also concluded that although flooding and predation likely are the major cause of the low recruitment, the results of their study "indicate that selenium and mercury may contribute to low reproduction." Neither Ruelle (1993) nor Allen and Blackford (1997) reported what the waterborne selenium levels were at their study sites. Other authors have reported selenium concentrations averaging about 2-4 µg/L (ppb) for major tributaries of the Missouri River system (North Platte River, See *et al*. 1992; James River, USDI-FWS 1989).

Results from studies of the Interior least tern suggest that selenium concentrations in California least tern eggs would substantively exceed the 6 µg/g threshold for embryotoxicity established for black-necked stilts (Skorupa 1998) if selenium concentrations were permitted to rise to a 5 µg/L (ppb) concentration In combination with elevated mercury concentrations already noted for eggs of California least terns (Hothem and Zador 1995), significant reproductive impairment would be the expected outcome.

<u>Light-footed Clapper Rail</u>: The Service is not aware of any existing data for selenium concentrations in eggs of light-footed clapper rails, or for any other tissues. The Service is also not aware of any studies characterizing the selenium profile of marshes currently supporting populations of light-footed clapper rails. Insufficient information is available to determine the likelihood of the CTR-proposed selenium criterion of 5 µg/L (ppb) being fully met within marshes crucial to survival and recovery of the light-footed clapper rail.

Because light-footed clapper rails have declined to just a few remnant populations vulnerable to rapid extirpation (Baron and Jorgensen 1994), are relatively sedentary nonmigratory residents prone to maximum exposure to localized contamination of a marsh, and are linked to a benthic foodchain that would be very efficient at bioaccumulating selenium, a worst-case scenario for potential impacts associated with a proposed 5 µg/L (ppb) selenium criterion must be assumed. Based on data for the California clapper rail and the Yuma clapper rail (summarized in this final biological opinion) a worst-case scenario of environmental selenium contamination up to the limits allowed by the proposed CTR criteria would include *in ovo* exposure to selenium substantially above best estimates of the embryotoxic threshold. Particularly if elevated levels of environmental selenium were established in the presence of elevated levels of mercury, selenium-induced or selenium/mercury interactively-induced reproductive failure could occur.

Marbled Murrelet: During the breeding season marbled murrelets forage in nearshore environments including bays and estuaries on small fish and euphasid shrimp. They have also been known to forage to a minor degree on salmonid fry in freshwater environments. As a piscivorous bird, much of the discussion provided above for the bald eagle regarding the inadequacy of the CTR-proposed selenium criterion may also apply to the marbled murrelet.

Adverse impacts from increased permissible concentrations of contaminants as proposed in the CTR to prey species such as the Pacific sardine, herring, topsmelt, and northern anchovies, has the potential to significantly reduce long-term reproductive success of marbled murrelets (USDI-FWS, 1997b). Adverse effects to prey species spawning and nursery habitats have the potential to impair population size and reduce recruitment throughout their range in California. The vulnerability of marbled murrelet populations in conservation zones 5 and 6, coupled with elevated concentrations of contaminants in spawning and nursery areas for murrelet prey species increase the risk of bioaccumulation of mercury and selenium. The synergistic effects of these contaminants pose a significant threat to marbled murrelet reproduction throughout conservation zones 5 and 6 and to a lesser degree in conservation zone 4.

Consequently, until species-specific data are collected or species-specific modeling is conducted for the marbled murrelet, a selenium criterion on the order of 1.4 µg/L (ppb) (generic piscivorous bird model; Lillebo *et al.* 1988) to 1.9 µg/L (ppb) (bald eagle model; Peterson and Nebeker 1992) must be viewed as the applicable guidance for protection of marbled murrelets. Foraging in environments with between 2 and 5 µg/L (ppb) selenium during the breeding season would likely present a reproductive hazard to the murrelet. The Services therefore conclude that the CTR-proposed criterion of 5 µg/L (ppb) must be viewed as unprotective of marbled murrelets foraging in enclosed bays and estuaries in the State of California.

Western Snowy Plover: Interior populations of the western snowy plover have been studied at breeding sites averaging about 5 µg/L (ppb) waterborne selenium in California's Tulare Lake Basin (Skorupa *et al.* unpubl. data). At those sites, eggs averaged about 9 µg/g selenium. That exceeds the 6 µg/g threshold for embryotoxicity among black-necked stilts, but species-specific data for snowy plover eggs containing a wide range of selenium concentrations (egg selenium from 2-50 µg/g) suggest that snowy plovers are less sensitive to selenium exposure than black-necked stilts (Skorupa *et al.* unpubl. data; Page *et al.* 1995; Washington Dept. of Fish and Wildlife 1995). Western snowy plovers appear to be about as tolerant of selenium exposure as American avocets (*Recurvirostra americana*) (cf. Skorupa 1996; 1998) which suggests that they would not be at risk of reproductive impairment when nesting at sites with up to 5 µg/L (ppb) waterborne selenium. The study sites producing this data for interior-nesting snowy plovers were uniformly uncontaminated with mercury (Skorupa *et al.* unpubl. data).

Unless coastal populations would be exposed to significant selenium-mercury interaction effects (cf. Heinz and Hoffman 1998), the results documented for populations of interior-nesting snowy plovers are expected to apply to the listed Pacific Coast populations of the snowy plover. Therefore, the western snowy plover is considered not likely to be adversely affected by the CTR-proposed selenium criterion of 5 µg/L (ppb).

<u>Yuma Clapper Rail</u>: With a biological profile very similar to the California clapper rail, the Yuma clapper rail is similarly vulnerable to selenium bioaccumulation via a benthic foodchain pathway. For backwaters of the lower Colorado River system in California, Lonzarich *et al.* (1992) reported a mean selenium concentration of 12.5 µg/g selenium for eggs from two abandoned clutches of Yuma clapper rails. They also stated that this level of exposure was "..believed to be associated with low hatching success and embryo deformities..." (Lonzarich *et al.* 1992:151). A mean of 12.5 µg/g *in ovo* selenium substantively exceeds the 6 µg/g threshold for embryotoxicity rigorously established for another benthic-foraging species of marshbird, the black-necked stilt (Skorupa 1998). The source water for the Colorado River backwaters where these Yuma clapper rail eggs were sampled averages about 2 µg/L (ppb) selenium (e.g., Setmire and Schreder 1998). Clearly, if selenium in the source water increased to 5 µg/L (ppb) as would be allowable under the CTR-proposed selenium criterion, it could be expected that the selenium content of Yuma clapper rail eggs would very substantially exceed the best available estimate of the embryotoxic threshold point.

Agricultural drainage water in the Imperial Valley typically contains 2-10 µg/L (ppb) selenium (see review for Salton Sea in Skorupa 1998). When marshes in the Imperial Valley were supplied with agricultural drainwater in 1990, selenium concentrations in a sample of Yuma clapper rail eggs were as high as 7.8 µg/g (C. Roberts, pers. comm.). When the drainage water was replaced with water containing 2 µg/L (ppb) selenium, the concentrations of selenium measured in Yuma clapper rail foods (crayfish) were at safe levels (2.2 µg/g). The data from the Colorado River and from the Imperial Valley, the major extent of the Yuma clapper rail's geographic range, are consistent in indicating that a selenium criterion of 5 µg/L (ppb) would not be adequately protective.

Amphibians and Reptiles

Selenium is toxic to developing frog embryos and tadpoles (Browne and Dumont, 1979), however, testing of amphibians has been very limited. Browne and Dumont for example only tested sodium selenite and only in short term acute tests. Most field studies of selenium do not include amphibians and those that do generally report uninterpreted residues in frog liver. The Service is unaware of specific studies of amphibian egg residues and associated impacts to reproduction, however, it is likely that amphibian toxic response is similar to fish and birds where reproductive failure is associated with egg concentrations greater than 6 µg/g in birds and 10 µg/g in fish. It is also likely that aquatic food chain contamination by selenium would be the most significant pathway of exposure as would maternal transfer of organic selenium to the eggs. In the absence of selenium toxicity information the Service believes a fish risk model may be most appropriate for assessing selenium hazard to amphibians such as the red-legged frog. This assessment may however be overly simplistic. Development of amphibians is unique among vertebrates in the occurrence of hormone mediated ontogenetic metamorphosis within the water column (Duellman and Trueb, 1986) and selenium is a notorious developmental toxin and growth inhibitor (Skorupa, 1998). Dietary selenium exposure of tadpoles may thus be another significant route of exposure affecting development. California red-legged frogs spend most of their lives in and near sheltered backwaters of ponds, marshes, springs, streams, and reservoirs. These types of environments are particularly vulnerable to selenium contamination of the food chain at low to medium level selenium contamination in water, should a selenium source to water exist. Red legged frogs are now reduced to about 30 percent of their historical range with most of the remaining population limited to coastal drainages. The cretaceous shales of the coast range of California provide a bulk source of selenium whose release to water bodies is accelerated by anthropogenic activities such as cattle grazing, and irrigation drainage. The Service therefore concludes that a criterion of 5 µg/L (ppb) may not be sufficiently protective for the red-legged frog.

Toxicity information on reptiles such as the giant garter snake are even more scanty than the amphibian literature. The Service is unaware of any such information. Endemic to wetlands in the Sacramento and San Joaquin Valleys, the giant garter snake inhabits marshes, sloughs, ponds, small lakes, low gradient streams, and other waterways and agricultural wetlands, such as irrigation and drainage canals and rice fields. Giant garter snakes feed on small fishes, tadpoles, and frogs (Fitch 1941, Hansen 1980, Hansen 1988). These foraging habits and habitat preference put the giant garter snake at risk of selenium exposure. The current day absence of the giant garter snake from extensive wetland areas (the Grasslands Water District) of the San Joaquin Valley, which for the last twenty years have received seleniferous irrigation drainage water, may be circumstantial evidence of a selenium effect on this top aquatic predator. In the absence of a species specific selenium toxicity model for the giant garter snake the Service would recommend using an avian risk model for selenium based on the close phylogenetic relationship of birds to reptiles (e.g., Romer 1966; Porter 1972:216; Storer et al. 1972:312). The Service concludes that a selenium criterion of 5 µg/L (ppb) would not adequately protect the giant garter snake.

Fish

A tremendous amount of research regarding toxic effects of selenium on fish has been conducted since the late 1970's. Recently, this body of research was reviewed and summarized by Lemly (1996b). Lemly reports that salmonids are very sensitive to selenium contamination and exhibit toxic symptoms even when tissue concentrations are quite low. Survival of juvenile rainbow trout (*Oncorhynchus mykiss*) was reduced when whole-body concentrations of selenium exceeded 5 µg/g (dry wt.). Smoltification and seawater migration among juvenile chinook salmon (*Oncorhynchus tshawytscha*) were impaired when whole-body tissue concentrations reached about 20 µg/g. However, mortality among larvae, a more sensitive life stage, occurred when concentrations exceeded 5 µg/g. Whole-body concentrations of selenium in juvenile striped bass (*Morone saxitilis*) collected from areas in California impacted by irrigation drainage ranged from 5 to 8 µg/g.

Summarizing studies of warm-water fish Lemly reports that growth was inhibited at whole-body tissue concentrations of 5 to 8 µg/g selenium or greater among juvenile and adult fathead minnows (*Pimephales promelas*). Several species of centrarchids (sunfish) exhibited physiologically important changes in blood parameters, tissue structure in major organs (ovary, kidney, liver, heart, gills), and organ weight-body weight relations when skeletal muscle tissue contained 8 to 36 µg/g selenium. Whole-body concentrations of only 4 to 6 µg/g were associated with mortality when juvenile bluegill (*Lepomis macrochirus*) were fed selenomethionine-spiked commercial diets in the laboratory. When bluegill eggs contained 12 to 55 µg/g selenium, transfer of the selenium to developing embryos during yolk-sac absorption resulted in edema, morphological deformities, and death prior to the swim-up stage. In a laboratory study of "winter stress syndrome" juvenile bluegill exposed to a diet containing 5.1 µg/g selenium and water containing 4.8 µg/L (ppb) selenium exhibited hematological changes and gill damage that reduced respiratory capacity while increasing respiratory demand and oxygen consumption. In combination with low water temperature (4 degrees centigrade) these effects caused reduced activity and feeding, depletion of 50 to 80 percent of body lipid, and significant mortality within 60 days. Winter stress syndrome resulted in the death of about one-third of exposed fish at whole-body concentrations of 5 to 8 µg/g selenium.

Based on Lemly's review of more than 100 papers, he recommended the following toxic effects thresholds for the overall health and reproductive vigor of freshwater and anadromous fish exposed to elevated concentrations of selenium: 4 µg/g whole body; 8 µg/g skinless fillets; 12 µg/g liver; and 10 µg/g ovary and eggs. He also recommended 3 µg/g as the toxic threshold for selenium in aquatic food-chain organisms consumed by fish. Lemly reported that when waterborne concentrations of inorganic selenium (the predominant form in aquatic environments) are in the 7- to 10-µg/L (ppb) range bioconcentration factors in phytoplankton are about 3,000. Consequently, he concluded that patterns and magnitudes of bioaccumulation are similar enough among various aquatic systems that a common number, 2 µg/L (ppb) (for filtered samples of water), could be given as a threshold for conditions "highly hazardous to the health and long-term survival of fish".

Recently, Hamilton (1998) reviewed the demonstrated and potential effects of selenium on six species of endangered fish in the Colorado River basin, including the humpback chub (*Gila cypha*), Colorado squawfish (*Ptychocheilus lucius*), bonytail chub (*Gila elegans*), razorback sucker (*Xyrauchen texanus*), flannelmouth sucker (*Catostomus latipinnis*), and roundtail chub (*Gila robusta*). Hamilton presents historical data supporting a hypothesis that long-term selenium contamination of the lower Colorado River basin may have been one of the factors contributing to the disappearance of endangered fish in the early 1930's. Contemporary issues of concern included the unusually high incidence of abnormal lesions on fish in the San Juan River, especially flannelmouth sucker, attributed to pathogens requiring inducement by stressors such as high contaminant concentrations or poor body condition; and concentrations of selenium in fish eggs as high as 28 µg/g in razorback sucker from the Green River and as high as 73 µg/g in eggs of rainbow trout collected from the mainstem Colorado River between Glen Canyon Dam and Lee's Ferry. In controlled studies of larval razorback suckers fed food organisms collected from the wild, Hamilton found 2.3 µg/g or more of selenium in the diet to be sufficient to cause reduced survival. In an enclosure study where razorback suckers were held in selenium-contaminated aquatic environments (Adobe Creek, 9-90 µg/L (ppb) selenium, and North Roadside Pond of Ouray National Wildlife Refuge, 40 µg/L (ppb) selenium) for 9 months, muscle plugs contained 17 and 12 µg/g selenium respectively and eggs contained 44 and 38 µg/g selenium. Finally, Hamilton stressed that consideration of selenium effects was an important component of recovery planning for the Colorado River basin endangered endemics.

Selenium effects on Delta Fishes: In November of 1996 the Service issued an approved Recovery Plan for the Sacramento/San Joaquin Delta Native Fishes (USDI-FWS 1996c). The plan addressed recovery requirements for eight species of fish native to the Delta including one species currently listed as threatened, the Delta Smelt (*Hypomesus transpacificus*), and the proposed threatened Sacramento Splittail (*Spirinchus thaleichthys*). Other species addressed by the plan are Longfin Smelt (*Spirinchus thaleichthys*), Green Sturgeon (*Acipenser medirostris*), the Sacramento Spring-run chinook salmon (*Oncorhynchus tshawytscha*), which has been petitioned for listing as endangered, the Sacramento Late Fall-run chinook salmon (*Oncorhynchus tshawytscha*), the San Joaquin Fall-run chinook salmon (*Oncorhynchus tshawytscha*), and the extirpated Sacramento Perch (*Archoplites interruptus*). The Sacramento-San Joaquin River Delta and San Francisco Bay estuary are subject to elevated levels of environmental selenium, and the introduction of high levels of contaminants (including selenium) is cited in the Recovery Plan as one of the more recent potential factors affecting Delta fishes.

Lillebo *et al*. (1988) calculated that a selenium criterion of 0.9 µg/L (ppb) waterborne selenium was necessary to adequately protect fish associated with the San Joaquin River system, including the southern Delta. The CTR-proposed selenium criterion of 5 µg/L (ppb) substantially exceeds the criterion calculated by Lillebo *et al*. (1988). The Recovery Plan states that Delta Smelt are ecologically similar to larval and juvenile Striped Bass (*Morone saxitilis*). Saiki and Palawski (1990) sampled juvenile striped bass in the San Joaquin River system including three sites in the San Francisco Bay estuary. Striped Bass from the estuary contained up to 3.3 µg/g whole-body selenium, a value just below Lemly's 4 µg/g toxicity threshold, even though waterborne selenium typically averages <1 µg/L (ppb) and has been measured no higher than 2.7 µg/L (ppb) within the estuary (Pease *et al*. 1992). Striped Bass collected from Mud Slough in 1986, when the annual median selenium concentration in water was 8 µg/L (ppb) (Steensen *et al*. 1997), contained up to 7.9 µg/g whole-body selenium and averaged 6.9 µg/g whole-body selenium. Saiki and Palawski's results suggest that water fully meeting the CTR-proposed 5 µg/L (ppb) criterion could result in Delta Smelt with whole-body selenium concentrations exceeding the toxic threshold of 4 µg/g. Delta Smelt spawning sites are almost entirely restricted to the north-Delta channels associated with the selenium-normal Sacramento River and are nearly absent from the south-Delta channels associated with the selenium-contaminated San Joaquin River (USDI-FWS 1996c).

White Sturgeon (*Acipenser transmontanus*), a representative surrogate species for the Green sturgeon, have been the subject of detailed studies within the San Francisco Bay estuary (e.g., Kohlhorst *et al*. 1991). White Sturgeon are long-lived, large-bodied, and demersal (bottomdwelling) fish. For most species of sturgeon, females require several years for eggs to mature between spawnings (Conte *et al*. 1988). White Sturgeon in the San Francisco Bay estuary congregate in Suisun and San Pablo Bays where they remain year-round except for a small fraction of the population that moves up the Sacramento River, and to a lesser extent the San Joaquin River, to spawn in late winter and early spring (Kohlhorst *et al*. 1991). Thus, many individuals of this species remain year-round in San Pablo Bay, the part of the San Francisco Bay estuary with the highest selenium concentrations (up to 2.7 µg/L (ppb)). Kroll and Doroshov (1991) report that developing ovaries of White Sturgeon from San Francisco Bay contained as much as 71.8 µg/g selenium, or 7-times over the threshold for reproductive toxicity (Lemly 1996a, 1996b) of 10 µg/g. Sampling of Pallid Sturgeon (*Scaphirhynchus albus*) in the Missouri River system suggests that normal selenium levels in sturgeon eggs are 2-3 µg/g (Ruelle and Keenlyne 1993) as has been found for many other fish species (see review in Skorupa *et al*. 1996 and in USDI-BOR/FWS/GS/BIA 1998). Thus, White Sturgeon in the San Francisco Bay estuary are producing eggs with as much as 35-times normal selenium content. Based on studies regarding toxicity response functions for avian and fish eggs (e.g., Lemly 1996a,b; Skorupa *et al*. 1996; USDI-BOR/FWS/GS/BIA 1998) it is highly probable that these fish are severely reproductively impaired due to selenium exposure. For example, bluegill embryos resulting from ovaries containing 38.6 µg/g selenium exhibited 65 percent mortality (Gillespie and Bauman 1986).

It is quite plausible that a waterborne concentration of 5 µg/L (ppb) selenium in the San Francisco Bay estuary, as would be allowable for effluent-dominated waters under the CTR-proposed selenium criterion, would result in complete reproductive collapse of sturgeon populations as well as elevated tissue concentrations in Delta Smelt above the 4 µg/g whole-body toxicity threshold.

<u>Selenium effects to Salmonids</u>: Salmonid species considered in this opinion are coho salmon, including Central California Coast and Southern Oregon/Northern California Coast ESUs; chinook salmon, including the Central Valley Spring-Run, the California Coastal, and the Sacramento River Winter-Run ESUs; steelhead trout, including the Central Valley, the Southern California, the South-Central California Coast, the Central California Coast, and Northern California ESUs; Lahontan cutthroat trout; Paiute cutthroat trout, and Little Kern golden trout. Salmonids are considered sensitive to selenium contamination (see review in Lemly 1996a,b). Depending on the form of selenium and the life-stage of fish considered, waterborne concentrations of selenium less than the CTR-proposed 5 µg/L (ppb) concentration can have direct toxic impacts on salmonids (Hodson *et al*. 1980; Moore *et al*. 1990). Hodson *et al*. reported that rainbow trout (*O. mykiss*) eggs respond physiologically (reduced median time to hatch) at selenium (as selenite) concentrations above 4.3 µg/L (ppb).

However, the most dangerous exposure pathway for salmonids, as with other fish, is via dietary bioaccumulation of selenium. As little as 3.2 µg/g selenium in the diet was sufficient to adversely affect early life stages of chinook salmon under controlled conditions (Hamilton *et al*. 1989; 1990). Based on a bioaccumulation factor for dry weight concentrations of selenium in aquatic invertebrates (compared to water) of 1,800 (Pease *et al*. 1992), a concentration of as little as 1.8 µg/L (ppb) selenium could result in salmonid foods averaging more than 3.2 µg/g selenium. That water concentration is already exceeded at times in San Pablo Bay (Pease *et al*. 1992), in the San Joaquin River (Steensen et al. 1997), in the Santa Ynez River (Westcot et al. 1990), in the Pajaro River (Westcot et al. 1990), and in the Salinas River (Westcot et al. 1990). If California's water bodies that currently support salmonid populations were allowed to have concentrations which meet the CTR-proposed selenium criterion of 5 µg/L (ppb), salmonid food organisms would be expected to contain an average of about 9 µg/g selenium (based on a bioaccumulation factor of 1,800). That value exceeds even the 6.5 µg/g dietary toxicity threshold for older life stages of chinook salmon in brackish-water (Hamilton et al. 1989; 1990). Hamilton et al. (1990) also found that dietary exposure of swim-up chinook salmon to 9.6 µg/g selenium resulted in reduced survival after 90 days. The Services thus conclude that currently available data for salmonids do not support the CTR-proposed selenium criterion of 5 µg/L (ppb) as adequately protective of salmonids.

Desert Pupfish: Specific data exist to support a conclusion that the desert pupfish would be unprotected by a chronic selenium criterion of 5 µg/L (ppb). Setmire and Schroeder (1998) report on a field study of sailfin mollies in the Salton Sea area of California. The mollies were chosen as surrogate species in order to assess contaminant threats to the co-occurring endangered desert pupfish. Mollies and pupfish were simultaneously collected from one site and found to contain virtually identical whole-body selenium concentrations (Bennett 1997), which verified the utility of mollies as a surrogate indicator of pupfish exposure. During 1994, mollies were collected from 13 agricultural drains. For 10 of the 13 drains, whole-body selenium concentrations were in the range of 3 to 6 µg/g, a level designated by a panel of selenium researchers as "of concern" for warmwater fishes (USDI-BOR 1993; also see Gober 1994; CAST 1994; Ohlendorf 1996). Two of the other three drains that were sampled yielded mollies averaging >6 µg/g, a level designated by the panel of researchers as exceeding the toxic threshold for warmwater fishes. Unfortunately, contemporaneous measures of waterborne selenium in the sampled drains were not obtained for comparison to the mollie tissue data.

An inquiry with California's Colorado River Basin Regional Water Quality Control Board yielded file data on waterborne selenium for one of the 13 drains sampled for mollies in 1994; however the file data is for water samples collected in 1996 (R. Lukens, Regional Water Board, pers. comm.). Ten monthly (March to December, 1996) measures of waterborne selenium in the Trifolium 12 drain averaged 4.96 µg/L (ppb). Sailfin mollies collected from Trifolium 12 drain in 1994 averaged 3.6 µg/g whole-body selenium, with a maximum of 3.8 µg/g (n=3). If the concentrations of selenium in the drain were roughly the same in 1994 as in 1996, then the CTR-proposed selenium criterion of 5 µg/L (ppb) would be associated with expected pupfish tissue concentrations of selenium at the "level of concern". As discussed in the species effect account for brown pelicans, borderline exposures for direct toxic effects may be particularly hazardous at the Salton Sea because of the recent record of diverse and frequent epizootic events documented for fish and birds at the Sea. It is well established for birds that selenium-induced immune dysfunction occurs at exposure levels below those required for direct selenium-poisoning. Until comparable studies are completed for fish, the safest default assumption is that the results for selenium-induced immune dysfunction documented for birds may also apply to fish.

The CTR-proposed selenium criterion of 5 µg/L (ppb) does not provide the margin of safety necessary to confidently conclude that the criterion would adequately safeguard survival and recovery of desert pupfish. It is also clear that selenium routes of exposure exist for the desert pupfish which put them at risk. The Services therefore conclude that the CTR-proposed selenium chronic criterion for selenium of 5 µg/L (ppb) does not adequately protect the desert pupfish.

Given the above effects analysis, the Services, in our draft opinion dated April 10, 1998 concluded that the selenium criteria as described by EPA in their August 1997 proposed CTR would be insufficiently protective. Implementation of these selenium criteria without future modification could jeopardize the continued existence of the following species: marbled murrelet, California clapper rail, California least tern, light-footed clapper rail, Yuma clapper rail, bonytail chub, coho salmon (California ESUs), delta smelt, desert pupfish, steelhead (California ESUs) Razorback sucker, Chinook salmon (California ESUs), Sacramento splittail, Giant garter snake, and California red-legged frog. It was the Services' opinion that a criterion of 2 µg/L or less would be necessary for protection of these species, that the proposed speciation based acute criterion should not be promulgated and that a selenium criteria revision which considered the bioaccumulative nature and long term persistence of selenium in aquatic sediments and food chains was necessary in the development of new criteria and a site specific guidance for criteria modification.

EPA modifications addressing the Services' April 9, 1999 draft Reasonable and Prudent Alternatives for selenium:

The above effect analysis considers the draft CTR as originally proposed in August of 1997.

EPA has agreed by letter dated December 16, 1999 to modify its action for selenium criteria per the following to avoid jeopardizing listed species.

A. EPA will reserve (not promulgate) the proposed acute aquatic life criterion for selenium in the final CTR.

B. EPA will revise its recommended 304(a) acute and chronic aquatic life criteria for selenium by January 2002. EPA will propose revised acute and chronic aquatic life criteria for selenium in California by January of 2003. EPA will work in close cooperation with the Services to evaluate the degree of protection afforded to listed species by the revisions to these criteria. EPA will solicit public comment on the proposed criteria as part of its rulemaking process, and will take into account all available information, including the information contained in the Services' opinion, to ensure that the revised criteria will adequately protect federally listed species. If the revised criteria are less stringent than those proposed by the Services in the opinion, EPA will provide the Services with a biological evaluation/assessment on the revised criteria by the time of the proposal to allow the Services to complete a biological opinion on the proposed selenium criteria before promulgating final criteria. EPA will *provide the Services with updates regarding the status of EPA's revision of the criterion and any draft biological evaluation/assessment associated with the revision. EPA will promulgate final criteria as soon as possible, but no later than 18 months, after proposal. EPA will continue to consult, under section 7 of ESA, with the Services on revisions to water quality standards contained in Basin Plans, submitted to EPA under CWA section 303, and affecting waters of California containing federally listed species and/or their habitats. EPA will annually submit to the Services a list of NPDES permits due for review to allow the Services to identify any potential for adverse effects on listed species and/or their habitats. EPA will coordinate with the Services on any permits that the Services identify as having potential for adverse effects on listed species and/or their habitat in accordance with procedures described in the draft MOA published in the Federal Register at 64 Fed. Reg. 2755 (January 15, 1999) or any modifications to those procedures agreed to in a finalized MOA.*

C. EPA will utilize existing information to identify water bodies impaired by selenium in the State of California. Impaired is defined as water bodies for which fish or waterfowl consumption advisories exist or where water quality criteria necessary to protect federally listed species are not met. Pursuant to Section 303(d) of the CWA, EPA will work, in cooperation with the Services, and the State of California to promote and develop strategies to identify sources of selenium contamination to the impaired water bodies where federally listed species exist, and use existing authorities and resources to identify, promote, and implement measures to reduce selenium loading into their habitat.

Services' assumptions regarding EPA's modifications for removing jeopardy.

The Services assume the following:

Contaminant threats to listed species can be reduced through application of appropriately protective water quality criteria to the water bodies occupied by listed species.

The presumptive adverse effect threshold for identifying effects to listed species, is either the exceedance of the criteria proposed in this opinion to protect listed species, or demonstrated effects below those proposed criteria concentrations for the priority pollutant under consideration.

The adjustments of criteria as proposed in the CTR by EPA for water bodies occupied by species considered in this opinion will be consistent with the effects analysis in this biological opinion unless new information is developed by EPA.

EPA adjustments of criteria will occur within agreed upon time frames.

The future adjustment of the selenium criteria will consider the bioaccumulative nature of selenium in aquatic systems, not just the waterborne toxicity and will result in a lowering of the criteria. Thus listed fish and wildlife species which are aquatic system foragers will be protected by the future criteria and the procedures for site specific adjustments.

The reservation of the acute aquatic life criterion for selenium will result in the criterion being withheld from use for regulation by the State and Regional boards.

Mercury

Assessment of Adequacy of Proposed Mercury Criteria to protect listed species

Aquatic Life Criteria for Mercury

The EPA has proposed an acute aquatic life criterion (criterion maximum concentration or CMC) for mercury of 1,400 ng/L and a chronic aquatic life criterion (criterion continuous concentration or CCC) of 770 ng/L. These criteria are based upon dissolved concentrations. EPA's proposed mercury criteria for aquatic life are based on the assumed waterborne toxicity of dissolved forms of mercury to aquatic organisms that exclusively live within the water column. The Services believe the proposed CTR aquatic life criteria for mercury will not protect listed fish from either dietary toxicity or maternal transfer of methylmercury to young. Promulgation of a dissolved mercury criteria also fails to consider the effects upon biota of particulate methylmercury and particulate inorganic mercury. Regulation of mercury on a dissolved basis only for aquatic life ignores the role of particulate mercury in the cycling of mercury in aquatic ecosystems and the need to consider the dietary pathway for mercury accumulation in aquatic life.

The aquatic life mercury criteria of 770 ng/L(chronic) and 1,400 ng/L (acute) are so high as to effectively be without value for controlling mercury in even the most severely mercury-impaired California water bodies. Concentrations above the chronic criterion concentration in the dissolved form are virtually unmeasured in the California environment, even though those environments contain numerous water bodies with direct mercury discharges. In a broad survey of mercury in freshwater systems in California and other areas, Gill and Bruland (1990) failed to locate any water bodies containing levels of mercury above or approaching these dissolved criteria although many of these same water bodies were mercury impaired due to elevated mercury concentrations in fish.

Two California examples illustrate why the chronic and acute criteria for mercury are unreasonably high with no potential to impact or control mercury concentrations. Walker Creek is potential habitat for both steelhead and the California red-legged frog and discharges into Tomales Bay. The Gambonini mine, an abandoned mercury mine, produces concentrations of total mercury in unfiltered water from Walker Creek as great as 100,000 ng/L, yet dissolved concentrations in the creek only range from 20 to 100 ng/L (Whyte 1998). These concentrations are of great concern as evidenced by Regional Board activity to cleanup and restore the mine site, but obviously well below EPA's proposed chronic aquatic life criterion of 770 ng/L. The aquatic life criteria of EPA would likely be controlling for Walker Creek as fish consumption from the creek is not a beneficial use and Walker Creek lacks a MUN designation (use for municipal drinking water purposes). Long *et al*. (1990) unexpectedly found toxicity to three species in sediments of Tomales Bay (their control site) and found the sediments of Tomales Bay devoid of the more sensitive crustaceans corroborating toxicity test results. This toxicity was best explained by the mercury as it was the only toxicant present at elevated concentrations.

Davis Creek Reservoir in the Cache Creek watershed is another example. This site is highly contaminated by mercury. This reservoir is also potential foraging habitat for the bald eagle as up to 60 eagles winter in this drainage. Davis Creek Reservoir has dissolved organo-mercury concentrations of 60 picomoles (12 ng/L) associated with a total dissolved mercury concentration of 16 ng/L and total unfiltered mercury concentrations of 26 to 32 ng/L. These concentrations of mercury in water were associated with fish tissue concentrations of 2.5 µg/g (ppm) wet weight (Gill and Bruland 1990). The fish mercury concentrations present significant risk to any foraging eagles. The proposed chronic aquatic life criterion for mercury at this reservoir, which probably is not covered by human health criteria as it is a water supply for processing gold ores, are an order of magnitude above all concentrations observed at this site.

Human Health Mercury Criterion (for Protection of Fish and Wildlife)

Since the aquatic life criteria clearly are not protective of fish and wildlife, the Services have evaluated whether the lower human health criterion of 50 ng/L would be protective. The Services find that the human health criterion for mercury will not protect listed fish or wildlife species. The EPA's biological evaluation (BE) (EPA 1997a) states that the human health criterion of 50 ng/L (total mercury), will offer protection of aquatic life in the water column and to non-aquatic piscivorous birds and mammals. Footnote a, page 42204 of the August 5, 1997, Federal Register (EPA 1997c) notes that for mercury "The fish tissue bioconcentration factor (BCF) from the 1980 documents was retained..." Unfortunately these bioconcentration factors were derived prior to modern developments in analytical chemistry that permit more accurate determination of concentrations of mercury in water. The resulting 1980 bioconcentration factor of 7,342.6 used to derive the proposed mercury criterion is neither appropriate, accurate, or reflective of real world environmental mercury concentrations in water. As a result of improvements after 1988 in water chemistry for mercury, it is now clear that mercury concentrations are far lower than was thought in 1980, and consequently bioconcentration factors and bioaccumulation factors have been revised and are now known to be far higher than those used by EPA in the CTR. This scientific information is well known and has been available for a decade (EPA 1997b; Bloom 1989; Bloom and Fitzgerald 1988). The Services therefore find the statement within the biological evaluation for the CTR that "the human health criteria for mercury will protect listed wildlife" is not supported by the best scientifically and commercially available data. In addition the Services also anticipate the criterion will not be sufficiently protective of the potential for maternal transfer of harmful concentrations of mercury to vertebrate eggs and embryos.

EPA indicated during informal consultation that the human health criterion for mercury may be changed in the near future. Should an appropriate bioaccumulation factor for mercury be applied at some future date to develop a human health criterion either in water or in fish tissue, it is not necessarily clear that such a criterion designed for protection of human health alone would also afford adequate protection to listed species. Because fish and wildlife typically have more restricted diets than humans, they are more susceptible to local contamination. Wildlife, particularly piscivorous wildlife, are often at greatest risk from mercury exposure within any ecosystem (EPA 1997b). Even with appropriate bioaccumulation factors for evaluating human fish consumption, the use of humans as the surrogate species to represent the bioaccumulation hazards presented to wildlife is not scientifically supported. "Fish-eating wildlife are more vulnerable to the adverse effects of mercury than are humans for two reasons: (1) fish compose a higher proportion of their diet: and (2) wildlife are more dependent on their reflexes to survive." (A. Kuzmack, EPA, pers comm., February 17, 1998).

Hazards to Species: Toxicity and Bioaccumulation

Toxicity

Mercury is a trace element with no known essential biological function. Mercury in environmental waters can exist in many forms including elemental form (Hg^0), dissolved and particulate ionic forms, and dissolved and particulate methylmercury (Gill and Bruland 1990; Vandal et al 1991; Mason and Fitzgerald 1993). Methylmercury may be formed either in the water column or in sediment.

Methylmercury is the most toxic and the most bioaccumulated form of mercury. Intestinal absorption of inorganic mercury is limited to a few percent while absorption of methyl mercury is nearly complete (Scheuhammer 1987). Inorganic mercury appears to have the greatest effect upon the kidneys, while methylmercury is a potent embryo and nervous system toxicant. Methylmercury readily penetrates the blood brain barrier, produces brain lesions, spinal cord degeneration, and central nervous system dysfunctions. The proportion of total mercury which is found as methylmercury in biota increases with trophic level approaching 100% at trophic levels 3 and 4. Methylmercury is biomagnified between trophic levels in aquatic systems and in proportion to its supply in water (Wattras and Bloom, 1992). It is appropriate therefore to focus attention on the toxicity of methylmercury, particularly in higher trophic level organisms (Nichols et al., 1999).

Fish: In the 1995 update to Water Quality Criteria Documents for Mercury, EPA stated that the estimated chronic value for effects to coho salmon was 370 ng/L and 420 ng/L for rainbow trout. EPA further explicitly acknowledged that the CCC of 908 ng/l (the CCC in favor as of 1995) might not adequately protect these species (EPA 1995b). In the subsequent CTR, EPA has reduced the proposed CCC for mercury to 770 ng/L. However, this revised number also remains unprotective for federally listed salmonid species. For example, in flow through bioassays, fertilized eggs of rainbow trout suffered 100 percent mortality after 8 day exposures to 100 ng/L concentrations of inorganic mercury (Birge *et al.* 1979). In a review of mercury toxicity to fish, Wiener and Spry (1996) noted direct adverse effects in a variety of fish species on behavior, growth, histology, reproduction, development and survival of fish at concentrations well below the proposed chronic criterion. Fish species tested with adverse effects below criteria concentrations include trout and fathead minnows.

Amphibians and Reptiles: Reptiles and amphibians remain the least studied vertebrates for mercury toxicity. Amphibian eggs and embryos may be the most vulnerable to direct waterborne concentrations. A dose of 50 µg/L applied to the embryos of the frog (*Xenopis laevis*) reduced survival by 50 percent after 4 days of treatment, and to 0 percent after 7 days. Surviving embryos showed disruption of morphogenesis, neurophysiology, and neuroimmune regulation (Ide et al, 1995). Rao and Madhyastha (1987) reported that the LC_{50} (the lethal concentration in water that kills 50 percent of the test organisms) of mercuric chloride to the tadpoles of (*Microhyla ornata*) ranged from 2.04 mg/L (24 hour) to 1.12 mg/L (96 hour). In leopard frog (*Rana pipiens)* embryos methylmercury concentrations of 40 µg/L and above were lethal (Dial 1976). Adverse affects were seen at concentrations as low as 10 µg/L. While these concentrations are well above the current criteria, they are also acute exposures of four to five days exposure and reflect no maternal transfer of methylmercury. Chronic studies in frogs of the effects of mercury contamination are generally lacking. The Service was not able to locate any published acute or chronic studies of mercury in snakes.

Birds: Symptoms of acute methylmercury poisoning in birds include reduced food intake leading to weight loss, progressive weakness in wings and legs, difficulty flying, walking, and standing, and an inability to coordinate muscle movements (Scheuhammer 1987). In addition to well-identified acute effects of mercury at high concentrations, there are also significant adverse effects at lower tissue-mercury concentrations representing chronic mercury exposures. Embryological exposure may possibly lead to impaired hearing, or altered behavior (Heinz 1979). Impaired or tunnel vision has been demonstrated in other adult vertebrate species (humans, and monkeys) (Wolfe *et al*. 1998). These sensory deficits could lead to reduced ability to locate and catch prey for the bald eagle or least tern, to impaired ability to find a mate through auditory clues in the clapper rail and an impaired ability to detect and escape predators in all species. In great white herons liver-mercury contamination > 6 µg/g correlated with mortality from chronic diseases (Sundloff *et al*. 1994).

Reproduction is one of the most sensitive toxicological responses, with effects occurring at very low dietary concentrations. Concentrations in the egg are typically most predictive of mercury risk to avian reproduction, but concentrations in liver have also been evaluated for predicting reproductive risk. The documented effects of mercury on reproduction range from embryo lethality to sublethal behavioral changes in juveniles at low dietary exposure. Reproductive effects in birds typically occur at only twenty percent of the dietary concentrations which produce lethal effects in adult birds (Scheuhammer 1991). Effects of mercury on reproduction are likely occurring in San Francisco Bay populations of birds due to concentrations of mercury observed in eggs including the least tern and the California clapper rail (Schwarzbach, et al, 1997).

Embryos of birds are extremely sensitive and vulnerable to relatively minute concentrations of mercury in the egg. Almost all of the mercury in bird eggs is thought to be methylmercury (Wolfe et al, 1998). Toxic effects of mercury in bird eggs have been documented by many investigators in both laboratory and field studies (Barr 1986; Birge *et al*. 1976; Fimreite 1971; Fimreite 1974; Heinz 1974; Heinz, 1975; Heinz 1979; Hoffman and Moore 1979; Finley and Stendell 1978; Tejning 1967; etc.). Fimreite estimated the threshold level in eggs for toxic effects to nest success in a field study of common terns to be between 1.0 and 3.6 µg/g. Heinz (1979) was able to examine more subtle behavioral effects in mallard ducklings fed methylmercury. Heinz fed ducks 0.5 µg/g mercury over 3 generations and found decreased reproductive success and altered behavior of ducklings. The Heinz study, remains the benchmark study which establishes the lowest observed adverse effect concentration in avian diet of 0.064 mg mercury/kg (body weight)/day (Sample *et al*. 1996). The mean mercury concentration in eggs associated with these observations was 0.86 µg/g fresh wet weight (fww). Fimreite in a 1971 mercury feeding study with ring-necked pheasants found significant reduction in hatchability associated with mercury levels between 0.5 and 1.5 µg/g. The Fimreite study establishes the lowest adverse concentration observed in avian eggs. Hoffman and Moore (1979) externally applied mercury to mallard eggs and found a dose related effects on survival, growth and abnormal development. The lowest dose applied which effected survival was 27 micrograms. Given an average mallard egg weight of 55 grams this dose corresponds to about 0.5 µg/g.

Reproductive effects may extend beyond the embryo to adversely effect the juvenile survival rates. Mercury in the eggs of mallards caused brain lesions in hatched ducklings. Mallards were fed 3.0 µg/g methylmercury dicyandiamide over two successive years. Mercury was accumulated in the eggs to an average of 7.18 and 5.46 µg/g on a wet weight basis in 2 successive years. Lesions included demyelination, neuron shrinkage, necrosis and hemorrhage in the meninges overlying the cerebellum (Heinz 1975). Bouton *et al*. (1999) reported significant behavioral effects on juvenile egrets in captive feeding studies at both high (5,000 µg/g) and low (500 µg/g) dose concentrations of mercury in the diet. Effects in the low dose group included lethargy, reduced motor skills, reduced packed cell volume, decreased appetite and changes in time spent standing vs. sitting. Low dose birds were also less likely to hunt and more likely to seek shade. An observation of significance in the Everglades appears to be that once feather growth ceases, mercury may pose a greater threat to fledgling birds as circulating levels of mercury in the blood are no longer sequestered in the growing feathers. This may be a critical stage for birds as they must learn to hunt and survive on their own at this time.

