UNITED STATES DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
NATIONAL MARINE FISHERIES SERVICE
Silver Spring, MD 20910

JUL 2 2012

Dr. Steven Bradbury
Director, Office of Pesticide Programs
U.S. Environmental Protection Agency
One Potomac Yard
2777 S. Crystal Drive
Arlington, VA 22202

Dear Dr. Bradbury:

Enclosed is the National Oceanic Atmospheric Administration National Marine Fisheries Service's (NOAA Fisheries) final biological opinion (BiOp), issued under the authority of section 7(a)(2) of the Endangered Species Act (ESA) of 1973 as amended (16 U.S.C. 1536(a)(2)), on the effects of the U.S. Environmental Protection Agency's (EPA) registration of the pesticide thiobencarb on endangered species, threatened species, and critical habitat that has been designated for those species.

After considering the status of the listed resources, the environmental baseline, and the direct, indirect, and cumulative effects of EPA's proposed action on listed salmonid species, NOAA Fisheries concludes that thiobencarb may adversely affect, but is not likely to jeopardize three listed Pacific salmonid Evolutionarily Significant Units (ESUs)/Distinct Population Segments (DPSs) or adversely modify their designated critical habitat. California is the only state within the range of listed Pacific salmonids that has approved the use of thiobencarb, and use is only approved for rice. As a result, this BiOp addresses the exposure and effects of thiobencarb on three listed Pacific salmonid ESUs/DPSs located in California's Central Valley where rice is grown. Reinitiation of this consultation will be necessary if thiobencarb is approved for use by other states with listed Pacific salmonids, if thiobencarb use expands to other parts of California within the range of listed Pacific salmonids, or if thiobencarb is approved for other uses within California.

Molinate, another herbicide, was originally paired with thiobencarb for this consultation as it shares the same mechanism of toxic action and was also used on rice. However, registration of molinate was cancelled and its use has not been allowed since August 2009. Cancellation renders consultation on molinate moot.

This BiOp assesses effects of thiobencarb on listed Pacific salmonids pursuant to the ESA. It does not address EPA's obligation under the Magnuson-Stevens Fishery Conservation and Management Act to consult on effects to essential fish habitat (EFH) for salmonids and other federally-managed species. Please contact Mr. Tom Bigford or Ms. Susan-Marie Stedman in NOAA Fisheries' Office of Habitat Conservation at 301-713-4300 regarding the EFH consultation process.

Thiobencarb's narrow use as documented by DPR's Pesticide Use Reporting System (no other state has such a system) provided a unique opportunity to more precisely assess the spatial and temporal overlap of thiobencarb-containing pesticide products with salmonids and their habitats. Within the range of Pacific salmon, thiobencarb is exclusively registered for use on rice, and rice is grown solely in California's Central Valley. DPR's reporting system encompasses more than 15 years of data on thiobencarb applications including when, where, and how much was applied. We found robust documentation of distinct application seasons. Thiobencarb is almost exclusively applied during the months of April, May, and June. In aggregate, the information allowed us to conduct a more targeted spatial and temporal analysis for this BiOp compared to previous BiOps. For these reasons, this approach is likely a unique case for most national FIFRA pesticide registration consultations.

NOAA Fisheries would like to thank EPA for their engagement with us and with California Department of Pesticides Regulation (DPR) during EPA's draft comment period. California rice farmers have implemented local programs that reduce thiobencarb loading into the surface waters of California's Central Valley. We developed Reasonable and Prudent Measures and Terms and Conditions to minimize incidental take of listed salmonids that reflect California's existing regulatory practices. DPR noted that "these conditions are practicable for applicators and growers and effective in reducing thiobencarb residues in surface waters and salmonid habitats." We believe that collaborating with EPA as well as state and local stakeholders, as occurred during this consultation, benefits listed species and their habitats.