<u>Mammals</u>: Methylmercury toxicity in mammals is primarily manifested as central nervous system damage, sensory and motor deficits, and behavioral impairment (Wren et al, 1988; Wren et al., 1986). Animals initially become anorexic and lethargic. Muscle ataxia, motor control deficits, and visual impairment develop as toxicity progresses, with convulsions preceding death (O’Conner and Nielsen, 1981; Wobeser et al., 1976). Smaller carnivores are more sensitive to methylmercury toxicity than larger species, as reflected in the shorter time to onset of toxic signs and time to death. Dietary concentrations of 4,000 to 5,000 µg/g methylmercury were lethal to mink and ferrets within 26 to 58 days, whereas otters receiving the same concentration survived an

average of 117 days (Wren et al., 1988; Wren, 1986). Methylmercury is readily transferred across the placenta and concentrates selectively in the fetal brain. Mercury concentrations in the fetal brain were twice as high as in the maternal brain for rodents fed methylmercury (Yang et al., 1972). Reproductive effects of methylmercury in mammals range from developmental alterations in the fetus, which produce physical or behavioral deficits after birth, to fetal death (Eccles and Annau,1987; Chang and Annau, 1984).

The behavioral deficits produced by prenatal exposure to methylmercury are known mostly from work with rodents and monkeys. Rats and mice exposed via the diet or by gavage at various times during gestation period showed retarded righting reflex, impaired or retarded swimming ability, decrease in spontaneous activities, impaired maze and avoidance learning, and deficits in operant learning (Shimai and Satoh, 1985). The use of primates to study the behavioral teratology of methylmercury has permitted more extensive investigations. Infant crab-eating macaques (*Macaca nemestrina*) born to females exposed to 50 or 70 µg/g/day of methylmercury had blood methylmercury levels of 1,690 µg/L at birth and 1,040 µg/L at the time of testing. The exposed macaques had significant deficits of visual recognition memory compared to controls (Gunderson et al., 1988). Cynomolgus monkeys (*Macaca fascicularis*) born to females given 50 µg/kg/day methylmercury showed more non-social passive behavior and less social play than non-exposed monkeys (Burbacher et al., 1990). Adult macaques dosed with 0.24 to 1.0 µg/g methylmercury at twice-weekly intervals for up to 73 weeks first experienced constriction of visual field, as has been reported by methylmercury-intoxicated humans, an effect that was reversible if exposure was discontinued. At higher or more prolonged doses, visual field constriction became permanent, and visual thresholds were altered, reflecting damage to neurons in the visual cortex (Merigan et al., 1983).

Bioaccumulation of mercury

Both organic and inorganic mercury bioaccumulate, but methylmercury accumulates at greater rates than inorganic mercury. Most mercury in fish or wildlife organisms is in the form of methylmercury (Bloom, 1995) as this form is more efficiently absorbed (Scheuhammer, 1987) and preferentially retained (Weiner, 1995). Much of the inorganic mercury found in some organisms such as procellariiform birds (albatrosses, shearwaters, and petrels) may have actually been originally accumulated as methylmercury and then demethylated by the organism. The bacterial rates of production of methylmercury in water and sediment matrices ultimately determines the potential of an aquatic system to develop a mercury bioaccumulation problem. Food chain transfer is the most important exposure pathway in all ecosystems (EPA, 1997b). Methylmercury is one of the rare compounds which not only bioaccumulates but also biomagnifies across trophic levels such that field measured BAFs for methylmercury are commonly in the millions for top trophic level fish (Nichols et al., 1999).

Table 5. Median bioaccumulation factors for fish presented in the Mercury Study Report to Congress (EPA, 1997b).

Hg form	BAF trophic level 3 fish*	BAF trophic level 4 fish*
Total mercury	124,800	530,400
Methyl mercury	1,600,000	6,800,000

Aquatic ecosystems tend to have higher rates of bioaccumulation and biomagnification than do terrestrial ecosystems (EPA, 1997b). Explanations for this phenomenon include the fact that fish store most mercury as methylmercury in their muscle while mammals and birds store much of their methylmercury burden in feathers and fur, items poorly digested or rarely eaten. Aquatic systems have more complex food webs and more trophic levels, and the primary producers in aquatic systems may themselves accumulate more mercury from water and sediment than do soil based primary producers in terrestrial systems (EPA, 1997b). Top predators in aquatic systems therefore are at greatest risk from mercury bioaccumulation. Mercury concentrations in blood greater than 1,000 µg/L and in eggs greater than 0.5 µg/g are considered harmful. In liver 5 µg/g is considered a conservative threshold for potential adverse effects to waterbirds (Wolfe *et al.*, 1998).

Listed wildlife species which are high trophic level predators in aquatic systems of California include one mammal, six birds, and two reptiles. These are the southern sea otter, bald eagle, California least tern, California brown pelican, California clapper rail, light-footed clapper rail, Yuma clapper rail, giant garter snake, and San Francisco garter snake.

Bioaccumulation Hazards of Mercury to Fish: Diet is the primary route of methylmercury uptake by fish in natural waters, contributing more than 90 percent of the methylmercury accumulated. The assimilation efficiency for uptake of dietary methylmercury in fish is probably 65 to 80 percent or greater. To a lesser extent, fish may obtain mercury from water passed over the gills, and fish may also methylate inorganic mercury in the gut (Wiener and Spry, 1996). Developing embryos are the most vulnerable life stage to mercury exposure. In all vertebrates, including fish, the transfer of methylmercury to the embryo represents the greatest hazard. In addition to the hazard to top avian reptilian and mammalian predators in aquatic systems, fish and amphibian species, particularly long lived species, may be at risk from mercury bioaccumulation and biomagnification. Even in fish, "methylmercury derived from the adult female probably poses greater risk than waterborne mercury for embryos in natural waters" (Wiener, 1995). This is likely true for amphibians, including the federally listed California red-legged frog. For this reason alone mercury criteria needed to protect aquatic life must consider maternal bioaccumulation rates in adult fish. Sublethal and lethal effects on fish embryos are associated with mercury residues in eggs that are perhaps 1 percent to 10 percent of the residues associated with toxicity in adult fish (Weiner, 1995). Mercury intoxicated rainbow trout have between 4 and 30 µg/g in whole bodies, while intoxicated embryos contain 0.07 to 0.1 µg/g (Weiner, 1995). Listed fish species with long life spans are potentially at risk from mercury bioaccumulation. Listed fish species potentially at risk of mercury bioaccumulation at concentrations permissible under the CTR criteria include listed salmonids, as well as the bonytail chub, razorback sucker, shortnose sucker, Lost River sucker and the Sacramento splittail.

While the Mercury Study Report to Congress (EPA, 1997b) generated data on a range of national bioaccumulation factors, that report emphasized the value of developing site specific and field derived bioaccumulation factors when developing criteria for specific regions. Factors which affect the site specific bioaccumulation factors within a given ecosystem are many and varied. Factors proposed to effect bioaccumulation rates include the number of trophic levels present and food web structure of the aquatic ecosystem, the abundance of sulfur reducing bacteria and the concentration of sulfates, dissolved oxygen, water temperature, organic carbon availability, pH, the nature of the mercury source and other parameters (Porcella *et al.*, 1995).

In the absence of site-specific bioaccumulation factors for mercury, EPA recommended default BAFs using the bioaccumulation factors in (EPA 1997b) (see table 5). In order to develop a site-specific bioaccumulation factor, concomitant measurements of mercury in fish and water are needed. Water measurements need to employ ultra clean sampling techniques (Gill and Bruland, 1990) and picomolar quantification methods of mercury determination in water (Bloom, 1989). In this regard there is a clear need for EPA to promulgate a new analytical method for mercury under the CWA which will have appropriate detection limits in water and address the problems of sample contamination in the current method.

While EPA's current human health criterion per the draft CTR continue to use bioconcentration factors from older lab studies, the EPA used bioaccumulation factors to assess ecological and human health risk for the Mercury Study Report to Congress. That report recommended the use of field derived bioaccumulation factors. The Services are aware of currently available, scientifically defensible field data which may likely permit calculation of site-specific bioaccumulation factors for mercury at a number of California locations. These locations include Clear Lake, Lake Nacimiento, Cache Creek, Walker Creek, Marsh Creek, Lake New Almaden, the New Almaden Mine area, Marsh Creek, the Sacramento River, the Petaluma River, Central San Francisco Bay, South San Francisco Bay (Cal/EPA, 1997), Davis Creek Reservoir, Snake Creek, Lake San Antonio and Las Tablas Creek (Gill and Bruland, 1990) as well as the Yuba River, the Feather River, the American River, and the Cosumnes River (Slotten et al., various reports to Central Valley Regional Board 1999). Ongoing studies funded by CalFed may support the development of such bioaccumulation factors for the Sacramento/San Joaquin delta area within the next two years.

<u>Bioaccumulation Hazards of Mercury to Reptiles and Amphibians</u>: The maternal transfer of methylmercury is likely to occur in amphibians and reptiles as it does in fish and birds. The Service is not aware of any available data on adverse effect residue concentrations in amphibians or reptiles which would at this time permit a calculation of an effect threshold for the red-legged frog, giant garter snake or San Francisco garter snake. The USFWS has conducted a study with the Biological Resource Division of United States Geologic Survey (USGS) within the Cache Creek drainage on mercury bioaccumulation within the watershed. Results from this study show

maximum whole body mercury concentrations in foothill yellow-legged frogs (*Rana boylii*) of 0.79 µg/g ww and 1.29 µg/g in bull frogs *(Rana catesbeiana*). In the absence of specific amphibian data the Services would recommend applying a fish model to assessing the risk to amphibian eggs laid in water and an avian risk model to evaluate impacts to predatory snakes in aquatic environments.

Bioaccumulation Hazards of Mercury to Birds: Mercury is transferred to avian eggs in proportion to the maternal dose (Walsh, 1990). Almost all of this mercury is methylmercury (Schwarzbach et al., 1997; Wolfe et al., 1998). While some of this egg mercury represents maternal body burden, much of it reflects maternal diet during the immediate pre-laying period. Trophic position, and mercury sources of contamination on the breeding grounds are the most significant factors in predicting mercury concentrations in bird eggs. Only relatively minute mercury concentrations are required to impair eggs.

There is substantial data on mercury in avian eggs of a number of species throughout California. A few of these are federally listed species. These data are summarized in the mercury appendix of this document. These data show that exclusively piscivorous birds typically face the greatest risk, followed by partially piscivorous birds. Clapper rails, a benthic omnivore and partially piscivorous bird, can also achieve very high levels of egg mercury where sediment methylmercury production is high. The California clapper rail in south San Francisco Bay has the maximum single egg concentration of mercury measured in any California egg at 2.5 µg/g (fww) (Schwarzbach et al., 1997). Other listed species for which egg mercury data exist in California include the light-footed clapper rail, the Yuma clapper rail, and the least tern. Data for eleven different bird species (Schwarzbach et al., 1997) overwhelmingly show that birds nesting in San Francisco Bay, including the least tern and the California clapper rail, are at much greater risk of mercury bioaccumulation than their cohorts nesting elsewhere in California. Data also indicate that Elkhorn Slough is nearly equally mercury impaired with regard to excessive bioaccumulation of mercury in fish eating birds (Caspian terns). The effects of the CTR mercury criteria, as proposed, will leave this condition unchanged.

We are unaware, at this writing, of bald eagle egg data for California. The only mercury data available to the Services is blood mercury data from the Klamath Basin (Frenzel and Anthony, 1989). These data showed a mean concentration of 2,290 µg/L. This is a concentration 7.5 times higher than bald eagles kept in captivity (Frenzel and Anthony, 1989) and well over the concentration of 1,000 µg/L suggested as harmful.

Bald eagles in California are likely to be the species with the greatest concentrations of mercury in eggs as nesting pairs occur at mercury contaminated reservoirs throughout the Coast Range and eagles occupy the highest trophic position in those systems. The proposed CTR mercury criteria will leave this condition unchanged, and likely not protect eagles from bioaccumulation. This conclusion is supported by the Mercury Study Report to Congress (EPA, 1997b) which developed an estimated total (as dissolved) mercury water concentration of 1.05 ng/L to protect the bald eagle from the bioaccumulation of mercury throughout its range. While site-specific factors may vary, it is unlikely that site specific bioaccumulation factors would lead to a new criterion above EPA's 50 ng/L human health criterion proposal.

Reproduction is the most sensitive endpoint and mercury accumulated in egg is the best predictor of mercury risk to embryo survival. Egg mercury measurements are superior to measurements of mercury concentration in potential prey items as proportions of possible prey in diet are not always known. One of the significant factors enhancing risk of mercury to the avian embryos is the lack of any protective detoxification mechanism in the avian egg once mercury is deposited there. The lowest adverse effect concentration in avian eggs is 0.5 µg/g (fww) (Fimreite, 1971).
The no adverse effect concentration in avian eggs is unknown. Mean fresh wet weight mercury concentration in failed eggs of the California least tern in San Francisco Bay in 1994 was 0.74 µg/g (fww). California clapper rail failed eggs in 1992 had a mean of 0.63 µg/g mercury in eggs.

A mercury bioaccumulation factor (BAF) for water or sediment to egg may be derived on a site- and species-specific basis. The USFWS has derived a mercury BAF for water to least tern eggs in San Francisco Bay (described below). A sediment BAF of 1,435 (on a ww basis) for methylmercury accumulation in California clapper rail eggs from sediment has been previously described elsewhere (Schwarzbach et al., 1996). These BAFs can be used in equations together with an estimated no observable adverse effect level (NOAEL) for mercury in avian eggs to estimate a safe concentration in water or sediment for the respective species. Alternatively, one may use the equations described and used in the Mercury Study Report to Congress (USEPA, 1997b) to derive an estimate of a safe concentration for mercury in water. These equations rely on dietary concentrations and bioaccumulation factors to fish together with a safe dietary daily dose estimate. These two methods are compared below to derive a water criterion for mercury protective of the least tern in San Francisco Bay. All of these methods suggest that for the mercury criterion to be protective of wildlife the concentrations would need to be substantially lower than proposed in the CTR.

Bioaccumulation Hazards of Mercury to Mammals: Mammals that forage within aquatic ecosystems are at greatest risk of mercury bioaccumulation. In mammalian tissues the greatest concentrations of mercury are usually found in liver and kidney. Mammals that consume fish, or mammals that consume mammals that consume fish are generally at greatest risk.

O'Conner and Nielsen (1981) found that length of exposure was a better predictor of tissue residue level than dose in otters but higher doses produced an earlier onset of clinical signs. A dose of 0.09 µg/g body weight (2 µg/g in diet as methylmercury) for 181 days was enough to produce anorexia and ataxia in two of three otters in a feeding study of river otters (*Lutra candensis*). Associated liver residues were 32.6 µg/g (O'Conner and Nielsen, 1981). Concentrations of 21 to 23 µg/g in kidney and liver were associated with liver and kidney histologic alterations in Rhesus monkeys (Rice *et al*., 1989). Muscle ataxia, motor control deficits, and visual impairment develop as toxicity progresses with convulsions preceding death. River otters fed 8 µg/g methylmercury died within a mean time of 54 days. Associated liver concentrations were 32.3 µg/g (O'Conner and Nielsen, 1981). While 8 µg/g or even 2 µg/g seems a higher concentration than what southern sea otters are likely to encounter in their prey, the duration of sea otter exposure in the wild is life-long. As indicated by mercury residues in sea otter livers, and laboratory feeding studies showing the importance of duration of dose, life long multi-generation exposures to elevated mercury in diet may produce elevated mercury in tissues and the attendant adverse effects. A long term exposure to mercury in the diet may result in the most exposed individuals experiencing decreased motor coordination, reduced sensory and mental acuity, impaired kidney function, ataxia, anorexia and even death.

In California the listed mammal which may be at greatest risk from mercury is the southern sea otter. The California sea otter population is endangered and population levels are declining. Sea otters forage in the nearshore marine environment, from the intertidal to depths exceeding 60 feet. At Elkhorn Slough, otters are often found foraging well within the slough. While sea otters, unlike river otters, are not exclusively piscivorous, they are opportunistic foragers on mussels, snails, clams, crabs, squids, sea urchins, star fishes and slow moving fish among other organisms (Estes, 1980; Zeiner, 1990). In captivity sea otters consume 15 to 35 percent of their body weight in food daily (Lensink 1962). The metabolic demands of sea otter existence may thus result in elevated risk of sea otter contaminant loading, although a lower fraction of the mercury consumed by omnivores is likely to be methylmercury. Wren (1986) suggested normal mercury concentrations in river otter livers were 4 µg/g (fww) or below. Livers collected from sea otters found dead along the central California coast had a maximum mercury concentration of (60 µg/g) (Mark Stephenson pers comm 1998). Of 125 sea otter livers examined for mercury on the California coast, 56 had concentrations greater than 4 µg/g and 30 had concentrations over 10 µg/g. Four had concentrations over 30 µg/g.

Estimates of Mercury Criteria Protective of listed Fish and Wildlife Species:

The proposed CTR as published in the federal register states: "This rule is important for several . . . reasons. Control . . . is necessary to achieve the CWA's goals and objectives. Many of California's . . . waters have elevated levels of toxic pollutants. Recent studies . . . indicate that elevated levels of toxic pollutants exist in fish tissue which result in fishing advisories or bans." Many of these advisories exist due to mercury bioaccumulation which is elevated in a number of water bodies in California. San Francisco Bay trophic level 3 fish average 0.140 µg/g (San Francisco Regional Water Quality Control Board, 1995), a level 2.7 times the national average and 1.8 times the concentration of methylmercury in trophic level 3 fish of 0.077 µgHg/g, (Nichols et al., 1999) associated with EPA's wildlife value in water. It is the Services' opinion that the effect of the proposed action (CTR) would be to effectively leave this condition (fish advisories and elevated mercury in trophic level 3 fish) unlikely to change. Further it is the opinion of the Services that sufficient data is available to allow preliminary calculation of protective criteria in California which take into account site-specific bioaccumulation to fish.

Below we calculate a bioaccumulation based mercury criterion to protect salmonids and a bioaccumulation based criterion to protect the California least tern in San Francisco Bay. While additional research would no doubt improve the confidence in the calculations below, it is readily apparent from both the Mercury Study Report to Congress, and our calculations with available data, that the proposed criteria in the draft CTR would be too high to protect many top predators in aquatic systems, including some listed species.

Estimating a Bioaccumulation Based Effect Concentration in Salmonids:

Neither the aquatic life criteria nor the human health criterion for mercury address the hazard of bioaccumulation of mercury to fish themselves, but only to the human consumers of fish. Where fish effects are considered in the aquatic life criterion it is only through direct waterborne toxicity. Mercury residue concentrations have been observed in mercury intoxicated trout of 4 µg/g (Wiener, 1995). Brook trout with whole body concentrations of 2.7 µg/g exhibited reproductive impairment (McKim et al., 1976). Using the default BAF_4 from USEPA (1997b) we derive below a water concentration of 5 ng/L total dissolved mercury which could be associated with reproductive impairment.

$$\text{Adverse effect concentration [T-Hg] in water for trophic level 4 fish} = \frac{\text{Toxic to fish Hg whole body conc.}}{BAF_4}$$

$$= \frac{2{,}700 \text{ ng/g}}{530{,}400 \text{ ng/g/µg/L}}$$

$$= 0.005 \text{ µg/L}$$

$$= 5 \text{ ng/L}$$

An examination of the data from rivers tributary to San Francisco Bay in 1996 (SFEI, 1997b) indicates that the dissolved component of total mercury varies seasonally but averages 19 percent 13 percent and 7.5 percent for the Sacramento, Napa, and Petaluma Rivers respectively. Using these mean ratios, corresponding total mercury effect concentrations in unfiltered water of these northern California rivers would be estimated at 26, 38, and 66 ng/L. Appropriately protective criteria should be below the effect concentrations. EPA's 51 ng/L criterion for human health would be below only the Petaluma River effect estimate. Dividing the effect concentrations by a safety factor of 2 would result in a fish protective criterion lower than the CTR human criterion (51 ng/L) in all three rivers.

Estimating a Bioaccumulation Based Mercury Criterion for Wildlife Species: Comparison of Two Estimates Using an Oral Dose Model and an Egg BAF Model in the California Least Tern in San Francisco Bay.

A wildlife criterion is defined by EPA to be the highest concentration of a substance that causes no significant reduction in growth, reproduction, viability or usefulness of a population of animals exposed over multiple generations. For a species listed as endangered the failure to achieve concentrations at or below an appropriate wildlife criterion may be critical to future survival of the species. While the final Mercury Study Report to Congress (USEPA, 1997b) developed a wildlife criterion for the bald eagle, the Services offer the following calculations using California specific data for the least tern and San Francisco Bay to illustrate that EPA's Great Lakes wildlife criteria are more nearly appropriate than the human health criterion suggested by EPA as protection for California's listed wildlife species.

For the purposes of example in this opinion, the Services have taken mercury data in water and trophic level 3 fish (shiner surf perch, a prey item of the California least tern) from the San Francisco Bay Regional Monitoring Program. Water mercury data collected by the San Francisco Estuary Institute (SFEI) in the spring of 1994 from 6 locations within central San Francisco Bay were also used. Fish mercury concentrations in shiner surf perch were matched with the two or three closest water sampling locations due to the fact that fish are mobile and water concentrations vary. Springtime water values were used because this is when California least terns are nesting in the bay (April BAFs also appear to generally be intermediate between February and August values in the Central Bay). Dry weight and wet weight bioaccumulation factors for mercury in shiner surf perch were calculated from the Regional Monitoring Program's data and are presented in Table 6.

Table 6. Dry weight and wet weight bioaccumulation factors for trophic level 3 (BAF_3)[@] fish in Central San Francisco Bay.

Fish Collection Location	Representative Water Collection Points	BAF_3 (DW) Total unfiltered Hg	BAF_3 (WW) Total unfiltered Hg
Richmond Harbor	Point Isabel, Red Rock ,Yerba Buena	137,311	30,483
Berkeley Pier	Point Isabel, Red Rock, Yerba Buena	118,098	27,163
Oakland Inner Harbor	Yerba Buena, Alameda	181,840	42,551
Oakland Middle Harbor	Yerba Buena, Alameda	72,290	20,530
Double Rock	Alameda, Oyster Point	76,319	18,088
Islais Creek	Yerba Buena, Alameda, Oyster Point	53,917	13,425
Geometric Mean for central SF Bay		97,723 [!]	23,659

@ Trophic level 3 fish are non-piscivorous foraging fish.

+ Mercury Data from 1994 Regional Monitoring Program (RMP) in SF Bay winter and spring of 1994(SFEI, 1997).

! Geometric mean dry weight factor is used in least tern criterion equation because the diet estimate for terns was based upon allometric equations using dry weight.

The following equation is used to calculate a wildlife criterion for least terns. This equation is identical to the one described in the Mercury Study Report to Congress, Volume VI (USEPA 1997b).

$$WC = \frac{TD \times (1/UF) \times Wt_A}{W_A + [(FD_3)(F_A \times BAF_3) + (FD_4)(F_A \times BAF_4)]}$$

WC = Wildlife criterion (units as calculated will be in mg/L; convert to µg/L)

Wt_A = Average species weight (kg)

W_A = average daily volume of water consumed (L/d)

F_A = average daily amount of food consumed (kg/d) (dry weight)

FD_3 = fraction of the diet derived from trophic level 3

FD_4 = fraction of the diet derived from trophic level 4

BAF_3 = aquatic life bioaccumulation factor for trophic level 3 (dry weight)

BAF_4 = aquatic life bioaccumulation factor for trophic level 4

TD = Threshold dose (mg/kg Body Wt/day). Ideally the threshold dose should be a bounded NOAEC (No observed adverse effect concentration). If however a NOAEC is not known then an uncertainty factor may be appropriately applied to a LOAEC (Lowest observed adverse effect concentration).

UF = Uncertainty Factor

The EPA procedure provides that in the absence of a NOAEC a LOAEC may be used with the addition of an uncertainty factor. Other uncertainty factors may be applied where there is interspecies uncertainty and when extrapolating from subchronic to chronic exposures.

Equation Values used for Least Tern

California least terns, a federally listed species, are the smallest members of the subfamily Sterninae (family Laridae), measuring about 22.9 cm (nine inches) long with a 50.8 cm (20 inch) wingspread and body weights ranging between 45 and 55 g. They are exclusively piscivorous and typically consume such trophic level 3 fish as topsmelt, anchovy, surf perch and jacksmelt.

Trophic level 3 fish are those which consume aquatic invertebrates, and planktivores. Thus, for the least tern in this analysis:

FD_4 = 0 and **FD_3 = 1.0**.

Using an average body weight of 0.05 Kg the F_a value for food consumption per day (dry weight) may be calculated using allometric equations for seabirds found in Nagy (1987) :

$g/d = 0.495(Bw)^{0.704}$. This results in **F_a = 0.0078 kg/day.**

Allometric equations are also used to generate an estimate of W_A . The following equation is from Calder and Braun, 1983:

$L/day = 0.059(BW)^{0.67}$. This results in **W_A = 0.007 L/day**.

A field derived BAF from central SF Bay for total mercury (for comparative purposes) was derived from synoptic sampling of fish (shiner surf perch) and water using ultra clean techniques at 6 central bay locations by the Regional Monitoring program in 1994 (Table 6). This BAF was derived from the geometric mean of these 6 sites. While field BAFs vary somewhat, USEPA (1997b) recommends using the geometric mean BAF where exposure concern is for repeated ingestion. The dry weight geometric mean BAF for total unfiltered mercury to shiner surf perch in Central SF Bay is 97,723 (Table 6). The allometric equations estimating food consumption of the tern are dry weight based, thus dry weight mercury concentrations were used to derive the dry weight BAF.

BAF_3 (dw) = 97,723 as total Hg (field derived, Central SF Bay).

(Note: A total mercury criterion is developed here to allow comparison of a sample wildlife criterion with the human health criterion proposed by EPA. Future development of wildlife criteria for California should probably be based upon a dissolved mercury or dissolved methylmercury concentration in water.)

The threshold dose value is from a three generation study feeding study in mallards with methylmercury dicyandiamide (Heinz, 1979). The lowest dose resulted in adverse effects on reproduction and behavior, therefore, this concentration represents a LOAEC not a NOAEC. This is the value used by EPA to calculate wildlife criteria in the final Mercury Study Report to Congress (USEPA, 1997b).

TD = 0.078 mg/kg/day

UF = 3 The EPA procedure provides that in the absence of a NOAEC a LOAEC may be used with the addition of an uncertainty factor. Other uncertainty factors may be applied where there is interspecies uncertainty and when extrapolating from subchronic to chronic exposures. Because the field species in this case, the least tern, is a piscivorous bird and fish eating birds may have greater capacity to demethylate mercury, no interspecies uncertainty factor is applied. Because the tested threshold dose was derived from a chronic 3 generation exposure no uncertainty factor for exposure duration is applied. An uncertainty factor of 3 is applied because the TD is a LOAEC not a NOAEC. The detailed reasoning behind the uncertainty factor of 3 is provided in USEPA (1997b) and Nichols et al. (1999).

Completing the equation yields:

$$WC = \frac{0.078 mg/kg/day \times [1/3] \times 0.05 kg}{0.007 L/d + [1.0(0.0078 \times 97{,}723)]} = 0.000001705 \text{ mg/L as dissolved total Hg}$$

WC = 0.00171 µg/L or **1.71 ng/L** total unfiltered Hg

Without using the uncertainty factor of three, the equation produces an effect threshold concentration for mercury in water where "take" may be estimated to occur for the least tern. This concentration is 5.11 ng/L as a geometric mean.

We conclude that using an oral dose model per the methods of USEPA, 1997b, a wildlife criterion that might be protective of California least terns would be 1.71 ng/L total unfiltered mercury.

Tern egg bioaccumulation method: An alternative method to calculate a wildlife criterion is to use the egg residues from the field and divide by the associated water mercury concentrations to develop an egg/water bioaccumulation factor. The egg/water BAF can then be used with established values of egg residues associated with embryo toxicity to determine a wildlife criterion. This method can then be assessed and compared with the dietary method of EPA for independent validation.

Six fail-to-hatch California least tern eggs from the nesting colony at Alameda Naval Air Station in 1994 were analyzed for mercury content. The wet weight mean concentration was 740 ng/g and concentrations ranged from 390 ng/g to 1,300 ng/g (Schwarzbach et al., 1997). Water mercury data in 1994 was collected as part of the Regional Monitoring Program by the San Francisco Estuary Institute (SFEI) at a number of stations in San Francisco Bay. The mean mercury concentration in unfiltered water in April among the following five central bay sites (Point Isabel, Red Rock, Yerba Buena, Alameda and Oyster Point) was 4.7 ng/L. This value is used to estimate the water mercury concentration for the BAF calculation. The April data was selected because of their proximity to the egg laying season for terns.

The following equations are used to calculate a protective criterion for total mercury in water. Wet weight values are used because toxic thresholds for mercury in eggs are typically expressed in wet weight.

species-specific field BAF = <u>measured egg concentrations</u>

for Ca. least terns measured water concentration

$$= \frac{740 \text{ ng/g}}{4.7 \text{ ng/L}} = 157 \text{ ng/g/ng/L}$$

A water criterion can now be derived by dividing the avian egg NOAEL by the field BAF. Unfortunately there is no known bounded avian egg NOAEL. The LOAEL however is 500 ng/g (ww). Using a LOAEL/NOAEL ratio for mercury concentrations in avian egg of two, one obtains a calculated NOAEL of 250 ng/g.

$$\frac{\text{NOAEL concentration in egg}}{\text{Field egg/water BAF}} = \frac{250 \text{ ng/g}}{157 \text{ ng/g/ng/L}} = 1.59 \text{ ng/L total mercury}$$

Dividing the NOAEL by the BAF results in a calculated water criterion concentration of 1.59 ng/L total mercury, a value comparable to the 1.71 ng/L result of the oral dose model presented above.
Without the uncertainty factor of 2, an effect threshold of 3.2 ng/L is calculated as total mercury (in unfiltered water).

EPA has calculated a piscivorous wildlife criterion value of 0.05 ng/L as methylmercury or 0.641 ng/L total "aqueous" (dissolved) mercury for protection of piscivorous wildlife (USEPA, 1997b). The wildlife criterion calculated by EPA in the Mercury Study Report to Congress was not released as a final report prior to the publication of the draft CTR in the federal register (USEPA, 1997c) and the mercury criterion for California water bodies as proposed in the CTR does not reflect this now available science. This "criterion value" has thus far been officially issued only in a report to Congress, not as guidance to the states as a basis for regulating water quality.

The criteria calculations presented above were done to evaluate the degree of protectiveness of EPA's CTR mercury criteria for a listed piscivorous species using site-specific bioaccumulation factors; to compare that site-specific criterion with criteria developed in the Mercury Study Report to Congress; and to evaluate the comparative usefulness of the egg bioaccumulation model with the oral dose model used by EPA in predicting mercury toxicity to avian reproduction. If comparable, this method may serve as a valuable alternative to the oral dose model for avian species where egg mercury and water data are available but dietary concentrations are not known. This model is most useful in predicting toxicity of bioaccumulated compounds to birds when the most sensitive endpoint is embryo toxicity.

The California least tern is exclusively piscivorous, or nearly so, and therefore tern mercury bioaccumulation, unlike clapper rail, is most directly dependent upon mercury concentrations in the water column. Another advantage of using the tern as a model species for estimating a water based criterion is that mercury data in fish, water and eggs exist from the same time period which allow a calculation of mercury criteria using both models. The three sub-species of clapper rails

(Yuma, light-footed, and California subspecies) have a mercury exposure pattern complicated by their benthic foraging habits and minor piscivory. For the bald eagle EPA has already developed a criterion (USEPA, 1997b). The California least tern diet overlaps in significant ways the potential diet and mercury exposure levels of the federally protected marbled murrelet.

The wildlife criteria calculated in the Mercury Study Report to Congress (1997b) was based on risk estimates to six species, two species of fish eating mammals (mink and river otter) and four species of fish eating birds (loons, bald eagles, kingfisher and osprey). Criteria were calculated on a methylmercury basis using an oral dose model similar to that used in the Great Lakes Initiative (USEPA, 1995b). Table 7 compares results of the two models with the various wildlife criteria developed by the USEPA (1997b) and the mercury criteria for California water bodies as proposed in the CTR.

Calculated water concentrations protective of terns from each of the two methods produce similar numbers for total mercury. The calculated wildlife criterion using EPA's oral dose model is 1.71 ng/L (oral dose model) while the egg bioaccumulation model estimates 1.59 ng/L (BAF model). These numbers are also in close agreement with EPA's overall number of 2.3 ng/L for piscivorous mammals and birds and clearly indicate that mercury criteria as proposed in the CTR are between one and three orders of magnitude under protective for listed wildlife species including the least tern and bald eagle. The Services conclude that the egg BAF model is capable of calculating a criterion comparable to the oral dose model prediction. The Services further conclude that criteria developed in the Mercury Study Report to Congress (1997b) would likely be sufficiently protective for the least tern and other piscivorous wildlife species in California.

Table 7. Mercury criteria concentrations in fresh water.

Source	"protected entity"	dis. methyl Hg	dis. total Hg	unfiltered total Hg	basis of criteria
USEPA,1997b.	loon	0.067 ng/L	0.859 ng/L^	3.09 ng/L*	Oral dose model
"	eagle	0.082 ng/L	1.051 ng/L^	3.78 ng/L*	"
"	kingfisher	0.027 ng/L	0.346 ng/L^	1.24 ng/L*	"
"	osprey	0.067 ng/L	0.859 ng/L^	3.09 ng/L*	"
"	mink	0.057 ng/L	0.73 ng/L^	2.63 ng/L*	"
"	river otter	0.042 ng/L	0.54 ng/L^	1.94 ng/L*	"
"	Piscivorous Wildlife	0.05 ng/L	0.641 ng/L^	2.3 ng/L*	"
FWS (oral dose)	Ca. least tern		0.46 ng/L*	1.71 ng/L	oral dose model

FWS (egg BMF)	Ca. least tern		0.44 ng/L*	1.59 ng/L	egg BAF model
CTR	aquatic life (chronic)		770 ng/L	2,772 ng/L*	waterborne toxicity
CTR	aquatic life (acute)			5,040 ng/L*	waterborne toxicity
CTR	human health			50 ng/L	1980 BCFs
Former CA Standards	Aquatic Life (chronic)			12 ng/L	literature evaluation
Former CA Standards	Aquatic Life (acute)			2,400 ng/L	literature evaluation

^ EPA methylmercury values are converted to dissolved total mercury by using 0.078 as an estimate of the fraction of methylmercury as a proportion of total mercury. This was EPA's "best" estimate (USEPA, 1997b). Methylmercury data for waters in San Francisco Bay is not available.

*Dissolved total mercury is converted to total unfiltered mercury and vice versa for all values by multiplying or dividing as appropriate by the ratio of total to dissolved (3.6) mercury to be consistent with conversion factor used in developing tern criteria. Values from 1994 RMP data from central San Francisco Bay (SFEI, 1997a).

Summary of Mercury Effects to Listed Species

Birds

Bald Eagle: The bald eagle is a generalized predator/scavenger primarily adapted to edges of aquatic habitats. Its primary foods, in descending order of importance, are fish (taken both alive and as carrion), waterfowl, mammalian carrion, and small birds and mammals.

The Klamath Basin in northern California and southern Oregon supports the largest wintering population of eagles in the lower 48 states, where up to 1000 birds may congregate at one time. Elevated mean mercury concentrations of 2.25 µg/L in the blood of bald eagles has been documented in the Klamath Basin (Frenzel and Anthony, 1989). Bald eagle exposure to elevated concentrations of mercury in California is likely, particularly in eagles wintering and breeding at coastal mountain reservoirs and associated watersheds. This exposure however, is poorly documented in eagle tissue and egg residues of mercury.

Scattered smaller groups of wintering eagles occur near reservoirs, and in close proximity to large concentrations of overwintering migratory waterfowl. In recent years San Antonio Reservoir has become an important wintering area for bald eagles. An estimate of 50+ eagles regularly winter there. These eagles may be exposed to hazardous mercury concentrations in the diet by foraging at nearby Lake Nacimiento. Important breeding sites for bald eagles include Lake Nacimiento. Lake Nacimiento is mercury impaired, and has a human health fish consumption advisory due to mercury: women are cautioned against consuming any large mouth bass and no one should eat more than 24 ounces of large mouth bass per month from this lake (Cal EPA public health warnings). USEPA (1997b) has developed a mercury criterion for water protective of bald eagles of 1.05 ng/L (as dissolved total mercury) but this recommendation was published after the CTR. The Service concludes EPA's proposed aquatic life and human health mercury criteria of 770 ng/L and 50 ng/L, respectively, in the CTR are not protective of bald eagles.

California Least Tern: California least terns are an exclusively piscivorous bird. Information presented above demonstrates that permissible concentrations of mercury in water under the CTR would produce elevated concentrations in tern eggs and prey sufficient to impair least tern reproduction. In the case of terns nesting in San Francisco Bay, mercury has already been measured in eggs with concentrations high enough to impair avian reproduction (> 0.5 µg/g). Concentrations in fail to hatch tern eggs from Alameda Naval Air Station in 1994 ranged from 0.4 to 1.24 µg/g fww with a mean of 0.74 µg/g. The current mercury threat is lower to least terns nesting in southern California. Eggs in 1994 from San Diego had mercury concentrations ranging from 0.12 to 0.26 µg/g with a mean concentration of 0.19 µg/g ww. However, permissible concentrations under the CTR could allow mercury concentrations in Southern California bays and estuaries sufficient to adversely effect tern reproduction. The Service has calculated a criterion value for the least tern of 1.71 ng/L using EPA methodology (EPA 1997b) and site specific bioaccumulation factors from central San Francisco Bay. Alternatively the Service has used tern egg data to calculate a criterion of 1.59 ng/L using an egg bioaccumulation model. These two criteria calculations developed independently confirm that EPA's criterion of 50 ng/L will not protect the least tern. The Service further concludes the mercury status of terns in San Francisco Bay would not be improved by the CTR.

California Clapper Rail: The extant range of the California clapper rail is restricted to marshes of the San Francisco Bay Estuary. California clapper rails feed almost exclusively on benthic invertebrates, are non-migratory and vulnerable to local particulate and waterborne mercury inputs. Mercury contamination in rails summarized above and in the mercury appendix of this document indicates California clapper rails have the highest concentration of mercury measured in a single egg of any species nesting within San Francisco Bay (Schwarzbach et al, 1997). Mean concentrations in 36 fail to hatch eggs in 1992 was 0.63 µg/g (fww). The percentage of non-viable eggs among south bay marshes in 1992 ranged from 24 to 38 percent. Based upon current mercury impairment, and the range of wildlife criteria values for mercury between 1 and 3 ng/L total mercury summarized above, the Service concludes that neither the proposed dissolved numeric aquatic criterion of 770 ng/L nor the total mercury criterion of 50 ng/L for human health, would improve the current mercury status of the rail. The Service further concludes the promulgation and adoption of these criteria for San Francisco Bay could reduce incentives for mercury emission control strategies that would benefit the rail.

Yuma Clapper Rail: With a biological profile very similar to the California clapper rail, the Yuma clapper rail is similarly vulnerable to mercury contamination of prey and eggs. There is reason to suspect potential for mercury contamination of Yuma Rail habitat in tributaries of the Colorado

River downstream of discharges into Bat Cave Wash. Additionally the elevated selenium concentrations, the interactive potential for selenium and mercury toxicity to avian embryos and the lack of protection afforded by the human health criterion for mercury to Yuma clapper rails leads the Service to conclude protective mercury criteria are needed for the Yuma clapper rail.

Light-footed Clapper Rail: With a biological profile very similar to the California clapper rail, the light-footed clapper rail is similarly vulnerable to mercury contamination of prey and eggs. While the Service knows of no current mercury threat to the light-footed clapper rail, the potential for future mercury concentrations to increase with adoption of the CTR leads the Service to conclude more protective criteria are needed for the light-footed rail. The non-migratory, benthic foraging niche and fragmented habitat of light footed rails places them at great risk of locally elevated concentrations of mercury within tidal marshes.

Marbled Murrelet: During the breeding season marbled murrelets forage in near shore environments including bays and estuaries on small fish and euphasid shrimp. They have also been known to forage to a minor degree on salmonid fry in freshwater environments. As a piscivorous bird, much of the discussion provided above for the least tern regarding the inadequacy of the CTR-proposed mercury criteria may also apply to the marbled murrelet.

Adverse impacts from increased permissible concentrations of contaminants as proposed in the CTR to prey species such as the Pacific sardine, herring, topsmelt, and northern anchovies, has the potential to significantly reduce long-term reproductive success of marbled murrelets (USDI-FWS, 1997). Adverse effects to prey species spawning and nursery habitats have the potential to impair population size and reduce recruitment throughout their range in California. The vulnerability of marbled murrelet populations in conservation zones 5 and 6, coupled with elevated concentrations of contaminants in spawning and nursery areas for murrelet prey species increase the risk of bioaccumulation of mercury and selenium. The synergistic effects of these contaminants pose a significant threat to marbled murrelet reproduction throughout conservation zones 5 and 6 and to a lesser degree in conservation zone 4.

Consequently, until species-specific data are collected or species-specific modeling is conducted for the marbled murrelet, a mercury criterion similar to that developed in this opinion for the California least tern or the Mercury Study Report to Congress must be viewed as the applicable guidance for protection of marbled murrelets.

Amphibians and Reptiles

Reptiles and amphibians remain the least studied vertebrates for mercury toxicity. It is also likely that aquatic food chain contamination by mercury would be the most significant pathway of exposure as would maternal transfer of methylmercury to the eggs. The Service believes a fish risk model may be most appropriate for assessing mercury hazard to amphibians such as the red-legged frog. This assessment may however be overly simplistic. Development of amphibians is unique among vertebrates in the occurrence of hormonally mediated ontogenetic metamorphosis within the water column (Duellman and Trueb, 1986). Chronic studies in frogs of the effects of mercury contamination are generally lacking.

California red-legged frogs spend most of their lives in and near sheltered backwaters of ponds, marshes, springs, streams, and reservoirs. These types of environments are particularly vulnerable to mercury contamination due to favorable conditions for the conversion of inorganic mercury to methylmercury. Red-legged frogs are reduced to about 30 percent of their historical range with most of the remaining population limited to coastal drainages. Several hundred abandoned mercury mines of varying sizes and states of remediation or disrepair currently contaminate this region with both inorganic and methylmercury. These mines and associated contaminated landscapes present potential exposure pathways for mercury to the habitat of the red-legged frog. Mercury residue data in yellow-legged frogs downstream from abandoned mines in the Cache creek data cited above and provided in the mercury appendix indicate ranid frogs may bioaccumulate mercury in the vicinity of these mines. The Service therefore concludes appropriate mercury criteria are needed for protection of red-legged frogs.