EPA's June 18, 2012, letter on the draft BiOp expressed concerns over referencing state programs on labels or Endangered Species Program (ESPP) Bulletins. As a result, and after interagency meetings with EPA staff and general counsel, we revised the Terms and Conditions by listing the DPR's permit requirements and the PRESCRIBE database use limitations necessary to minimize take of listed Pacific salmonids. The RPM provides EPA with nine months to revise thiobencarb labels. The rationale for this duration ensures that Federal protections will be in place prior to the next rice growing season, i.e., spring 2013. It is our understanding that EPA's process for amending labels or ESPP

Bulletins requires a six-month public comment period. To that end, please let us know if we can advise on the implementation of the RPMs in this BiOp.

In your letter you also indicated that the definition of salmon habitat, i.e., salmon bearing waters, has been inconsistent across pesticide BiOps. The definition of salmon habitat has remained the same across all BiOps, however we designed risk reduction measures, i.e., RPAs and RPMs, to be pesticide-specific reflecting differences in risk. Aquatic habitats articulated in our GIS shape files are how NOAA Fisheries has delineated salmon habitat for all the FIFRA consultations since the first pesticide BiOp was released in 2008. DPR's Endangered Species Program has incorporated our salmon habitat GIS shape files into their PRESCRIBE database, which allows pesticide users to determine use limitations that apply to their geographic location based on the proximity of the application site to endangered species habitat. For your convenience, the shape files are attached to this transmittal letter for use in implementing the RPM for thiobencarb. More detailed maps that delineate when and where each life stage occurs for the three species of listed salmonids in the Central Valley are currently not available and are not necessary to implement the RPMs in this BiOp.

EPA also requested that we provide supporting documentation and decision rules for assigning low/medium/high rankings to toxicity assessment endpoints found on pages 250 and 251. The rationale was described in the draft BiOp on p, 249 Section 8.5.14 titled Evaluation of Data Available for Response Analysis (inserted below) and is also included in the final BiOp. We have followed this protocol for each of the completed BiOps covering 27 active ingredients. This process has allowed us to acquire, review, and apply the best scientific and commercial data available. The protocol was part of a legal challenge to the first BiOp on diazinon, chlorpyrifos, and malathion wherein the court upheld our analyses and findings (currently under appeal).

> We summarize the available toxicity information by assessment endpoint in Table 52. Data and information reviewed for each assessment endpoint was assigned a general qualitative ranking of either "low," "moderate," or "high." To achieve a high confidence ranking, the information stemmed from direct measurements of an assessment endpoint, conducted with a listed species or appropriate surrogate, and was from a well-conducted experiment with stressors of the action or relevant chemical surrogates. A moderate ranking was assigned if one of these three general criteria was absent, and low ranking was assigned if two criteria were absent.

We appreciate efforts made by EPA during this consultation to work diligently with NOAA Fisheries, DPR, and Valent (applicant and registrant for thiobencarb) to discuss RPMs in this BiOp so that EPA can proceed with implementation of measures that protect listed species. As expressed in your comment letter, we also look forward to the National Academy of Sciences/National Research Council review of our respective practices.

If you have questions regarding this BiOp please contact me or Ms. Gina Shultz, Chief of our Endangered Species Act Interagency Consultation Division, at (301) 427-8495.

Sincerely,

Helen M. Golde
Acting Director, Office of Protected Resources

Attachment: GIS shape files for Central Valley ESA listed Pacific salmonids.

cc: Tom Bigford
Susan-Marie Stedman
Will Stelle
Rod Mcinnis
Maria Rae
Joe Dillon

Biological Opinion

National Marine Fisheries Service Endangered Species Act
Section 7 Consultation on Environmental Protection Agency's
Registration of Thiobencarb

Reprinted with permission from the photographer, Chris Haile, chrishailephotography.com, Marysville, CA.