The Service was not able to locate any published acute or chronic studies of mercury in snakes. Studies of mercury in garter snakes are needed to better evaluate the protection afforded to these species of proposed mercury criteria.

Fish

Based on the information presented above on the toxicity of mercury to salmonid fish at 100 ng/L concentrations, it would appear the aquatic life criterion is unprotective of listed salmonids and possibly other fish species as well (Weiner and Spry 1995). Based on the review of mercury bioaccumulation factors in fish, it appears that harmful degrees of maternal transfer of mercury to fish eggs and young could occur at concentrations below the lowest CTR criteria number for mercury (50 ng/L). Mercury intoxicated rainbow trout have between 4 and 30 µg/g in whole bodies, while intoxicated embryos contain 0.07 to 0.1 µg/g (Weiner 1995). Application of EPA bioaccumulation factors predicts reproductive adverse effect concentrations at 5 ng/L total aqueous mercury. Due to the potential for elevated concentrations of mercury in water and/or biota in a number of California water bodies, and due to the life history characteristics, the Services believe an exposure pathway exists for the following listed or proposed fish species: all runs and ESUs of coho and chinook salmon and steelhead trout, Little Kern Golden trout, Paiute cutthroat trout, Lahontan cutthroat trout, bonytail chub, unarmored threespine stickleback, shortnose sucker, Lost River sucker and the Sacramento splittail.

Mammals

<u>Southern</u> <u>Sea</u> <u>Otter</u>: Southern sea otters are known to forage at the mouths of freshwater systems as well as in shallow marine waters adjacent to the coast. California has abundant geologic sources of mercury and a long history of mercury contamination associated with mercury mining, particularly in the Coast Range. These sources of mercury often are coincidental with headwaters

of streams discharging to the central California coast. Livers collected from sea otters found dead along the central California coast range as high as 60 µg/g (Mark Stephenson, CDFG, pers comm 1998). Of 125 California coast sea otters examined for mercury in liver, 45 percent had concentrations greater than what may be considered a normal river otter ambient concentration of 4 µg/g. One fourth of these salvaged individuals had concentrations over 10 µg/g and 3 percent had concentrations over the 30 µg/g hepatic concentration associated with lethality. Acute mercury poisoning in mammals is primarily manifested in central nervous system damage, sensory and motor deficits, and behavioral impairment. Animals initially become anorexic and lethargic.

Sea otters are voracious consumers eating as much as 35 percent of their body weight per day. This high forage rate leaves them potentially vulnerable to dietary contaminant loading. The diet of sea otters consists of slow moving fish and invertebrates (Estes, 1980). Sea otters obtain about 23 percent of their water needs from sea water, making them vulnerable to impaired kidney function from inorganic mercury and cadmium. The proximity of otter foraging to elevated coast range discharges of mercury and cadmium places the otter at risk of dietary mercury and cadmium exposure. Given the potential for exposure and the documentation of elevated concentrations in a significant fraction of dead otters the Service concludes a mercury wildlife criterion comparable to that developed for piscivorous wildlife in the Mercury Study Report to Congress is needed for sea otter protection.

EPA modifications addressing the Services' April 9, 1999 draft Reasonable and Prudent Alternatives for mercury:

The above effect analysis evaluates the draft CTR as originally proposed in August of 1997. EPA has agreed by letter dated December 16, 1999, to modify its action for mercury per the following to avoid jeopardizing listed species.

A. *EPA will reserve (not promulgate) the proposed freshwater and saltwater acute and chronic aquatic life criteria for mercury in the final CTR.*

B. *EPA will promulgate a human health criterion of 50 ng/l or 51 ng/l as designated within the final CTR for mercury <u>only</u> where no more restrictive federally-approved water quality criteria are now in place (e.g., the promulgation will not affect portions of San Francisco Bay).*

C. *EPA will revise its recommended 304(a) human health criteria for mercury by January 2002. EPA will propose revised human health criteria for mercury in California by January 2003. These criteria should be sufficient to protect federally listed aquatic and aquatic-dependent wildlife species. EPA will work in close cooperation with the Services to evaluate the degree of protection afforded to federally listed species by the revised criteria. EPA will solicit public comment on the proposed criteria as part of its rulemaking process, and will take into account all available information, including the information contained in the Services' opinion, to ensure that the revised criteria will* *adequately protect federally listed species. If the revised criteria are less stringent than those proposed by the Services in the opinion, EPA will provide the Services with a biological evaluation/assessment on the revised criteria by the time of the proposal to allow the Services to complete a biological opinion on the proposed mercury criteria before promulgating final criteria. EPA will provide the Services with updates regarding the status of EPA's revision of the criterion and any draft biological evaluation/assessment associated with the revision. EPA will promulgate final criteria as soon as possible, but no later than 18 months, after proposal. EPA will continue to consult, under section 7 of ESA, with the Services on revisions to water quality standards contained in Basin Plans, submitted to EPA under CWA section 303, and affecting waters of California containing federally listed species and/or their habitats. EPA will annually submit to the Services a list of NPDES permits due for review to allow the Services to identify any potential for adverse effects on listed species and/or their habitats. EPA will coordinate with the Services on any permits that the Services identify as having potential for adverse effects on listed species and/or their habitat in accordance with procedures described in the draft MOA published in the Federal Register at 64 FR 2755 (January 15, 1999) or any modifications to those procedures agreed to in a finalized MOA.*

D. EPA will utilize existing information to identify water bodies impaired by mercury in the State of California. Impaired is defined as water bodies for which fish or waterfowl consumption advisories exist or where water quality criteria necessary to protect federally listed species are not met. Pursuant to Section 303(d) of the CWA, EPA will work, in cooperation with the Services, and the State of California to promote and develop strategies to identify sources of mercury contamination to the impaired water bodies where federally listed species exist, and use existing authorities and resources to identify, promote, and implement measures to reduce mercury loading into their habitat. (See also "Other Actions B." below.)

E. EPA promulgated a new more sensitive analytical method for measuring mercury (see 40 CFR Part 136).

Services' assumptions regarding EPA's modifications to the proposed action for removing jeopardy.

In modifying our April 1998 jeopardy opinion and the modified draft RPAs considered in April 1999, the Services have assumed the following regarding EPA's proposed modifications:

Contaminant threats to listed species can be reduced through application of appropriately protective water quality criteria to the water bodies occupied by listed species.

The presumptive adverse effect threshold for identifying effects to listed species, is either the exceedance of the criteria proposed in this opinion to protect listed species, or demonstrated effects below those proposed criteria concentrations for the priority pollutant under consideration.

The adjustments of criteria as proposed in the CTR by EPA for water bodies occupied by species considered in this opinion will be consistent with the effects analysis in this biological opinion unless new information is developed by EPA.

EPA adjustments of criteria will occur within agreed upon time frames.

Promulgations by EPA of the new mercury human health criterion will apply to all water bodies in California containing listed species and /or their habitats considered in this opinion by June of 2004.

The modification of 304a human health criterion for mercury which precedes EPA's promulgation of criteria in California will serve as the scientific guidance to permit writers for those permits with mercury discharges into waters occupied by listed species after January 2002

The revision of the human health mercury criterion will employ field derived bioaccumulation factors and this will result in a substantial lowering of the present criterion. The Services thus assume this revision will represent a substantial improvement statewide in the mercury water quality objectives for both listed aquatic species and wildlife species that forage within aquatic systems.

The draft CTR human health criterion of 51 ng/L will apply only where no more restrictive criteria are in effect, including San Francisco Bay.

The reservation of the acute and chronic aquatic life criteria for mercury means these criteria will not be used for regulatory purposes in California.

Pentachlorophenol (PCP)

Adequacy of Proposed Criteria

Aquatic Life Criteria

The EPA has proposed a pH-dependent freshwater acute criterion of 19 µg/L at pH= 7.8 (CMC = exp(1.005(pH)-4.830)), and a pH-dependent freshwater chronic criterion of 15 µg/L at pH=7.8 (CCC = exp(1.005(pH)-5.290)) for PCP (USEPA, 1997c). If the CTR is promulgated as proposed, salmonids and other listed fish could be exposed to ambient levels of PCP at or below the proposed acute and chronic criteria. After a review of the available data the Services conclude that the proposed acute and chronic water quality criteria for PCP are not protective of endangered and threatened fish. Current literature indicates adverse effects of commercial (technical grade) PCP on reproduction, early life stage survival, growth, or behavior of salmonid species at concentrations at or below the proposed criteria. EPA has not included within the criteria interactive effects of pH, dissolved oxygen or temperature on toxicity of PCP to fish. These factors exacerbate the deleterious effect of PCP toxicity on salmonids at the proposed criteria concentrations. The criteria also do not consider bioconcentration of PCP or its impurities into aquatic organisms and subsequent ingestion by wildlife.

EPA has suggested to the Services that drinking water standards for PCP (0.28 µg/L) could serve to protect salmonids. These standards, however, do not apply in water bodies without the appropriate MUN designation. MUN is the beneficial use designation for water bodies that serve as municipal and domestic water supply. The following water bodies serve as habitat for listed fish species and do not have the MUN designation. Listed salmonids and other fish species in these water bodies are dependent upon the aquatic life criterion alone for protection. Therefore, adverse effects to listed species occurring within these water bodies are likely to occur.

Region 1:	Laguna de Santa Rosa
Region 2:	First Valley Creek (tributary to Drake's Estero)
	Coast Creek
	Alamere Creek
	Bolinas Bay tributaries
	Rodeo Creek (tributary to Rodeo Lagoon)
	Millerton Gulch (tributary to Tomales Bay)
	Walker Creek and tributaries
	Bear Valley Cr., Devil's Gulch, and Gulch Creek (tributaries to Olema Creek)
	Frenchman's Creek
	Purisima Creek
	Lobitas Creek
	Tunitas Creek
	San Gregorio Creek and tributaries
	Pomponio Creek
	Butano Creek
	San Rafael Creek
	Corte Madera Creek and tributaries
	Coyote Cr., Old Mill Cr., and Arroyo Corte Madera del Presidio (tributaries to Richardson Bay)
	San Leandro Creek and tributaries
	Alameda Creek and tributaries
Region 3:	Watsonville Slough and tributary sloughs
Region 5:	Battle Creek
	Thomes Creek
	Big Chico Creek
	Stony Creek
	Butte Creek (below Chico)

Lower Yuba River (below Engelbright Dam to Feather River)
Mokelumne River (Comanche Reservoir to Delta)

Hazards of PCP

PCP Sources, Chemistry, and Environmental Fate

PCP at one time, was one of the most widely used biocides. In 1986, approximately 28 million pounds were used in the United States. It was registered for use as a molluscide, fungicide, herbicide, insecticide, disinfectant, wood preservative, slimicide in pulp and paper products, and paint preservative. Its use was restricted by EPA since 1984, consequently it is no longer available for home and garden use (ATSDR 1993). Approximately 80% of the total technical grade PCP use is for wood preservation. The majority of wood treated with PCP is done so commercially, using pressurized treatment. Treatment with PCP results in a 5 to 8-fold increased useful life of wood products. The aqueous form, sodium pentachlorophenate (NaPCP) has been used in pressboard, insulation, and industrial cooling water, among other uses (Crosby 1981; Eisler 1989).

In the U.S., PCP is produced by the chlorination of phenols in the presence of catalysts. The alternative production process, hexachlorobenzene hydrolysis, is not used in the U.S. Commercial grades of pentachlorophenol, also referred to as technical PCP, are generally about 86% pure. Reagent grade and purified forms of PCP have been used extensively in toxicity testing in order to differentiate the toxicity of PCP in relationship to the numerous impurities found in commercial preparations. However, the Services assume that technical grades of PCP are the forms more commonly released to the environment.

Impurities found in commercial preparations of PCP include relatively high concentrations of chlorophenols, polychlorinated dibenzodioxins and dibenzofurans (PCDD/Fs), hexachlorobenzene, chlorinated phenoxyphenols, and chlorinated diphenyloxides (USEPA 1980; Eisler1989; Cleveland *et al*. 1982; Hamilton *et al*. 1986). Chlorinated phenoxyphenols and other compounds found in PCP can be precursors to the formation of PCDD/Fs (Cleveland *et al*. 1982; Hamilton *et al*. 1986). PCDD/Fs are known to bioaccumulate in the environment and are also highly toxic to avian and mammalian wildlife. The bioaccumulation and chronic toxicity to wildlife of the other impurities found in commercial PCPs are not well known. The commercial preparations of PCP have been found to be 5 to 6-fold more toxic to fathead minnow than are purified PCP forms. It is believed that the impurities in commercial PCPs are largely responsible for the enhanced toxicity (Cleveland *et al*. 1982).

PCP can be released into the aquatic environment in runoff and in wood-treatment effluents. The majority of wood treatment plants evaporate their waste water, so they do not discharge to surface waters. The rest of the wood treatment plants discharge to waste-water treatment facilities. Prior to EPA restricting its use, discharges to water totaled approximately 37,000 pounds annually. Releases to the aquatic environment now are expected to be less. In 1991, Toxics Release

Inventory data indicates total releases to the environment (including discharge to water, air and soil) from certain facilities were 16,296 pounds. Total releases to the environment are likely higher than reported by the Toxic Release Inventory, because data are available for only certain types of facilities required to report releases (ATSDR 1993).

PCP is soluble in most solvents, and slightly soluble in water. In contrast, the sodium salt of PCP, NaPCP, is very water soluble. However, the chemical properties of PCP are closely related to the pH of the aqueous solution. PCP has a pK_A of 4.7, which means that at a pH of 4.7, aqueous solutions will contain 50% ionized PCP. At pH 6.7, in the range of many natural waters, PCP is 99% ionized. However the toxicity of PCP increases as the pH of the water decreases, because the un-ionized form (which is favored at low pH) passively diffuses across the gill membrane (USEPA 1986). The proposed criteria are pH-dependent because PCP ionization in water increases with an increase in pH (i.e., PCP is more toxic at lower pH because the un-ionized form which crosses the membrane is predominant over the ionized form).

Once released to water, the half-life of PCP ranges from less than one day to 15 days. The degree of degradation is controlled by amount of incident radiation (sunlight penetration), dissolved oxygen, and pH of the water. Photolysis and degradation by microorganisms are considered the major mechanisms by which PCP is degraded in water. Degradation of PCP in water forms other compounds, primarily pentachloroanisole, 2,3,4,5-tetrachlorophenol, 2,3,4,6-tetrachlorophenol, and 2,3,5,6-tetrachlorophenol (ATSDR 1993).

Ambient surface water concentrations of PCP have been reported to generally be between 0.1 to 10 µg/L (as of 1979, ATSDR 1993). These values are within the range of the proposed chronic criterion for PCP (assuming a neutral pH = 6.7, the chronic criterion is 4.95 µg/L). Industrialized areas, and areas near paper mills and wood treatment facilities, have levels at the high end of that range, or even higher. However, much of the existing published data on surface water concentrations is from the 1970's, prior to its use restrictions by EPA. Collecting additional data on ambient PCP concentrations in streams supporting federally listed fish would help identify locations where PCP may be a problem for listed fish species.

Toxicity

The mechanism of PCP toxic action is regarded to be via reduced production of adenosine triphosphate (ATP) and alteration of liver enzymes, which control energy metabolism. The response to this effect is an increased basal metabolism, resulting in increased oxygen consumption and high fat utilization (Webb and Brett 1973; Chapman and Shumway 1978; Johansen *et al.* 1985; Nagler *et al.* 1986; Eisler 1989). Growth parameters and locomotion/activity have been found to be sensitive endpoints for salmonids and other fish exposed to PCP (Hodson and Blunt 1981; Webb and Brett 1973; Dominquez and Chapman 1984; Brown *et al.* 1985; Johansen *et al.* 1987; Brown *et al.* 1987). The fact that the mechanism of action affects energy metabolism is support for use of growth parameters (e.g., reduced growth rate, reduced biomass) and activity parameters (reduced swimming activity, reduced prey consumption, reduced predator avoidance) to be used as sensitive and appropriate endpoints in sublethal toxicity tests. This mechanism also supports the conclusion that early fry, which have just finished utilizing the yolk sac and have begun to feed on exogenous sources of food, are among the most sensitive life stages tested.

In general, fish are more sensitive to PCP than are other aquatic organisms. Salmonids have been found to be the most sensitive fish species tested under acute exposure conditions (Choudhury *et al*. 1986; Eisler 1989; USEPA 1980, 1986b, 1995b, 1996c). Warmwater species are generally less sensitive than coldwater species in acute lethal toxicity tests (USEPA 1995c). Evaluation of threatened or endangered salmonid species against the rainbow trout, a typical test organism, found that the Apache trout (*Oncorhynchus apache*) was more sensitive than the rainbow trout in acute lethality tests with PCP, indicating an additional margin of safety may be needed to protect listed salmonids when using rainbow trout test data in toxicity assessments (USEPA 1995c). EPA (1995) also recommends that "further testing be done on listed species or their FWS-identified surrogate species before definitive policy decisions concerning the protection of endangered and threatened species are made. In addition, chronic toxicity assessments should be conducted in order to compare chronic responses between listed and surrogate species."

Early life stage of salmonids, such as sac fry and early fry, have been found to be more sensitive than later life stages and even more sensitive than embryos, to acute exposures of PCP. Similarly, early life stage of largemouth bass have varying sensitivity to acute exposures of PCP (Johansen 1985). Acute toxicity of PCP to fathead minnow also varies with life-stage, but adults appear to be more sensitive than juveniles or fry to PCP (Hedtke *et al*. 1986). In a study by Adema and Vink (1981) 96-hour lethal concentrations for 50 percent of the populations tested (LC_{50}s) in guppy ranged between 450 to 1,600 µg/L (life stage only specified as young or adult). Early life stages of the plaice (*Pieuronectes platessa*) were more sensitive with 96-hour LC_{50}s ranging from 60 to 750 µg/L at pH of 8; the larval stage was the most sensitive and the egg the least sensitive of the life stages tested. LC_{50}s for early life stage salmonids are lower at between 18 to 160 µg/L (Table 8a). Thus, non-salmonid fish appear to be less sensitive at early life stages than salmonids to acute toxicity of PCP.

<u>Summary of Effects of PCP on Listed Species</u>

Salmonids

Salmonid species evaluated include: all ESUs and runs of listed or proposed coho and chinook salmon and steelhead trout, Lahontan cutthroat trout, Paiute cutthroat trout, and Little Kern golden trout.

Tables 8a and 8b summarize the critical acute and chronic studies conducted on salmonid species used in this analysis. The proposed EPA criteria are dependent upon pH. To compare the water concentrations of PCP used in the study to the criteria, the final column in Tables 8a and 8b derives an acute and chronic water quality criterion using equations described in USEPA (1995b) for the pH at which the study was conducted. (There appears to be an error in footnote "f" in the Federal Register table. We based our pH corrections on the pH-dependent equations listed on pp. M-1, M-2 of USEPA 1995b).

Acute Studies: The first study listed in Table 8a is an acute study on rainbow trout conducted by Little et al. (1990). These researchers evaluated behavioral effects with implications for survival in the environment. Chapman's review of the draft biological opinion criticized this study stating that the acetone could artificially enhance uptake of PCP impurities (Chapman 1998). Although this may occur no studies have been done to evaluate the hypothesis. Since acetone was also in the control group, the effects of acetone itself is not at issue. Chapman (1998) goes on to recommend that proper studies be done to resolve the issues regarding differences in toxicity between commercial PCPs and the purified forms of PCP. Another limitation of the Little et al. (1990) study is that only nominal concentrations of PCP in test water are reported; water samples do not appear to have been analyzed to confirm the test concentrations. The evaluated behaviors of the Little *et al.* (1990) study included swimming activity, swimming capacity, feeding, and vulnerability as prey. Swimming capacity was not affected. Survival from predation did not show a clear dose-response curve; greater survival was observed in the 2 µg/L compared to the 0.2 µg/L group. Similarly, there was not a clear dose-response for number of prey consumed and swimming activity. There was significantly reduced swimming activity and prey consumption observed at 2 µg/L of technical grade PCP after 4 days of exposure, compared to controls. As Chapman (1998) points out, determining safe levels from this study is difficult given the experimental design and the lack of clear dose-response for many of the endpoints evaluated. Also, Chapman (1998) indicates that this study does not report whether pH was monitored during the tests. However, even if the pH of the static test solutions were a full pH unit lower than measured in the well water (i.e., pH = 6.8 instead of 7.8), the acute criterion of 7.13 µg/L and the chronic criterion of 5.47 µg/L (at pH = 6.8) would still be greater than the concentrations at which effects on behavior were observed. Therefore, the proposed acute criterion for PCP of about 19.5 µg/L (pH-adjusted to pH = 7.8) is not protective of salmonid behavior relative to growth and survival.

Table 8a: Summary of Critical Acute Studies on the Effects of PCP in Salmonids.

Citation	**Life Stage and Species#**	**Exposure Duration, days**	**Test Solution**	**Test Type**	**Effect**	**pH**	**Effect concentration, µg/L**	**pH Adjusted Criteria*, µg/L**
Little et al. (1990)	0.5 - 1.0 g *O. mykiss*	4	Tech. grade PCP	static	reduced swimming activity and reduced prey consumption	7.8	LOAEL = 2 NOAEL = 0.2	19.5
Van Leeuwen et al. (1985)	early fry (77 days) *O. mykiss*	4	97 percent purified PCP	static renewal	50 percent mortality	7.2	18 (96 hr. LC_{50})	10.6
Van Leeuwen et al. (1985)	sac fry (42 days) *O. mykiss*	4	97percent purified PCP	static renewal	50 percent mortality	7.2	32 (96 hr. LC_{50})	10.6
Dominguez and Chapman (1984)	fry (70 days) *O. mykiss*	4	99 percent purified PCP	flow-through	50 percent mortality	7.4	66 (96 hr LC_{50})	13
Davis & Hoos (1975)	1-3 g *O. mykiss*	4	NaPCP	static	50 percent mortality	5.7 - 7.0	45 - 100 (96 hr LC_{50})	2.3 - 8.7
Davis & Hoos (1975)	1-3 g *O. kisutch*	4	NaPCP	static	50 percent mortality	7.0	32 - 96 (96 hr LC_{50})	8.7
Davis & Hoos (1975)	1-3 g *O. nerka*	4	NaPCP	static	50 percent mortality	7.2 - 7.7	50 - 130 (96 hr LC_{50})	10.6 - 17.6
U.S. FWS (1986)	0.3g fry *O. tshawytscha*	4	96 percent Technical Grade PCP	static	50 percent mortality	7.4	31 (96 hr LC_{50})	13
U.S. FWS (1986)	1.0g fry *O. tshawytscha*	4	96 percent Technical Grade PCP	static	50 percent mortality	7.4	68 (96 hr LC_{50})	13
U.S. FWS (1986)	yolk-sac fry *O. mykiss*	4	96 percent Technical Grade PCP	static	50 percent mortality	7.4	121 (96 hr LC_{50})	13
U.S. FWS (1986)	1.0g fry *O. mykiss*	4	96 percent Technical Grade PCP	static	50 percent mortality	7.4	34 - 52 (96 hr LC_{50})	13
U.S. FWS (1986)	1.0g fry *O. mykiss*	4	NaPCP	static	50 percent mortality	7.4	55 - 58 (96 hr LC_{50})	13

Citation	Life Stage and Species#	Exposure Duration, days	Test Solution	Test Type	Effect	pH	Effect concentration, µg/L	pH Adjusted Criteria*, µg/L
U.S. FWS (1986)	yolk-sac fry *O. mykiss*	4	NaPCP	flow-through	50 percent mortality	7.4	160 (96 hr LC_{50})	13
U.S. FWS (1986)	swim-up fry *O. mykiss*	4	NaPCP	flow-though	50 percent mortality	7.4	165 (96 hr LC_{50})	13
U.S. FWS (1986)	eyed-egg *O. mykiss*	4	NaPCP	flow-through	50 percent mortality	7.4	>300 (96 hr LC_{50})	13
U.S. FWS (1986)	0.3g fry *O. tshawytscha*	4	NaPCP	flow though	50 percent mortality	7.4	165 (96 hr LC_{50})	13
U.S. FWS (1986)	swim-up fry *O. tshawytscha*	4	NaPCP	flow-through	50 percent mortality	7.4	>250 (96 hr LC_{50})	13
U.S. FWS (1986)	1.0g fry *O. tshawytscha*	4	NaPCP	static	50 percent mortality	7.4	67.5 (96 hr LC_{50})	13
U.S. FWS (1986)	yolk-sac fry *O. tshawytscha*	4	NaPCP	static	50 percent mortality	7.4	30.5 (96 hr LC_{50})	13
U.S. EPA (1995)	0.5 - 1.0g fry *O. mykiss*	4	99 percent purified PCP	static	50 percent mortality	8.2	160 (96 hr LC_{50})	30
U.S. EPA (1995)	0.5 - 1.0g fry *O. apache*	4	99 percent purified PCP	static	50 percent mortality	8.2	110 (96 hr LC_{50})	30
U.S. EPA (1995)	0.5 - 1.0 fry *O. clarki stomias*	4	99 percent purified PCP	static	50 percent mortality	8.2	>10 (96 hr LC_{50})	30
U.S. EPA (1995)	0.5 - 1.0 fry *O. clarki henshawi*	4	99 percent purified PCP	static	50 percent mortality	8.2	170 (96 hr LC_{50})	30

* acute criterion (µg/L) = $e^{(1.005\,(pH) - 4.869)}$

\# *O. mykiss* = rainbow trout
O. apache = Apache trout
O. clarki stomias = Greenback cutthroat trout
O. clarki henshawi = Lahontan cutthroat trout
O. kisutch = Coho salmon
O. nerka - sockeye salmon

O. tshawytscha = Chinook salmon

Table 8b: Summary of Critical Chronic Studies on the Effects of PCP in Salmonids

Citation	Life Stage and Species#	Exposure Duration, days	Test Solution	Test Type	Effect	pH	Effect concentration, µg/L	pH Adjusted Criteria*, µg/L
Dominguez and Chapman (1984)	egg through day 72 *O. mykiss*	72	99 percent purified PCP	flow-through	34 percent mortality; decreased weight and length; increased fin erosion and mild malformations	7.4	19	10
Dominguez and Chapman (1984)	egg through day 72 *O. mykiss*	72	99percent purified PCP	flow-through	NOAEL for mortality, growth	7.4	11	10
Chapman and Shumway (1978)	fertilization of egg through complete yolk absorption *O. mykiss*	chronic	Tech. grade Na PCP	flow-through	little or no mortality compared to control at D.O. = 10 mg/L	7.8	10	15
Chapman	‘’	chronic	Tech. grade	flow-through	27.4 percent	7.8	10	15
Chapman and Shumway (1978)	‘’	chronic	Tech. grade Na PCP	flow-through	100 percent mortality at D.O. = 3 mg/L	7.8	10	15
Chapman (1969)	alevin *O. mykiss*	20-35	Tech. grade Na PCP	flow-through	15% reduction in weight gain	7.8?	30	15
Webb and Brett (1973)	subyearling *O. nerka*	14 - 56 (+ 4 weeks post-exposure exam)	Na PCP	flow-through	growth rate and food conversion efficiency	6.8	EC50 for growth rate = 1.74 EC50 for conversion efficiency = 1.8	5.5
Matida *et al.* (1971)	fry (2.1 - 2.5 g) *O. mykiss*	28	Tech. grade Na PCP	flow-through	27 percent growth inhibition	7.2	8	8.2
Nagler et al. (1986)	adult female *O. mykiss*	18	99 percent purified PCP	flow-through	reduced number of viable oocytes	7.5	LOAEL = 21.8 NOAEL = 11.5	11

Citation	Life Stage and Species#	Exposure Duration, days	Test Solution	Test Type	Effect	pH	Effect concentration, µg/L	pH Adjusted Criteria*, µg/L
Iwama et al. (1986)	juvenile (15 g) *O. tshawytscha*	40	Na PCP	flow-through	changed in blood BUN and GLU	?	3.9	15 @ pH = 7.8
Hodson and Blunt (1981)	embryo and alevin (after hatch to early fry) *O. mykiss*	exposed from embryo or alevin through fry feeding for 4 weeks	99percent purified NaPCP	flow-through	reduced wet weight, growth rate, and biomass at 20°C	7.78 - 8.08	11-16	18.2 @ pH = 8.0

F. chronic criterion (µg/L) = $e^{(1.005(pH) - 5.134)}$ (USEPA 1995b)

\# *O. mykiss* = rainbow trout
O. nerka = sockeye salmon
O. tshawytscha = Chinook salmon

One of the more comprehensive papers on the lethal effects of PCP on salmonids described a series of acute toxicity tests conducted on a range of early life stage rainbow trout (Van Leeuwen *et al.* 1985). LC_{50} 96-hour values for six early life stages (from egg through early fry) were determined. LC_{50} values ranged over 167 fold, with eggs being the least sensitive and early fry, the most sensitive life stages. Table 8a lists the LC_{50} value of 18 µg/L for the most sensitive life stage tested, early fry. The second most sensitive life stage was sac fry, with an LC_{50} of 32 µg/L. Van Leeuwen *et al.* did not develop a NOAEL for these life stages, so we cannot assess whether the proposed chronic criterion of 8.2 µg/L and acute criterion of 10.6 µg/L (adjusted for pH) would be protective against significant mortality of sensitive early life stage salmonids. As Chapman (1998) indicates, one problem with this study design is that acetone, which may or may not enhance toxicity of impurities in PCP, was used in the test groups but not in the control. Chapman (1998) also notes another flaw of this study is that pH was not monitored, so it is unclear what the pH was during the test. Nonetheless, the Van Leeuwen *et al.* (1985) study indicates the relative sensitivities in mortality between various early life stage of salmonids due to short-term exposures of PCP.

There are differences in the 96-hour LC_{50} calculated for early life stage salmonids between the Van Leeuwen *et al.* (LC_{50} = 18 µg/L) and the Dominquez and Chapman (66 µg/L) studies. The early fry stage (approximately 77 days), found to be the most sensitive in the Van Leeuwen study, appears to have been tested in the Dominquez and Chapman study. Chapman (1998) maintains that the fry used in their study were "probably farther advanced" than the developmental stage of the fry found to be most sensitive in the Van Leeuwen *et al.* study; this contention is difficult to verify given that neither Van Leeuwen et al (1985) or Dominquez and Chapman (1984) provide specific information on state of yolk sac absorption in the fry tested, and the studies test different forms of the same species (anadromous steelhead versus rainbow trout). Chapman (1998) suggests that factors responsible for the differences in LC_{50} s include the use of acetone as a carrier in the Van Leeuwen *et al.* study, or differences in pH not measured in the Van Leeuwen study. Other experimental design differences between the two studies include: static renewal versus flow-through design, differences in purity of the PCP compound , and variety of salmonid (steelhead versus rainbow trout). Nevertheless, the essential point is that both studies indicate that PCP causes significant lethality in early life stage salmonids after exposures as short as 4 days. The narrow range between the proposed acute and chronic criteria is insufficient to protect early life stage, since the chronic criterion is a four-day average concentration limit which is also the duration of these acute studies. There is only a 2-fold difference between the chronic criterion and the LC_{50} for early fry determined by Van Leeuwen *et al.* (1985) (8.2 versus 18 µg/L). There is only a 6-fold difference between the chronic criterion and the LC_{50} for fry determined by Dominquez and Chapman (1984) (10 versus 66 µg/L). Since the LC_{50} is the concentration at which half of the organisms die, both these studies suggest it is likely that some mortality would occur at PCP concentrations at or below the proposed chronic criterion.

An interlaboratory bioassay testing program was conducted using rainbow trout, coho salmon, and sockeye salmon (Davis and Hoos 1975). The pH of the test water varied with lab, as did the LC_{50} values which ranged from 37 µg/L to 130 µg/L sodium pentachlorophenate. No apparent species sensitivity in acute lethality was observed, and the authors concluded that any major variation in toxicity value were explained by physical and chemical characteristics of the bioassay (pH, water temperature, etc.)

The U.S. Fish and Wildlife Service (USFWS 1986) conducted a series of acute bioassays using technical grade PCP and the sodium salt (Na PCP), on various life stages of chinook salmon and rainbow trout. The results of these studies indicate that swim-up, sac fry and eyed embryos of chinook and rainbow trout are less sensitive than the 1.0 g- size fry to the acute exposures of both technical grade PCP and NaPCP. The lowest LC_{50} was for a 0.3 g chinook salmon: the 0.3 g fry was twice as sensitive as the 1.0 g fry (LC_{50} s of 31 µg/L vs. 68 µg/L technical grade PCP). For 1.0 g fry, chinook were somewhat less sensitive than rainbow trout to technical grade PCP (LC_{50} s of 68 µg/L and 34 to 52 µg/L, respectively). Similarly for NaPCP, chinook fry were somewhat less sensitive than rainbow trout (LC_{50} s of 67 µg/l and 55 to 58 µg/L respectively). It is interesting to note that the 24-hour LC_{50} values for 1.0 g-size fry are very close, or identical to, the 96-hour LC_{50}. This suggests that short-term exposures of PCP to ELS salmonids are as detrimental as 4-day exposures. In other words, the exposure time for mortality to occur is very short.

A series of acute lethality studies on salmonids (USEPA 1995c) evaluated three different listed salmonid species against the rainbow trout. This study found that there were species differences in sensitivity under acute exposures, with the Apache trout being more sensitive than the other species tested. The 96-hour LC_{50} s from these studies were higher by a factor between 3 to 9 than the other acute studies listed in Table 8a. During the test, there was a variation in pH, and some of the test runs had dissolved oxygen levels below 60% saturation at 48 hours or below 40% saturation at 96 hours. USEPA (1985) found that there was no apparent trend in results for test with varying water quality, and did not eliminate any tests or modify calculation of LC_{50} s. As was found in the USFWS (1986) studies, the 24-hour LC_{50} s were close to the 96-hour LC_{50}, indicating the exposure time for mortality to occur is very short. USEPA (1995) concluded , “Further [acute] testing should be conducted with other listed species or their FWS-identified surrogate species before definitive policy decisions concerning the protection of endangered and threatened species are made”.

To summarize the various acute lethality studies conducted on ELS salmonids, the LC_{50} s on rainbow trout fry (0.5 to 1.0 g) using technical grade PCP (USFWS 1986) were lower than similar studies using purified PCP (Dominguez and Chapman 1984). The results of the Van Leeuwen et al (1985) on 97 percent purified PCP had the lowest LC_{50} of 18 µg/L. The studies conducted by USEPA (1995) on acute lethality of similar -size rainbow trout fry were from 3 to 9 times higher (indicating less sensitivity) than either of the previous studies. The 96-hour LC_{50} s for early fry rainbow trout (which appears to be one of the most sensitive life stages) varies between 18 to 160 µg/L, or almost an order of magnitude. Factors that may contribute to the variation in LC_{50} values include differences in form of PCP tested and the pH of the test solution.

As Table 8a indicates, the acute criterion at the pH of the test solution is below the LC_{50} value.

However, by definition the LC_{50} is the concentration at which half of the organisms are expected to die, and cannot be used to determine the concentration that would be lethal to low numbers of salmonid trout exposed for a short period of time. Therefore, due to the uncertainty as to the true LC_{50} for ELS salmonids using commercial grades of PCP, there is an apparent need for EPA to conduct additional acute bioassays. Also, due to the uncertainty as to the true LOAEL and NOAEL for sublethal effects for ELS salmonids using commercial grades of PCP under acute exposures, there is an apparent need for EPA to conduct additional acute bioassays using sensitive sublethal endpoints.

<u>Chronic</u> <u>Studies</u>: Chronic studies are summarized in Table 8b. A chronic exposure study on early life stage salmonids was conducted by Dominguez and Chapman (1984) using purified PCP instead of commercial grade PCP. They exposed rainbow trout from the embryo stage through 72 days of development. Dominguez and Chapman found 34 percent mortality at 19 µg/L PCP at the end of the test. A significant reduction in weight of the trout at 19 µg/L PCP was observed compared to controls (32% reduction in weight). At 11 µg/L PCP level, weight was reduced 15% compared to controls, but was not statistically significant. Other effects observed included increased fin erosion, mild malformations, and lethargy. A NOAEL for mortality of 11 µg/L was also determined. The pH-adjusted chronic criterion would be 10 µg/L, which is essentially the same as the acute NOAEL. One limitation of the Dominquez and Chapman study is that only nominal concentrations of PCP in test water are reported; water samples do not appear to have been analyzed to confirm the test concentrations. Another limitation with this study is that purified PCP, not commercial PCP was used in the test. As discussed in more detail below, purified PCP formulations are believed to be less toxic than commercial PCP formulations. Therefore, the Dominquez and Chapman (1984) NOAEL of 11 µg/L using purified PCP suggests that the chronic criterion of 10 µg/L at pH =7.4 would not be protective of salmonids exposed to commercial forms of PCP.

Early work by Chapman (1969) found an average of 15% reduced weight gain compared to controls in alevins (sac-fry) exposed to 30 µg/L PCP for between 20 and 35 days at 10 and 15 °C. Juvenile steelhead had a 17% reduction in weight gain compared to controls after a 3 week exposure to 30 µg/L PCP. A NOAEL could not be determined from these experiments because 30 µg/L was the lowest concentration tested and because Chapman did not statistically evaluate the data for differences. Chapman (1969) concludes that alevin growth decreased by 6% for each 10 µg/L increase in PCP. These observed effects on growth in both sac-fry and juvenile salmonids after a few weeks of exposure indicate that growth is a sensitive sublethal endpoint for early life stage salmonids.

In a study using young-of-the-year sockeye salmon, Webb and Brett (1973) derived median effect concentrations for growth rate and food conversion efficiency. The EC50 for growth effects was calculated to be 1.74 µg/L, and for food conversion efficiency was calculated as 1.8 µg/L (Webb and Brett 1973). This concentration was approximately 2.8 percent of the 96-hour LC 50. Chapman (1998) notes that the graphical techniques used by Webb and Brett provide a best estimate of an effect-no effect threshold concentration, and not an EC50 as is commonly interpreted (the concentration at which 50 percent of the organisms are expected to exhibit the sublethal response). The study design also varied the exposure duration for different test concentrations, making comparisons between various test concentrations and controls difficult. The control and 3.42 µg/L PCP exposure had the same exposure duration of 56 days; a 10% reduction in growth was observed at that concentration compared to controls. Whether that level of reduced growth was statistically significant was not determined by the authors. Effects on growth rate and conversion efficiency continued post-exposure at greater than 2 µg/L PCP, although some recovery from effects was observed. Swimming performance was not affected in this test, leading these researchers to conclude that growth responses are more sensitive indicators than swimming**.** Chapman (1998) criticized this study as being unrealistic because the flowrate of 20 cm/sec during the tests may have unrealistically increased the energy demands of the fish, making them more sensitive than usual to the effects of PCP. However, Webb and Brett (1973) concluded that feeding and assimilation efficiency were unaffected by PCP, which implies that unusual energy demands were not placed on the fish at the flowrate of the study. Additionally, 20 cm/sec is within the range of swimming speeds reported for underyearling coho salmon of 6 to 30 cm/second (Sandercock 1991). Since the observed effects were seen during PCP exposure, in contrast to a control group that also experienced the same flowrate, the Services conclude that this study is relevant.

In a study by Matida *et al.* (1970), rainbow trout fry were exposed to 3, 8 and 20 µg/L PCP for 28 days. At 20 µg/L PCP mortality was greater than in the controls (13.3% vs. 3.3%), and there was decreased weight gain compared to controls (39.7% versus 98.3%). At 8 µg/L PCP, mortality also appeared elevated compared to controls (16.7% versus 3.3%), and weight gain was apparently decreased (70.4% versus 98.3%). At 3 µg/L PCP, mortality was elevated compared to controls (16.7% versus 3.3%), and weight gain was decreased slightly (92.8% versus 98.3%). Use of this study to set criteria is problematic because the study design did not allow for evaluating the statistical significance of the results, and it does not appear that pH was measured during the test. There appears to be a dose-response to PCP for weight gain, but not for mortality. This study, along with the study by Webb and Brett (1973) indicate that growth is a more sensitive endpoint than mortality for young salmonids, and that effects on growth occur at concentrations at or below the proposed chronic criterion.

One of the few studies to date on reproductive effects in adult salmonids was conducted by Nagler *et al.* (1986). This study revealed adverse impacts on ovarian development at 22 µg/L after an 18-day exposure. Effects on ovarian development were not seen at 11 µg/L, the adjusted chronic criterion (rounded). However, this study was conducted on purified PCP, not technical grade PCP, the formulation released into the environment. Cleveland *et al.* (1982) demonstrated that contaminants in technical grade PCP increased the sublethal toxicity to fathead minnow by a factor of 6 compared to purified PCP. Therefore, it has not been shown that the proposed chronic criterion would be protective against reproductive effects in adult salmonids chronically exposed to technical grade PCP. PCP has been shown to affect reproduction in adult salmonids, as well as having lethal and sublethal effects on early life stage salmonids. The cumulative effect of both reduced reproductive success in adults along with reduced survival or fitness of young, is not addressed by the proposed chronic criterion.

It has been established that commercial PCPs are significantly more toxic to aquatic organisms than are the purified forms of PCP (Cleveland *et al*. 1982; Eisler 1989). Chapman (1998) criticizes the Cleveland *et al*. 1982 study, which demonstrated that the commercial PCP was more toxic than purified forms to fathead minnow in a partial life-cycle test, because small amounts of acetone were used to solubilize the PCP. However, as previously stated, no studies have been performed to confirm this hypothesis. Chapman (1998) cites his own work as not indicating a difference in toxicity between pure and technical grade PCP. However, in the Dominquez and Chapman (1984) study, fry that were past yolk sac absorption and exogenous feeding were exposed to purified PCP, while Chapman (1969) exposed fry to commercial PCP prior to onset of exogenous feeding. Thus, the differences in life-stage tested between the two studies confounds the interpretation of toxicity due to either purified or commercial PCP. Chapman (1998) suggests that technical grade PCP can vary in the nature and toxicity of impurities, and proposes using Whole Effluent Toxicity (WET) testing as a regulatory option for discharges of PCP. Therefore, there is a need for EPA to evaluate using WET in permitted discharges. However, WET would be less useful for evaluating non-point sources of commercial PCPs in the environment, or in establishing ambient water quality criteria.

In summary, the papers cited above indicate that the proposed chronic criterion for PCP would not be protective against lethal or sublethal effects on early life stage salmonid species. Because of the effects on adult reproduction, and effects on early life stage salmonids observed at concentrations at or below the proposed chronic criterion, there is an apparent need for EPA to conduct critical life-cycle tests on salmonids in a manner which meets their requirements for deriving a chronic value, using commercial preparations of PCP. Such tests should include the effects of pH, elevated temperatures, and low dissolved oxygen on lethal and sublethal effects to salmonids, and should include sensitive endpoints such as growth and behavior.

Chapman (1998) concludes, "Overall, the Services are justifiably concerned that the current EPA criterion for PCP might not be sufficiently conservative to provide protection for endangered species of salmonid fish and perhaps other nonsalmonid species. It appears that the most defensible means of providing this protection is to use a more conservative acute-to-chronic ratio and include further protection to account for expected conditions of dissolved oxygen reduction and/or temperature elevation." Chapman (1998) also reviews the literature and the acute-to-chronic ratio used by EPA and concludes, " The Services' comments regarding the EPA's derivation of an acute-to-chronic ratio are apt. I agree with their finding that a larger ACR [acute-to-chronic ratio] is suggested by the available data." Chapman derives an acute-to-chronic ratio for the protection of fish species of 5.219 for PCP (in contrast to an acute-to-chronic ratio of 2.608 cited in USEPA 1995b). Therefore, there is an apparent need for EPA to re-evaluate the basis for the acute-to-chronic ratio.

Cumulative and Interactive Effects: Another study on early life stage steelhead trout, conducted by Chapman and Shumway (1978), examined the effects of low dissolved oxygen in conjunction with PCP exposure. These researchers found significant mortality in early life stage salmonids at 10 µg/L PCP under low dissolved oxygen conditions. This study indicates the importance of other water quality parameters in addition to pH in establishing water quality criteria. Chapman (1998) concludes that the Chapman (1969) and Chapman and Shumway (1978) studies "probably understate the effects that would be observed in a true early life stage study." Thus, exposure to the chronic criterion for PCP is likely to result in increased mortality of early life stage salmonids under low dissolved oxygen conditions.

A study on juvenile chinook salmon was conducted by Iwama *et al.* (1986). Chronic exposure to 3.9 µg/L resulted in alteration of blood chemistry parameters (blood urea nitrogen (BUN) and glucose (GLU)). As noted by Chapman (1998), the significance of the altered blood chemistry is uncertain as to impacts on growth, survival and behavior. However, Iwama *et al.* (1986) indicate that these altered blood chemistry are indicative of hyperglycemia and suggest the effect is due to the stress of PCP exposure, though they do not rule out handling as a possible factor causing the stress. The altered blood chemistry is further evidence that adverse biochemical effects on salmonids may occur at levels below the proposed chronic criterion. Results of this study also suggest, but are not conclusive, that there may be an interaction between infectious agents and PCP in the concentration range of the proposed water quality criteria, with PCP exposure possibly enhancing the effects on infected fish. No changes in feeding or schooling behavior were observed at either test concentration.

Hodson and Blunt (1981) investigated the interactive effects of PCP and temperature on early life stages of rainbow trout. The study found that at 20°C, biomass of fish exposed to 11 to 16 µg/L NaPCP was reduced compared to controls. Reduced biomass, wet weight, and growth rate were observed both for fish exposed as embryos and for fish exposed at day of hatch, through 4 weeks of feeding as fry. In contrast, under a colder temperature regime (10°C), biomass of early life stage was not reduced until PCP concentrations were greater than 20 µg/L. At PCP concentrations greater than 20 µg/L (10°C), mortality of embryos and larvae, delayed hatching and reduced yolk sac resorption efficiency were observed, in addition to effects on biomass and growth rate. Hodson and Blunt also observed that early life stage salmonids exposed from fertilization were more sensitive to the effects of PCP than salmonids exposed only after hatch. Mortality of early life stage was determined to be a function of PCP concentration, temperature, and life-stage exposed. Effects on growth rate of early life stage were a function of PCP concentration and temperature, but not the life-stage exposed. Thus, this study demonstrates that temperature and life-stage are important considerations in developing a chronic criterion for PCP, in addition to pH. This study indicates that in warm water environments the proposed chronic criterion would not be protective of salmonids to sublethal effects of reduced growth rate and weight.

In summary, the proposed chronic criterion does not address the cumulative and interactive effects of PCP toxicity through the critical life-cycle, or under conditions of elevated temperatures or reduced dissolved oxygen. There is an apparent need for EPA to revise the proposed chronic criterion to address the cumulative and interactive effects of PCP toxicity under conditions of elevated temperatures or reduced dissolved oxygen.

Alternative Chronic Criteria**:** In the EPA's consultant review of the draft biological opinion (Chapman 1998), the reviewer proposed several different alternative chronic criteria. One proposal was to use acute toxicity values for carp (Verma *et al*. 1981, Hashimoto *et al*. 1982, and Matida *et al*. 1970). The study by Verma *et al*. (1981) on 3-day old carp larvae (*Cyprinus carpio*) found a 96-hour TL50 of 9.5 µg/L PCP, and a maximum acceptable threshold concentration (MATC) of between 0.5 to 0.6 µg/L PCP (based on survival and growth after 60 day exposure). However, the PCP in the test was not measured, nor was the pH. Because of the uncertainty in the pH and PCP concentration, we disagree that this study demonstrates that carp are more sensitive than salmonids to the acute effects of PCP. This study does however suggest that growth and mortality after chronic exposures is a sensitive endpoint for fish, given the low MATC derived. A study by Matida *et al*. (1971) further calls into question the contention by Chapman (1998) that carp are more sensitive than trout to PCP. In this study, both trout and carp fry were exposed to technical grade PCP under both acute and chronic exposures. The results of the acute study indicated that the 96-hour LC_{50} for trout are almost a factor of 3 lower than for carp. The differences in sensitivity were even more pronounced in the chronic study evaluating growth and mortality over 28 days for the trout, and 70 days for the carp. At 20 µg/L PCP, growth and mortality of carp fry were similar to that of the control after 70 days. In contrast, 20 µg/L PCP exposure to trout fry for only 28 days resulted in greater mortality than in the controls (13.3% vs. 3.3%), and decreased weight gain (39.&% versus 98.3%). At 8 and 3 µg/L PCP, mortality also appeared elevated compared to controls, and 8 µg/L appeared to affect growth. Use of this study to set criteria is problematic because the study design did not allow for evaluating the statistical significance and it does not appear that pH was measured during the test. Finally, the study by Hashimoto *et al*. (1982) using early life stage carp to test the acute toxicity of a commercial emulsifiable concentrate of PCP found little difference in sensitivity between the early life stage tested. This is in contrast to the findings of Van Leeuwen *et al*. (1985) who found sensitivity of salmon early life stage varied over 160-fold. In summary, the Services are unconvinced that using the carp studies to revise the final acute value and then derive a chronic criterion, as suggested by Chapman (1998), would be protective of early life stage salmonids.

Dr. Chapman (1998) also proposed revising the chronic criterion by using the existing final acute value of 10.56 µg/L PCP (at pH=6.5), along with two different revised acute-to-chronic ratios, to yield values of 2.02 µg/L and 2.94 µg/L (at pH = 6.5). This compares to an EPA proposed criterion of 4.04 µg/L (at pH = 6.5). Such an approach may protect early life stage salmonids from significant mortality, although it is unclear if the greater toxicity of commercial PCPs, as compared to purified PCP, is accounted for in the final acute value. This approach would not be protective of sublethal effects on early life stage salmonids. Alternatively, Dr. Chapman proposes that the chronic criterion be 5.8 µg/L (at pH=7.4), based upon the highest concentration showing no adverse effect on mortality or growth (Chapman and Dominquez 1984). However, this study was conducted on purified PCP, and therefore it is not clear that this alternative criterion would be protective of early life stage salmonids exposed to commercial forms of PCP. The study by Little *et al.* (1990), finding behavioral effects at 2 µg/L after only 4 days exposure and no effect at 0.2 µg/L of commercial PCP, suggests that a chronic criterion protective of both lethal and sublethal effects would be in the range of 0.2 to 2.0 µg/L (at pH=7.8). This range for the chronic criterion is also supported by the studies of Webb and Brett (1973) which found the threshold for effects on growth rate and food conversion efficiency to be around 2 µg/L (at pH=6.8).

The essential difficulty in devising an appropriate chronic criterion for protection of endangered salmonids is due to the apparent dearth of chronic toxicity tests which meet the EPA's exacting guidelines. The EPA has defaulted to using the approach of altering the final acute value by an acute-to-chronic ratio. It is clear from the numerous studies previously cited that sublethal effects on growth and behavior are the most sensitive endpoints for chronic exposure of PCP to salmonids, and that the approach of deriving a chronic criterion by adjusting the final acute value is inadequate. Therefore, there is an apparent need for EPA to conduct critical life-cycle, tests on salmonids in a manner which meets their requirements for deriving a chronic value, using commercial preparations of PCP. Such tests should include the effects of pH, elevated temperatures, and low dissolved oxygen on lethal and sublethal effects to salmonids, and should include sensitive endpoints such as growth and behavior. In the interim, the Services conclude that the existing data support a chronic criterion of between 0.2 to 2.0 µg/L PCP to be protective of early life stage salmonids (at pH 7.8).

Non-salmonid fish

There is limited information available on the acute toxicity of PCP to other federally listed fish species such as the Delta smelt, Lost River sucker, Modoc sucker, shortnose sucker, tidewater goby, unarmored three-spine stickleback, and Sacramento splittail. A study by Hedtke *et al.* (1986) determined a 96-hour LC_{50} of 85 µg/L for the white sucker (*Catastomus commersoni*) at a pH range of 7.4 to 8.4. The life stage or age of the fish was not provided. The sucker was more sensitive than the other two fish species tested, the fathead minnow (96hr. LC_{50} s = 120-510), and the bluegill (96hr. LC_{50} s = 200 and 270). A study by Adema and Vink (1981) found both the 48 hour and the 7 day LC_{50} of 450 µg/L for adult saltwater goby (*Gobus minutus*) at pH of 8.

To evaluate the early life stage effects on growth and behavior seen in salmonids, it is useful to compare those studies to other studies using similar endpoints with non-salmonid fish. Data on chronic toxicity to early life stage fish are also available for the fathead minnow, largemouth bass, and guppy. In a study by Brown *et al.* (1985), juvenile guppies were exposed to PCP (form not specified) for 4 weeks and general behavior, predator efficiency, and predator-prey response were observed. No effect was observed at 100 µg/L PCP, while behaviors indicative of decreased response to predators were observed at 500 and 700 µg/L. The lowest observable adverse effect level (LOAEL) of 500 µg/L is approximately 50 percent of the 96-hour LC_{50} of 1020 µg/L. In contrast, for salmonids the LOAEL for swimming activity of 2 µg/L is approximately 4 percent of the 96-hour LC_{50} value of 53 µg/L (Little *et al.* 1990). In a study on largemouth bass fry, Johansen *et al.* (1987) determined the chronic thresholds for food conversion efficiency and growth to be both approximately 24 µg/L of reagent grade PCP. These chronic values are about 15 percent of the 96 hr. LC_{50} of 159 µg/L (Johansen *et al.* 1985). In a related study larval largemouth bass were exposed to reagent grade PCP for 8 weeks. The LOAEL for reduced feeding and growth was 45 µg/L, or approximately 16 percent of the 96-hour LC_{50} of 281 µg/L (Johansen *et al.* 1985; Brown *et al.* 1987). In a study on fathead minnows, embryos were exposed to PCP (93.7 percent pure) for 32 days and hatchability, weight, and survival were observed. No effects on hatchability or weight were seen at concentrations ranging from 16.9 to 176 µg/L. However, none of the early life stage minnows survived in the 176 µg/L test concentration, which is about 37 percent of the 96-hour LC_{50} determined for the egg. It appears, therefore, that chronic effects observed in early life stage salmonids occur at lower concentrations relative to the LC_{50} in other fish species tested. This is stated with caution however, because some of the chronic early life stage tests on non-salmonid fish were done with purified forms of PCP, which have been shown to be less toxic. For example, a 90 day study of early life stage fathead minnows conducted by Cleveland *et al.* (1982) using a composite commercial PCP determined a LOAEL for growth of 13 µg/L at pH=7.4, which is near the level of the proposed chronic criterion of 10 µg/L at that pH. Therefore, the limited literature on early life stage non-salmonid fish suggest that criteria which are protective of salmonids are likely to be protective of non-salmonids.

Bioaccumulative Effects

The proposed criteria for PCP use a BCF from water to fish tissue of 11. Eisler (1989) cites several studies showing much greater BCFs in fish. At 25 µg/L PCP, the BCF for trout muscle was 40 (as cited by Eisler). In studies cited in USEPA (1986b; Table 5) using non-salmonid fresh and saltwater fish, BCFs ranged from 7.3 to over 1000. It appears from the summary table in USEPA (1986b) that the BCF may be inversely related to the water concentration, with higher BCFs occurring at lower water concentrations of PCP. Chapman (1998) notes that a perusal of this same summary table suggests that BCFs seem to increase with decreasing pH. This phenomenon was demonstrated in goldfish exposed for 5 hours to PCP (Kishino and Kobayashi 1995). In that study, a BCF for PCP of 584 was determined at pH = 6; a BCF of 118 was found at pH=8; and a BCF of 8.9 was reported at pH=10. The duration of exposure may also determine the BCF; longer exposure durations may result in higher BCFs.

A study conducted by Niimi and McFadden (1982) found that PCP uptake from water is an important pathway for accumulation in fish over 115 days exposure. Water concentrations were less than 1 µg/L PCP, or well below the proposed water quality criteria. In their protocol, concentrations in fish were determined by removing intestinal content and discarding liver and gall bladder. BCFs in the study were in the range of 200 to 240, which are about 20-fold greater than the BCF used in the proposed water quality criteria.

The EPA consultant who reviewed the Services' draft biological opinion concurred, stating "Certainly the BCF of 11 does not appear to be appropriate based upon the information currently available" (Chapman 1998). Chapman notes that the Final Residue Value (FRV) approach was not used in the Great Lakes Initiative (USEPA 1995b), nor is a FRV identified in this proposed rule or by the Services. While true, the choice of BCF should be based upon a more thorough review of the literature. Moreover, the higher BCF for PCP suggests that wildlife ingesting contaminated food may be at risk. Therefore, there is an apparent need for EPA to reevaluate the BCF, and to evaluate the effect of PCP on wildlife that ingest aquatic organisms exposed to PCP.

It has been established that commercial PCPs are significantly more toxic to aquatic organisms than are the purified forms of PCP (Eisler 1989). Also of concern is that impurities occurring in commercial preparations of PCP have been found to contain relatively high concentrations of polychlorinated dibenzodioxins and dibenzofurans (PCDD/Fs), hexachlorobenzene, chlorinated phenoxyphenols, and chlorinated diphenyloxides. Chlorinated phenoxyphenols and other compounds found in PCP can be precursors to the formation of PCDD/Fs (Cleveland *et al.* 1982; Hamilton *et al.* 1986). PCDD/Fs are known to bioaccumulate in the environment and are also highly toxic to avian and mammalian wildlife. The bioaccumulation and chronic toxicity to wildlife of the other impurities found in commercial PCPs are not addressed by the proposed criteria. Therefore, there is an apparent need for EPA to also evaluate bioaccumulation and chronic toxicity to wildlife of the other impurities found in commercial PCPs.

Summary of Pentachlorophenol Effects on Listed Species

Based on the documented toxicity of pentachlorophenol to early life stage salmonids, with adverse effects seen at water concentrations between 2.5 to 7.5 times below the proposed chronic criterion, together with the potential for exposure of anadromous salmonids to occur, the Services conclude that the proposed numeric criteria are likely to significantly impair the survival and recovery of all listed anadromous salmonids, and are likely to adversely affect populations of the Lahontan cutthroat trout, Paiute cutthroat trout, and Little Kern golden trout if an exposure pathway is created within the habitat for these species.

The toxicity of PCP to non-salmonids, particularly the chronic toxicity, is difficult to assess due to a paucity of testing with the more toxic commercial grade PCP. In one of the few studies to use commercial PCP with non-salmonids the LOAEL for fathead minnow was within a few µg/L of the proposed chronic criterion for PCP. The Services therefore believe that chronic exposures at concentrations approaching the chronic criterion may also pose a potential hazard to some non-salmonid species. Among the non-salmonids, suckers and minnows appear more sensitive. The chronic criterion for PCP also fails to consider highly variable bioconcentration factors, an appropriate acute to chronic ratio, and differences in toxicity between commercial and purified PCP with regard to the acute to chronic ratio. The Sacramento splittail, delta smelt, Modoc sucker, shortnose sucker, and Lost River sucker all reside within watersheds in which pentachlorophenol exposure could occur. The Services therefore conclude that chronic exposure to PCP at concentrations below the criteria concentrations could have the potential to produce toxic effects in these species.

EPA Modifications Addressing the Services' April 9, 1999 draft Reasonable and Prudent Alternatives for Pentachlorophenol (PCP):

The above effect analysis considers the draft CTR as originally proposed in August of 1997. EPA has agreed by letter dated December 16, 1999, to modify its action for PCP per the following to avoid jeopardizing listed species.

A. *"By March of 2001, EPA will review, and if necessary, revise its recommended 304(a) chronic aquatic life criterion for PCP sufficient to protect federally listed species and/or their critical habitats. In reviewing this criterion, EPA will generate new information on chronic sub-lethal toxicity of commercial grade PCP, and the interaction of temperature and dissolved oxygen, to protect early life-stage salmonids. If EPA revises its recommended 304(a) criterion, EPA will then propose the revised PCP criterion in California by March 2002. If the proposed criterion is less protective than proposed by the Services in their opinion or if EPA determines that a proposed criterion is not necessary, EPA will provide the Services with a biological evaluation/assessment by March 2002 and will reinitiate consultation. EPA will keep the Services informed regarding the status of EPA's review of the criterion and any draft biological evaluation/assessment associated with the review. If EPA proposes a revised PCP criterion by March 2002, EPA will promulgate a final criterion as soon as possible, but no later than 18 months, after proposal."*

B. *"EPA will continue to use existing NPDES permit information to identify water bodies which contain permitted PCP discharges and Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) and Resource Conservation and Reclamation Act (RCRA) sites that potentially contribute PCP to surface waters. EPA, in cooperation with the Services, will review these discharges and associated monitoring data and permit limits, to determine the potential for the discharge to impact federally listed species and/or critical habitats. If discharges are identified that have the potential to adversely affect federally listed species and/or critical habitat, EPA will work with the Services and the State of California to address the potential effects to the species. EPA will give priority to review data for fresh water bodies within the range of federally listed salmonids that currently lack a MUN designation as specified in the Regional Water Quality Control Boards' Basin Plans."*

Services' Assumptions regarding EPA modifications to the proposed action for removing jeopardy for PCP.

The Services anticipate the 304(a) criteria guidance for PCP will be revised by EPA to be sufficiently protective of salmonids by March 2001 and that criteria will be applied to all the appropriate water bodies within California no later than September 2003.

The Services recognize there are some scientific uncertainties and additional research is needed to determine the appropriate PCP criteria revision. Therefore, while EPA proposes to revise the criteria after generating new data, the Services assume that if new criteria are not developed, the new information generated regarding the toxicity of commercial grade PCP and the interaction of temperature, pH and DO on sublethal acute and chronic toxicity to early life stage salmonids would conclusively demonstrate that the criteria as originally proposed by EPA (in the draft CTR) are sufficiently protective. The Services assume this information will be provided in sufficient detail to the Services in a biological assessment/evaluation to complete consultation by on the PCP criteria by March 2002, if necessary.

The Services assume a review of PCP monitoring and discharge data on existing hazards to salmonids in California water bodies will occur sometime during the year 2000 and that EPA will use existing authorities to identify and reduce PCP hazards to listed salmonids.

Cadmium

Adequacy of Proposed Chronic Criterion for Cadmium

The Services find that the chronic aquatic life criterion for cadmium proposed in the CTR does not protect listed salmonid and stickleback fish. The adequacy of cadmium criteria to protect certain sensitive species of aquatic organisms has apparently been in doubt for quite some time. In Eisler's (1985a) synoptic review of cadmium hazards, the author commented on the then current EPA 1980 cadmium criterion of 0.012 µg/L saying "even these comparatively rigorous criteria are not sufficient to protect the most sensitive species of freshwater insects, plants, crustaceans, and teleosts". (note to the reader: all cadmium concentrations discussed in this section are at 50 mg/L hardness unless noted otherwise). The EPA in their 1985 criteria document for cadmium (USEPA 1985b) raised the chronic criterion to 0.66 µg/L and noted that "if brook trout, brown trout, and striped bass are as sensitive as some data indicate, they might not be protected by this criterion". The 1985 criterion was also three to five times higher than the species mean chronic values for two cladoceran species which are important food sources for numerous juvenile and adult fish species. In 1995, the EPA again updated and increased the chronic cadmium criterion to 1.4 µg/L (USEPA 1996b) but did not make note of their own concerns that the previous criterion may not have been protective. In a ten year period the chronic cadmium criterion was increased 100-fold although there was doubt that certain salmonid species would be protected even with the lowest criterion. Pascoe and Mattey (1977) found in long-term tests that cadmium caused death in stickleback at concentrations measured at 0.8 µg/L (hardness of 103-111 mg/L as $CaCO_3$)and presumably causes toxic sub lethal effects at lower concentrations. Additional concerns of the Services over formulaic modifications of cadmium regulation on a dissolved basis are included in the formula-based metals section of this opinion.

Cadmium Criteria History

The EPA, in the 1976 criteria document, noted the sensitivity of salmonids and cladocerans (USEPA 1976). For soft water (0 - 75 mg/L), EPA recommended a 0.4 µg/L criterion specifically for salmonids and cladocerans. This was an order of magnitude below the recommended criterion for other nonsensitive species. The 1980 acute criterion was 1.2 µg/L and the chronic criterion was 0.012 µg/L using a hardness dependent formulas. Eisler did not consider these criteria sufficiently protective of the most sensitive aquatic species (Eisler, 1985a).

In the document "Ambient Water Quality Criteria for Cadmium - 1984" (USEPA 1985b), the EPA had difficulties in determining final acute and chronic values. The acute data ranged widely with a salmonid being 3,400 times more sensitive than goldfish. When the final acute value was calculated, the value (8.917 µg/L) was higher than the acute toxicity to several trout species. To protect these commercially and recreationally important species, EPA lowered the value to 3.589 µg/L. This value was then divided by two for the acute aquatic criterion of 1.8 µg/L.

If sufficient data on chronic toxicity are available, the chronic criterion can be calculated using the same method as that used to develop the acute criterion or the chronic criterion can be determined by dividing the final acute value by the final acute to chronic ratio(ACR). In most cases for metals the EPA has used the ACR method (see USEPA metal criteria documents). The ACR is an acute effects concentration divided by a chronic effects concentration for the same species. However, the thirteen cadmium ACRs ranged from 0.9 to 433.8 and did "not seem to follow a pattern"(i.e. did not increase or decrease as the acute values increased or decreased, were not within a factor of ten). Based on the data, EPA decided that it was not "reasonable" to use a final ACR to determine a final chronic value. As an alternative, EPA took the thirteen genus mean chronic values and used the final acute value procedure to calculate a final chronic value. The chronic value initially calculated was 0.0405 µg/L. Although this value is over three times higher than the 1980 criterion of 0.012 µg/L it is still three to four times lower than the chronic toxicity concentrations for the most sensitive species tested. EPA then stated "however, because the thirteen genus mean chronic values contain values for five of the six freshwater genera that are acutely most sensitive to cadmium, it seemed more appropriate to calculate the final chronic value using N = 44, rather than N = 13...". N is the number of data points available and is used in one of the formulas to calculate the final acute or chronic values. In this case EPA used the acute N value (number of acute data points) to calculate the chronic value. It is not clear to the Services why using the acute N value to calculate the chronic criterion is "more appropriate". After making these adjustments a final chronic criterion of 0.66 µg/L was calculated. This value is higher than the chronic toxicity values for two cladocerans (see discussion below), is 16.5 times higher than the value calculated using the chronic N value, and is 55 times higher than the previous chronic criterion.

In 1995, EPA updated criteria for several pollutants including cadmium (USEPA 1996b). While some new acute data on cadmium were included and some older data were eliminated, it is unclear to the Services why a 1995 update did not use post 1986 cadmium references. The result of this recalculation was an acute value of 2.1 µg/L, a slight increase over the older value of 1.8 µg/L. For the chronic value, three of the old data points were eliminated because two values were determined using river water and in the other the cadmium concentration had not been directly measured. Two of the eliminated data points were for the second and fourth most sensitive genera. This had a significant effect on the calculations since the data for the four most sensitive genera are ultimately used in the final chronic value calculation. Three new data points were added to the original 1985 chronic data set. One became the highest chronic value in the data set at 20 µg/L for an oligochaete (an "aquatic earthworm") but this value does not directly affect the calculations. The other two new values were eventually not used in the calculations because data for a more sensitive species in that genera was used as the genus mean chronic value for the final calculations. The update again used the acute N value to calculate the chronic criterion although the elimination of data made the EPA's reason for using the acute N value rather than the chronic N value less "appropriate" because the twelve genus mean chronic values now contain values for four (rather than five) of the six freshwater genera that are acutely most sensitive to cadmium. The 1995 recalculation doubled the chronic value to 1.4 µg/L from the old 0.66 µg/L and is over 100 times higher than the 0.012 µg/L criterion of 1980. If EPA had used the chronic N value to calculate the chronic criterion a value of 0.096 µg/L would have been obtained.

As previously noted, EPA did not use the ACR method to determine the chronic criterion because the ratios did not follow any clear trends. If the ACR method had been used there are several options that can be considered: 1) use all fresh and salt water ACRs available, 2) use all fresh water ACRs, or 3) use the fresh water ACRs of those species with mean acute values closest to the final acute value. Taking the 1985 data as updated in 1995 the ACR chronic values would be 1) 0.11 µg/L, 2) 0.07 µg/L, and 3) 0.18 µg/L. For the third method, three ACR values were used and included the two most chronicly sensitive species (daphnia and chinook salmon) which were also two of the four most acutely sensitive species. Also, the three species mean acute values were within a factor of ten.

Based on the evaluations above using the chronic N value and looking at several ACR methods, it appears that a continuous concentration criterion for cadmium that would be protective of salmonids and stickleback is somewhere between 0.096 and 0.180 µg/L, but probably would still not protect cladocerans.

Considering that the 1985 criteria document noted that the chronic criterion may not be protective of some cladoceran and trout species, it appears unusual that the 1995 update, which doubles the chronic criterion, makes no mention of this lack of protection. Since the original 1985 chronic cadmium criterion may not have been protective of cladocerans and several trout species, the Services conclude the 1995 updated chronic criterion will not be protective of listed salmonid species either and therefore the proposed CTR chronic criterion for cadmium will not be protective.

Considering that the only data available on cadmium toxicity to threespine stickleback shows that the species is highly sensitive at concentrations below the proposed criterion, the Services conclude that the proposed chronic criterion will not be protective of this species.

The Services also conclude that the additional loss of protection due to the proposed regulation of cadmium on a dissolved basis using a formula-based criterion, as discussed elsewhere in this opinion, adds to the likelihood of adverse effects to listed salmonid species and the unarmored threespine stickleback.

Cadmium Hazards to Aquatic Organisms

Sources

Eisler's synoptic review (1985a), EPA's criteria document (USEPA 1985b), Sorensen (1991), and Moore and Ramamoorthy (1985) provide a good summary of cadmium sources and pathways. Cadmium is not a biologically essential metal. It is a soft metal with properties similar to zinc. Cadmium is most often found with sulfide ores and is frequently associated with other metals such as zinc, copper, and lead. Mining and ore smelting are significant sources of cadmium to the environment via direct discharge of mine drainage and atmospheric deposition. Cadmium is frequently associated with industrial discharges and stormwater runoff. Uses of cadmium include electroplating, pigments, plastic stabilizers, batteries, and electronic components. Background concentrations of cadmium in freshwater ranges from <0.01 to 0.2 µg/L and are usually less than 0.05 µg/L in waters unimpacted by man (USEPA 1985b, Eisler 1985a, Wren et al., 1995). The maximum background concentrations are close to or at concentrations that can be harmful to sensitive aquatic species. Human activities can raise cadmium concentrations to levels >1 µg/L.

Pathways

For cadmium and other dissolved metals the most direct pathway to aquatic organisms is via the gills. Cadmium is also directly taken up by bacteria, algae, plants, and planktonic and benthic invertebrates. Another biologically significant pathway for exposures of aquatic organisms to cadmium is through consumption of contaminated aquatic detritus, plants, invertebrates, and other food items. Dietary exposure and association with sediment is significant in cadmium accumulation in fish species (Sorensen 1991). Omnivorous fish tend to accumulate higher levels of cadmium than carnivorous fish and bottom feeding fish tend to accumulate more cadmium than free-swimming fish feeding in the water column.

General Toxicity of Cadmium

Cadmium damages gill, liver, kidney, and reproductive tissue (Eisler 1985a; Sorensen 1991; Moore and Ramamoorthy 1984). Acute mechanisms of cadmium toxicity to fish do not appear to be the same as chronic mechanisms. In acute tests cadmium accumulates in gill tissue to a greater extent than elsewhere, whereas, in chronic tests at lower concentrations, cadmium accumulates more in liver and kidney tissue. The principle acute effect is gill toxicity leading to an aquatic organism's inability to breath. Long term effects include the inability to regulate plasma constituents, produce healthy bones, and reproduce. Cadmium will compete with essential metals such as zinc for enzyme binding sites, thus disrupting normal enzyme functions. Hypocalcemia also occurs due to exposure to cadmium thus causing muscular and neural abnormalities. Cadmium is considered a teratogenic substance.

The toxicity of cadmium varies greatly among aquatic species (USEPA 1985b). Mean acute values for sensitive life stages of freshwater fish range from 1.6 µg/L for brown trout to 7,685 µg/L for mosquitofish. The most sensitive species being salmonids, striped bass, and cladocerans. Acute toxicity for chinook salmon is 4.254 µg/L. Mean acute values for less sensitive species range as high as 1,200 µg/L for midge larvae to 12,755 µg/L for crayfish. The goldfish mean acute value is 8,325 µg/L. Hardness, pH, alkalinity, salinity, and temperature can significantly affect cadmium toxicity.

USEPA (1985b) shows mean chronic toxicity concentrations for two cladocerans at 0.1918 µg/L and 0.1354 µg/L. USEPA (1996b), noted additional low chronic values for cladocerans at 0.12, 1.25, 3.919, 4.0, and 6.096 µg/L. Four of the cladoceran values were not used in the calculation of the 1995 criterion for reasons noted above. As sensitive as cladocerans seem to be it is possible that the life stage of cladocerans being used in most bioassays are not the most sensitive. Shurin and Dodson (1997) found that sexual reproduction in cladocerans is more sensitive to toxicants than the asexual reproductive stage and that most bioassays utilize daphnia during the asexual phase because they are well fed and cultured under low stress situations. Under stress (low temperature, drought, low food supply) cladocerans and other zooplankton use sexual reproduction to produce resting eggs that can remain dormant for months to years until more favorable conditions return. The loss or a decrease in the production of resting eggs can have a significant long-term effect on the populations these species. Snell and Carmona (1995) found that for a rotifer zooplankton, sexual reproduction was more strongly affected by several toxicants, including cadmium, than asexual reproduction. The authors concluded that the “level of toxicants presently allowable in surface waters...may expose zooplankton populations to greater ecological risks than is currently believed.”

Mean chronic values in fish range from 2.362 µg/L for the brook trout to 16.32 µg/L for bluegill while the mean chronic value for early life stage chinook salmon is 2.7 µg/L. Pascoe and Mattey (1977) found that cadmium at concentrations as low as 1 µg/L can be toxic to the three-spined stickleback after 33 days. Acute to chronic ratios also vary greatly among test organisms and range from 0.9 to 433.8.

There is very little information on the toxicity of cadmium to amphibians. USEPA (1985b) notes data on three species. The EC_{50} (death and deformity) of embryo and larval narrow-mouthed toads (*Gastrophyryne carollnensis*) after seven days at a hardness of 195 mg/L was 40 µg/L. The 48 hr LC_{50} (death) of African clawed frogs (*Xenopus laevis*) at 209 and 170 mg/L hardness was 11,700 and 3,200 µg/L respectively. After 100 days African clawed frogs showed signs of inhibited development at 650 µg/L at a hardness of 170 mg/L. Finally, marbled salamander (*Ambystoma opacum*) embryos and larvae had an EC_{50} (death and deformity) of 150 µg/L at a hardness of 99 mg/L after eight days. The sensitive life stages of these species appear to be similar in their sensitivity to cadmium as adult goldfish and fathead minnows. Concentrations of cadmium that would be protective of salmonids would protect amphibians.

<u>Summary of Cadmium Criteria Effects to Listed Species</u>

Fish

Salmonid species are particularly sensitive to cadmium. USEPA (1996c) shows mean acute toxicity values of sensitive life stages for coho salmon at 5.894 µg/L, chinook salmon at 4.254 µg/L, rainbow trout at 3.589 µg/L, and brown trout at 1.638 µg/L. Chronic values for coho salmon, chinook salmon, brown trout, and brook trout are 2.324 µg/L, 2.694 µg/L, 7.372 µg/L, and 2.194 µg/L respectively. These low concentrations reduce growth, survival, and fecundity.

Increased water temperature increases cadmium toxicity (Eisler 1985a; USEPA 1985b; Sorensen 1991; and Moore and Ramamoorthy 1985). Increased temperature is a major problem for listed salmonids in California due, in part, to logging activities decreasing riparian shading of streams and dams increasing water temperatures in reservoirs.

Cladocerans and other invertebrates are very sensitive to cadmium. They also provide significant food sources for early life stage salmonids and other aquatic organisms that are themselves prey items for salmonids. It also appears that the least sensitive reproductive stage of zooplankton such as cladocerans is more often used for bioassays leading to an underestimate of their sensitivity to various toxicants including cadmium (Shurin and Dodson 1997, Snell and Carmona 1995). A loss of this prey base can indirectly impact salmonids and stickleback.

Pascoe and Cram (1977) found lethal chronic toxicity of cadmium to the three-spined stickleback (*Gaserosteus aculeatus L.*) at all tested concentrations with the lowest concentration tested being 300 µg/L. An interaction was also found between the incidence of parasitism and sensitivity to cadmium. Subsequently Pascoe and Mattey (1977) performed a long-term (89 day) study on three-spined stickleback at concentrations of cadmium from 100,000 µg/L to 1 µg/L. Lethality to the stickleback was again found at all concentrations tested. The authors determined a 96 h LC_{50} of 23,000 µg/L but went on to say; "The results confirm earlier work (Pascoe & Cram 1977) that cadmium is highly toxic to sticklebacks. It is now seen to cause death at concentrations as low as 0.001 mg l^{-1} [1 µg/L] in water of total hardness 103-111 mg l^{-1} as $CaCO_3$ at 15° C, and presumably causes toxic sub lethal effects at lower concentrations." The median period of survival at 1 µg/L was 48,000 minutes (33.3 days). At 3.2 µg/L the median survival time was 23,000 minutes (16 days). The nominal concentration at this low level was 0.001 mg l^{-1} while the measured concentration was 0.0008 mg l^{-1} (0.8 µg/L). This chronic data, while cited, was not used by EPA in criteria calculations. However, the Services and EPA must consider this relevant and available data for evaluation of potential effects of permissible cadmium concentrations to the listed subspecies of the stickleback *(G. aculeatus williamsonii).*

The Services believe that all ESUs and runs of coho and chinook salmon and steelhead trout, Lahontan cutthroat trout, Paiute cutthroat trout, Little Kern golden trout, along with the unarmored threespine stickleback are likely to be adversely affected by concentrations of cadmium at or below those that would be allowed in the proposed CTR.

EPA modifications addressing the Services' April 9, 1999 draft Reasonable and Prudent Alternatives for Cadmium:

The above effect analysis evaluates the draft CTR as originally proposed in August of 1997. EPA has agreed by letter dated December 16, 1999, to modify its action for cadmium per the following to avoid jeopardizing listed species.

"EPA will develop a revision to its recommended 304 (a) chronic aquatic life criterion for cadmium by January 2001 to ensure the protection of federally listed species and/or critical habitats and will propose the revised criterion in California by January 2002. However, if EPA utilizes the revised metals criteria model referred to below, EPA will develop a revision to its recommended 304(a) criterion by January 2002 and will propose the revised criterion in California by January 2003. EPA will solicit public comment on the proposed criteria as part of its rulemaking process, and will take into account all available information, including the information contained in the Services' opinion, to ensure that the revised criterion will adequately protect federally listed species. If the revised criterion is less stringent than that proposed by the Services in the opinion, EPA will provide the Services with a biological evaluation/assessment on the revised criterion by the time of the proposal to allow the Services to complete a biological opinion on the proposed cadmium criterion before promulgating final criteria. EPA will provide the Services with updates regarding the status of EPA's revision of the criterion and any draft biological evaluation/assessment associated with the revision. EPA will promulgate final criteria as soon as possible, but no later than 18 months, after proposal. EPA will continue to consult, under section 7 of ESA, with the Services on revisions to water quality standards contained in Basin Plans, submitted to EPA under CWA section 303, and affecting waters of California containing federally listed species and/or their habitats. EPA will annually submit to the Services a list of NPDES permits due for review to allow the Services to identify any potential for adverse effects on listed species and/or their habitats. EPA will coordinate with the Services on any permits that the Services identify as having potential for adverse effects on listed species and/or their habitat in accordance with procedures agreed to by the Agencies in the draft MOA published in the Federal Register at 64 F. R. 2755 (January 15, 1999) or any modifications to those procedures agreed to in a finalized MOA."

Services' Assumptions Regarding EPA's Modifications for Removing Jeopardy for Cadmium.

The Services assume the 304(a) cadmium chronic aquatic life criterion can and will be revised by EPA to be sufficiently protective of sticklebacks and salmonids in California by no later than January 2001. The Services assume that this revision will result in lowering the permissible concentrations of cadmium. Further, the Services assume this scientific guidance can and will be used in revising permits during the interim period prior to promulgation of this criterion in California.

If, however, the criterion proposed by EPA is less stringent than that suggested by the effects analysis of the Services, EPA will provide a new biological assessment with new information that indicates why a criterion less stringent than that suggested by the Services will be sufficiently protective.

The Services assume that because EPA offered to revise the chronic aquatic life criterion for cadmium by January 2001 that this is achievable by EPA. There is a discrepancy in EPA's letter about when a new criteria model for metals will be developed per paragraphs IV and V in EPA's December 16, 1999 letter. June of 2003 is presented as the date of the model revision for metals criteria, but paragraph IV states the 304a criterion for cadmium per the new model would be ready by January 2002. The Services' view is that an earlier revision as proposed by EPA without the new metals model that protects these listed species is preferable and should be pursued by EPA to provide the earliest possible increase in protection.

Metals

Adequacy of Proposed Criteria

Metals addressed in the CTR include: arsenic (As), cadmium (Cd), trivalent chromium (CrIII), hexavalent chromium (CrVI), copper (Cu), lead (Pb), mercury (Hg), nickel (Ni), selenium (Se) in saltwater, silver (Ag), and zinc (Zn). Although mercury, cadmium and selenium are discussed in separate sections of this biological opinion, this section on conversion factors and water effect ratios also applies to proposed mercury and saltwater selenium criteria. The formula-based metals are included in this single discussion as a group because the key issues of how dissolved metal criteria are derived and the implications are similar for each of them. That is, the formula-based metal method does not sufficiently consider the environmental fate, transport, and transformations of metals in natural environments.

Use of Formulas

The EPA proposes to promulgate within the CTR aquatic life criteria that are formula-based for the following metals: As, Cd, Cr(III), Cr(VI), Cu, Pb, Hg, Ni, Se (in saltwater), Ag, and Zn. To determine criteria for these metals that are applicable to a given water body, site-specific data must be obtained, input to a formula, and numeric criteria computed. There are three types of site-specific data that may be necessary to determine and/or modify the criterion for a metal at a site: water hardness, conversion factors and translators, and water effect ratios. The following is a brief description of these types of data.

1. Formulas for Cd, Cu, Cr(III), Pb, Ni, Ag, and Zn are water hardness dependent. The Services assume that the measure of hardness referred to in the CTR is a measure of the water hardness due to calcium and magnesium ions. By convention, hardness measurements are expressed in terms of the mg/L of $CaCO_3$ required to contribute that amount of calcium + magnesium hardness. Therefore, the site-specific hardness is determined at a site, expressed as mg/L of $CaCO_3$, then input to the criteria formulas for each metal. Originally criteria were determined using data on the total metal concentration (dissolved and particulate) in the test water. Thus, the general formula for a hardness based chronic criterion or Criterion Continuous Concentration (CCC) on a total metal basis is:

$$CCC = e^{(m[\ln(hardness)]+b)}.$$

As an example, for Cu, the following data can be input to the general formula above: a site hardness of 40 mg/L and the slope (m) and intercept (b) for copper hardness dependent chronic toxicity (from CTR Table 2). The Criterion Continuos Concentration (CCC) for Cu, on a total basis would be:

$$\begin{aligned} CCC\ (total) &= e^{(0.8545[\ln(40)]+(-1.702))} \\ &= 4.3\ \mu g/L \end{aligned}$$

Criteria for Cd, Cu, Cr(III), Pb, Ni, Ag, and Zn can not be found directly by seeking out a reference like the CTR, because numbers listed in such tables are usually based on the assumption that the site-specific hardness is 100 mg/L (the CCC for Cu at this hardness is 9.3 µg/L). Criteria for these metals require that site-specific hardness is measured and input to the formula, as demonstrated above.

2. Formulas for all the metals also include a total-to-dissolved conversion factor (CF) based on the fraction of the metal that was in a dissolved form during the laboratory toxicity tests used to develop the original total based criteria. Criteria as proposed in the CTR would be on a dissolved basis. Table 1 in the CTR lists the CFs for the metals. The modified formula becomes:

$$CCC\ (dissolved) = CF\ x\ e^{(m[\ln(hardness)]+b)}.$$

Using the hardness, slope, and intercept values from above and the CF from Table 1 in the CTR, the dissolved Cu chronic criterion would be:

$$\begin{aligned} CCC\ (dissolved) &= 0.96\ x\ e^{(0.8545[\ln(40)]+(-1.702))} \\ &= 4.1\ \mu g/L \end{aligned}$$

There is an added level of complexity in the computations of criteria for Cd and Pb because the CFs for these metals are themselves hardness dependent. For example, the formula to derive the hardness-dependent CF for the chronic (CCC) Cd criterion is:

$$CF = 1.101672-[(\ln\{hardness\})(0.041838)]$$

This hardness-specific CF would then be entered into the formula for Cd and the criterion would be calculated similar to the example above.

If a total maximum daily load (TMDL) is needed to regulate discharges into an impaired water body, the dissolved criterion must be converted or translated back to a total value so that the TMDL calculations can be performed. The translator can simply be the CF (divide the dissolved criterion by the CF to get back to the total criterion) or site-specific data on total and dissolved metal concentrations in the receiving water are collected and a dissolved-to-total ratio is used as the translator.