June 30, 2012

Table of Contents

List of Tables

List of Figures

National Marine Fisheries Service

Endangered Species Act Section 7 Consultation

Biological Opinion

Agency:	United States Environmental Protection Agency
Activities Considered:	Authorization of pesticide products (as described by product labels) containing the active ingredient thiobencarb[1], and their formulations in the United States and its affiliated territories.
Consultation Conducted by:	Endangered Species Division of the Office of Protected Resources, National Marine Fisheries Service
Approved by:	
Date:	JUL 2 2012

Section 7(a)(2) of the Endangered Species Act of 1973, as amended (ESA; 16 U.S.C. §1531 *et seq.*) requires each federal agency to insure that any action they authorize, fund, or carry out is not likely to jeopardize the continued existence of any endangered or threatened species or result in the destruction or adverse modification of critical habitat of such species. When a federal agency's action "may affect" a protected species or designated critical habitat, that agency is required to consult formally with the National

[1] EPA paired molinate and thiobencarb for this consultation, However, EPA has cancelled all registrations of molinate and has not allowed its use after August 2009. Cancellation of this active ingredient renders the need for consultation moot.

Marine Fisheries Service (NMFS) or the U.S. Fish and Wildlife Service (USFWS)[2], depending upon the endangered species, threatened species, or designated critical habitat that may be affected by the action (50 CFR §402.14(a)). Federal agencies are exempt from this general requirement if they have concluded, with written concurrence from the U.S. Fish and Wildlife Service, NMFS or both, that an action "may affect but is not likely to adversely affect" endangered species, threatened species or designated critical habitat (50 CFR §420.14(b)).

The United States (U.S.) Environmental Protection Agency (EPA) initiated consultation with NMFS on its proposals to authorize use, pursuant to the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA), 7 U.S.C. 136 *et seq.*, of pesticide products containing the active ingredients (a.i.) thiobencarb and molinate from August 1, 2002. EPA authorization of pesticide uses are categorized as FIFRA sections 3 (new product registrations), 4 (re-registrations and special review), 18 (emergency use), or 24(c) [Special Local Needs (SLN)]. At that time, EPA determined that uses of pesticide products containing molinate and thiobencarb would have "no effect" on most listed salmonids and "may affect, but is not likely to adversely affect" three species (Central Valley spring-run Chinook, Sacramento River winter-run Chinook, and California Central Valley steelhead) and designated critical habitat for these ESUs. NMFS does not concur with EPA's three not likely to adversely affect determinations and conducted formal consultation. This document represents NMFS' biological opinion (Opinion) on the impacts of EPA's authorization of pesticide products containing thiobencarb on the listed ESUs. This is a partial consultation intended to comply with a court order[3] requiring EPA to make a determination on the effect of thiobencarb and 53 other active ingredients on listed Pacific Salmonids. [4] Consultation with NMFS will not be complete

[2] Generally, NMFS conducts consultation for marine and anadromous species, while FWS conducts consultations for freshwater and terrestrial species.
[3] *Northwest Coalition for Alternatives to Pesticides vs. National Marine Fisheries Service*, Civ. NO 07-1791 (W.D. Wa).
[4] Two additional Pacific salmonids species have been listed since the court order. Although the court's order did not address these two species, NMFS would have analyzed the effects of EPA's action to them because they belong to the same taxon and require consideration of the same information. However,

for registration of this a.i. until EPA makes effect determinations on all other species and designated critical habitat under NMFS jurisdiction and consults with NMFS as necessary.

This Opinion is prepared in accordance with section 7(a)(2) of the ESA and implementing regulations at 50 CFR Part 402. However, consistent with the decision in Gifford Pinchot Task Force v. USFWS, 378 F.3d 1059 (Ninth Cir. 2004), we did not apply the regulatory definition of “destruction or adverse modification of critical habitat” at 50 CFR §402.02. Instead, we relied on the statutory provisions of the ESA to complete our analysis of the effects of the action on designated critical habitat.

This Opinion is based on NMFS’ review of the package of information the EPA submitted with its 2003 and 2004 requests for formal consultation on the proposed authorizations of the above a.i.s. It also includes our review of recovery plans for listed Pacific salmonids, past and current research, monitoring reports from prior research, previous Opinions , published and unpublished scientific information on the biology and ecology of threatened and endangered salmonids in the action area, and other sources of information gathered and evaluated during the consultation on the proposed authorization of the a.i. thiobencarb. NMFS also reviewed pesticide labels, available monitoring data and other local, county, and state information, online toxicity databases, incident reports, data generated by pesticide registrants, and exposure models run by NMFS. NMFS also considered information and comments provided by EPA, by the registrants identified as applicants by EPA and information and comments submitted by others during EPA’s public comment process.