3. Formulas for all the metals listed above also include a Water Effects Ratio (WER), a number that acts as a multiplication factor. If no site-specific WER is determined, then the WER is presumed to be 1 and would not modify a formula result. A WER purportedly accounts for the difference in toxicity of a metal in a site water relative to the toxicity of the same metal in reconstituted laboratory water. The contention is that natural waters commonly contain constituents which "synthetic" or "reconstituted" laboratory waters lack, such as dissolved organic compounds, that may act to bind metals and reduce their bioavailability. Where such constituents act to modify the toxicity of a metal in a site water compared to the toxicity of the same metal in laboratory water, a "water effect" is observed.

Example WER calculation:

Suppose the LC_{50} of Cu in site water is 30 µg/L.
Suppose the LC_{50} of Cu in laboratory water is 20 µg/L.
As before assume a site hardness of 40 mg/L.
The freshwater conversion factor (CF) for Cu = 0.96.

$$\text{WER} = \frac{\text{Site } \mathbf{LC_{50}}}{\text{Lab } \mathbf{LC_{50}}} = \frac{30\ \mu g/L}{20\ \mu g/L} = 1.5$$

$$\begin{aligned}\text{Cu Site-Specific CCC} &= \text{WER x CF x } e^{(m[\ln(40)]+b)}\\ &= 1.5 \text{ x } 0.96 \text{ x } 4.3\\ &= 6.2\ \mu g/L\end{aligned}$$

What follows are discussions of the Services' concerns regarding the applications of WER, CF and the attendant translators, and deficiencies of the hardness-dependent factors in formula-based determinations of criteria for As, Cd, Cr (III), Cr (VI), Cu, Pb, Hg, Ni, Se (in saltwater), Ag, and Zn.

Water Effect Ratios

Except in waters that are extremely effluent-dominated, WERs are > 1 and result in higher numeric criteria. Note that, in the examples above, use of a site-specific WER for copper raised the criterion concentration allowed at the site from 4.1 µg/L to 6.2 µg/L, an increase of 50 percent. A WER may be more important than site water hardness or metal-specific conversion factors and translators in determining a criterion and hence the metal loading allowed (see hardness and ading discussions below).

EPA has published guidelines for determining a site-specific WER, which outline procedures for water sampling, toxicity testing, acclimating test organisms, etc. (USEPA 1994). When site water toxicity is lower than laboratory water toxicity, criteria may be raised because: 1) differences in calcium to magnesium ratios in hardness between laboratory water and site water can significantly alter the WER; 2) toxicity testing for WER development is not required across the same range of test organisms used in criteria development; and 3) the inherent variabilities associated with living organisms used in toxicity testing can be magnified when used in a ratio.

EPA guidelines for WER determinations (USEPA 1994) instruct users to reconstitute laboratory waters according to protocols that result in a calcium-to-magnesium ratio of ~0.7 across the range of hardness values (USEPA 1989, 1991). This proportion (~0.7) of calcium to magnesium is far less than the ratio found in most natural waters (Welsh *et al*. 1997). The Services agree with Welsh *et al*. (1997) that imbalances in Ca-to-Mg ratios between site waters and dilution waters may result in WERs which are overestimated because calcium ions are more protective of metals toxicity than are magnesium ions. The EPA has noted this problem with determining WERs but limits the suggested correction of matching the laboratory Ca-to-Mg ratio and the site ratio to a single sentence at the end of the proposed rule. Thus, the significance and correction of this problem is not adequately addressed.

EPA metal criteria are based on over 900 records of laboratory toxicity tests (USEPA 1992) using hundreds of thousands of individual test organisms, including dozens of species across many genera, trophic levels, and sensitivities to provide protection to an estimated 95 percent of the genera most of the time (USEPA 1985f). The use of a ratio based WER determined with 2 or 3 test species limits the reliability of the resultant site-specific criteria and calls into question the level of protection provided for families or genera not represented in the WER testing

The inherent variability of toxicity testing can also have a significant effect on the final WER determination, especially because it is used in a ratio. As discussed above, the EPA has developed its criteria based on a relatively large database. However, even with such a large database variability in test results can still cause difficulty in determining a criteria value. For example, Cd data were so variable that EPA abandoned the acute to chronic ratio method of determining the chronic criterion (USEPA 1985b). Instead, EPA applied the acute method to derive a chronic value. The EPA criteria document for Cd (USEPA 1985b) notes a chronic value for chinook salmon of 1.563 µg/L with a range of 1.3 to 1.88 µg/L. This is a variability of 17 percent in either direction, which is rather good (inter and intra laboratory variability higher than 17 percent is not unusual). Therefore, if this data is used in a ratio such as a WER, the variability alone could result in a 34 percent difference in the values used. A potential WER using such data could range from 0.7 to 1.4. Thus, a site-specific criteria could increase by 40 percent due to natural variability in the toxicity testing alone. In development of a site-specific WER, fewer tests are conducted and with fewer species, increasing the likelihood that natural variation in toxicity test results could affect the outcome. Care should also be taken to make sure that test results between lab and site water are significantly different. If 95 percent confidence intervals for the tests overlap then they are likely not significantly different and should not be used to determine a WER. Thus, toxicity tests should be conducted and carefully evaluated to minimize experimental variance when collecting data to calculate WERs.

Zooplankton such as cladocerans (*Daphnia sp.*) are commonly used in bioassays to determine national and site-specific criteria or develop WERs and translation factors. As sensitive as cladocerans seem to be it is possible that the life stage of cladocerans being used in most bioassays are not the most sensitive. Shurin and Dodson (1997) found that sexual reproduction in cladocerans is more sensitive to toxicants than the asexual reproductive stage and that most bioassays utilize daphnia during the asexual phase because they are well fed and cultured under low stress situations. Under stress (low temperature, drought, low food supply) cladocerans and other zooplankton use sexual reproduction to produce resting eggs that can remain dormant for months to years until more favorable conditions return. The loss or a decrease in the production of resting eggs can have a significant long-term effect on the populations of these species. Snell and Carmona (1995) found that for a rotifer zooplankton, sexual reproduction was more strongly affected by several toxicants, including cadmium, than asexual reproduction. The authors concluded that the "level of toxicants presently allowable in surface waters . . . may expose zooplankton populations to greater ecological risks than is currently believed." Other metals may also be more toxic to the sexual stage of zooplankton adding additional doubt to the protectiveness of some criteria and WERs.

Procedures for acclimation of test organisms prior to toxicity testing may also be inadequate to assure meaningful comparisons between site and laboratory waters. For the reasons stated above, the Services believe that the EPA procedures for determining WERs for metals may result in criteria that are not protective of threatened or endangered aquatic species. Thus, WERs of three (3) or less are unacceptable because they are likely within the variance of the toxicity tests. WERs over three must be carefully developed and evaluated to ensure that listed species will be protected.

Conversion Factors and Translators

EPA derived ambient metals criteria from aquatic toxicity tests that observed the dose-response relationships of test organisms under controlled (laboratory) conditions. In most of these studies, organism responses were plotted against nominal test concentrations of metals or concentrations determined on unfiltered samples. Thus, until recently metals criteria have been expressed in terms of total metal concentrations. Current EPA metals policy (USEPA 1993a) and the CTR in particular propose that criteria be expressed on a dissolved basis because particulate metals contribute less toxicity than dissolved forms. EPA formulas for computing criteria thus are adjusted via a conversion factor (CF), so that criteria based on total metal concentrations can be "converted" to a dissolved basis. Metals for which a conversion factor has been applied include arsenic, cadmium, chromium, copper, lead, mercury, nickel, silver, and zinc.

The CF is a value that is used to estimate the ratio of dissolved metals to total recoverable metals to adjust the former criteria based on total metal to yield a dissolved metal criterion. A CF based on the premise that the dissolved fraction of the metals in water is the most bioavailable and therefore the most toxic (USEPA 1993a, 1997c). The presumption is that the dose/response relationships found in toxicity tests would be more precise if "dissolved" metal concentrations were determined in test solution samples that have been filtered to remove the larger-sized, particulate metal fraction. The term "total" metal refers to metal concentrations determined in unfiltered samples that have been acidified (pH < 2) before analysis. The term "dissolved" metal refers to metal concentrations determined in samples that have been filtered (generally a 0.45-micron pore size) prior to acidification and analysis. Although it is clear that concentrations determined in a procedurally-defined dissolved sample are not accurate measures of dissolved metals, it may be premature to recommend immediate changes to the current procedure (Chapman 1998). Particulate metals can be single atoms or metal complexes adsorbed to or incorporated into silt, clay, algae, detritus, plankton, etc., which can be removed from the test water by filtration through a 0.45 micron filter. A CF value is always less than 1 (except for As which is currently 1.0) and is multiplied by a total criterion to yield a (lower) dissolved criterion. For example, CF values for Cd, Cu, Pb, and Zn, are 0.944, 0.960, 0.791, and 0.978 respectively (USEPA 1997c). The CF values approach 100 percent for several metals because they are ratios determined in laboratory toxicity-test solutions, not in natural waters where relative contributions of waterborne particulate metals are much greater. The California Department of Fish and Game (CDFG 1997) has commented that particulate fractions in natural waters in California are often in the range of 80 percent, which would equate to a dissolved-to-total ratio of 0.2.

To convert metals criteria, EPA reviewed test data that reported both total and dissolved concentrations in their test waters and also conducted simulations of earlier experiments to determine the dissolved-to-total ratios (USEPA 1992, 1995a, 1997c). In this way, the historical toxicity database could be preserved and a large number of new toxicity tests would not have to be performed. Overall, the CFs proposed in the CTR are based upon roughly 10% of the historical database of toxicity tests. CF values for As and Ni were based on only 1 study each, comprising 11 records. CF values for Cr were based on only 2 studies, while the estimated CF for Pb was based on 3 studies, comprised of only 3 records. Although additional confirmatory studies were performed to develop the CFs, the database available appears to be limited and calls into question the defensibility of the CFs determined for these metals.

Ultimately the scientifically most defensible derivation of dissolved metals criteria should be based on reviews of new laboratory investigations because:

1. the several water quality variables that modulate metal toxicity may not have been properly controlled, measured, reported, or manipulated over ranges that are environmentally realistic and necessary to consider if site-specific criteria are to be proposed (see section on hardness);

2. it is likely that most toxicity tests measured organism responses in terms of traditional endpoints such as mortality, growth, reproductive output. These may not be sufficient for

determining the toxic effects of metals in test waters manipulated to reflect environmental (site) conditions (see section on hardness);

3. the test waters contained very low contributions from particulate metals to the total metal concentrations. These proportions are not environmentally realistic; and

4. the present EPA criteria for metals lack meaningful input and modification from metals toxicity research done in the last decade.

Points 1 and 2 above are discussed in this final biological opinion in the hardness section dealing with the use of water hardness as a general water quality "surrogate". Point 3 is illustrated by the fact that the CF's proposed in the CTR for several metals are near a value of 1.0. This indicates that the toxicity tests reviewed to derive dissolved-based criteria exposed test organisms in waters that contained very low concentrations of particulate metals. For example, the CF values for Cd, Cu, Pb, and Zn, are 0.944, 0.960, 0.791, and 0.978 respectively (USEPA 1997c), meaning that particulate metal percentages were (on average) 5.6%, 4.0%, 20.9%, and 2.2%. These percentages are much lower than found in many natural waters. The California Department of Fish and Game, in their comments to the EPA on the proposed CTR, has stated that particulate fractions in natural waters in California are often in the range of 80 percent (CDFG 1997), which would equate to a dissolved-to-total ratio of 0.2. It is clear that the historical toxicity database does not include studies of the toxic contributions of particulate metals under environmentally realistic conditions. Improved assessments are necessary to develop adequately protective, site-specific criteria.

The EPA Office of Water Policy and Technical Guidance has noted that particulate metals contribute some toxicity and that there is considerable debate in the scientific community on this point (USEPA 1993a). While the Services agree that dissolved metal forms are generally more toxic, this is not equivalent to saying that particulate metals are non-toxic, do not contribute to organism exposure, or do not require criteria guidance by the EPA. Few studies have carefully manipulated particulate concentrations along with other water constituents, to determine their role(s) in modulating metals toxicity. Erickson *et al*. (1996) performed such a study while measuring growth and survival endpoints in fish and suggested that copper adsorbed to particulates cannot be considered to be strictly non-toxic. Playle (1997) cautions that it is premature to dismiss particulate-associated metals as biologically unavailable and recommends the expansion of fish gill-metal interaction models to include these forms. The Service is particularly concerned that investigations have not been performed with test waters that contain both high particulate metal concentrations and dissolved concentrations near the CTR-proposed criteria concentrations. Despite a paucity of information about the aquatic toxicity of particulate metals, the CTR proposes that compliance would be based on removing (filtering) these contaminants from a sample prior to analysis. It would be prudent to first conduct short-term and longer term studies, as well as tests that expose organisms other than fish.

Particulates may act as a sink for metals, but they may also act as a source. Through chemical, physical, and biological activity these metals can become bioavailable (Moore and Ramamoorthy 1984). Particulate and dissolved metals end up in sediments but are not rendered entirely nontoxic nor completely immobile, thus they still may contribute to the toxicity of the metal in natural waters.

Particulate metals have been removed from the regulatory "equation" through at least two methods: the use of a CF to determine the dissolved metal criteria, and the use of a translator to convert back to a total metal concentration for use in waste load limit calculations. When waste discharge limits are to be developed and TMDLs are determined for a receiving waterbed, the dissolved criterion must be "translated" back to a total concentration because TMDLs will continue to be based on total metals.

EPA provides three methods in which the translation of dissolved criteria to field measurements of total metal may be implemented. These three methods may potentially result in greatly different outcomes relative to particulate metal loading. These methods are:

1. Determination of a site specific translator by measuring site specific ratios of dissolved metal to total metal and then dividing the dissolved criterion by this translator. As an example: a site specific ratio of 0.4 (40% of the metal in the site water is dissolved) would result in a 2.5 fold increase in the discharge of total metal. The higher the fraction of particulate metal in the site water the greater the allowable discharge of total metal. See the discussion and Table 9 below. This is EPA's preferred method.

2. Theoretical partitioning relationship. This method is based on a partitioning coefficient determined empirically for each metal and when available the concentration of total suspended solids in the site specific receiving water.

3. The translator for a metal is assumed to be equivalent to the criteria guidance conversion factor for that metal (use the same value to convert from total to dissolved and back again).

Since translators are needed to calculate discharge limits they become important in determining the total metals allowed to be discharged (see also loading discussion for individual metals below. In the economic analysis performed by the EPA and evaluated by the State Board (SWRCB 1997), it was estimated that translators based on site-specific data will decrease dischargers costs of implementing the new CTR criteria by 50 percent. This cost savings is "directly related to the less stringent effluent limitations that result from the use of site-specific translators." This implies a strong economic incentive for dischargers to reduce costs by developing site-specific translators and ultimately being allowed to discharge more total metals. This conclusion regarding the impact of site specific translators is supported by documents received from EPA (USEPA 1997d). EPA performed a sensitivity analysis on the effect of the site specific translator, which relies on determining the ratio of metal in water after filtration to metal in water before filtration in downstream waters. EPA's analysis indicated that use of a site-specific translators to calculate criteria would result in greater releases of toxic-weighted metals loads above the option where the

Cfs are used as the translators. The potential difference was estimated to be between 0.4 million and 2.24 million “toxic weighted” pounds of metals discharged to California waterways.

The Services believe that the current use of conversion factors and site specific translators in formula-based metal criteria are not sufficiently protective of threatened and endangered aquatic species because:

1. particulate metals have been removed from the regulatory equation even though chemical, physical, and biological activity can subsequently cause these particulate metals to become bioavailable;

2. the criteria are developed using toxicity tests that expose test organisms to metal concentrations with very low contributions from particulate metals;

3. toxicity tests do not assess whether the toxic contributions of particulate metals are negligible when particulate concentrations are great and dissolved concentrations are at or near criteria levels;

4. this method has the potential to significantly increase the discharge of total metal loads into the environment even though dissolved metal criteria are being met by a discharger; and

5. the premise ignores the fact that water is more than a chemical medium, it also physically delivers metals to the sediments.

Hardness

The CTR should more clearly identify what is actually to be measured in a site water to determine a site-specific hardness value. Is the measure of hardness referred to in the CTR equations a measure of the water hardness due to calcium and magnesium ions only? If hardness computations were specified to be derived from data obtained in site water calcium and magnesium determinations alone, confusion could be avoided and more accurate results obtained (APHA 1985). Site hardness values would thus not include contributions from other multivalent cations (e.g., iron, aluminum, manganese), would not rise above calcium + magnesium hardness values, or result in greater-than-intended site criteria when used in formulas. In this Biological opinion, what the Services refer to as hardness is the water hardness due to calcium + magnesium ions only.

The CTR should clearly state that to obtain a site hardness value, samples should be collected upstream of the effluent source(s). Clearly stating this requirement in the CTR would avoid the computation of greater-than-intended site criteria in cases where samples were collected downstream of effluents that raise ambient hardness, but not other important water qualities that affect metal toxicity (e.g., pH, alkalinity, dissolved organic carbon, calcium, sodium, chloride, etc.). Clearly, it is inappropriate to use downstream site water quality variables for input into criteria formulas because they may be greatly altered by the effluent under regulation. Alterations in receiving water chemistry by a discharger (e.g., abrupt elevation of hardness, changes in pH, exhaustion of alkalinity, abrupt increases in organic matter etc.) should not result, through application of hardness in criteria formulas, in increased allowable discharges of toxic metals. If the use of downstream site water quality variables were allowed, discharges that alter the existing, naturally-occurring water composition would be encouraged rather than discouraged. Discharges should not change water chemistry even if the alterations do not result in toxicity, because the aquatic communities present in a water body may prefer the unaltered environment over the discharge-affected environment. Biological criteria may be necessary to detect adverse ecological effects downstream of discharges, whether or not toxicity is expressed.

The CTR proposes criteria formulas that use site water hardness as the only input variable. In contrast, over twenty years ago Howarth and Sprague (1978) cautioned against a broad use of water hardness as a "shorthand" for water qualities that affect copper toxicity. In that study, they observed a clear effect of pH in addition to hardness. Since that time, several studies of the toxicity of metals in test waters of various compositions have been performed and the results do not confer a singular role to hardness in ameliorating metals toxicity. In recognition of this fact, most current studies carefully vary test water characteristics like pH, calcium, alkalinity, dissolved organic carbon, chloride, sodium, suspended solids, and others while observing the responses of test organisms. It is likely that understanding metal toxicity in waters of various chemical makeups is not possible without the use of a geochemical model that is more elaborate than a regression formula. It may also be that simple toxicity tests (using mortality, growth, or reproductive endpoints) are not capable of discriminating the role of hardness or other water chemistry characteristics in modulating metals toxicity (Erickson *et al.* 1996). Gill surface interaction models have provided a useful framework for the study of acute metals toxicity in fish (Pagenkopf 1983; Playle *et al.* 1992; Playle *et al.* 1993a; Playle *et al.* 1993b; Janes and Playle 1995; Playle 1998), as have studies that observe physiological (e.g. ion fluxes) or biochemical (e.g. enzyme inhibition) responses (Lauren and McDonald 1986; Lauren and McDonald 1987a; Lauren and McDonald 1987b; Reid and McDonald 1988; Verbost *et al.* 1989; Bury *et al.* 1999a; Bury *et al.* 1999b). Even the earliest gill models accounted for the effects of pH on metal speciation and the effects of alkalinity on inorganic complexation, in addition to the competitive effects due to hardness ions (Pagenkopf 1983). Current gill models make use of sophisticated, computer-based, geochemical programs to more accurately account for modulating effects in waters of different chemical makeup (Playle 1998). These programs have aided in the interpretation of physiological or biochemical responses in fish and in investigations that combine their measurement with gill metal burdens and traditional toxicity endpoints.

The Services recognize and acknowledge that hardness of water and the hardness acclimation status of a fish will modify toxicity and toxic response. However the use of hardness alone as a universal surrogate for all water quality parameters that may modify toxicity, while perhaps convenient, will clearly leave gaps in protection when hardness does not correlate with other water quality parameters such as DOC, pH, Cl^- or alkalinity and will not provide the combination of comprehensive protection and site specificity that a multivariate water quality model could provide. In our review of the best available scientific literature the Services have found no conclusive evidence that water hardness, by itself, in either laboratory or natural water, is a consistent, accurate predictor of the aquatic toxicity of all metals in all conditions.

Hardness as a predictor of copper toxicity: Lauren and McDonald (1986) varied pH, alkalinity, and hardness independently at a constant sodium ion concentration, while measuring net sodium loss and mortality in rainbow trout exposed to copper. Sodium loss was an endpoint investigated because mechanisms of short-term copper toxicity in fish are related to disruption of gill ionoregulatory function. Their results indicated that alkalinity was an important factor reducing copper toxicity, most notably in natural waters of low calcium hardness and alkalinity. Meador (1991) found that both pH and dissolved organic carbon were important in controlling copper toxicity to *Daphnia magna*. Welsh *et al*. (1993) demonstrated the importance of dissolved organic carbon in affecting the toxicity of copper to fathead minnows and suggested that water quality criteria be reviewed to consider the toxicity of copper in waters of low alkalinity, moderately acidic pH, and low dissolved organic carbon concentrations. Applications of gill models to copper binding consider complexation by dissolved organic carbon, speciation and competitive effects of pH, and competition by calcium ions, not merely water hardness (Playle *et al*. 1992; Playle *et al*. 1993a; Playle *et al*. 1993b). Erickson *et al*. (1996) varied several test water qualities independently and found that pH, hardness, sodium, dissolved organic matter, and suspended solids have important roles in determining copper toxicity. They also suggested that it may difficult to sort out the effects of hardness based on simple toxicity experiments. It is clear that these studies question the use of site calcium + magnesium hardness only as input to a formula to derive a criterion for copper because pH, alkalinity, and dissolved organic carbon concentrations are key water quality variables that also modulate toxicity. In waters of moderately acidic pH, low alkalinity, and low dissolved organic carbon, the use of hardness regressions may be most inaccurate. Also, it is not clear that the dissolved organic carbon in most or all waters render metals unavailable. This is because dissolved organic carbon from different sources may vary in both binding capacity and stability (Playle 1998).

Hardness as a predictor of silver toxicity: While there is strong evidence that ionic silver is the form responsible for causing acute toxicity in freshwater fish, recent science (Wood et al, 1999; Bruy eta al, 1999; Karen et al, 1999; Galvez and Wood, 1997; Hogstrand and Wood, 1998) challenges the EPA concept of hardness as having a large ameliorating effect on aquatic toxicity of silver. These studies indicate that chloride and dissolved organic carbon concentrations must be accounted for in the criterion formula for this metal. Bury *et al*. (1999) exposed rainbow trout to silver nitrate and measured physiological (Na^+ influx) and biochemical (gill Na^+/K^+-ATPase activity) endpoints, as well as silver accumulations in gills. They found that chloride and dissolved organic carbon concentrations, but not calcium hardness, ameliorated the inhibition of Na^+ influx and gill Na^+/K^+-ATPase activity. Dissolved organic carbon greatly reduced gill accumulations of silver through complexation. Chloride ion did not reduce gill accumulations of silver because it bound with free silver (Ag^+) and accumulated in gills as AgCl, but reduced toxicity because the AgCl did not enter chloride cells and disrupt ionoregulation.

Calcium, the hardness ion thought to modify metals toxicity to the greatest degree is, by itself, not that protective in the case of silver. Karen *et al*. 1999 found DOC more important than hardness for predicting the toxicity of ionic silver in natural waters to rainbow trout, fathead minnows and *Daphnia magna*. These authors suggested incorporating an organic carbon coefficient into the silver criterion equation to enhance the site specificity of criterion. Wood et al (1999) noted chloride ion and DOC were influential in ameliorating silver toxicity and that in ammonia rich waters silver might be more than additively toxic with ammonia to fish.

<u>Hardness as a predictor of cadmium toxicity</u>: Our review of acute cadmium toxicity in fish indicates that calcium hardness does exhibit ameliorating effects (Reid and McDonald 1988; Verbost *et al*. 1989; Playle and Dixon 1993). However, most studies that manipulated hardness ions varied only calcium and so there is little evidence that magnesium ions ameliorate cadmium toxicity. Investigations of the differences between these two hardness constituents (Carroll *et al*. 1979; Davies *et al*. 1993) revealed that magnesium ions provide little or no protection against acute cadmium toxicity in fish. Hunn (1985) suggested that calcium binds to biological molecules in ways that magnesium does not, due to differences in the coordination geometry of the ions. Mechanistic studies of cadmium toxicity in fish reveal that cadmium inhibits enzyme-mediated calcium uptake in the gills (Verbost *et al*. 1989). Dissolved organic carbon, if present in sufficient concentrations and binding strengths, may also modulate cadmium toxicity. In natural waters hardness, pH, alkalinity, salinity, and temperature may also interact to affect cadmium toxicity but these factors may not always correlate to hardness measures at a given waterbed.

Loading

The Services are concerned that particulate metals discharges from municipal and industrial effluents will likely increase under the CTR proposed criteria. Current guidance for waste load allocation calculations (USEPA 1996b) consists of simple dilution formulations using effluent metal loads, receiving water flows, and dissolved-to-total metals ratios in the receiving waters. To illustrate our concerns, we expanded upon a hypothetical example contained in *The Metal Translator: Guidance For Calculating a Total Recoverable Permit Limit From a Dissolved Criterion* (USEPA 1996b). In this document, EPA provides a procedure for determining the concentration of total Cu that could be discharged in an effluent without exceeding the ambient criterion for dissolved Cu in the receiving water (i.e., a waste load allocation). In order to include additional metals in our analyses (not just Cu), we retained the assumptions of the EPA example for effluent flow, receiving stream flow, and ratio of dissolved metal to total metal in the receiving stream (f_d). For metals other than Cu, we assumed that the total metal in the receiving water, upstream of the discharge, was the same percentage of the National Toxics Rule (NTR) criterion as was assumed for Cu in the EPA example (~23 percent). For the 1992 NTR we assumed the same conditions as the EPA example but the total metal criteria was used.

Table 9 compares the concentration of total metals that could be discharged in an effluent without exceeding the ambient criterion for dissolved metals in the receiving water using: 1) total metal criteria from the 1992 NTR; 2) dissolved metal criteria from the CTR using a 40 percent dissolved-to-total metal ratio ($f_d = 0.4$) in the receiving water body; and 3) dissolved metal criteria from the CTR using a 20 percent dissolved-to-total metal ratio ($f_d = 0.2$). The dissolved-to-total ratio of 0.4 is the same as that used in the EPA example and a ratio of 0.2 is not unusual for natural waters in California (CDFG 1997). It is evident that substantial increases in total metals would be permitted in this hypothetical discharge under proposed CTR criteria. If the dissolved fraction of total metals in the receiving water was 40 percent, then under the CTR, the total metal concentrations that would be allowed to be discharged would increase by 51 to 203 percent compared to the 1992 National Toxics Rule (Table 1). Nickel is the only metal under this scenario that would decrease (-21 percent). If the dissolved fraction of total metals in the receiving water was 20 percent, then under the CTR the total metal concentrations in allowable discharge would increase by 78 to 524 percent, including nickel (78 percent).

It also appears that as the fraction of particulate metal in the receiving water increases, the allowable discharge of particulate metals will increase, rather than decrease. The Services expect that increases similar to our examples would occur in allowable TMDLs under CTR criteria because a TMDL is the instream total metal concentration that equates to the dissolved metal criteria concentration (USEPA 1996b). Under the CTR, total metal discharges may increase as long as the dissolved criteria are not exceeded. Economic analyses of the draft CTR performed by the EPA and SWRCB (1997) show that implementing the new CTR criteria will decrease discharger costs by 50 percent because of "less stringent effluent limitations that result [from] the use of site-specific translators." Therefore, it would be incorrect to assume that TMDLs limit total metal loadings simply because they are expressed as total metal concentrations. Moreover, increases in permitted, point-source metal discharges will be incremental to discharges from agricultural or urban non-point sources, which are largely uncontrolled through the discharge-permitting process. Metals criteria based only on dissolved concentrations provide little in the way of incentives for reducing non-point sources, which are largely particulate forms. The Services are concerned that metals criteria based on dissolved concentrations in the absence of sediment criteria linked to total metals will not effectively prevent sediment contamination by metals and may lead to increased allowable loads of metals to sediments. The dissolved approach ignores the fact that water is more than a chemical medium; it also physically delivers metals to the sediments.

The Services believe that the CTR proposed formula-based metal criteria is not protective of threatened or endangered aquatic species because total metal discharges will likely increase and the criteria development methods do not adequately consider the environmental fate, transport, and transformation of metals in natural environments.

Table 9. Comparison of total metal concentrations permitted in a hypothetical point-source discharge under the 1992 National Toxics Rule that regulated metals on a total basis and the 1997 California Toxics Rule that proposes to regulate metals on a dissolved basis. The CTR concentrations are based on a receiving waterbed's percent dissolved to total metals of 40 and 20 percent. Values in parentheses are percent increase over 1992 NTR. Values are in µg/L total metal.

Receiving Water Percent Dissolved Metals	As	Cd	Cr (III)	Cr (VI)	Cu	Pb	Hg	Ni	Ag	Zn
NTR Total	973	11	1,487	43	48	230	7	3,835	9	318
CTR 40 percent	2,561 (163)	33 (203)	4,150 (179)	122 (182)	100 (106)	488 (112)	10 (51)	3,043 (-21)	26 (179)	888 (179)
CTR 20 percent	5,311 (446)	67 (524)	8,588 (478)	251 (483)	208 (331)	1,011 (339)	22 (219)	6,831 (78)	54 (478)	1,835 (476)

The Services find that the regulation of metals on a dissolved basis using the formulas proposed by the EPA in the CTR does not assure adequate protection of threatened or endangered species and their potential for exposure to dissolved and particulate metals in the water column because:

1. Criteria are based on toxicity tests that expose test organisms to metal concentrations with very low contributions from particulate metals and do not assess exposures under environmentally realistic conditions;

2. Particulate metals have been removed from the equation even though chemical, physical, and biological activity can cause these metals to become bioavailable. While the Services agree that dissolved metal forms are more toxic, this is not equivalent to saying that metals in the particulate fraction are not toxic, will not become toxic, are not being exposed to organisms, and do not require criteria guidance by the EPA;

3. Toxicity tests do not assess whether the toxic contributions of particulate metals are negligible when particulate concentrations are great and dissolved concentrations are at or near criteria levels;

4. The proposed criteria have the potential to significantly increase the discharge of total metal loads into the environment even though dissolved metal criteria are being met by a discharger;

5. The role of major cations (sodium, potassium), anions (nitrate, sulfate, chloride), and other water quality parameters (pH, temperature, dissolve organic matter) that modify metal toxicity may not be assumed to be negligible, thus hardness alone does not fully address site water effects

on toxicity;

6. The regulation of metals on a dissolved basis ignores the fact that water is more than a chemical medium, it also physically delivers metals to the sediments;

7. Larger databases with a wider range of test species used to derive the criteria can be nullified by use of smaller databases with fewer test species to adjust criteria on a site-specific basis via WER and CF translator determinations that use ratios which can greatly modify the final criteria; and

8. Aquatic criteria based on the dissolved metal fraction without concurrent wildlife criteria and sediment criteria fail to address a wide variety of exposure scenarios and effects such as bioaccumulation through the diet and synergism.

For these reasons the Services believe that the proposed formula-based method for developing metal criteria is not sufficiently protective of threatened or endangered aquatic species.

<u>Metal Hazards to Aquatic Organisms</u>

Sources

Eisler's series of synoptic reviews, EPA's criteria documents, Sorensen (1991), and Moore and Ramamoorthy (1985) provide a good summary of sources, pathways, and toxic effects of these metals. Metals in general are widely distributed and frequently, (as in the case of cadmium, copper, lead, and zinc) are found in the same ore deposits. Thus, activities such as mining can be a source of several metals at once. Metals are rarely found alone in discharges or the environment. Several metals are frequently associated with mining discharges, industrial discharges, and stormwater runoff. A variety of inorganic and organic forms of each metal are found in the environment and toxicity among these compounds varies widely.

There is a multitude of uses for these metals in the economy. Past and current uses include the production of numerous alloys, pigments, printing, wood preservatives, batteries, pesticides, electronics, electroplating, plastic stabilizers, tanning, furnaces, dyes, wiring, roofing, anticorrosion, plumbing, solders, ammunitions, gasoline additives, and currency.

Pathways

Because of the wide variety of uses, these metals can and will enter the environment through many pathways. The most direct routes are through acid mine drainage from active and abandoned mines and point-source discharges from industrial activities such as plating, textile, tanning, and steel industries. Municipal waste water treatment plants and urban runoff are also significant source of metals to the environment. Arsenic, copper, and zinc used as pesticides and wood preservatives enter the environment via drift, erosion, surface runoff, and leaching. Copper used as an aquatic herbicide is directly applied to the water under controlled situations. Particulate metals from combustion and dust can be transported through the air.

Metals can enter the aquatic environment in a dissolved form or attached to organic and inorganic particulate matter. The amount of metal in the dissolved versus particulate form in natural waters can vary greatly, but the particulate form is usually found in greater concentrations. Metals can flux between different states and forms in an aquatic environment due to changes in pH, temperature, oxygen, presence of other compounds, and biological activity. These transformations can occur within and between water, sediment, and biota as the cycles of nature change.

As dissolved metals in the water, the most direct pathway to aquatic organisms is via the gills. Dissolved metals are also directly taken up by bacteria, algae, plants, and planktonic and benthic invertebrates. The dissolved forms of metals can adsorb to particulate matter in the water column and enter organisms through various routes. Metals adsorbed to particulates can also be transferred across the gill membranes (Lin and Randall 1990; Playle and Wood 1989; Sorensen 1991; Wright *et al*. 1986). Planktonic and benthic invertebrates can ingest particulate metals from the water column and sediments and then be eaten by other organisms. Thus, dietary exposure is a significant source of metals to aquatic and aquatic dependent organisms.

Although metals bound to sediments are generally less bioavailable to organisms, they are still present, and changes in the environment (e.g., dredging, storm events, temperature, lower water levels, biotic activity) can alter the bioavailability of these metals. The feeding habits of fish can determine the amount of uptake of certain metals. Piscivorous fish are exposed to different levels of metals than omnivorous and herbivorous fish. For example, copper is more commonly found in herbivorous fish than carnivorous fish from the same location (Mathis and Cummings 1971). In general, these metals do not biomagnify in the food chain as do mercury or selenium, thus impacts to resources tend to be limited to aquatic organisms.

General Toxicity of Metals

The toxicity of metals varies greatly depending on the chemical form and valence. Trivalent arsenic and hexavalent chromium are more toxic than other forms of arsenic and chromium, while chelated forms of metals are less toxic than the unbound ions. The various metals can have a wide variety of effects on organisms. They can cause enzyme inhibition due to reactions with the sulfhydryl groups of proteins. Some metals such as cadmium will compete with essential metals such as zinc for enzyme binding sites. Metal exposure can result in damage to gill and gut tissues, disrupt nervous system operation, and alter liver and kidney functions. Some metals can affect olfactory responses which are important to migrating salmonid species. Elevated metal concentrations can cause growth inhibition and impaired reproduction resulting in decreased primary production. An alteration of primary production can then impact growth and survival farther up the foodchain. Impacts from metal contamination can shift species composition and abundance towards more pollution-tolerant species. Copper is highly toxic to most freshwater invertebrates with LC 50s as low as 6 µg/L (Moore and Ramamoorthy 1984). The California freshwater shrimp recovery plan notes this species is at particular risk from copper exposures relative to non-point sources associated with dairy operations and cow foot-baths using copper based compounds (USDI-FWS 1997a).

The toxicity of each metal to different organisms varies greatly. Copper is generally more toxic to aquatic organisms than the other metals. Complex synergistic effects among the metals can occur as well as antagonistic effects. The toxicity of metals can be altered by hardness, salinity, alkalinity, pH, and temperature. For most of the metals in the proposed rule, the criteria are formula based and hardness dependent because increasing hardness decreases the toxicity of the metal.

Particulate Toxicity

In the biological evaluation for the CTR, EPA determined that exposures to ambient concentrations of dissolved metals at or below the proposed CTR aquatic life criteria are unlikely to adversely affect threatened or endangered aquatic organisms (USEPA 1997a). While the CTR criteria proposed for metals are based on the dissolved fractions of these metals only, aquatic organisms in natural waters are exposed to additional, waterborne, particulate metal forms. As discussed in the CF section, the CTR will likely increase particulate metal loading even though dissolved criteria are being met. Dredging and disposal operations can result in substantial suspension and re-suspension of particulates in the water column, including those contaminated with metals.

Through respiratory uptake, aquatic organisms are exposed to metals in addition to those measured in the dissolved fraction of ambient waters. As fish ventilate, a nearly continuous flow of water passes across their gills (Moyle and Cech 1988) and particulate metals suspended in the water column may become entrapped. At the lowered pHs occurring near gill surfaces (Lin and Randall 1990; Playle and Wood 1989; Wright *et al*. 1986) entrapped particulate metals may release soluble metal ions (Sorensen 1991), which are the forms EPA considers most bioavailable and efficiently taken up by aquatic organisms (USEPA 1993a,1997a). Although most research has been done on particulate exposures to fish gills (primarily salmonids), it is reasonable to conclude that other fish and gill breathing organisms are affected in the same way.

Newly developed models seem well suited to assessments of the toxic contribution from suspended particulate metals and could be used to establish safe levels that do not substantially increase respiratory exposures. A panel of toxicologists has recently reviewed metals bioavailability and criteria issues and recommended replacing the current EPA approach to acute criteria development with a mechanistic approach such as a fish gill model (Bergman and Dorward-King 1997). Gill-model approaches have been used to successfully investigate how metal binding at fish gills is influenced by water hardness, pH, alkalinity, and dissolved organic carbon (Playle and Dixon 1993), as well as to estimate how effectively the gill competes with abiotic ligands for metals (Playle *et al*. 1993).

The Services believe that the proposed EPA metals criteria in the CTR for aquatic life should not exclude particulate forms of any metal, unless and until EPA demonstrates that exposures of threatened or endangered species to these contaminants are unlikely to cause adverse effects in natural waters.

Dietary Exposure

A biologically significant pathway for exposures of aquatic organisms to metals is through consumption of contaminated aquatic detritus, plants, invertebrates, and other food items. EPA has not assessed whether the food base of aquatic organisms may accumulate excessive metal residues under CTR proposed criteria. As the CTR preamble quotes from the CWA and EPA's 1985 guidelines, a criterion is the "highest concentration of a substance in water which does not present a significant risk to the aquatic organisms in the water and their uses." Their uses include "consumption by humans and wildlife." Certainly, an ecologically significant use of aquatic invertebrates is their consumptive use by fish. Invertebrates may accumulate appreciable body burdens of metals in aquatic systems and are prey consumed by salmonids and other fish species (Anderson 1977; Cain *et al*. 1992; Cain *et al*. 1995; Clements *et al*. 1994; Dallinger 1994; Elwood *et al*. 1976; Gerhardt and Westermann 1995; Ingersoll *et al*. 1994; Kiffney and Clements 1993; Luoma and Carter 1991; Lynch *et al*. 1988; McKnight and Feder 1984; Moore *et al*. 1991; Phillips 1978; Rainbow and Dallinger 1993; Smock 1983; Smock 1983a; Timmermans 1993; Saiki 1995; Zanella 1982; Moyle 1976; Saiki 1995).

The regulation of water quality criteria on a dissolved basis, as EPA proposes, does not consider particulates, sediment, and dietary exposure routes. In a recent experiment (Woodward *et al.* 1994) age-0 rainbow trout that were held in clean water and fed a diet of metals-contaminated invertebrates (for 91 days) exhibited reduced survival and growth. After 91 days, whole-body metal concentrations were similar to those in trout inhabiting the stream where the contaminated invertebrates were collected. In concurrent treatments, trout exposed to waterborne metals (at concentrations meeting criteria established by the EPA) and fed a diet of uncontaminated invertebrates exhibited no reductions in survival or growth. These results and those of similar studies of diet-borne metal exposures to salmonids collectively suggest that to reduce dietary hazards to salmonids, water quality criteria should protect invertebrate forage from excessive metal residue accumulations (Dallinger and Kautzky 1985; Dallinger *et al*. 1987; Farag *et al*. 1994; Giles 1988; Harrison and Klaverkamp 1989; Harrison and Curtis 1992; Miller *et al*. 1993; Mount *et al*. 1994; Thomann and Harrison 1997; Spry *et al*. 1988; Woodward *et al*. 1995).

The Services believe that without due consideration of dietary exposure of metals to aquatic organisms, the proposed CTR criteria for metals are not protective of threatened and endangered aquatic species. Criteria that are not protective of aquatic invertebrates from contamination and result in subsequent loss of beneficial use by fish and other aquatic organisms are not consistent with the CWA, nor are they protective of listed invertebrates considered in this biological opinion.

Bioaccumulation

As discussed throughout the formula based metals section, organisms are exposed to metals through many routes. These metals do bioaccumulate in the lower trophic levels of aquatic systems (Moore and Ramamoorthy 1984). The Services understand that EPA criteria development guidelines include a component designed to assure that the water quality criterion for a substance is sufficiently low that residue accumulations will not impair the use of aquatic organisms (USEPA 1985c). Data from residue studies are to be considered alongside acute and chronic toxicity data in the criteria development process (USEPA 1985c). However, it appears that the proposed metals criteria are based solely on results of aquatic toxicity tests (USEPA 1997c), where metal exposures occur only across gills or other respiratory surfaces. This is because toxicity tests used to develop the criteria are performed with controlled laboratory water with little particulate metals and do not include realistic dietary or other exposures.

Criteria documents for metals include the discussion of bioaccumulation studies but final criteria are based on acute and chronic toxicity studies. EPA has not considered results of investigations, similar to the studies discussed in the dietary exposure section, which indicate that exposures of salmonids to metals-contaminated invertebrate diets may result in adverse effects. Because EPA is now proposing criteria on a dissolved basis, and for the many reasons discussed throughout the formula-based metal discussion, bioaccumulation becomes even more important in evaluating the protectiveness of those criteria. A panel of toxicologists has recently reviewed metals bioavailability and criteria issues and recommended that ambient water criteria development include a tissue residue/toxicity model (Bergman and Dorward-King 1997).

The Services believe that without due consideration of the bioaccumulation potential of metals in aquatic systems the proposed CTR criteria for metals are not protective of threatened and endangered aquatic species.