---

since both of these species are outside of the area considered by NMFS because of limited use of thiobencarb (see Consultation History, infra), NMFS did not evaluate the effects of thiobencarb application to these two newly listed species.

1. Background

On January 30, 2001, the Washington Toxics Coalition, Northwest Coalition for Alternatives to Pesticides, Pacific Coast Federation of Fishermen's Associations, and Institute for Fisheries Resources filed a lawsuit against EPA in the U.S. District Court for the Western District of Washington, Civ. No. 01-132. This lawsuit alleged that EPA violated section 7(a)(2) of the ESA by failing to consult on the effects of its continuing approval of 54 pesticide a.i.s on 26 listed Pacific salmonid ESUs.

On July 2, 2002, the court ruled that EPA had violated ESA section 7(a)(2) and ordered EPA to initiate interagency consultation and make determinations regarding effects of all 54 a.i.s on listed salmonids by December 2004. *Washington Toxics Coalition v. EPA*, C01-132C (W.D. Wash. 7/2/2002).

On January 22, 2004, the court enjoined application of pesticides within 20 (for ground) and 100 (for aerial) feet (ft) of streams supporting salmon. *Washington Toxics Coalition v. EPA*, C01-132C (W.D. Wash. 1/22/2004). The court imposed several additional restrictions on pesticide use in specific settings.

On November 5, 2007, the Northwest Coalition for Alternatives to Pesticides and others filed a legal complaint in the U.S. District Court for the Western District of Washington, Civ. No. 07-1791, against NMFS for its unreasonable delay in completing the section 7 consultations for EPA's registration of 54 pesticide a.i.s.

On July 30, 2008, NMFS and the plaintiffs entered into a settlement agreement with the Northwest Coalition for Alternatives to Pesticides. NMFS agreed to complete consultation within four years on 37 a.i.s. EPA had concluded that 17 of the 54 a.i.s at issue in the first litigation would not affect any listed salmonid species or any of their designated critical habitat, and so did not initiate consultation on those a.i.s.

On November 18, 2008, NMFS issued its first Opinion for three organophosphates: chlorpyrifos, diazinon, and malathion.

On April 20, 2009, NMFS issued its second Opinion for three carbamates: carbaryl, carbofuran, and methomyl.

On August 31, 2010, NMFS issued its third Opinion. This third consultation evaluated 12 organophosphate insecticides: azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, methamidophos, methidathion, methyl parathion, naled, phorate, and phosmet.

On June 30, 2011, NMFS issued its fourth Opinion. This fourth consultation evaluated four herbicides: 2,4-D, triclopyr BEE, diuron and linuron; and 2 fungicides: captan and chlorothalonil.

On May 31, 2012, NMFS issued its fifth Opinion. This fifth consultation evaluated herbicides: oryzalin, trifluralin, and pendimethalin.

2. Consultation History

On August 1, 2002 EPA transmitted a letter to NMFS' Office of Protected Resources (OPR) requesting section 7(a)(2) consultation for the registration of the two a.i.'s: thiobencarb and molinate providing their effects determinations on 26 ESUs of Pacific salmonids listed at that time (Puget Sound steelhead and Lower Columbia River coho were not evaluated). In the BE, and summarized in Table 1, EPA's Office of Pesticide Programs (OPP) determined that the use of thiobencarb and molinate may affect, but is not likely to adversely affect, three salmonid species: Central Valley (CV) spring-run Chinook, Sacramento River winter-run Chinook ESUs, and CV steelhead DPS. EPA determined that these products would have no effect on the remaining salmonid ESUs and DPSs.

On June 28, 2005, NMFS listed the Lower Columbia River coho salmon ESU as threatened.

On May 22, 2007, NMFS listed the Puget Sound Steelhead Distinct Population Segment (DPS) as threatened.