Summary of Metal Criteria Effects to Listed Species

In summary, the effects of metals may be generalized to include: central nervous system disruption, altered liver and kidney function, impaired reproduction, decreased olfactory response, delayed smoltification, impaired ability to avoid predation and capture prey, growth inhibition, growth stimulation, changes in prey species community composition increasing foraging budgets, and lethality. The Services believe that all ESUs and runs of coho and chinook salmon and steelhead trout, Lahontan cutthroat trout, Paiute cutthroat trout, Little Kern golden trout, delta smelt, Sacramento splittail, Mohave tui chub, Lost River sucker, Modoc sucker, shortnose sucker, tidewater goby, and unarmored threespine stickleback are likely to be adversely affected by concentrations of particulate and/or dissolved metals at or below those that would be allowable under criteria procedures provided in the proposed CTR.

EPA Modifications to Address the Services' April 9, 1999 draft Reasonable and Prudent Alternatives for Dissolved Metals:

The above effect analysis evaluates the draft CTR as originally proposed in August of 1997. EPA has agreed by letter dated December 16, 1999, to modify its action for metals criteria per the following to avoid jeopardizing listed species.

A. *"By December of 2000, EPA, in cooperation with the Services, will develop sediment criteria guidelines for cadmium, copper, lead, nickel and zinc, and by December of 2002, for chromium and silver. When the above guidance for cadmium, copper, lead, nickel and zinc is completed, Region 9, in cooperation with the Services, will draft implementation guidelines for the State of California to protect federally listed threatened and endangered species and critical habitat in California."*

B. *"EPA, in cooperation with the Services, will issue a clarification to the Interim Guidance on the Determination and Use of Water-Effect Ratios for Metals (USEPA 1994) concerning the use of calcium-to-magnesium ratios in laboratory water, which can result in inaccurate and under-protective criteria values for federally listed species considered in the Services' opinion. EPA, in cooperation with the Services, will also issue a clarification to the Interim Guidance addressing the proper acclimation of test organisms prior to testing in applying water-effect ratios (WERs). "*

C. *"By June of 2003, EPA, in cooperation with the Services, will develop a revised criteria calculation model based on best available science for deriving aquatic life criteria on the basis of hardness (calcium and magnesium), pH, alkalinity, and dissolved organic carbon (DOC) for metals. This will be done in conjunction with "Other Actions A." below."*

D. *"In certain instances, the State of California may develop site-specific translators, using EPA or equivalent state/tribe guidance, to translate dissolved metals criteria into total recoverable permit limits. A translator is the ratio of dissolved metal to total recoverable metal in the receiving water downstream, from a discharge. A site-specific translator is determined on site-specific effluent and ambient data."*

"Whenever a threatened or endangered species or critical habitat is present within the geographic range downstream from a discharge where a State developed translator will be used and the conditions listed below exist, EPA will work, in cooperation with the Services and the State of California, to use available ecological safeguards to ensure protection of federally listed species and/or critical habitat. Ecological safeguards include: (1) sediment guidelines; (2) biocriteria; (3) bioassessment; (4) effluent and ambient toxicity testing; or (5) residue-based criteria in shellfish."

"Conditions for use of ecosystem safeguards:

1. A water body is listed as impaired on the CWA section 303(d) list due to elevated *metal concentrations in sediment, fish, shellfish or wildlife; or,*

2. A water body receives mine drainage; or,

3. Where particulate metals compose a 50% or greater component of the total metal measured in a downstream water body in which a permitted discharge (subject to translator method selection) is proposed and the dissolved fraction is equal to or within 75% of the water quality criteria."

"Whenever a threatened or endangered species is present downstream from a discharge where a State developed translator will be used, EPA will work with the permitting authority to ensure that appropriate information, which may be needed to calculate the translator in accordance with the applicable guidance, will be obtained and used. Appropriate information includes:

4. *Ambient and effluent acute and chronic toxicity data;*
5. *Bioassessment data; and/or*
6. *An analysis of the potential effects of the metals using sediment guidelines, biocriteria and residue-based criteria for shellfish to the extent such guidelines and criteria exist and are applicable to the receiving water body."*

"EPA, in cooperation with the Services, will review these discharges and associated monitoring data and permit limits, to determine the potential for the discharge to impact federally listed species and/or critical habitats. If discharges are identified that have the potential to adversely affect federally listed species and/or critical habitat, EPA will work with the Services and the State of California in accordance with procedures agreed to by the Agencies in the draft MOA published in the Federal Register at 64 FR 2755 (January 15, 1999) or any modifications to those procedures agreed to in a finalized MOA."

Other [EPA] Actions

A. *"EPA will initiate a process to develop a national methodology to derive site-specific criteria to protect federally listed threatened and endangered species, including wildlife, in accordance with the draft MOA between EPA and the Services concerning section 7 consultations."*

Services' Assumptions Regarding EPA's CTR Modifications for regulating dissolved metals that result in Removing Jeopardy to listed species.

FORMULA BASED METALS CRITERIA

The Services assume EPA sediment guidelines for cadmium, copper, lead, nickel and zinc will be in place by December 2000 and sediment guidelines for chromium and silver will be in place by December 2002. The Services assume that these guidelines when implemented will increase protection for federally listed species and critical habitat. We also assume sediment guidelines will be used to limit particulate metal loadings into aquatic ecosystems in California.

The Services assume that the revised guidance on the use of water effect ratios for metals will reduce chances for inaccurate or under protective criteria.

The Services assume that a revised criteria calculation model for metals based on more than hardness, (pH, alkalinity, DOC) will actually result in more accurately protective criteria for federally listed species. The Services assume that use of such a model will require the use of more water quality parameter data (in addition to hardness) from water bodies where criteria are applied and that this supporting information will decrease the likelihood of under protective criteria.

The Services assume the use of site specific translators in metals discharge permits will not be used to allow significant increases in metal loadings in water bodies with mine drainage, or where water bodies are listed as impaired due to metals where listed species may be effected by such increases.

The Services also assume that where particulate metals are being transported to sediments under EPA approved discharge permits, these sediment locations will not exceed EPA guidelines for metals in sediment, especially where these water bodies contain federally listed species or critical habitat.

The Services assume the use of “ecosystem safeguards” such as ambient and effluent toxicity testing, biocriteria, sediment guidelines, and tissue based criteria, will increase the protection afforded federally listed species where metals are regulated on a dissolved basis.

CUMULATIVE EFFECTS

Cumulative effects include the effects of future State, Tribal, local, or private actions that are reasonably certain to occur in the action area considered in this biological opinion. Future Federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to section 7 of the Act.

Cumulative effects on aquatic species including bonytail chub, coho salmon (all California ESUs), delta smelt, desert pupfish, Lahontan cutthroat trout, little Kern golden trout, Lost River sucker, Modoc sucker, Mohave tui chub, Owens pupfish, Owens tui chub, Paiute cutthroat trout, razorback sucker, Sacramento splittail, shortnose sucker, steelhead trout (all California ESUs), tidewater goby, unarmored threespine stickleback, and chinook salmon (all California ESUs) and their designated critical habitat within the aquatic ecosystems considered in this biological opinion include:

1. Water management such as diversions, levee maintenance, channel dredging, channel enlargement, flood control projects, drainage pumps, diversion pumps, siphons, non-Federal pumping plants associated with water management in the Sacramento-San Joaquin Delta, intrusion of brackish water, continuing or future non-Federal diversions of water, flood flow releases, and changes in water management;

2. Introduction of non-native fish, wildlife and plants, hybridization with non-native fishes, inbreeding of small populations, and genetic isolation;

3. Discharges into surface waters including point source discharges (permitted), non-point source runoff (e.g., mining runoff), runoff from high-density confined livestock production facilities, runoff from copper sulfate foot baths associated with dairy farms, agricultural irrigation drainwater discharges (surface and subsurface), runoff from overgrazed rangelands, municipal and industrial stormwater discharges (permitted and non-permitted), release of contaminated ballast and spills of oil and other pollutants into enclosed bays, and illegal, non-permitted discharges;

4. Overfishing and overutilization for scientific, commercial, and educational purposes;

5. Wildland fires and land management practices such as timber harvest practices and improper rangeland management resulting in sedimentation of surface waters; and application of pesticides, herbicides, fungicides,, fumigants, fertilizers and other soil/water amendments, urban development, and conversion and reclamation of wetland habitats;

6. Recreational disturbances including water sports, illegal fishing, and off-road vehicle use.

Cumulative effects for the semi-aquatic, piscivorous, and terrestrial wildlife including, Aleutian Canada goose, bald eagle, California brown pelican, California clapper rail, California least tern, light-footed clapper rail, marbled murrelet, western snowy plover, Yuma clapper rail, southern sea otter, Arroyo toad, California red-legged frog, giant garter snake, San Francisco garter snake, Santa Cruz long-toed salamander, California freshwater shrimp, conservancy fairy shrimp, longhorn fairy shrimp, Riverside fairy shrimp, San Diego fairy shrimp, Shasta crayfish, vernal pool fairy shrimp, and vernal pool tadpole shrimp and their designated critical habitat considered in this biological opinion include:

1. Water management such as diversions, levee maintenance, channel dredging, channel enlargement, flood control projects, installation of pumps, wells, and drains, non-Federal pumping plants associated with water management in the Sacramento-San Joaquin Delta, intrusion of brackish water, continuing or future non-Federal diversions of water, flood flow releases, and changes in water management;

2. Introduction of non-native fish, wildlife and plants, inbreeding of small populations, and genetic isolation;

3. Discharges into surface waters including point source discharges (permitted), non- point source runoff (e.g., mining runoff), runoff from high-density confined livestock production facilities, agricultural irrigation drainwater discharges (surface and subsurface), runoff from overgrazed rangelands, municipal stormwater runoff, and illegal, release of contaminated ballast and spills of oil and other pollutants into enclosed bays, non-permitted discharges;

4. Overutilization for scientific, commercial, and educational purposes;

5. Logging, wildland fire and land management practices including fluctuations in agricultural land crop production, plowing, discing, grubbing, improper rangeland management, timber harvest practices, irrigation canal clearance and maintenance activities, levee maintenance, permitted and non-permitted use and application of pesticides, herbicides, fungicides, rodenticides, fumigants, fertilizers and other soil/water amendments, urban development, urban refuse disposal, land conversions, illegal fill of wetlands and conversion and reclamation of wetland habitats; and

6. Recreational disturbances, vandalism, road kills, off-road vehicle use, chronic disturbance, noise, disturbances from domestic dogs and equestrian uses.

The adoption of the CTR is certain to affect listed species dependent on the aquatic ecosystem. These effects are prolonged and pose significant threats to species already threatened or endangered throughout their range. Continued growth and development in the State of California is likely to exacerbate existing environmental conditions for species already in peril. It is the summation of the direct, indirect, and cumulative effects of the proposed action that the Services conclude are likely to adversely affect these species and their habitats throughout the State.

CONCLUSION

Findings of Not Likely to Jeopardize

After reviewing the current status of the species, the environmental baseline for the action area, the effects of EPA's proposed action and its modifications to the proposed action for selenium, mercury, PCP, cadmium, and formula based dissolved criteria and the cumulative effects, it is the Services' biological opinion that the promulgation of the CTR, as modified by EPA's December 16, 1999 letter, is not likely to jeopardize the continued existence of, or adversely modify critical habitats for species listed in Table 3. The Services reached these conclusions for the following reasons: (1) adverse effects associated with the modified proposed action will be sufficiently minimized by NPDES permit evaluation and early coordination and consultation with the Services on all other CWA programs subject to section 7 consultation; (2) the time frames and procedural commitments proposed by EPA in their December 16, 1999, letter provide assurance that future criteria will be adequately protective of listed species and critical habitat; and (3) that EPA will promulgate such criteria in a manner that will provide protection to listed species and/or critical habitat. The modifications proposed by EPA in their December 16, 1999 letter, and revised by the Services are incorporated in the "Incidental Take Statement" section of this document and presented as non-discretionary terms and conditions.

INCIDENTAL TAKE STATEMENT

The Act prohibits take of endangered and threatened species without a special exemption. "Take" is defined as harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or attempt to engage in any such conduct. "Harm" is further defined by the Services to include significant habitat modification or degradation that actually kills or injures a listed species by significantly impairing essential behavioral patterns, including breeding, spawning, rearing, feeding, migrating or sheltering. "Harass" is defined by the Service as an action that creates the likelihood of injury to a listed species by annoying it to such an extent as to significantly disrupt normal behavioral patterns which include, but are not limited to, breeding, feeding, or sheltering. "Incidental take" is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. Under the terms of section 7(b)(4) and section 7(o)(2), such incidental taking is not considered to be a prohibited taking under the Act provided that such taking is in compliance with this Incidental Take Statement.

The measures described below are non-discretionary and must be implemented by EPA so that they become binding conditions of any grant or permit issued to the applicant, as appropriate, in order for the exemption in section 7(o)(2) to apply. EPA has a continuing duty to regulate the activity that is covered by this incidental take statement. If the Federal agency (1) fails to require the applicant to adhere to the terms and conditions of the incidental take statement through enforceable terms that are added to the permit or grant document, and/or (2) fails to retain oversight to ensure compliance with these terms and conditions, the protective coverage of section 7(o)(2) may lapse. In order to monitor the impact of incidental take, the EPA must report the progress of the action and its impact on the species to the Services as specified in the terms and conditions in this incidental take statement.

The Services have developed the following incidental take statement based on the premise that the reasonable and prudent measures and terms and conditions will be implemented.

Amount or Extent of Take

In the accompanying biological opinion, the Services determined that this level of anticipated take is not likely to result in jeopardy to the species or destruction or adverse modification of critical habitat when the reasonable and prudent measures and terms and conditions are implemented. The Services anticipate that take of listed species in the form of kill and harm is likely to occur as a result of the proposed implementation and compliance schedules for the CTR. Take may occur in the five year timelag that is likely to occur after the State adopts the CTR, and dischargers are granted a five year grace period within which they are to come into compliance with new criteria. Therefore, the Services anticipate the following levels of take may occur as a result of the implementation of and compliance with the CTR, as modified in this opinion and by EPA's December 16, 1999, letter.

The Services are not including an incidental take authorization for marine mammals at this time because the incidental take of marine mammals has not been authorized under section 101(a)(5) of the Marine Mammal Protection Act and/or its 1994 Amendments. Following issuance of such regulations or authorizations, the Services may amend this biological opinion to include an incidental take statement for marine mammals, as appropriate.

The Services anticipate that take for the bald eagle, and California brown pelican, will be difficult to detect since these species (1) often transport prey items to their nests to feed their young; (2) may travel great distances, or are wide-ranging, and are not likely to be recovered following lethal or sublethal exposures; (3) after consuming a lethal or sublethal doses of contaminants may fly some distance from the aquatic ecosystem before being incapacitated and its death may go undetected; (4) sublethal doses of contaminants ingested may significantly impair essential behavioral patterns including feeding, sheltering, breeding, or immune response; and (5) young fed poisoned prey species by the adult or nestling may die at the nest site without being discovered. Therefore, the incidental take of bald eagles, California brown pelicans is expected to be in the form of killing or harming (as previously defined) as a result of lethal or sublethal exposure to environmental contaminants considered herein.

All bald eagles, California brown pelicans, California clapper rails, California least terns, light-footed clapper rails, marbled murrelets, and Yuma clapper rails that forage in the state that are associated with the proposed action are likely to be adversely affected as a result of the proposed action. The Service expects the likelihood of detecting take to be extremely low. Therefore, in order to insure the protection of listed species, reinitiation of formal consultation is required if a total of three (3) dead or sublethally affected bald eagles; or three (3) California clapper rails, or three (3) California least terns, or three (3) light-footed clapper rails, or three (3) marbled murrelets, or three (3) Yuma clapper rails, or if 1,000 or more California brown pelicans are found dead or sublethally affected by contaminants considered in this biological opinion.

The Services anticipate that incidental take of arroyo toad, California red-legged frog and Santa Cruz long-toed salamander will be difficult to detect since these species (1) are most vulnerable to the effects of mercury selenium or metals during their egg and/or larvae stage whose death may go undetected; (2) may experience undetected reduced hatchability, survival, and growth due to exposure to sublethal concentrations of mercury selenium or metals; (3) as juveniles may disperse from natal areas and are not likely to be recovered following lethal or sublethal early life stage exposures; (4) sublethal doses of mercury selenium or metals ingested may adversely affect them by significantly impairing essential behavioral patterns including feeding, sheltering, breeding, or immune response. Therefore, the incidental take of arroyo toad, California red-legged frog, and Santa Cruz long-toed salamander are expected to be in the form of killing or harming (as previously defined) as a result of lethal or sublethal exposure to environmental contaminants.

All arroyo toads, California red-legged frogs, southern California population of the mountain yellow-legged frog, and Santa Cruz long-toed salamanders occurring in California waterbodies are likely to be adversely affected as a result of the proposed action. The Service expects the likelihood of detecting take to be extremely low. In order to insure the protection of listed species, reinitiation of formal consultation is required if more than 10 toads, frogs, or salamanders are found dead or sublethally affected and pollutants considered in this biological opinion are found to be the causative agent.

The Service anticipates that incidental take of San Francisco garter snakes and giant garter snakes will be difficult to detect since the species (1) utilizes water and small mammal burrows for escape cover; (2) after consuming a lethal or sublethal doses of contaminants may travel some distance from the aquatic ecosystem before its death and may go undetected; and (3) sublethal doses of contaminants ingested may adversely affect them by significantly impairing essential behavioral patterns including feeding, sheltering, breeding, or immune response. Therefore, the incidental take of San Francisco garter snakes is expected to be in the form of killing or harming (as previously defined) as a result of lethal or sublethal exposure to environmental contaminants considered herein.

All San Francisco garter snakes and giant garter snakes in the action area are likely to be adversely affected as a result of the proposed action. The Service expects the likelihood of detecting take to be extremely low. Therefore, in order to insure the protection of listed species, reinitiation of formal consultation is required if one (1) dead or sublethally affected San Francisco garter snake or giant garter snake is discovered and contaminants considered in this biological opinion are confirmed to be the causative agent.

The Services anticipate that incidental take of all listed fish and invertebrate species considered in this opinion will be difficult to detect since these species (1) are aquatic in nature, and there is a low likelihood of discovering sublethally or lethally affected individuals; (2) may be directly lost to other environmental and human-caused conditions due to a reduced capacity to escape predation or other human induced habitat conditions; (3) are small bodied and/or affected at an early life stage and are not likely to be detected; and (4) losses may be masked by seasonal or inter-annual fluctuation in numbers or by other causes such as ocean conditions that lie outside the action area.

All aquatic fish and invertebrate species in California waterbodies are likely to be adversely affected as a result of the proposed action. The Services expect the likelihood of detecting take to be extremely low. In order to insure the protection of listed species, reinitiation of formal consultation is required if fish kills of any listed non-salmonid species considered in this biological opinion exceed 1,000 individuals and contaminants considered in this biological opinion are confirmed to be the causative agent. In addition, reinitiation of formal consultation is required if 10 or more dead or sublethally affected anadromous salmonids are discovered and contaminants considered in this biological opinion are confirmed to be the causative agent. This requirement shall apply whenever the combined total of anadromous fish from all ESUs exceeds

10 in any given year.

In the event of exceedance of allowed take the EPA must immediately provide an explanation of the causes of the taking and shall review with the Services the need for possible modification of the reasonable and prudent measures listed below. Take of an individual of any non-fish species is not in violation of the Act as long as the terms and conditions as specified in this biological opinion were adhered to at the time of the incident.

REASONABLE AND PRUDENT MEASURES

The Service believes the following reasonable and prudent measures are necessary and appropriate to minimize impacts of incidental take of the species described below:

1. Minimize the incidental take associated with the proposed numeric criteria for selenium for the following listed species:

BIRDS
Aleutian Canada goose
Bald eagle
California brown pelican
California clapper rail
California least tern
Light-footed clapper rail
Yuma clapper rail

MAMMALS
Southern sea otter

FISH
Bonytail chub
Chinook salmon (California ESUs)
Coho salmon (California ESUs)
Delta smelt
Desert pupfish
Lahontan cutthroat trout
Little Kern Golden Trout
Lost River Sucker
Modoc Sucker
Mohave tui chub
Owens pupfish
Owens tui chub
Paiute cutthroat trout
Razorback sucker
Sacramento splittail
Shortnose sucker
Steelhead trout(California ESUs)
Tidewater goby
Unarmored threespine stickleback

REPTILES AND AMPHIBIANS
Arroyo toad
California red-legged frog
Giant garter snake
San Francisco garter snake
Santa Cruz long-toed salamander

INVERTEBRATES
California freshwater shrimp
Conservancy fairy shrimp
Longhorn fairy shrimp
Riverside fairy shrimp
San Diego fairy shrimp
Shasta crayfish
Vernal pool fairy shrimp
Vernal pool tadpole shrimp

2. Minimize the incidental take associated with the proposed numeric criteria for Mercury for the following listed species:

BIRDS
Aleutian Canada goose
Bald eagle
California brown pelican
California clapper rail
California least tern
Light-footed clapper rail
Yuma clapper rail

MAMMALS
Southern sea otter

FISH
Chinook salmon (California ESUs)
Coho salmon (California ESUs)
Delta smelt
Desert pupfish
Lahontan cutthroat trout
Little Kern Golden Trout
Lost River Sucker
Modoc Sucker
Mohave tui chub
Owens pupfish
Owens tui chub
Paiute cutthroat trout
Sacramento splittail
Shortnose sucker
Steelhead trout(California ESUs)
Tidewater goby
Unarmored threespine stickleback

REPTILES AND AMPHIBIANS
Arroyo toad
California red-legged frog
Giant garter snake
San Francisco garter snake
Santa Cruz long-toed salamander

INVERTEBRATES
California freshwater shrimp
Conservancy fairy shrimp
Longhorn fairy shrimp

Riverside fairy shrimp
San Diego fairy shrimp
Shasta crayfish
Vernal pool fairy shrimp
Vernal pool tadpole shrimp

3. Minimize the incidental take associated with the proposed numeric criteria for PCP on the following listed species:

FISH
Chinook salmon (California ESUs)
Coho salmon (California ESUs)
Delta smelt
Lahontan cutthroat trout
Little Kern golden trout
Lost River sucker
Modoc sucker
Paiute cutthroat trout
Sacramento splittail
Shortnose sucker
Steelhead (California ESUs)

4. Minimize the incidental take associated with the proposed numeric criteria for cadmium on the following listed species:

Chinook salmon (California ESUs)
Coho salmon (California ESUs
Lahontan cutthroat trout
Little Kern golden trout
Paiute cutthroat trout
Steelhead (California ESUs)
Unarmored threespine stickleback

5. Minimize the incidental take associated with the proposed formula based dissolved metals criteria on the following listed species:

FISH
Bonytail chub
Chinook salmon (California ESUs)
Coho salmon (California ESUs)
Delta smelt
Lahontan cutthroat trout
Little Kern golden trout
Lost River sucker

INVERTEBRATES
California freshwater shrimp
Conservancy fairy shrimp
Longhorn fairy shrimp
Riverside fairy shrimp
San Diego fairy shrimp
Shasta crayfish

Modoc sucker
Mohave tui chub
Owens pupfish
Owens tui chub
Paiute cutthroat trout
Razorback sucker
Sacramento splittail
Shortnose sucker
Steelhead (California ESU's)
Tidewater goby
Unarmored threespine stickleback
Vernal pool fairy shrimp
Vernal pool tadpole shrimp

REPTILES AND AMPHIBIANS
Arroyo toad
California red-legged frog
Giant garter snake
San Francisco garter snake
Santa Cruz long-toed salamander

MAMMALS
Southern sea otter

Terms and Conditions

In order to comply with the Act, EPA must comply with the following terms and conditions, which implement the reasonable and prudent measures described above and outline required reporting/monitoring requirements. These terms and conditions are non-discretionary.

1. The following terms and conditions implement reasonable and prudent measure number one for the proposed numeric criteria for selenium.

b) EPA will reserve (not promulgate) the proposed acute aquatic life criterion for selenium in the final CTR.

b) EPA will revise its recommended 304(a) acute and chronic aquatic life criteria for selenium by January 2002. In revising these criteria EPA will work in close cooperation with the Services, inviting scientists from each Service to participate on peer review panels and as observers on criteria revision teams.

c) EPA will propose revised acute and chronic aquatic life criteria for selenium in California by January 2003.

d) If EPA's proposed acute or chronic criterion for selenium in California are less stringent than the criteria suggested in this opinion ($\leq$ 2 µg/L), EPA will provide the Services with a biological evaluation/assessment and request for formal consultation on the revised criterion (or criteria) by January 2003. EPA's biological evaluation/assessment on the revised criterion (or criteria) will specifically address semi-aquatic wildlife species.

e) EPA will promulgate final acute and chronic criteria for selenium in California no later than June 2004.

f) EPA will provide the Services in California with semi-annual reports regarding the status of EPA's revision of the selenium criteria and accompanying draft biological evaluation/assessment associated with the revision. The first report will be provided by June 30, 2000.

g) EPA will identify water bodies in the State of California where selenium criteria necessary to protect federally listed species are not met (selenium-impaired water bodies), and will annually submit to the Services a list of NPDES permits due for review to allow the Services and EPA to identify any potential for adverse effects on listed species and/or their habitats. A list of selenium-impaired water bodies and the first NPDES permit review shall occur prior to October 2000. EPA will annually submit to the Services a list of NPDES permits due for review to allow the Services and EPA to identify any potential for adverse effects on listed species and/or their habitats. The first NPDES permit review shall occur prior to October 2000.

h) EPA will coordinate with the Services on any permits containing limits for selenium that the Services (or EPA) identify as having potential for adverse effects on listed species and/or their habitat in accordance with procedures agreed to by the Agencies in the draft MOA published in the Federal Register at 64 FR 2755 (January 15, 1999). If discharges are identified that have the potential to adversely affect federally listed species and/or critical habitat, EPA will work with the Services and the State of California to address the potential effects to the species. This will include, where appropriate, decreasing the allowable discharge of selenium consistent with this opinion. Among other options to resolve the issue, the EPA may make a formal objection to a permit and federalize the permit where consistent with EPA's CWA authority. If EPA objects to a NPDES permit, EPA will follow the permit objection procedures outlined in 40 CFR 123.44 and coordinate with the Services. If EPA assumes permit issuing authority for a NPDES permit, EPA will consult with the Services prior to issuance of the permit (as a federal action) as appropriate under section 7 of the ESA. Under such circumstances EPA would prepare and submit a biological evaluation/assessment on those permits for purposes of completing consultation.

2. The following terms and conditions implement reasonable and prudent measure number two for the proposed numeric criteria for mercury.

a) EPA will reserve (not promulgate) the proposed freshwater and saltwater acute and chronic aquatic life criteria for mercury in the final CTR.

b) EPA will promulgate a human health criterion of 50 ng/l or 51 ng/l as designated within the final CTR for mercury <u>only</u> where no more restrictive federally-approved water quality criteria are now in place (e.g., the promulgation will not affect portions of San Francisco Bay).

c) EPA will revise its recommended 304(a) human health criteria for mercury by January

2002. These criteria should be sufficient to protect federally listed aquatic and aquatic-dependent wildlife species. If the revised criteria are less stringent than the range of criteria concentrations suggested by the Services to protect listed species in this opinion or the EPA's mercury report to Congress piscivorous wildlife values, EPA will provide the Services with a biological evaluation/assessment and request for formal consultation on the revised criteria by the time of the proposal. The Services believe protective concentrations for mercury in water are generally on the order of ≤ 2.0 ng/L as total Hg or equivalent methylmercury concentration as determined by site specific data.

d) EPA will propose revised human health criteria for mercury in California by January 2003.

e) EPA will work in close cooperation with the Services to evaluate the degree of protection afforded to federally listed species by the revised criterion. EPA will provide the Services in California with semi annual reports regarding the status of EPA's revision of the mercury criterion and/or any draft biological evaluation/assessment associated with the revision. The first report will be provided by June 30, 2000. EPA will invite scientists representing the Services to participate in efforts to jointly evaluate mercury concentrations protective of fish and wildlife.

f) EPA will identify water bodies in the State of California where mercury criteria necessary to protect federally listed species are not met (mercury-impaired water bodies), and will annually submit to the Services a list of NPDES permits due for review to allow the Services and EPA to identify any potential for adverse effects on listed species and/or their habitats. EPA will annually submit to the Services a list of NPDES permits due for review to allow the Services and EPA to identify any potential for adverse effects on listed species and/or their habitats from mercury. A list of mercury-impaired water bodies and the first NPDES permit review shall occur prior to October 2000.

g) EPA will coordinate with the Services on any permits containing limits for mercury that the Services (or EPA) identify as having potential for adverse effects on listed species and/or their habitat in accordance with procedures agreed to by the Agencies in the draft MOA published in the Federal Register at 64 FR 2755 (January 15, 1999). If discharges are identified that have the potential to adversely affect federally listed species and/or critical habitat, EPA will work with the Services and the State of California to address the potential effects to the species. This will include, where appropriate, decreasing the allowable discharge of mercury consistent with this opinion. Among other options to resolve the issue, the EPA may make a formal objection to a permit and federalize the permit where consistent with EPA's CWA authority. If EPA objects to a NPDES permit, EPA will follow the permit objection procedures outlined in 40 CFR 123.44 and coordinate with the Services. If EPA assumes permit issuing authority for a NPDES permit, EPA will consult with the Services prior to issuance of the permit (as a federal action) as appropriate under section 7 of the ESA. Under such circumstances EPA would prepare and submit a biological evaluation/assessment on those permits for purposes of completing consultation.

3. The following terms and conditions implement reasonable and prudent measure number three for the proposed numeric criteria for PCP.

a) By March of 2001, EPA will review, and if necessary, revise its recommended 304(a) chronic aquatic life criterion for PCP sufficient to protect federally listed species and/or their critical habitats. In reviewing this criterion, EPA will generate new information on PCP regarding the toxicity of commercial grade PCP and the interaction of temperature and dissolved oxygen on sublethal acute and chronic toxicity to early life stage salmonids. These tests will include at least one anadromous species and produce data on chronic toxicity of PCP to listed species.

b) If as a result of these new studies EPA, revises its recommended 304(a) criterion, EPA will then propose the revised PCP criterion in California by March 2002. If the revised criterion is less stringent than the range of criterion concentrations suggested by the Services to protect listed species in this opinion (0.2 to 2.0 µg/L at pH of 7.8) or if EPA determines that a criterion revision is not necessary, EPA will provide the Services with a biological evaluation/assessment and request for formal consultation by March 2002.

c) If EPA proposes a revised PCP criterion by March 2002, EPA will promulgate a final criterion as soon as possible, but no later than 18 months, after proposal.

d) EPA will keep the Services informed regarding the status of EPA's review of the PCP chronic aquatic life criterion and any draft biological evaluation/assessment associated with the review with semi-annual reports.

e) EPA will continue to use existing NPDES permit information to identify water bodies which contain permitted PCP discharges and Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) and Resource Conservation and Reclamation Act (RCRA) sites that potentially contribute PCP to surface waters. EPA, in cooperation with the Services, will review these discharges and associated monitoring data and permit limits, to determine the potential for the discharge to impact federally listed species and/or critical habitats. The first review of PCP information by EPA shall occur prior to October 2000.

f) If discharges are identified that have the potential to adversely affect federally listed species and/or critical habitat, EPA will work with the Services and the State of California to address the potential effects to these species. This will include, where appropriate, decreasing the allowable discharge of PCP to protective concentrations consistent with this opinion. Among other options to resolve the issue, the EPA may make a formal objection to a permit and federalize the permit where consistent with EPA's CWA authority. If EPA objects to a NPDES permit, EPA will follow the permit objection procedures outlined in 40 CFR 123.44 and coordinate with the Services. If EPA assumes permit issuing authority for a NPDES permit, EPA will consult with the Services prior to issuance of the permit (as a federal action) as appropriate under section 7 of the ESA. Under such circumstances EPA would prepare and submit a biological evaluation/assessment on those permits for purposes of completing consultation. EPA will give priority to review data for fresh water bodies within the range of federally listed salmonids that currently lack a MUN designation as specified in the Regional Water Quality Control Boards' Basin Plans.

4. The following terms and conditions implement reasonable and prudent measure number four for the proposed numeric criteria for Cadmium.

a) EPA will revise the 304(a) chronic aquatic life criterion for cadmium such that it will be protective of sticklebacks and salmonids, by no later than January 2001 and will propose the revised criterion in California by January 2002. EPA will not wait for new criteria models to be developed in revising the criterion, but may use these models if they are available by this date. EPA will promulgate final criteria as soon as possible, but no later than 18 months, after proposal.

b) If the revised criterion is less stringent than the range of protective criteria concentrations proposed by the Services in this opinion (0.096 µg/L to 0.180 µg/L), EPA will provide the Services with a biological evaluation/assessment and request for formal consultation on the revised criterion by the time of the proposal.

c) EPA will provide the Services with semi-annual updates regarding the status of EPA's revision of the chronic aquatic life criterion revision for cadmium and any draft biological evaluation/assessment associated with the revision.

d) EPA will continue to consult, under section 7 of ESA, with the Services on revisions to water quality standards contained in Basin Plans submitted to EPA under CWA section 303 and affecting waters of California containing federally listed species and/or their habitats.

e) EPA will annually submit to the Services a list of NPDES permits due for review and RCRA or CRCLA sites where cadmium is a pollutant of concern. EPA, in cooperation with the Services, will review these discharges and associated monitoring data and permit limits to identify any potential for adverse effects on listed species and/or their habitats. EPA will coordinate with the Services on any permits that the Services or EPA identify as having potential for adverse effects on listed species and/or their habitat. By December 2000 EPA will identify all cadmium discharges from point sources and cadmium contaminated RCRA or CERCLA sites in California that may affect listed species and will provide a report to the Services by December 31, 2000.

f) If discharges are identified that have the potential to adversely affect federally listed species and/or critical habitat, EPA will work with the Services and the State of California to address the potential effects to the species. This will include, where appropriate, reducing the permissible concentrations of cadmium consistent with this opinion. Among other options to resolve the issue, the EPA may make a formal objection to a permit and federalize the permit where consistent with EPA's CWA authority. If EPA objects to a NPDES permit, EPA will follow the permit objection procedures outlined in 40 CFR 123.44 and coordinate with the Services. If EPA assumes permit issuing authority for a NPDES permit, EPA will consult with the Services prior to issuance of the permit (as a federal action) as appropriate under section 7 of the ESA. Under such circumstances EPA would prepare and submit a biological evaluation/assessment on those permits for purposes of completing consultation.

5. The following terms and conditions implement reasonable and prudent measure number five for the proposed formula based dissolved metals criteria.

a) By December of 2000, EPA, in cooperation with the Services, will develop sediment criteria guidelines for cadmium, copper, lead, nickel and zinc, and by December of 2002, for chromium and silver. When the sediment guidance for cadmium, copper, lead, nickel and zinc is completed, Region 9, in cooperation with the Services, will draft implementation guidelines for the State of California to protect federally listed threatened and endangered species and critical habitat in California. EPA will submit semi-annual reports to the Services in California on the status of sediment guideline development. The first report will be due June 30, 2000.

b) Before the end of 2000, EPA, in cooperation with the Services, will issue two clarifications to the *Interim Guidance on the Determination and Use of Water-Effects Ratios for Metals* (EPA 1994) concerning the use of calcium-to-magnesium ratios in laboratory water and the proper acclimation of test organisms prior to testing in applying water-effects ratios (WERs). The EPA shall also allow the use of WERs only when the site specific LC_{50} and the laboratory LC_{50} are significantly different using a 95% confidence interval.

c) By June of 2003, EPA, in cooperation with the Services, will develop a revised criteria calculation model based on best available science for deriving aquatic life criteria on the basis of hardness (calcium and magnesium), pH, alkalinity, and dissolved organic carbon (DOC) for metals. This will be done in conjunction with “Other Actions.” below. EPA will submit semi-annual reports to the Services on the status of the development of the revised criteria calculations model for metals. The first report will be provided by June 30, 2000.

d) In certain instances, the State of California or specific dischargers may develop site-specific translators, using EPA or equivalent state/tribe guidance, to translate dissolved metals criteria into total recoverable permit limits. A translator is the ratio of dissolved metal to total recoverable metal in the receiving water downstream from a discharge. A site-specific translator is determined on site-specific effluent and ambient data. Whenever a threatened or endangered species or critical habitat is present within the geographic range downstream from a discharge where a State developed translator will be used and the conditions listed below exist, EPA will work, in cooperation with the Services and the State of California, to use available ecological safeguards to ensure protection of federally listed species and/or critical habitat. Ecological safeguards include: (1) sediment guidelines; (2) biocriteria; (3) bioassessment; (4) effluent and ambient toxicity testing; or (5) residue-based criteria in shellfish.

(i) Conditions for use of ecosystem safeguards:

1. A water body is listed as impaired on the CWA section 303(d) list due to elevated metal concentrations in sediment, fish, shellfish or wildlife; or,

2. A water body receives mine drainage; or,

3. Where particulate metals compose a 50% or greater component of the total metal measured in a downstream water body in which a permitted discharge (subject to translator method selection) is proposed and the dissolved fraction is equal to or within 75% of the water quality criteria.

(ii) Whenever a threatened or endangered species is present downstream from a discharge where a State developed translator will be used, EPA will work with the permitting authority to ensure that appropriate information, which may be needed to calculate the translator in accordance with the applicable guidance, will be obtained and used. Appropriate information includes:

1. Ambient and effluent acute and chronic toxicity data;
2. Bioassessment data; and/or
3. An analysis of the potential effects of the metals using sediment guidelines, biocriteria and residue-based criteria for shellfish to the extent such guidelines and criteria exist and are applicable to the receiving water body.

(iii) EPA, in cooperation with the Services, will review these discharges and associated monitoring data and permit limits, to determine the potential for the discharge to impact federally listed species and/or critical habitats. If discharges of metals are identified that have the potential to adversely affect federally listed species and/or critical habitat, EPA will work with the Services and the State of California to address these adverse impacts in accordance with procedures agreed to by the Agencies in the draft MOA published in the Federal Register at 64 FED REG. 2755 (January 15, 1999). Among other options to resolve the issue, the EPA may make a formal objection to a permit, and federalize the permit where consistent with EPA's CWA authority. If EPA objects to a NPDES permit, EPA will follow the permit objection procedures outlined in 40 CFR 123.44 and coordinate with the Services. If EPA assumes permit issuing authority for a NPDES permit, EPA will consult with the Services prior to issuance of the permit (as a federal action) as appropriate under section 7 of the ESA.

Other Actions

EPA will initiate a process to develop a national methodology to derive site-specific criteria to protect federally listed threatened and endangered species, including wildlife, in accordance with the draft MOA between EPA and the Services concerning section 7 consultations. EPA will invite input and participation from the Services in developing this methodology and will share reports and written products as this methodology progresses. Annual reports on the status of this methodology development will be provided to both the Divisions of Environmental Contaminants and Endangered Species of the Fish and Wildlife Service's Arlington Office, and to the Silver Springs Office of Protected Resources of the National Marine Fisheries Service.

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, help implement recovery plans, or to develop information.

The Services recommend the following additional actions to promote the recovery of federally listed species and their habitats:

1. The EPA should quantify the toxic effects of selenium and mercury individually and in combination to listed reptiles and amphibians using appropriate surrogate species. Research should include the most toxic forms of selenium and mercury and include full life cycle exposure protocols including dietary routes of exposure and maternal transfer as a route of embryonic exposure.

2. The EPA should conduct research on mercury residues in amphibian tissues which would allow prediction of adverse effects from mercury residues found in field collected frogs.

3. The EPA should consider developing a tissue based criteria for mercury and selenium protective of reproduction of aquatic dependent species of fish and wildlife in California.

4. The EPA should, in cooperation with the Service and USGS, conduct research on the toxic effects of selenium and mercury, individually and in combination, to the reproduction of

fish-eating birds using appropriate surrogate species. Research should include the most toxic forms of selenium and mercury and include sensitive life stages and exposure protocols that include dietary routes of exposure to females and maternal transfer as a route of embryonic exposure.

5. The EPA should use existing authorities to develop or require testing to develop site-specific bioaccumulation factors for mercury to assess risk of mercury exposure to bald eagles throughout California.

6. The EPA in conjunction with the San Francisco Bay Regional Water Quality Control Board and Central Valley Regional Water Quality Control Board should assess the influx, fate, and transport of mercury into the San Francisco Bay Estuary to facilitate the development of mercury control strategies.

7. The EPA should conduct toxicity tests in waters where particulate concentrations are great and dissolved metal concentrations are low. These studies should ideally include a dietary exposure component (*in situ* studies) to determine the effects of these discharges on the growth, survival, and reproduction on listed fishes and crustaceans.

In order for the Services to be kept informed of actions that either minimize or avoid adverse effects or that benefit listed species or their habitats, we request notification of the implementation of any conservation recommendations.

REINITIATION NOTICE

This concludes formal consultation and conference on the proposed CTR as outlined in your August 5, 1997, Federal Register notice and your October 27, 1997, request for initiation of formal consultation. 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 proposed 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 listed species or critical habitat that was not considered in this opinion; or (4) a new species is listed or critical habitat is designated that may be affected by the proposed action. In instances where the amount or extent of incidental take is exceeded, any operations causing such take must cease pending reinitiation.

The incidental take statement provided with this conference opinion does not become effective for the Northern California steelhead ESU, the Southern California population of the mountain yellow-legged frog, Santa Ana sucker, or the Southern California population of the California tiger salamander, until the species are listed and the conference opinion is adopted as the biological opinion. No take of the Northern California steelhead ESU, Southern California population of the mountain yellow-legged frog, Santa Ana sucker, or the Southern California population of the California tiger salamander is allowed between the time they are listed and the adoption of the conference opinion as a biological opinion is authorized. You may request the Services to immediately adopt this conference opinion as a biological opinion if these species are listed. The request must be in writing. Provided none of the reinitiation criteria apply, the Services will agree with EPA's request.

If you have any questions regarding this response please feel free to contact Mr. Wayne White at the Service's Sacramento Fish and Wildlife Office at (916) 979-2710, or Mr. Jim Lecky at the National Marine Fisheries Service Southwest Regional Office at (562) 980-4015.

Sincerely,

Michael J. Spear
Manager, California/Nevada Operations Office
U.S. Fish and Wildlife Service

Rodney R. Mc Innis
Acting Regional Administrator
Southwest Regional Office
National Marine Fisheries Service

Table 1: Species considered in the consultation for the California Toxics Rule		
COMMON NAME	**SCIENTIFIC NAME**	**STATUS**
BIRDS		
Aleutian Canada goose	Branta canadensis leucopareia	T
Bald eagle	Haliaeetus leucocephalus	PD
California brown pelican	Pelecanus occidentalis californicus	E
California clapper rail	Rallus longirostris obsoletus	E
California condor	Gymnogyps californianus	E
California least tern	Sterna antillarum (=albifrons) browni	E
Coastal California gnatcatcher	Polioptila californica californica	T
Inyo brown towhee	Pipilo fuscus eremophilus	T
Least Bell's vireo	Vireo bellii pusillus	E
Light-footed clapper rail	Rallus longirostris levipe	E
Marbled murrelet	Brachyramphus marmoratus	T
Northern spotted owl	Strix occidentalis caurina	T
San Clemente loggerhead shrike	Lanius ludovicianus mearnsi	E
San Clemente sage sparrow	Amphispiza belli clementeae	T
Short-tailed albatross	Diomedea albatrus	E
Southwestern willow flycatcher	Empidonax traillii extimus	E
Western snowy plover (coastal population)	Charadrius alexandrinus nivosus	T
Yuma clapper rail	Rallus longirostris yumanesis	E
FISH		
Bonytail chub	Gila elegans	E
Chinook salmon, Central Valley Spring ESU	Oncorhynchus tshawytscha	T
Chinook salmon, CA Coast ESU	Oncorhynchus tshawytscha	T
Chinook salmon, Winter Run	Oncorhynchus tshawytscha	E
Coho salmon, Central California ESU	Oncorhynchus kisutch	T
Coho salmon, So. Oregon/No. California ESU	Oncorhynchus kisutch	T
Delta smelt	Hypomesus transpacificus	T
Desert pupfish	Cyprinodon macularius	E
Lahontan cutthroat trout	Oncorhynchus clarki henshawi	T

Table 1: Species considered in the consultation for the California Toxics Rule		
Little Kern golden trout	Oncorhynchus aquabonita whitei	T
Lost River sucker	Deltistes luxatus	E
Modoc sucker	Catostomus microps	E
Mohave tui chub	Gila bicolor mohavensis	E
Owens pupfish	Cyprinodon radiosus	E
Owens tui chub	Gila bicolor snyderi	E
Paiute cutthroat trout	Oncorhynchus clarki seleniris	T
Razorback sucker	Xyrauchen texanus	E
Sacramento splittail	Pogonichthys macrolepidotus	T
Santa Ana sucker	Catostomus santanaanae	PT
COMMON NAME	**SCIENTIFIC NAME**	**STATUS**
Shortnose sucker	Chasmistes brevirostris	E
Steelhead, Northern California ESU	Onchorhynchus mykiss	PT
Steelhead, Central California Coast ESU	Oncorhynchus mykiss	T
Steelhead, Central Valley ESU	Oncorhynchus mykiss	T
Steelhead, South Central California Coast ESU	Oncorhynchus mykiss	T
Steelhead, Southern California ESU	Oncorhynchus mykiss	E
Tidewater goby	Eucyclogobius newberryi	E
Unarmored threespine stickleback	Gasterosteus aculeatus williamsoni	E
Warner sucker	Catostomus warnerensis	T
Winter-run chinook salmon	Oncorhynchus tshawytscha	E
REPTILES AND AMPHIBIANS		
Alameda whipsnake	Masticophis lateralis euryxanthus	T
Arroyo southwestern toad	Bufo microscaphus californicus	E
Blunt-nosed leopard lizard	Gambelia (=Crotaphytus) silus	E
California red-legged frog	Rana aurora draytonii	T
California tiger salamander, Santa Barbarbra	Ambystoma californniense	PE
Coachella Valley fringe-toed lizard	Uma inornata	T
Desert slender salamander	Batrachoseps aridus	E
Desert tortoise	Gopherus (=Xerobates) agassizii	E
Flat-tailed horned lizard	Phrynosoma meallii	PT

Table 1: Species considered in the consultation for the California Toxics Rule		
Giant garter snake	Thamnophis gigas	T
Green turtle	Chelonia mydas (incl. agassizi)	T
Island night lizard	Xantusia (=Klauberina) riversiana	T
Leatherback turtle	Dermochelys coriacea	E
Loggerhead turtle	Caretta caretta	T
Mountain yellow-legged frog, Southern California	Rana muscosa	PE
Olive (=Pacific) Ridley turtle	Lepidochelys olivacea	T
San Francisco garter snake	Thamnophis sirtalis tetrataenia	E
Santa Cruz long-toed salamander	Ambystoma macrodactylum croceum	E
INVERTEBRATES		
Bay checkerspot butterfly	Euphydryas editha bayensis	T
Behren's silverspot butterfly	Speyeria zerene behrensii	E
California freshwater shrimp	Syncaris pacifica	E
Callippe silverspot butterfly	Speyeria callippe callippe	E
Conservancy fairy shrimp	Branchinecta conservatio	E
Delhi Sands flower-loving fly	Rhaphiomidas terminatus abdominalis	E
Delta green ground beetle	Elaphrus viridis	T
El Segundo blue butterfly	Euphilotes (=Shijimiaeoides) battoides allyn	E
Kern primrose sphinx moth	Euproserpinus euterpe	T
Laguna Mountains skipper	Pyrgus ruralis lagunae	E
Lange's metalmark butterfly	Apodemia mormo langei	E
Longhorn fairy shrimp	Branchinecta longiantenna	E
Lotis blue butterfly	Lycaeides argyrognomon lotis	E
COMMON NAME	**SCIENTIFIC NAME**	**STATUS**
INVERTEBRATES (CONTINUED)		
Mission blue butterfly	Icaricia icarioides messionensis	E
Morro shoulderband snail	Helminthoglypta walkeriana	E
Mount Hermon June beetle	Polyphylla barbata	E
Myrtles's silverspot butterfly	Speyeria zerene myrtleae	E
Oregon silverspot butterfly	Speyeria zerene hippolyta	T

Table 1: Species considered in the consultation for the California Toxics Rule		
Palos Verdes blue butterfly	Glaucopsyche lygadamus palosverdesensis	E
Quino checkerspot butterfly	Euphydryas editha quino	E
Riverside fairy shrimp	Streptocepahlus woottoni	E
San Bruno elfin butterfly	Incisalia mossii bayensis	E
San Diego fairy shrimp	Branchinecta sandiegoensis	E
Santa Cruz rain beetle	Pleocoma conjungens conjungens	PE
Shasta crayfish	Pacifastacus fortis	E
Smith's blue butterfly	Euphilotes (=Shijimiaeoides) enoptes smithi	E
Valley elderberry longhorn beetle	Desmocerus californicus dimorphus	T
Vernal pool fairy shrimp	Branchinecta lynchi	T
Vernal pool tadpole shrimp	Lepidurus packardi	E
Zayante band-winged grasshopper	Trimerotropis infantilis	E
MAMMALS		
Amargosa vole	Microtus californicus scirpensis	E
Fresno kangaroo rat	Dipodomys nitratoides exilis	E
Giant kangaroo rat	Dipodomys ingens	E
Jaguar (U.S. population)	Panthera onca	PE
Morro Bay kangaroo rat	Dipodomys heermanni morroensis	E
Pacific little pocket mouse	Perognathus longimembris pacificus	E
Peninsular bighorn sheep	Ovis canadensis cremnobates	E
Point Arena mountain beaver	Aplodontia rufa nigra	E
Salt marsh harvest mouse	Reithrodontomys raviventris	E
San Joaquin kit fox	Vulpes macrotis mutica	E
Southern sea otter	Enhydra lutris nereis	T
Stellar (=Northern) sea lion	Eumetopias jubatus	T
Stephen's kangaroo rat	Dipodomys stephensi	E
Tipton kangaroo rat	Dipodomys nitratoides nitratoides	E

T = threatened, E = endangered, PT = proposed threatened, PE = proposed endangered, PD = proposed for delisting

Table 2: Species Not Likely to be Adversely Affected by the Proposed Action		
COMMON NAME	**SCIENTIFIC NAME**	**STATUS**

BIRDS		
Least Bell's vireo	Vireo bellii pusillus	E
Southwestern willow flycatcher	Empidonax traillii extimus	E
FISH		
Warner sucker	Catostomus warnerensis	T
MAMMALS		
Salt marsh harvest mouse	Reithrodontomys raviventris	E
San Joaquin kit fox	Vulpes macrotis mutica	E

T = threatened
E = endangered
PT = proposed threatened
PE = proposed endangered
PD = proposed for delisting

Table 3: Species Likely to be Adversely Affected by the Proposed Action			
			CRITICAL
COMMON NAME	SCIENTIFIC NAME	STATUS	HABITAT
BIRDS			
Aleutian Canada goose	Branta canadensis leucopareia	T	
Bald eagle	Haliaeetus leucocephalus	PD*	
California brown pelican	Pelecanus occidentalis californicus	E	
California clapper rail	Rallus longirostris obsoletus	E	
California least tern	Sterna antillarum (=albifrons) browni	E	
Light-footed clapper rail	Rallus longirostris levipe	E	
Marbled murrelet	Brachyramphus mamoratus mamoratus	T	Y
Western snowy plover (coastal population)	Charadrius alexandrinus nivosus	T	Y
Yuma clapper rail	Rallus longirostris yumanesis	E	
FISH			
Bonytail chub	Gila elegans	E	Y
Chinook salmon, Central Valley Spring ESU	Oncorhynchus tshawytscha	T	Y
Chinook salmon, CA Coast ESU	Oncorhynchus tshawytscha	T	Y
Chinook salmon, Winter Run	Oncorhynchus tshawytscha	E	Y
Coho salmon, Central California	Oncorhynchus kisutch	T	Y
Coho salmon, So. Oregon/No. California	Oncorhynchus kisutch	T	Y
Delta smelt	Hypomesus transpacificus	T	Y
Desert pupfish	Cyprinodon macularius	E	Y
Lahontan cutthroat trout	Oncorhynchus clarki henshawi	T	
Little Kern golden trout	Oncorhynchus aquabonita whitei	T	Y
Lost River sucker	Deltistes luxatus	E	P
Modoc sucker	Catostomus microps	E	Y
Mohave tui chub	Gila bicolor mohavensis	E	
Owens pupfish	Cyprinodon radiosus	E	
Owens tui chub	Gila bicolor snyderi	E	Y
Paiute cutthroat trout	Oncorhynchus clarki seleniris	T	

Razorback sucker	Xyrauchen texanus	E	Y
Sacramento splittail	Pogonichthys macrolepidotus	T	
Santa Ana sucker	Catostomus santanaanae	PT	
Shortnose sucker	Chasmistes brevirostris	E	P
Steelhead, Northern California ESU	Oncorhynchus mykiss	PT	
Steelhead, Central California Coast ESU	Oncorhynchus mykiss	T	Y
Steelhead, Central Valley ESU	Oncorhynchus mykiss	T	Y
Steelhead, South Central California Coast ESU	Oncorhynchus mykiss	T	Y
Steelhead, Southern California ESU	Oncorhynchus mykiss	E	Y
Tidewater goby	Eucyclogobius newberryi	E	P
Unarmored threespine stickleback	Gasterosteus aculeatus williamsoni	E	P

Table 3: Species Likely to be Adversely Affected by the Proposed Action (continued)			
			CRITICAL
COMMON NAME	**SCIENTIFIC NAME**	**STATUS**	**HABITAT**
REPTILES AND AMPHIBIANS			
Arroyo southwestern toad	Bufo microscaphus californicus	E	
California red-legged frog	Rana aurora draytonii	T	
California tiger salamander, Santa Barbara Co.	Ambystoma californiense	PE	
Giant garter snake	Thamnophis gigas	T	
Mountain yellow-legged frog, Southern CA DPS	Rana muscosa	PE	
San Francisco garter snake	Thamnophis sirtalis tetrataenia	E	
Santa Cruz long-toed salamander	Ambystoma macrodactylum croceum	E	
INVERTEBRATES			
California freshwater shrimp	Syncaris pacifica	E	
Conservancy fairy shrimp	Branchinecta conservatio	E	
Longhorn fairy shrimp	Branchinecta longiantenna	E	
Riverside fairy shrimp	Streptocephalus woottoni	E	
San Diego fairy shrimp	Branchinecta sandiegoensis	E	
Shasta crayfish	Pacifastacus fortis	E	
Vernal pool fairy shrimp	Branchinecta lynchi	T	
Vernal pool tadpole shrimp	Lepidurus packardi	E	
MAMMALS			
Southern sea otter	Enhydra lutris nereis	T	

T = threatened
E = endangered
PT = proposed threatened
PE = proposed endangered
PD = proposed for delisting

Table 4. Species/Critical Habitats the Services' Concluded were Likely to be Jeopardized/Adversely Modified by the August 5, 1997, Version of the California Toxics Rule (from our April 10, 1998 draft biological opinion).

COMMON NAME	SCIENTIFIC NAME	STATUS	ISSUES	CRITICAL HABITAT (Adverse Modification)
BIRDS				
California clapper rail	Rallus longirostris obsoletus	E	Se, Hg	
California least tern	Sterna antillarum (=albifrons) browni	E	Se, Hg	
Light-footed clapper rail	Rallus longirostris levipe	E	Se, Hg	
Marbled murrelet	Brachyramphus marmoratus	T	Se, Hg	Y (N)
Yuma clapper rail	Rallus longirostris yumanesis	E	Se, Hg	
FISH				
Bonytail chub	Gila elegans	E	Se, Hg, metals	Y (Y)
Chinook, Coastal California/ Southern Oregon	Oncorhynchus tshawytscha	PT	Se, Hg, PCP, metals	P (Y)
Chinook, Central Valley Spring Run	Oncorhynchus tshawytscha	PE	Se, Hg, PCP, metals	P (Y)
Chinook, Central Valley Fall/Late Fall	Oncorhynchus tshawytscha	PT	Se, Hg, PCP, metals	P (Y)
Coho salmon, Central California	Oncorhynchus kisutch	T	Se, Hg, PCP, metals	P (Y)
Coho salmon, So. Oregon/No. California	Oncorhynchus kisutch	T	Se, Hg, PCP, metals	P (Y)
Delta smelt	Hypomesus transpacificus	T	Se, Hg, metals	Y (Y)
Desert pupfish	Cyprinodon macularius	E	Se, Hg	Y (Y)
Lahontan cutthroat trout	Oncorhynchus clarki henshawi	T	Cd, Metals	N
Razorback sucker	Xyrauchen texanus	E	Se, Hg, metals	Y (Y)
Sacramento splittail	Pogonichthys macrolepidotus	T	Se, Hg, metals	
Steelhead, Central California ESU	Oncorhynchus mykiss	T	Se, Hg, PCP, metals	P(Y)
Steelhead, Central Valley ESU	Oncorhynchus mykiss	T	Se, Hg, PCP, metals	P(Y)
Steelhead, South Central California ESU	Oncorhynchus mykiss	T	Se, Hg, PCP, metals	P(Y)
Steelhead, Southern California ESU	Oncorhynchus mykiss	E	Se, Hg, PCP, metals	P(Y)
Unarmored threespine stickleback	Gasterosteus aculeatus williamsoni	E	Se, Hg, metals	P (Y)
Winter-run chinook salmon	Oncorhynchus tshawytscha	E	Se, Hg, PCP, metals	Y (Y)
REPTILES AND AMPHIBIANS				
California red-legged frog	Rana aurora draytonni			
Giant garter snake	Thamnophis gigas	T	Se, Hg	
MAMMALS				
Southern sea otter	Enhydra lutris nereis	T	Hg, metals	

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U.S. Fish & Wildlife Service

ECOS

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western ridged mussel (*Gonidea angulata*)

Range Information | Candidate Info | Federal Register | Recovery | Critical Habitat | SSA | Conservation Plans | Petitions | Biological Opinions | Life History

Search for images on digitalmedia.fws.gov

Taxonomy: View taxonomy in ITIS

Listing Status: Under Review

General Information

The species historical range included California, Idaho, Nevada, Oregon, Washington. See below for information about where the species is known or believed to occur.

Current Listing Status Summary

Show 10 entries

Status	Date Listed	Lead Region
Under Review	07-27-2021	Pacific Region (Region 1)

Showing 1 to 1 of 1 entries

‹ Previous 1 Next ›

» Range Information

Current Range

Zoom in! Some species' locations may be small and hard to see from a wide perspective. To narrow-in on locations, check the state and county lists (below) and then use the zoom tool.

Want the FWS's current range for all species? Click here to download a zip file containing all individual shapefiles and metadata for all species.

* For consultation needs do not use only this current range map, please use IPaC.

Current range maps are only shown within the jurisdictional boundaries of the United States of America. The species may also occur outside this region.

There is currently no Current Range data for this species.

- **Listing status: Under Review**
  - **States/US Territories** in which this population is known to or is believed to occur:
  - **US Counties** in which this population is known to or is believed to occur: View All
  - **USFWS Refuges** in which this population is known to occur:
  - **Countries** in which this population is known to occur: Canada, United States

» Candidate Information

No Candidate information available for this species.

No Candidate Assessments available for this species.
No Candidate Notice of Review Documents currently available for this species.
No Uplisting Documents currently available for this species.

» Federal Register Documents

Federal Register Documents

Show 10 entries

Date	Citation Page	Title
07/27/2021	86 FR 40186 40189	Endangered and Threatened Wildlife and Plants; 90-D Findings for Three Species

Showing 1 to 1 of 1 entries

‹ Previous 1 Next ›

» Species Status Assessments (SSAs)

Species Status Assessments (SSAs)

No Species Status Assessments (SSA's) are currently available for this species.

Special Rule Publications

No Special Rule Publications currently available for this species.

» Recovery

- Species with Recovery Documents Data Explorer

No Current Recovery Plans available for this species.

No Other Recovery Documents currently available for this species.

No Five Year Reviews currently available for this species.

No Delisting Documents currently available for this species.

» Critical Habitat

No Critical Habitat Documents currently available for this species.

» Conservation Plans

No Conservation Plans currently available for this species.

» Petitions

Show 10 entries

Petition Title	Date Received by the FWS	Where the species is believed to or known to occur	Petitioner Name	Requested Action	Petitio Finding

Petition to List the Western Ridged Mussel Gonidea angulata (Lea, 1838) as an Endangered Species Under the U.S. Endangered Species Act	08/21/2020	Canada, United States	• The Xerces Society for Invertebrate Conservation	• Listing: Endangered • APA: Designate Critical Habitat	• 90 pe fin Su on 07

Showing 1 to 1 of 1 entries

‹ Previous 1 Next ›

» Biological Opinions

To see all FWS Issued Biological Opinions please visit the BO Report.

» Life History

No Life History information has been entered into this system for this species.

» Other Resources

NatureServe Explorer Species Reports-- NatureServe Explorer is a source for authoritative conservation information on more than 50,000 plants, animals and ecological communtities of the U.S and Canada. NatureServe Explorer provides in-depth information on rare and endangered species, but includes common plants and animals too. NatureServe Explorer is a product of NatureServe in collaboration with the Natural Heritage Network.

ITIS Reports-- ITIS (the Integrated Taxonomic Information System) is a source for authoritative taxonomic information on plants, animals, fungi, and microbes of North America and the world.

FWS Digital Media Library -- The U.S. Fish and Wildlife Service's National Digital Library is a searchable collection of selected images, historical artifacts, audio clips, publications, and video." +

UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
REGION 10
1200 Sixth Avenue, Suite 900
Seattle, Washington 98101-3140

12 DEC 2008

Reply To
Attn Of: OWW-131

Barry Burnell
Water Quality Program Administrator
Idaho Department of Environmental Quality
1410 North Hilton
Boise, Idaho 83706-1255

Re: EPA Disapproval of Idaho's Removal of Mercury Acute and Chronic Freshwater Aquatic Life Criteria, Docket No. 58-0102-0302

Dear Mr. Burnell:

Pursuant to its authority under Section 303(c) of the Clean Water Act ("CWA") and the implementing regulations at 40 CFR Part 131, the U.S. Environmental Protection Agency ("EPA" or "Agency") has reviewed one provision of Idaho's revised water quality standards contained in Docket 58-0102-0302. In accordance with its authorities, EPA is disapproving the removal of acute and chronic numeric freshwater aquatic life criteria for mercury that were contained in columns B1 and B2 in the toxic criteria table and the addition of footnote "g" to that table found at IDAPA 58.01.02.210.01.

Background

In 2003, the Idaho Department of Environmental Quality ("IDEQ") began a negotiated rulemaking in response to a petition from Idaho Mining Association to update Idaho's mercury criteria (Docket No. 58-0102-0302). As a result of the negotiated rulemaking, IDEQ published a proposed rule on August 4, 2004, in the Idaho Administrative Bulletin. The rule proposed to update a selected group of numeric criteria for toxic pollutants. A 45-day public comment period was initiated and three public hearings were held. As part of this negotiated rulemaking, IDEQ proposed removing the acute and chronic numeric freshwater aquatic life criteria for mercury and adding footnote "g" to the toxic criteria table. Footnote "g" states in part that the narrative criteria for toxics apply and that the human health criterion for methyl mercury will be protective of aquatic life in most situations.

On September 20, 2004, EPA submitted comments to IDEQ on the proposed rule stating that the Agency's recommended 304(a) chronic freshwater aquatic life criterion (0.77 μg/l) for mercury may not be adequately protective in Idaho. EPA cited its *"1995 Updates: Water Quality Criteria Documents for the Protection of Aquatic Life in Ambient Water"* (September 1996), which stated that several important species of fish, including rainbow trout, coho salmon and bluegill may not be adequately protected by the recommended chronic criterion (0.77 µg/l) for mercury. Several species of trout and salmon are native to and important in Idaho. At that time, EPA informed IDEQ that the Agency was unlikely to revise its 304(a) chronic aquatic life criterion for mercury any time soon and made three recommendations.

First, EPA recommended IDEQ retain the current chronic value of 0.012 µg/l, which EPA considered protective of aquatic species in Idaho based on our evaluation during consultation under the Endangered Species Act. Second, EPA recommended IDEQ retain the chronic 0.012 µg/l criterion while the State of Idaho or EPA develops a new chronic aquatic life mercury criterion that is adequately protective of aquatic species in Idaho. Third, EPA recommended IDEQ adopt the Agency's current 304(a) freshwater acute aquatic life criterion for mercury (1.4 µg/l) until the State of Idaho or EPA develops a new acute aquatic life mercury criterion that is adequately protective of aquatic species in Idaho.

In addition to its comments and recommendations, EPA also provided IDEQ a detailed discussion clarifying the distinction between EPA's action on May 18, 2000, promulgating toxic criteria for the state of California (California Toxics Rule or "CTR"), and IDEQ's removal of the previously adopted freshwater acute and chronic mercury criteria. EPA provided the additional discussion concerning the CTR because IDEQ had stated that its decision to remove the acute and chronic numeric mercury criteria was based in part on the CTR. In the CTR, EPA reserved the acute and chronic freshwater aquatic life criteria for mercury because of the U.S. Fish and Wildlife Service's opinion that those values were not protective of species in California. Under the CTR reservation, EPA agreed not to promulgate new numbers; rather it would reserve the acute and chronic freshwater aquatic life criteria for mercury with the understanding that the mercury human health water column value of 0.05 µg/l and 0.051 µg/l would be implemented as an interim approach to protect aquatic species until a more adequately protective approach/criteria were developed. In contrast to the CTR, Idaho has no equivalent interim number because IDEQ has not demonstrated that its human health methyl mercury criterion would protect aquatic life.

IDEQ did not revise the proposed rule in accordance with EPA's recommendations, and submitted the rule to the Idaho Board of Environmental Quality on November 18, 2004. The Idaho Board of Environmental Quality adopted the rule and submitted it to the Idaho Legislature in January 2005. The Idaho Legislature adopted the rule as final and made it effective on April 6, 2005. On August 8, 2005, IDEQ submitted the rule, along with several other revisions of the state's water quality standards, to EPA for review and approval.

EPA reviewed the submission and on September 30, 2005, approved numeric criteria for eight different toxic pollutants. However, EPA did not act on the removal of acute and chronic numeric freshwater aquatic life criteria for mercury and replacement of these values with footnote "g" to the table.

EPA's Decision

EPA has reviewed the revision removing the acute and chronic numeric freshwater aquatic life criteria for mercury and replacement with footnote "g" contained in Docket 58-0102-0302. In addition, EPA has reviewed IDEQ's supporting justification entitled *"Technical Justification, Adoption of Mercury Fish Tissue Criterion and Update to Selected Metals Criteria Recommended by EPA as of 2002, Idaho Rulemaking 58-0102-0302."* EPA also reviewed Chapter 7 of Idaho's Implementation Guidance for the Mercury Water Quality Criteria, entitled *"Implications of Criterion Implementation for Aquatic Species and Aquatic-dependent Wildlife Species."*

Section 303(c)(2)(b) of the Clean Water Act states:

> Whenever a State reviews water quality standards...such State shall adopt criteria for all toxic pollutants listed pursuant to section 307(a)(1) of this Act for which criteria have been published under Section 304(a), the discharge or presence of which in the affected water could reasonably be expected to interfere with those designated uses adopted by the State, as necessary to support such designated uses. Such criteria shall be specific numerical criteria for such toxic pollutants.

In EPA's *"Guidance for State Implementation of Water Quality Standards for CWA Section 303(c)(2)(B)"* (December 1988), the Agency states:

> EPA believes that an effective State water quality standards program should include both the chemical specific (i.e., ambient criteria) and narrative approaches. ...The narrative standard can be the basis for limiting toxicity where a specific toxic pollutant can be identified as causing the toxicity, but there is no numeric criterion in State standards. The narrative standard can also be used to limit whole effluent toxicity where it is not known which chemical or chemicals are causing toxicity.
>
> Section 303(c)(2)(B) addresses only pollutants listed as "toxic" pursuant to section 307(a) of the Act, which are codified at 40 CFR §401.15. The section 307(a) list contains 65 compounds and families of compounds, which potentially include thousands of specific compounds. The Agency has interpreted that list to include 126 "priority" toxic pollutants for regulatory purposes.

In addition, water quality standards regulations at 40 CFR 131.11(a)(1) state in part that States must adopt water quality criteria that protect designated uses. Criteria must be based on sound scientific rationale and must contain sufficient parameters or constituents to protect the designated use. Regarding toxic pollutants, 40 CFR

131.11(a)(2) requires States to review water quality data and information on discharges to identify specific water bodies where toxic pollutants may be adversely affecting water quality or the attainment of the designated water use, or where the level of toxic pollutants warrant concern and to adopt criteria for such toxic pollutants applicable to the water body sufficient to protect the designated use. Lastly, 40 CFR 131.11(b) states that in establishing criteria, States should set numerical values based on EPA's 304(a) Guidance, 304(a) Guidance modified to reflect site-specific conditions, or other scientifically defensible methods.

EPA has determined that the removal of the acute and chronic numeric freshwater aquatic life criteria for mercury and replacement with footnote "g" is inconsistent with Clean Water Act Section 303(c) and 40 CFR 131.11. The supporting documentation that Idaho submitted does not provide specific information which would demonstrate that the designated aquatic life uses in Idaho are assured protection from discharges of mercury that would adversely affect water quality and/or the attainment of the aquatic life uses. Although Chapter 7 of Idaho's Implementation Guidance for the Mercury Water Quality Criteria provides a comparative evaluation of the potential effects of 0.3 mg/kg methylmercury on aquatic species and aquatic dependent wildlife relative to the effects anticipated from aquatic life exposure, it does not provide a procedure or detailed implementation for the translation from a human health criterion to protective aquatic life criteria. The Guidance does not contain definitive information on how the State would translate the fish tissue criterion developed to protect human health to a value which can be used to protect aquatic life.

Therefore, EPA is disapproving the removal of the acute and chronic numeric freshwater aquatic life criteria for mercury from columns B1 and B2, as well as the addition of footnote "g" to the table found in the Idaho Water Quality Standards at IDAPA 58.01.02.210.01.

Remedies to Address EPA's Disapproval

The federal water quality standards regulations at 40 CFR 131.21 state in part that when EPA disapproves a State's water quality standards, EPA shall specify changes which are needed to assure compliance with the requirements of Section 303(c) of the Clean Water Act and federal water quality standards regulations. There are several options Idaho could consider in establishing mercury criteria that are based on scientifically defensible methods and protect Idaho's designated aquatic life uses, including:

1) evaluate the protectiveness of EPA's current recommended 304(a) numeric acute freshwater aquatic life criterion for mercury (1.4 μg/l);
2) evaluate the protectiveness of Idaho's previous numeric chronic freshwater aquatic life criterion for mercury (0.012 μg/l);
3) evaluate development of Idaho-specific numeric acute and chronic freshwater aquatic life criteria for mercury; and

4) evaluate the use of a combination of protective numeric water column values and numeric wildlife criteria appropriate for Idaho species (this approach is being used in the Great Lakes Initiative).

Please note that EPA has not recommended using the 304(a) numeric chronic freshwater aquatic life criterion for mercury (0.77 µg/l) in the above list of options. As discussed above, the 304(a) criteria may not adequately protect such important fishes as the rainbow trout, coho salmon and bluegill, and there are several species of trout and salmon present in Idaho. EPA suggests that further evaluation is needed when considering adoption of the chronic criterion for waters where salmonids are present. EPA recommends further analysis of whether the 304(a) chronic criteria would be protective of designated uses in Idaho because salmonids are a key component of many of the aquatic communities in Idaho's waters. If IDEQ pursues this option, EPA recommends that any analysis is prepared prior to public comment and made available to the public for review at that time. Furthermore, if this approach is used EPA will require a scientifically sound demonstration of the protectiveness of any criterion to be provided at the time of submission.

Freshwater Aquatic Life Criteria for Mercury Currently in Effect in Idaho

Until Idaho develops and adopts and EPA approves revisions to numeric acute and chronic aquatic life criteria for mercury, the numeric aquatic life mercury criteria applicable to the designated aquatic life uses in Idaho that are effective for Clean Water Act Purposes are the previously adopted acute (2.1 µg/l) and chronic (0.012 µg/l) mercury criteria which EPA approved in 1997.

Please feel free to contact me at (206) 553-7151 if you have questions concerning this letter or Lisa Macchio, Idaho Water Quality Standards Coordinator, at (206) 553-1834.

Sincerely,

Michael F. Gearheard
Director, Office of Water and Watersheds

cc: Michael McIntyre, IDEQ
Don Essig, IDEQ

2002

Pharmaceuticals, Hormones, and Other Organic Wastewater Contaminants in U.S. Streams, 1999-2000: A National Reconnaissance

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Pharmaceuticals, Hormones, and Other Organic Wastewater Contaminants in U.S. Streams, 1999−2000: A National Reconnaissance

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To provide the first nationwide reconnaissance of the occurrence of pharmaceuticals, hormones, and other organic wastewater contaminants (OWCs) in water resources, the U.S. Geological Survey used five newly developed analytical methods to measure concentrations of 95 OWCs in water samples from a network of 139 streams across 30 states during 1999 and 2000. The selection of sampling sites was biased toward streams susceptible to contamination (i.e. downstream of intense urbanization and livestock production). OWCs were prevalent during this study, being found in 80% of the streams sampled. The compounds detected represent a wide range of residential, industrial, and agricultural origins and uses with 82 of the 95 OWCs being found during this study. The most frequently detected compounds were coprostanol (fecal steroid), cholesterol (plant and animal steroid), *N*,*N*-diethyltoluamide (insect repellant), caffeine (stimulant), triclosan (antimicrobial disinfectant), tri(2-chloroethyl)phosphate (fire retardant), and 4-nonylphenol (nonionic detergent metabolite). Measured concentrations for this study were generally low and rarely exceeded drinking-water guidelines, drinking-water health advisories, or aquatic-life criteria. Many compounds, however, do not have such guidelines established. The detection of multiple OWCs was common for this study, with a median of seven and as many as 38 OWCs being found in a given water sample. Little is known about the potential interactive effects (such as synergistic or antagonistic toxicity) that may occur from complex mixtures of OWCs in the environment. In addition, results of this study demonstrate the importance of obtaining data on metabolites to fully understand not only the fate and transport of OWCs in the hydrologic system but also their ultimate overall effect on human health and the environment.

Introduction

The continued exponential growth in human population has created a corresponding increase in the demand for the Earth's limited supply of freshwater. Thus, protecting the integrity of our water resources is one of the most essential environmental issues of the 21st century. Recent decades have brought increasing concerns for potential adverse human and ecological health effects resulting from the production, use, and disposal of numerous chemicals that offer improvements in industry, agriculture, medical treatment, and even common household conveniences (*1*). Research has shown that many such compounds can enter the environment, disperse, and persist to a greater extent than first anticipated. Some compounds, such as pesticides, are intentionally released in measured applications. Others, such as industrial byproducts, are released through regulated and unregulated industrial discharges to water and air resources. Household chemicals, pharmaceuticals, and other consumables as well as biogenic hormones are released directly to the environment after passing through wastewater treatment processes (via wastewater treatment plants, or domestic septic systems), which often are not designed to remove them from the effluent (*2*). Veterinary pharmaceuticals used in animal feeding operations may be released to the environment with animal wastes through overflow or leakage from storage structures or land application (*3*). As a result, there are a wide variety of transport pathways for many different chemicals to enter and persist in environmental waters.

Surprisingly, little is known about the extent of environmental occurrence, transport, and ultimate fate of many synthetic organic chemicals after their intended use, particularly hormonally active chemicals (*4*), personal care products, and pharmaceuticals that are designed to stimulate a physiological response in humans, plants, and animals (*1*, *5*). One reason for this general lack of data is that, until recently, there have been few analytical methods capable of detecting these compounds at low concentrations which might be expected in the environment (*6*). Potential concerns from the environmental presence of these compounds include abnormal physiological processes and reproductive impairment (*7*−*12*), increased incidences of cancer (*13*), the development of antibiotic-resistant bacteria (*14*−*17*), and the potential increased toxicity of chemical mixtures (*18*). For many substances, the potential effects on humans and aquatic ecosystems are not clearly understood (*1*, *2*, *19*).

The primary objective of this study is to provide the first nationwide reconnaissance of the occurrence of a broad suite of 95 organic wastewater contaminants (OWCs), including

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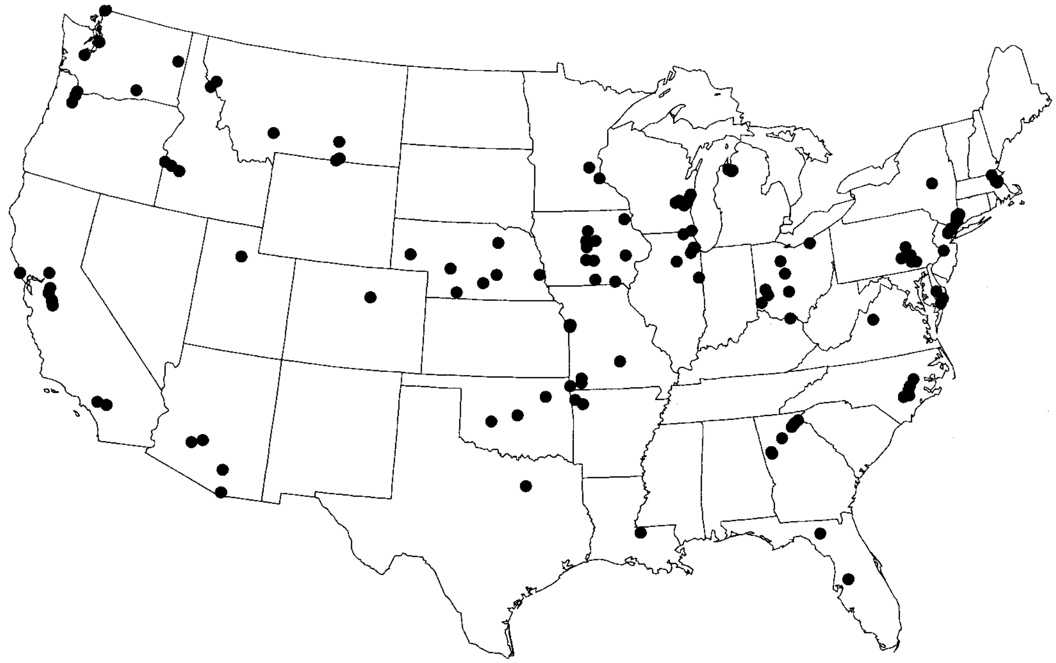

FIGURE 1. Location of 139 stream sampling sites.

many compounds of emerging environmental concern, in streams across the United States. These OWCs are potentially associated with human, industrial, and agricultural wastewaters and include antibiotics, other prescription drugs, nonprescription drugs, steroids, reproductive hormones, personal care products, products of oil use and combustion, and other extensively used chemicals. The target OWCs were selected because they are expected to enter the environment through common wastewater pathways, are used in significant quantities, may have human or environmental health implications, are representative or potential indicators of certain classes of compounds or sources, and/or can be accurately measured in environmental samples using available technologies. Although these 95 OWCs are just a small subset of compounds being used by society, they represent a starting point for this investigation examining the transport of OWCs to water resources of the United States.

This paper describes the analytical results available from 139 streams sampled during 1999−2000 (Figure 1). The results are intended to determine if OWCs are entering U.S. streams and to estimate the extent of their co-occurrence in susceptible waters. In addition, this study provides a focal point for the development and testing of new laboratory methods for measuring OWCs in environmental samples at trace levels, an interpretive context for future assessments of OWCs, and a means for establishing research priorities and future monitoring strategies. More complete interpretations, including an evaluation of the role of potential sources of contamination, will follow in subsequent papers.

Site Selection and Sampling

Little data were available on the occurrence of most of the targeted OWCs in U.S. streams at the onset of this investigation. Therefore, the selection of sampling sites primarily focused on areas considered susceptible to contamination from human, industrial, and agricultural wastewater. The 139 stream sites sampled during 1999−2000 (Figure 1) represent a wide range of geography, hydrogeology, land use, climate, and basin size. Specific information on the individual sampling sites is provided elsewhere (*20*).

All samples were collected by U.S. Geological Survey personnel using consistent protocols and procedures designed to obtain a sample representative of the streamwaters using standard depth and width integrating techniques (*21*). At each site, a composite water sample was collected from about 4−6 vertical profiles which was split into appropriate containers for shipment to the participating laboratories. For those bottles requiring filtration, water was passed through a 0.7 μm, baked, glass-fiber filter in the field where possible, or else filtration was conducted in the laboratory. Water samples for each chemical analysis were stored in precleaned-amber, glass bottles and collected in duplicate. The duplicate samples were used for backup purposes (in case of breakage of the primary sample) and for laboratory replicates. Following collection, samples were immediately chilled and sent to the laboratory. To minimize contamination of samples, use of personal care items (i.e. insect repellents, colognes, perfumes), caffeinated products, and tobacco were discouraged during sample collection and processing.

Each stream site was sampled once during the 1999−2000 study period. Samples collected in 1999 were analyzed for a subset of the OWCs based on the watershed land-use characteristics. Samples collected in 2000 were analyzed for the complete suite of OWCs. The analytical results for each stream sample are available elsewhere (*20*).

Analytical Methods

To determine the environmental extent of 95 OWCs (Table 1) in susceptible streams, five separate analytical methods were used. Each method was developed independently in different laboratories, with somewhat different data objectives, such as identifying hormones versus identifying antibiotics. As a result of these differing objectives, varying approaches were used in the development of the five analytical methods. For example, select methods (Methods 1−3 below) used filtered water for solid-phase extraction (SPE) with liquid chromatography/mass spectrometry positive-ion electrospray (LC/MS-ESI(+)) analysis, while others (Methods 4 and 5 below) used whole-water continuous liquid−liquid extraction (CLLE) with capillary gas chromatography/mass spectrometry (GC/MS) analysis.