On December 10-12, 2007, EPA, NMFS and the USFWS met and discussed approaches for moving forward with ESA consultations on pesticide registrations. The agencies agreed to develop methodologies for filling existing data gaps. In the interim, NMFS and USFWS agreed to develop approaches to address these gaps within their Opinions. The agencies identified communication and coordination mechanisms to address technical and policy issues and procedures for conflict resolution.

On February 11, 2008, NMFS listed the Oregon Coast coho salmon evolutionarily significant unit (ESU) as threatened. This ESU was considered in EPA's Biological Evaluation for the a.i.s. in the five previous Opinions referenced in the *Background* section above.

On August 20, 2008, NMFS met with EPA and requested EPA to identify applicants for this and subsequent pesticide consultations.

On September 17, 2008, NMFS requested EPA approval of Confidential Business Information (CBI) clearance for certain staff members in accordance with FIFRA regulations and access to EPA's incident database so NMFS staff could evaluate CBI materials from the applicants and incident reports for the a.i.s under consultation. EPA conveyed to NMFS that no access to the incident database would be authorized and the reports would be sent directly from EPA to NMFS.

On September 23, 2008, NMFS staff received notification of CBI clearance from EPA.

On September 26, 2008, NMFS sent correspondence to EPA regarding the roles of the federal action agency and applicants identified by such agency during formal consultation. NMFS also requested incident reports and label information for this and subsequent pesticide consultations from EPA. The specified timeline for NMFS' receipt of incident reports and label information for the two a.i.s (thiobencarb and molinate) considered in this Opinion was December 1, 2010.
On December 7, 2010, EPA sent to NMFS via email incident monitoring data for the remaining chemicals, including thiobencarb.

On December 8, 2010, NMFS received notification from EPA that registration of molinate and another a.i. (lindane) scheduled to be addressed in a subsequent Opinion were canceled. EPA and NMFS agreed that consultation on label reviews for molinate and lindane would not be necessary. EPA provided NMFS an internal EPA memo dated July 14, 2008 that stated no use of molinate will be permitted after August 31, 2009.

On December 20, 2010, NMFS sent EPA (via email) a request for the label on thiobenarb along with a copy of the Sept. 26, 2008, letter requesting that EPA provide NMFS incident data and labels for the remaining a.i.s by December 1, 2010.

On January 10, 2011, EPA called NMFS to give an update on the status of the labels for the last batch of a.i.s. They also indicated that they would not provide labels for molinate and lindane because their registration had been canceled.

On January 11, 2011, NMFS requested from EPA (via email) final cancelation orders documenting cancellation of all lindane and molinate products registered by EPA.

On January 11, 2011, EPA provided to NMFS final cancelation orders documenting the cancellation of lindane and molinate.

On March 1, 2011, NMFS received preliminary thiobencarb labels from EPA. EPA said that they would send official labels later.

On April 27, 2011, NMFS received the official thiobencarb labels from EPA.

On June 29, 2011, EPA and NMFS received a letter from Valent U.S.A. Corporation (Valent), which requesting they be considered an "applicant" for purposes of EPA's ESA consultation with NMFS regarding the effects of thiobencarb on threatened and endangered Pacific salmonids and their designated critical habitat.

On July 18, 2011, NMFS sent an email to EPA seeking clarification on thiobencarb uses. NMFS noticed in EPA's Registration Eligibility Decision (RED), thiobencarb was listed for other uses (e.g., celery, lettuce, endive).

On July 19, 2011, EPA confirmed (via email) that rice is the only use approved for thiobencarb on the west coast, within the range of the listed Pacific salmonids.

On July 19, 2011, NMFS met with applicant Valent and EPA to talk about the thiobencarb consultation and to exchange general information to assist with the consultation process. Valent, EPA and NMFS each provided presentations at this meeting. NMFS confirmed with Valent that thiobencarb is only approved for rice on the west coast, and confirmed that they are the only registrant for thiobencarb. Valent provided NMFS thiobencarb monitoring data and suggested it reflected current usage, stewardship programs, and permit conditions.