All methods use selected ion monitoring (SIM) for improved sensitivity, thus, only the target compounds were reported with no attempt to report data for nontarget

TABLE 1. Summary of Analytical Results of Streams Sampled for 95 Organic Wastewater Contaminants[i]

chemical (method)	CASRN	*N*	RL (μg/L)	freq (%)	max (μg/L)	med (μg/L)	use	MCL or *HAL* (*23*) (μg/L)	lowest LC_{50} for the most sensitive indicator species (μg/L)/no. of aquatic studies identified (*24*)
Veterinary and Human Antibiotics									
carbodox (1)	6804-07-5	104	0.10	0	ND	ND	antibiotic	–	–/1
chlortetracycline (1)	57-62-5	115	0.05	0	ND	ND	antibiotic	–	88000[a]/3
chlortetracycline (2)	57-62-5	84	0.10	2.4	0.69	0.42	antibiotic	–	88000[a]/3
ciprofloxacin (1)	85721-33-1	115	0.02	2.6	0.03	0.02	antibiotic	–	–/0
doxycycline (1)	564-25-0	115	0.1	0	ND	ND	antibiotic	–	–/0
enrofloxacin (1)	93106-60-6	115	0.02	0	ND	ND	antibiotic	–	40[b]/29
erythromycin-H_2O (1)	114-07-8	104	0.05	21.5	1.7	0.1	erythromycin metabolite	–	665000[b]/35
lincomycin (1)	154-21-2	104	0.05	19.2	0.73	0.06	antibiotic	–	–/0
norfloxacin (1)	70458-96-7	115	0.02	0.9	0.12	0.12	antibiotic	–	–/6
oxytetracycline (1)	79-57-2	115	0.1	0	ND	ND	antibiotic	–	102000[a]/46
oxytetracycline (2)	79-57-2	84	0.10	1.2	0.34	0.34	antibiotic	–	102000[a]/46
roxithromycin (1)	80214-83-1	104	0.03	4.8	0.18	0.05	antibiotic	–	–/0
sarafloxacin (1)	98105-99-8	115	0.02	0	ND	ND	antibiotic	–	–/0
sulfachloropyridazine (2)	80-32-0	84	0.05	0	ND	ND	antibiotic	–	–/0
sulfadimethoxine (1)	122-11-2	104	0.05	0	ND	ND	antibiotic	–	–/5
sulfadimethoxine (2)	122-11-2	84	0.05	1.2	0.06	0.06	antibiotic	–	–/5
sulfamerazine (1)	127-79-7	104	0.05	0	ND	ND	antibiotic	–	100000[c]/17
sulfamerazine (2)	127-79-7	84	0.05	0	ND	ND	antibiotic	–	100000[c]/17
sulfamethazine (1)	57-68-1	104	0.05	4.8	0.12	0.02	antibiotic	–	100000[c] 17
sulfamethazine (2)	57-68-1	84	0.05	1.2	0.22	0.22	antibiotic	–	100000[c]/17
sulfamethizole (1)	144-82-1	104	0.05	1.0	0.13	0.13	antibiotic	–	–/0
sulfamethoxazole (1)	723-46-6	104	0.05	12.5	1.9	0.15	antibiotic	–	–/0
sulfamethoxazole (3)	723-46-6	84	0.023	19.0	0.52	0.066	antibiotic	–	–/0
sulfathiazole (1)	72-14-0	104	0.10	0	ND	ND	antibiotic	–	–/0
sulfathiazole (2)	72-14-0	84	0.05	0	ND	ND	antibiotic	–	–/0
tetracycline (1)	60-54-8	115	0.05	0	ND	ND	antibiotic	–	550000[b]/3
tetracycline (2)	60-54-8	84	0.10	1.2	0.11	0.11	antibiotic	–	550000[b]/3
trimethoprim (1)	738-70-5	104	0.03	12.5	0.71	0.15	antibiotic	–	3000[c]/4
trimethoprim (3)	738-70-5	84	0.014	27.4	0.30	0.013	antibiotic	–	3000[c]/4
tylosin (1)	1401-69-0	104	0.05	13.5	0.28	0.04	antibiotic	–	–/0
virginiamycin (1)	21411-53-0	104	0.10	0	ND	ND	antibiotic	–	–/0
Prescription Drugs									
albuterol (salbutamol) (3)	18559-94-9	84	0.029	0	ND	ND	antiasthmatic	–	–/0
cimetidine (3)	51481-61-9	84	0.007	9.5	0.58[d]	0.074[d]	antacid	–	–/0
codeine (3)	76-57-3	46	0.24	6.5	0.019	0.012	analgesic	–	–/0
codeine (4)	76-57-3	85	0.1	10.6	1.0[d]	0.2[d]	analgesic	–	–/0
dehydronifedipine (3)	67035-22-7	84	0.01	14.3	0.03	0.012	antianginal	–	–/0
digoxin (3)	20830-75-5	46	0.26	0	ND[d]	ND[d]	cardiac stimulant	–	10000000[a]/24
digoxigenin (3)	1672-46-4	84	0.008	0	ND	ND	digoxin metabolite	–	–/0
diltiazem (3)	42399-41-7	84	0.012	13.1	0.049	0.021	antihypertensive	–	–/0
enalaprilat (3)	76420-72-9	84	0.15	1.2	0.046[d]	0.046[d]	enalapril maleate (antihypertensive) metabolite	–	–/0
fluoxetine (3)	54910-89-3	84	0.018	1.2	0.012[d]	0.012[d]	antidepressant	–	–/0
gemfibrozil (3)	25812-30-0	84	0.015	3.6	0.79	0.048	antihyperlipidemic	–	–/0
metformin (3)	657-24-9	84	0.003	4.8	0.15[d]	0.11[d]	antidiabetic	–	–/0
paroxetine metabolite (3)	–	84	0.26	0	ND[d]	ND[d]	paroxetine (antidepressant) metabolite	–	–/0
ranitidine (3)	66357-35-5	84	0.01	1.2	0.01[d]	0.01[d]	antacid	–	–/0
warfarin (3)	81-81-2	84	0.001	0	ND	ND	anticoagulant	–	16000[c]/ 33
Nonprescription Drugs									
acetaminophen (3)	103-90-2	84	0.009	23.8	10	0.11	antipyretic	–	6000[a]/ 14
caffeine (3)	58-08-2	84	0.014	61.9	6.0	0.081	stimulant	–	40000[e]/ 77
caffeine (4)	58-08-2	85	0.08	70.6	5.7	0.1	stimulant	–	40000[e]/ 77
cotinine (3)	486-56-6	84	0.023	38.1	0.90	0.024	nicotine metabolite	–	–/0
cotinine (4)	486-56-6	54	0.04	31.5	0.57	0.05	nicotine metabolite	–	–/0
1,7-dimethylxanthine (3)	611-59-6	84	0.018	28.6	3.1[d]	0.11[d]	caffeine metabolite	–	–/0
ibuprofen (3)	15687-27-1	84	0.018	9.5	1.0	0.20	antiinflammatory	–	–/0
Other Wastewater-Related Compounds									
1,4-dichlorobenzene (4)	106-46-7	85	0.03	25.9	4.3	0.09	deodorizer	*75*	1100[c]/190
2,6-di-*tert*-butylphenol (4)	128-39-2	85	0.08	3.5	0.11[d]	0.06[d]	antioxidant	–	–/2
2,6-di-*tert*-butyl-1,4-benzoquinone (4)	719-22-2	85	0.10	9.4	0.46	0.13	antioxidant	–	–/0
5-methyl-1H-benzotriazole (4)	136-85-6	54	0.10	31.5	2.4	0.39	anticorrosive	–	–/0
acetophenone (4)	98-86-2	85	0.15	9.4	0.41	0.15	fragrance	–	155000[e]/21
anthracene (4)	120-12-7	85	0.05	4.7	0.11	0.07	PAH	–	5.4[e]/188
benzo[*a*]pyrene (4)	50-32-8	85	0.05	9.4	0.24	0.04	PAH	0.2	1.5[a]/428
3-*tert*-butyl-4-hydroxy anisole (4)	25013-16-5	85	0.12	2.4	0.2[d]	0.1[d]	antioxidant	–	870[c]/14
butylated hydroxy toluene (4)	128-37-0	85	0.08	2.4	0.1[d]	0.1[d]	antioxidant	–	1440[a]/15
bis(2-ethylhexyl) adipate (4)	103-23-1	85	2.0	3.5	10[f]	3[f]	plasticizer	400	480[a]/9
bis(2-ethylhexyl) phthalate (4)	117-81-7	85	2.5	10.6	20[f]	7[f]	plasticizer	6	7500[a]/309

TABLE 1. (Continued)

chemical (method)	CASRN	*N*	RL (μg/L)	freq (%)	max (μg/L)	med (μg/L)	use	MCL or *HAL* (*23*) (μg/L)	lowest LC_{50} for the most sensitive indicator species (μg/L)/no. of aquatic studies identified (*24*)
					Other Wastewater-Related Compounds				
bisphenol A (4)	80-05-7	85	0.09	41.2	12	0.14	plasticizer	–	3600[e]/26
carbaryl (4)	63-25-2	85	0.06	16.5	0.1[d]	0.04[d]	insecticide	*700*	0.4[a]/1541
***cis*-chlordane** (4)	5103-71-9	85	0.04	4.7	0.1	0.02	insecticide	2	7.4[b]/28
chlorpyrifos (4)	2921-88-2	85	0.02	15.3	0.31	0.06	insecticide	*20*	0.1[a]/1794
diazinon (4)	333-41-5	85	0.03	25.9	0.35	0.07	insecticide	*0.6*	0.56[a]/1040
dieldrin (4)	60-57-1	85	0.08	4.7	0.21	0.18	insecticide	0.2	2.6[c]/1540
diethylphthalate (4)	84-66-2	54	0.25	11.1	0.42	0.2	plasticizer	–	12000[c]/129
ethanol,2-butoxy-phosphate (4)	78-51-3	85	0.2	45.9	6.7	0.51	plasticizer	–	10400[e]/7
fluoranthene (4)	206-44-0	85	0.03	29.4	1.2	0.04	PAH	–	74[e]/216
lindane (4)	58-89-9	85	0.05	5.9	0.11	0.02	insecticide	0.2	30[c]/1979
methyl parathion (4)	298-00-0	85	0.06	1.2	0.01	0.01	insecticide	*2*	12[a]/888
4-methyl phenol (4)	106-44-5	85	0.04	24.7	0.54	0.05	disinfectant	–	1400[a]/74
naphthalene (4)	91-20-3	85	0.02	16.5	0.08	0.02	PAH	*20*	910[c]/519
N,*N*-diethyltoluamide (4)	134-62-3	54	0.04	74.1	1.1	0.06	insect repellant	–	71250[c]/9
4-nonylphenol (4)	251-545-23	85	0.50	50.6	40[g]	0.8[g]	nonionic detergent metabolite	–	130[e]/135
4-nonylphenol monoethoxylate (4)	–	85	1.0	45.9	20[g]	1[g]	nonionic detergent metabolite		14450[a]/4
4-nonylphenol diethoxylate (4)	–	85	1.1	36.5	9[g]	1[g]	nonionic detergent metabolite	–	5500[a]/6
4-octylphenol monoethoxylate (4)	–	85	0.1	43.5	2[g]	0.2[g]	nonionic detergent metabolite	–	–/0
4-octylphenol diethoxylate (4)	–	85	0.2	23.5	1[g]	0.1[g]	nonionic detergent metabolite	–	–/0
phenanthrene (4)	85-01-8	85	0.06	11.8	0.53	0.04	PAH	–	590[a]/192
phenol (4)	108-95-2	85	0.25	8.2	1.3[f]	0.7[f]	disinfectant	*400*	4000[c]/2085
phthalic anhydride (4)	85-44-9	85	0.25	17.6	1[f]	0.7[f]	plastic manufacturing	–	40400[c]/5
pyrene (4)	129-00-0	85	0.03	28.2	0.84	0.05	PAH	–	90.9[a]/112
tetrachloroethylene (4)	127-18-4	85	0.03	23.5	0.70[d]	0.07[d]	solvent, degreaser	5	4680[c]/147
triclosan (4)	3380-34-5	85	0.05	57.6	2.3	0.14	antimicrobial disinfectant	–	180[e]/3
tri(2-chloroethyl) phosphate (4)	115-96-8	85	0.04	57.6	0.54	0.1	fire retardant	–	66000[b]/8
tri(dichlorisopropyl) phosphate (4)	13674-87-8	85	0.1	12.9	0.16	0.1	fire retardant	–	3600[b]/9
triphenyl phosphate (4)	115-86-6	85	0.1	14.1	0.22	0.04	plasticizer	–	280[c]/66
					Steroids and Hormones				
***cis*-androsterone** (5)	53-41-8	70	0.005	14.3	0.214	0.017	urinary steroid	–	–/0
cholesterol (4)	57-88-5	85	1.5	55.3	10[d]	1[d]	plant/animal steroid	–	–/0
cholesterol (5)	57-88-5	70	0.005	84.3	60[h]	0.83	plant/animal steroid	–	–/0
coprostanol (4)	360-68-9	85	0.6	35.3	9.8[d]	0.70[d]	fecal steroid	–	–/0
coprostanol (5)	360-68-9	70	0.005	85.7	150[h]	0.088	fecal steroid	–	–/0
equilenin (5)	517-09-9	70	0.005	2.8	0.278	0.14	estrogen replacement	–	–/0
equilin (5)	474-86-2	70	0.005	1.4	0.147	0.147	estrogen replacement	–	–/0
17α-ethynyl estradiol (5)	57-63-6	70	0.005	15.7	0.831	0.073	ovulation inhibitor	–	–/22
17α-estradiol (5)	57-91-0	70	0.005	5.7	0.074	0.03	reproductive hormone	–	–/0
17β-estradiol (4)	50-28-2	85	0.5	10.6	0.2[d]	0.16[d]	reproductive hormone	–	–/0
17β-estradiol (5)	50-28-2	70	0.005	10.0	0.093	0.009	reproductive hormone	–	–/0
estriol (5)	50-27-1	70	0.005	21.4	0.051	0.019	reproductive hormone	–	–/0
estrone (5)	53-16-7	70	0.005	7.1	0.112	0.027	reproductive hormone	–	–/11
mestranol (5)	72-33-3	70	0.005	10.0	0.407	0.074	ovulation inhibitor	–	–/0
19-norethisterone (5)	68-22-4	70	0.005	12.8	0.872	0.048	ovulation inhibitor	–	–/0
progesterone (5)	57-83-0	70	0.005	4.3	0.199	0.11	reproductive hormone	–	–/0
stigmastanol (4)	19466-47-8	54	2.0	5.6	4[d]	2[d]	plant steroid	–	–/0
testosterone (5)	58-22-0	70	0.005	2.8	0.214	0.116	reproductive hormone	–	–/4

[a] *Daphnia magna* (water flea) – 48 h exposure LC_{50}. [b] Other species and variable conditions. [c] *Oncorhynchus mykiss* (rainbow trout) – 96 h exposure LC_{50}. [d] Concentration estimated – average recovery <60%. [e] *Pimephales promelas* (fathead minnow) – 96 h exposure LC_{50}. [f] Concentration estimated – compound routinely detected in laboratory blanks. [g] Concentration estimated – reference standard prepared from a technical mixture. [h] Concentration estimated – value greater than highest point on calibration curve. [i] Compounds suspected of being hormonally active are in bold (*4, 22*). CASRN, Chemical Abstracts Service Registry Number; *N*, number of samples; RL, reporting level; freq, frequency of detection; max, maximum concentration; med, median detectable concentration; MCL, maximum contaminant level; HAL, health advisory level; LC_{50}, lethal concentration with 50% mortality; ND, not detected; –, not available; PAH, polycyclic aromatic hydrocarbon.

compounds. Target compounds within each method were selected from the large number of chemical possibilities based upon usage, toxicity, potential hormonal activity, and persistence in the environment. Some compounds that fit the above criteria, however, could not be included (such as amoxicillin, roxarsone, polybrominated diphenyl ethers) because they were either incompatible with the corresponding method or reference standards were not available. Positive identification of a compound required elution within the expected retention time window. In addition, the sample spectra and ion abundance ratios were required to match that of the reference standard compounds. The base-peak ion was used for quantitation, and, if possible, two qualifier ions were used for confirmation. After qualitative criteria were met, compound concentrations were calculated from 5 to 8 point calibration curves (generally from 0.01 to 10.0 μg/L) using internal standard quantitation. Methods 1 and 2 process calibration standards through the extraction procedure, which generally corrects concentrations for method losses but not matrix effects. Methods 3–5 do not extract calibration standards, thus the reported concentrations are not corrected for method losses. Reporting levels (RLs) were determined for each method by either an evaluation of instrument response, calculation of limit of detection, or from a previously published procedure (*25*). RLs were adjusted based on experience with the compounds in each method, known interferences, or known recovery problems.

The following descriptions are intended to provide a brief overview of the five analytical methods used for this study. More comprehensive method descriptions are provided elsewhere (*26*−*28*) or will be available in subsequent publications.

Method 1. This method targets 21 antibiotic compounds (Table 1) in 500-mL filtered water samples using modifications from previously described methods (*26*, *29*). The antibiotics were extracted and analyzed by tandem SPE and single quadrapole, LC/MS-ESI(+) using SIM. To prevent the tetracycline antibiotics from complexing with Ca^{2+} and Mg^{2+} ions and residual metals on the SPE cartridges, 0.5 mg of disodium ethylenediaminetetraacetate (Na_2EDTA; $C_{10}H_{14}O_8$-Na_2N_2−H_2O) was added to each water sample. Sample pH was adjusted to 3 using concentrated H_2SO_4. The tandem SPE included an Oasis Hydrophilic−Lipophilic-Balance (HLB) cartridge (60 mg) followed by a mixed mode, HLB-cation exchange (MCX) cartridge (60 mg) (Waters Inc., Milford, MA). The HLB and MCX cartridges were conditioned with ultrapure H_2O, CH_3OH, and CH_3OH with 5% NH_4OH. The HLB cartridge was attached to the top of the MCX cartridge, and the sample was passed through the SPE cartridges using a vacuum extraction manifold. The cartridges were eluted with CH_3OH, and the MCX cartridge was eluted separately using CH_3OH with 5% NH_4OH. The eluate was spiked with 500 ng of $^{13}C_6$-sulfamethazine (internal standard), vortexed, and evaporated to 20 μL using N_2 and a water bath of 55° C. Three hundred μL of 20 mM of $NH_4C_2H_3OO$ (pH 5.7) was added to sample eluate, vortexed, transferred to a glass chromatography vial, and frozen until analysis. Samples were extracted as a set of 11 environmental samples, one duplicate sample, two fortified ultrapure water spikes (check standards), and two ultrapure water blanks.

Method 2. This method targets eight antibiotic compounds (Table 1) in filtered water samples. Complete details of this method have been described previously (*26*). The antibiotics were extracted and analyzed using SPE and SIM LC/MS-ESI(+). Samples were prepared for extraction by adding $^{13}C_6$-sulfamethazine and meclocycline as surrogate standards, Na_2EDTA, and H_2SO_4. Target compounds were extracted using 60-mg HLB cartridges preconditioned with CH_3OH, NHCl, and distilled H_2O. Target compounds were eluted with CH_3OH into a test tube containing the internal standard, simatone. The extracts were then concentrated under N_2 to approximately 50 μL, and mobile phase A (10 mM $NH_4H_2O_2$ in 90/10 water/CH_3OH with 0.3% CH_2O_2) was added. The resulting solutions were transferred to amber autosampler vials to prevent photodegradation of tetracyclines (*30*). Mobile phase conditions are described in detail elsewhere (*26*).

For each compound, the proton adduct of the molecular ion (M + H)+ and at least one confirming ion were acquired using LC/MS-ESI(+). All mass spectral conditions are described in detail elsewhere (*26*). Quantitation was based on the ratio of the base peak ion (M + H)+ of the analyte to the base peak of the internal standard. Standard addition was used for quantitation where each sample was analyzed with and without the addition of a 0.5 μg/L spike to correct for suppression of the electrospray signal.

Method 3. This method targets 21 human prescription and nonprescription drugs and their select metabolites (Table 1) in filtered water samples. Compounds were extracted from 1 L water samples using SPE cartridges that contain 0.5 g of HLB (flow rate of 15 mL/min). After extraction, the adsorbed compounds were eluted with CH_3OH followed by CH_3OH acidified with $C_2HCl_3O_2$. The two fractions were reduced under N_2 to near dryness and then combined and brought to a final volume of 1 mL in 10% C_2H_3N:90% H_2O buffered with $NH_4H_2O_2$/CH_2O_2.

Compounds were separated and measured by high-performance liquid chromatography (HPLC) using a polar (neutral silanol) reverse-phase octylsilane (C8) HPLC column (Metasil Basic 3 μm, 150 × 2.0 mm; Metachem Technologies). The compounds were eluted with a binary gradient of mobile phase A (aqueous $NH_4H_2O_2$/CH_2O_2 buffer; 10 mM, pH 3.7) and mobile phase B (100% C_2H_3N).

Method 4. This method (*27*, *28*) targets 46 OWCs (Table 1) in unfiltered water. One-liter whole-water samples were extracted using CLLE with CH_2Cl_2. Distilled solvent was recycled through a microdroplet dispersing frit to improve extraction efficiency. Samples were extracted for 3 h at ambient pH and for an additional 3 h at pH 2. The extract was concentrated under N_2 to 1 mL and analyzed by capillary-column GC/MS. Available standards for the 4-nonylphenol compounds were composed of multiple isomers, and thus, laboratory standards for these compounds as well as octylphenol ethoxylates were prepared from technical mixtures.

Method 5. This method (*28*) targets 14 steroid compounds including several biogenic and synthetic reproductive hormones (Table 1). The CLLE extracts from the previously analyzed samples of Method 4 were derivatized and reanalyzed. Analysis of steroid and hormone compounds by GC/MS is enhanced by derivatization to deactivate the hydroxyl and keto functional groups. The technique used in this study is the formation of trimethylsilyl (TMS) ethers of the hydroxyl groups and oximes of the keto groups. Samples were stored in a silanizing reagent to prevent hydrolysis of the derivatives back to the free compound. Surrogate standards ($d_4$estradiol and $d_7$cholesterol) were added to the samples prior to derivatization to evaluate method performance. After derivatization, the samples were analyzed by GC/MS.

Quality Assurance Protocol. At least one fortified laboratory spike and one laboratory blank was analyzed with each set of 10−16 environmental samples. Most methods had surrogate compounds added to samples prior to extraction to monitor method performance. A summary of recoveries for target compounds and surrogate compounds in environmental samples (Table 2) indicates the general proficiency of the methods. The RL (Table 1) is equivalent to the lowest concentration standard that could be reliably quantitated. The compound concentrations reported below the RL or the lowest calibration standard were estimated as indicated in Figure 2. The concentration of compounds with <60% recovery, routinely detected in laboratory blanks, or prepared with technical grade mixtures, was also considered estimated (Table 1).

The laboratory blanks were used to assess potential sample contamination. Blank contamination was not subtracted from environmental results. However, environmental concentrations within twice the values observed in the set blank were reported as less than the RL.

A field quality assurance protocol was used to determine the effect, if any, of field equipment and procedures on the concentrations of OWCs in water samples. Field blanks, made from laboratory-grade organic free water, were submitted for about 5% of the sites and analyzed for all of the 95 OWCs. Field blanks were subject to the same sample processing, handling, and equipment as the stream samples. To date, one field blank had a detection of coprostanol and testosterone, one field blank had a detection of naphthalene and tri(dichlorisopropyl)phosphate, and one field blank had

TABLE 2. Summary of Quality Assurance/Quality Control Results for Target and Surrogate Compounds[b]

compound	spike concn (μg/L)	mean % recovery	% RSD
	Method 1		
target compounds	1.0	99.0	12.1
	Method 2		
target compounds	1.0	97.5	12.2
$^{13}C_6$-sulfamethazine	1.0	80.0	20.0
meclocycline	1.0	80.0	20.0
	Method 3		
target compounds	0.5	85.1	11.6
C_{13}-phenacetin	1.0	96.8	14.0
	Method 4		
target compounds	1.0	81.0	11.0
d_{21}-BHT	2.0	63.0	25.0
n-nonylphenol	2.0	83.0	20.0
	Method 5		
target compounds	NA	NA	NA
d_4-estradiol[a]	0.047	128.8	42.0
d_3-testosterone[a]	0.051	148.5	47.3
d_7-cholesterol[a]	0.053	116.9	55.9

[a] Surrogate standard added after CCLE extraction but prior to derivitization. [b] RSD, relative standard deviation; NA, not currently available.

a detection of naphthalene, 4-nonylphenol, phenol, 4-*tert*-octylphenol monoethoxylate, and ethanol,2-butoxy-phosphate. Most of these detections were near their respective RLs verifying the general effectiveness of the sampling protocols used for this study. In addition all field blanks had low level concentrations of cholesterol being measured using Method 5 (median concentration = 0.09 μg/L) documenting its ubiquitous nature in the environment. Cholesterol concentrations from 0.005 to 0.18 μg/L obtained through Method 5 were set to less than the RL.

Compounds that were measured by more than one analytical method (Table 1; Figure 3) also were used to evaluate the results for this study. The presence or absence of these compounds were confirmed in 100% of the determinations for sulfamerazine, and sulfathiazole; 98.8% for oxytetracycline, sulfadimethoxine, sulfamethazine, and tetracycline; 98.6% for cholesterol and coprostanol; 97.6% for chlortetracyline; 95.7% for 17β-estradiol; 94.4% for cotinine; 94.0% for trimethoprim; 89.1% for sulfamethoxazole; 86.4% for codeine; and 83.3% for caffeine. The comparisons for codeine, caffeine, and cotinine may have been affected by the differing extractions (SPE versus CLLE) as well as differing types of sample (filtered versus whole water).

An interlaboratory comparison of Methods 1 and 3 was conducted using two reagent water blanks and 24 reagent water spikes prepared at concentrations ranging from 0.5 to 1.1 μg/L for two frequently detected antibiotics (sulfamethoxazole and trimethoprim). The results demonstrated that both methods are accurately confirming the presence of sulfamethoxazole and trimethoprim in water, with the measured concentrations being within a factor of 3 or better of the actual concentrations for these compounds. No false positives or false negatives occurred for this experiment.

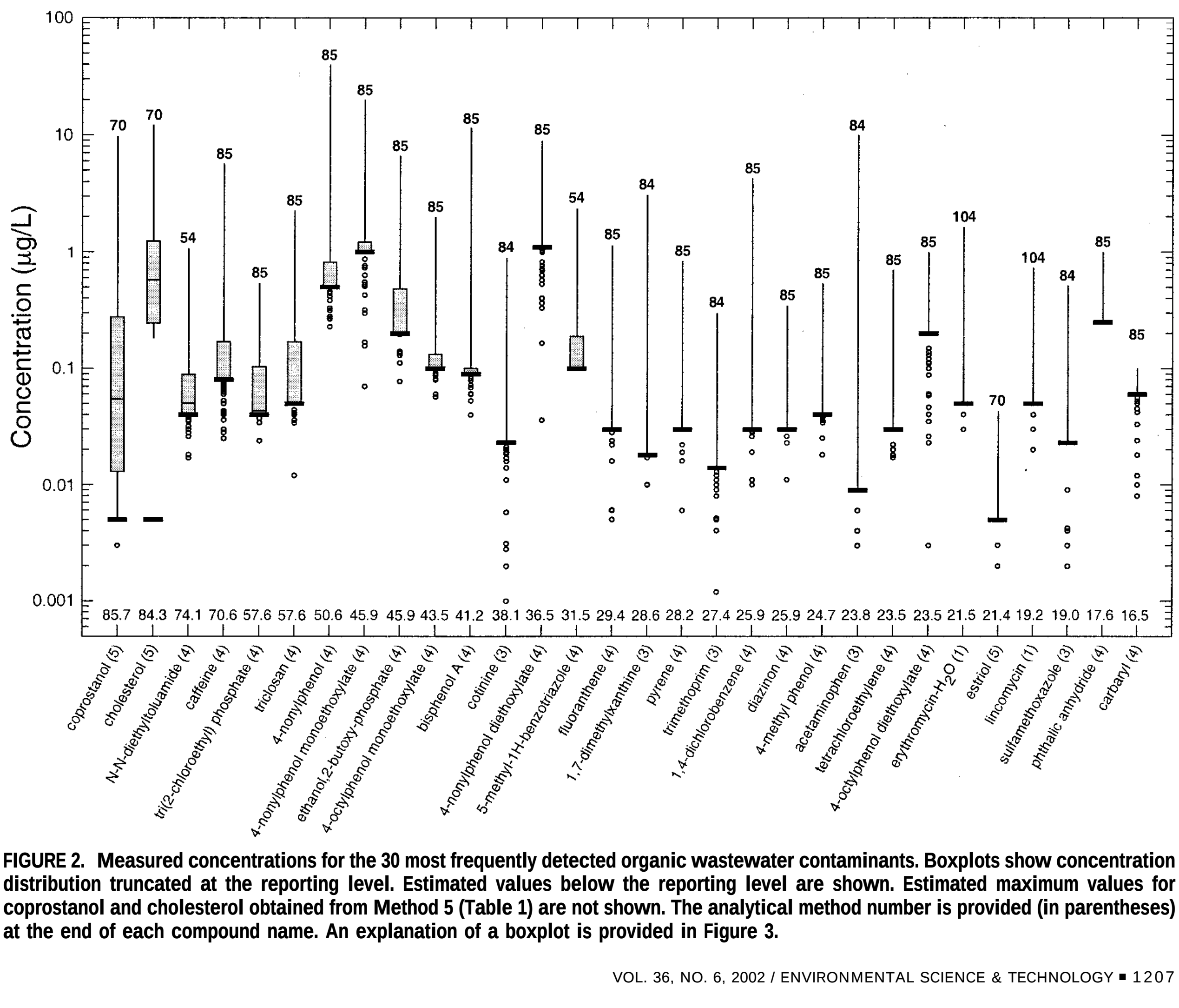

FIGURE 2. Measured concentrations for the 30 most frequently detected organic wastewater contaminants. Boxplots show concentration distribution truncated at the reporting level. Estimated values below the reporting level are shown. Estimated maximum values for coprostanol and cholesterol obtained from Method 5 (Table 1) are not shown. The analytical method number is provided (in parentheses) at the end of each compound name. An explanation of a boxplot is provided in Figure 3.

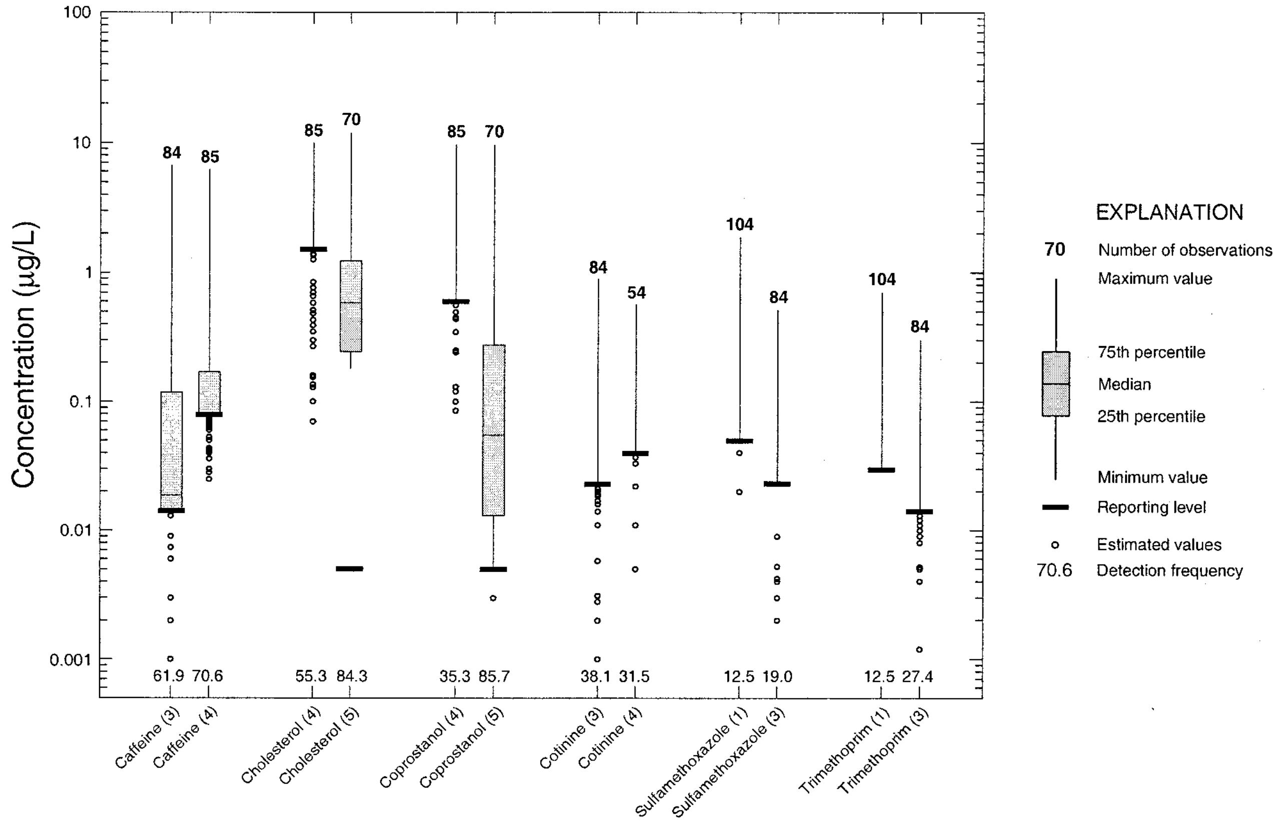

FIGURE 3. Comparison of concentrations of select compounds that were measured using two different methods with significantly different reporting levels. Boxplots show concentration distribution truncated at the reporting level. Estimated values below the reporting level are shown. Estimated maximum values for chloesterol and coprostanol obtained from Method 5 (Table 1) are not shown. The analytical method number is provided (in parentheses) at the end of each compound name.

Results and Discussion

One or more OWCs were found in 80% of the 139 streams sampled for this study. The high overall frequency of detection for the OWCs is likely influenced by the design of this study, which placed a focus on stream sites that were generally considered susceptible to contamination (i.e. downstream of intense urbanization and livestock production). In addition, select OWCs (such as cholesterol) can also be derived from nonanthropogenic sources. Furthermore, some of the OWCs were selected because previous research (*28*) identified them as prevalent in the environment. Thus, the results of this study should not be considered representative of all streams in the United States. A previous investigation of streams downstream of German municipal sewage treatment plants also found a high occurrence of OWCs (*31*).

A large number of OWCs (82 out of 95) were detected at least once during this study (Table 1). Only eight antibiotics and five other prescription drugs were not detected in the samples analyzed (Table 1). Measured concentrations were generally low (median detectable concentrations generally $<1\ \mu g/L$, Table 1), with few compounds exceeding drinking-water guidelines, health advisories, or aquatic-life criteria (Table 1). The concentration of benzo[*a*]pyrene exceeded its maximum contaminant level (MCL) of 0.2 $\mu g/L$ at one site and bis(2-ethylhexyl)phthalate concentrations exceeded its MCL of 6.0 $\mu g/L$ at five sites. In addition, aquatic-life criteria were exceeded for chlorpyrifos (Table 1) at a single site. However, many of the 95 OWCs do not have such guidelines or criteria determined (Table 1). In fact, much is yet to be known about the potential toxicological effects of many of the OWCs under investigation (*1*). For many OWCs, acute effects to aquatic biota appear limited because of the low concentrations generally occurring in the environment (*24, 32−34*). More subtle, chronic effects from low-level environmental exposure to select OWCs appear to be of much greater concern (*1*). Such chronic effects have been documented in the literature (*34−38*). In addition, because antibiotics are specifically designed to reduce bacterial populations in animals, even low-level concentrations in the environment could increase the rate at which pathogenic bacteria develop resistance to these compounds (*15−17, 39*).

The 30 most frequently detected compounds represent a wide variety of uses and origins including residential, industrial, and agricultural sources (Figure 2, Table 1). Only about 5% of the concentrations for these compounds exceeded 1 $\mu g/L$. Over 60% of these higher concentrations were derived from cholesterol and three detergent metabolites (4-nonyphenol, 4-nonylphenol monoethoxylate, and 4-nonylphenol diethoxylate). The frequent detection of cotinine, 1,7-dimethylxanthine, erythromycin-H_2O, and other OWC metabolites demonstrate the importance of obtaining data on degradates to fully understand the fate and transport of OWCs in the hydrologic system. In addition, their presence suggests that to accurately determine the overall effect on human and environmental health (such as pathogen resistance and genotoxicity) from OWCs, their degradates should also be considered. The presence of the parent compound and/or their select metabolites in water resources has previously been documented for OWCs (*40, 41*) as well as other classes of chemicals such as pesticides (*42, 43*).

Many of the most frequently detected compounds (Figure 2) were measured in unfiltered samples using Method 4. Thus, their frequencies of detection may be somewhat higher because concentrations being measured include both the dissolved and particulate phases, whereas concentrations measured by Methods 1−3 include just the dissolved phase. For example, about 90% of the coprostanol discharged from

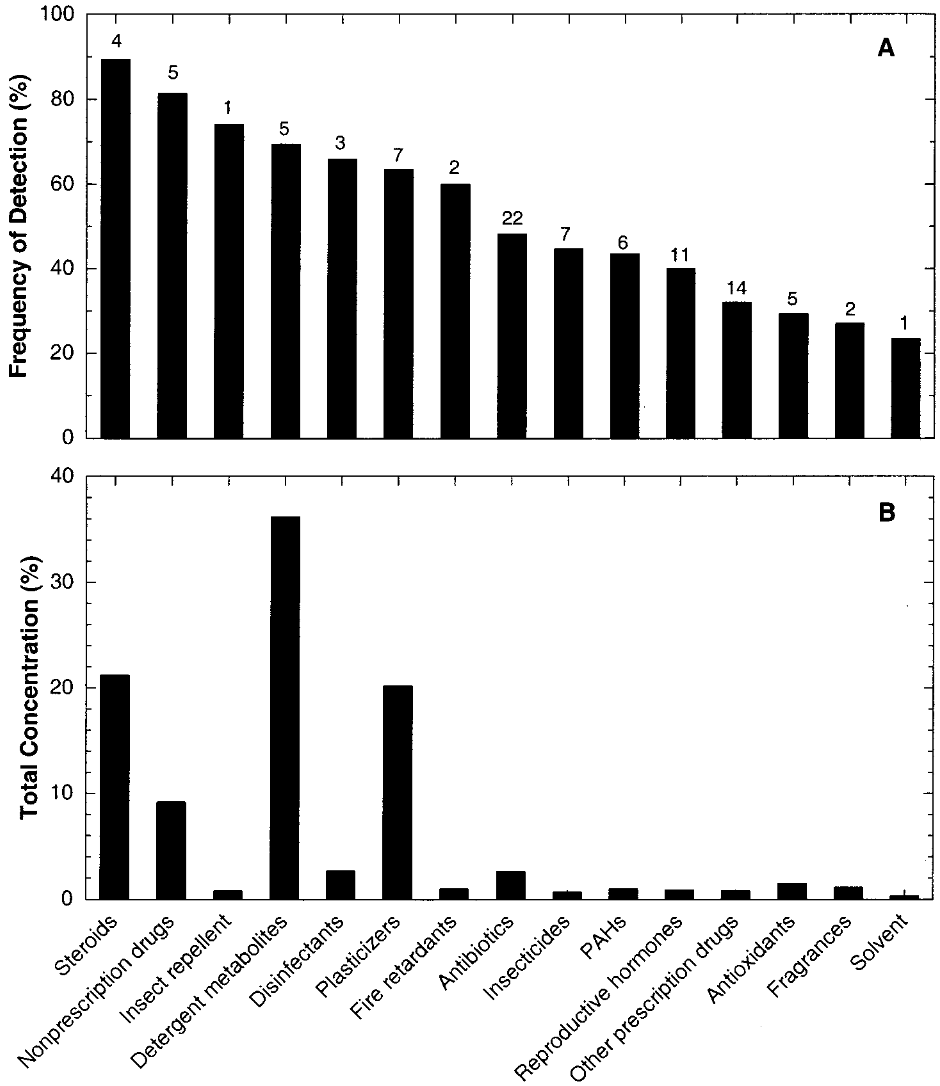

FIGURE 4. Frequency of detection of organic wastewater contaminants by general use category (4A), and percent of total measured concentration of organic wastewater contaminants by general use category (4B). Number of compounds in each category shown above bar.

sewage effluents has been shown to be associated with particulate matter (*44*). Thus, the concentration and frequency of detection for select compounds would likely have been reduced if sample filtration had taken place.

Variations in RL also influence the frequency of OWC detection (Figure 2). For example, the detection of 4-nonylphenol would likely have been much greater if an order of magnitude lower RL (similar to other OWCs) could have been achieved. The effect of RL on frequencies of detection is more clearly demonstrated by comparison of concentrations of select compounds that were measured using multiple analytical methods (Figure 3). As expected, the frequency of detection for a given compound was higher with the lower RL. The only exception being caffeine, where filtration of Method 3 may have reduced caffeine concentrations compared to that of the unfiltered Method 4. Figures 2 and 3 also demonstrate the importance of estimated values (*45*) below the RL. Clearly the numerous estimated concentrations illustrate that the current RLs are not low enough to accurately characterize the total range of OWC concentrations in the stream samples and that the frequencies of detection for this study are conservative.

To obtain a broader view of the results for this study, the 95 OWCs were divided into 15 groups based on their general uses and/or origins. The data show two environmental determinations: frequency of detection (Figure 4A) and percent of total measured concentration (Figure 4B) for each group of compounds. These two views show a vastly different representation of the data. In relation to frequency of detection, there were a number of groups that were frequently detected, with seven of the 15 groups being found in over 60% of the stream samples (Figure 4A). However, three groups (detergent metabolites, plasticizers, and steroids) contributed almost 80% of the total measured concentration (Figure 4B).

For those groups of compounds that have received recent public attention—namely antibiotics, nonprescription drugs, other prescription drugs, and reproductive hormones (*1, 2, 10*)—nonprescription drugs were found with greatest frequency (Figure 4A). Antibiotics, other prescription drugs, and reproductive hormones were found at relatively similar frequencies of detection. The greater frequency of detection for nonprescription drugs may be at least partially derived from their suspected greater annual use compared to these other groups of compounds. When toxicity is considered, measured concentrations of reproductive hormones may have greater implications for health of aquatic organisms than measured concentrations of nonprescription drugs. Previous research has shown that even low-level exposure ($<0.001\,\mu g/L$) to select hormones can illicit deleterious effects in aquatic species (*7, 46, 47*).

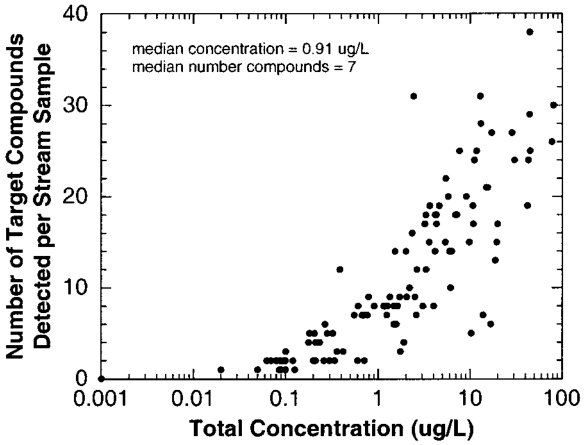

FIGURE 5. Relation between total concentration (summation from all detections) and number of organic wastewater contaminants found per water sample (Spearman's rank correlation coefficient = 0.94, *P* < 0.001).

Mixtures of various OWCs were prevalent during this study, with most (75%) of the streams sampled having more than one OWC identified. In fact, a median of seven OWCs were detected in these streams, with as many as 38 compounds found in a given streamwater sample (Figure 5). Because only a subset of the 95 OWCs were measured at most sites collected during the first year of study, it is suspected that the median number of OWCs for this study is likely underestimated. Although individual compounds were generally detected at low-levels, total concentrations of the OWCs commonly exceeded 1 μg/L (Figure 5). In addition, 33 of the 95 target OWCs are known or suspected to exhibit at least weak hormonal activity with the potential to disrupt normal endocrine function (*4, 7, 8, 10, 12, 22, 36, 37, 48−50*), all of which were detected in at least one stream sample during this study (Table 1). The maximum total concentration of hormonally active compounds was 57.3μg/L. Aquatic species exposed to estrogenic compounds have been shown to alter normal hormonal levels (*7, 48, 51*). Thus, the results of this study suggest that additional research on the toxicity of the target compounds should include not only the individual OWCs but also mixtures of these compounds. The prevalence of multiple compounds in water resources has been previously documented for other contaminants (*52, 53*). In addition, research has shown that select chemical combinations can exhibit additive or synergistic toxic effects (*54−56*), with even compounds of different modes of action having interactive toxicological effects (*57*).

The results of this study document that detectable quantities of OWCs occur in U.S. streams at the national scale. This implies that many such compounds survive wastewater treatment (*1, 6, 58*) and biodegradation (*59*). Future research will be needed to identify those factors (i.e. high use and chemical persistence) that are most important in determining the occurrence and concentration of OWCs in water resources.

Although previous research has also shown that antibiotics (*60*), other prescription drugs (*1, 2, 19, 61−63*), and non-prescription drugs (*1, 40, 62, 64*) can be present in streams, this study is the first to examine their occurrence in a wide variety of hydrogeologic, climatic, and land-use settings across the United States. Much is yet to be learned pertaining to the effects (particularly those chronic in nature) on humans, plants, and animals exposed to low-level concentrations of pharmaceuticals and other OWCs. Furthermore, little is known about the potential interactive effects (synergistic or antagonistic toxicity) that may occur from complex mixtures of these compounds in the environment. Finally, additional research also needs to be focused on those OWCs not frequently detected in this stream sampling. Select OWCs may be hydrophobic and thus may be more likely to be present in stream sediments than in streamwater (*65, 66*). For example, the low frequency of detection for the tetracycline (chlortetracycline, doxycycline, oxytetracycline, tetracycline) and quinolone (ciprofloxacin, enrofloxacin, norfloxacin, sarafloxacin) antibiotics is not unexpected given their apparent affinity for sorption to sediment (*66*). In addition, select OWCs may be degrading into new, more persistent compounds that could be transported into the environment instead of (or in addition to) their associated parent compound.

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