During this meeting, NMFS and EPA agreed that the consultation on thiobencarb would be based not on all federally authorized uses, as agreed upon in the December 2007 meeting and as occurred for NMFS' four previous pesticide biological opinions, but on a subset of authorized use, that of rice growing in California. This decision was based on the fact that, unlike for other pesticide active ingredients, there is currently no authorized use of thiobencarb in Washington, Oregon or Idaho. EPA and NMFS agreed that EPA would reinitiate consultation in the unlikely event that Washington, Oregon or Idaho authorized use of thiobencarb or that rice growing expanded beyond the identified counties in California. NMFS' evaluation will consider the impacts of application of thiobencarb up to the maximum labeled amount.

On August 25, 2011, NMFS sought to confirm that thiobencarb is only permitted for use on rice and that there are no thiobencarb products labeled for use in Washington or Oregon. This email was sent because monitoring data showed recent detections of thiobencarb in both states.

On September 15, 2011, EPA confirmed that there are no uses of thiobencarb in Washington or Oregon. EPA suggested thiobencarb detections in the monitoring data were either from misuse, atmospheric transport, or errors in the monitoring data.

On September 29, 2011, Valent sent NMFS (via email) a list of studies they had submitted to EPA. Valent indicated that they did not have information to characterize the potential for off-target drift of aerially applied granular products of thiobencarb. They stated that the drift potential for aerially-applied granular formulations is "significantly less" than for liquid formulations, however, they could provide no comparisons giving numerical values.

On September 30, 2011, NMFS requested (via email) EPA provide copies of the 15 studies Valent referenced in their September 29 email to NMFS.

On December 7, 2011, NMFS received (via email) one of the studies (MRID 43404005) requested on September 30, 2011.

On December 9, 20ll, NMFS received (via email) eight (MRID 46091402, MRID 46091401, MRID 42460401, MRID 43252001, MRID 42257801, MRID 45695101, MRID 44628601, MRID 44628602) additional studies NMFS requested on Sept. 30, 2011.

On December 9, 2011, NMFS sent a request (via email) to EPA for copies of the remaining six studies referenced in Valent's September 29, 2011 email.

On January 5, 2012, NMFS sent an email to EPA to ask about EPA's assessment of risks of thiobencarb to the California red-legged frog and Delta smelt.

On January 6, 2012, in response to NMFS Jan. 5, 2012, email, EPA explained that in the course their routine data quality review for risk assessment, the half-lives were recalculated from the raw data provided in the studies, using a non-linear first-order equation. EPA used this recalculation of half-lives in the effect determination, rather than the face value reported in the studies.

On January 9, 2012, NMFS sent EPA follow up questions and asked for additional information on the method EPA used for the recalculations. NMFS also requested a copy of the Louisiana study (MRID 434040-04) referenced in EPA's Jan. 6, 2012 email. EPA provided a response to NMFS' questions. NMFS sent EPA an email seeking clarifications to responses provided by EPA.

On January 10, 2012, NMFS received from EPA (via email) a copy of the Louisiana study referenced above (MRID 42003404).

On January 30, 2012, in response to an earlier inquiry by NMFS, the California Rice Commission (CRC) confirmed that representatives would be available to meet with NMFS in Sacramento on February 28, 2012 to discuss the CRC role and to introduce NMFS to rice growers in the region.

On February 7, 2012, NMFS received from EPA (via email) the remaining six studies (MRID 42680401, MRID 43252001 [already received on Dec. 9, 2011], MRID 43404001, MRID 43404004, MRID 42384701, MRID 43404003) requested by NMFS September 30, 2011 and December 9, 2011.

On February 13, 2012, NMFS requested EPA to send MRID 25179 (Lauck, JE. 1979) Final Report of Field Study: Ortho Bolero 8 EC-- Rice, 1979. (Unpublished study received Dec 11, 1979 under 239- 2450; submitted by Chevron Chemical Co., Richmond, Calif.; CDL:241490-F).

On February 14, 2012, NMFS requested (via email) another study from EPA (MRID 79986) cited in EPA's California Red-legged frog assessment.

On February 15, 2012, EPA informed NMFS (via email) that the studies requested on Feb 13 and Feb 14 were old studies and that it may take a week to get an electronic copy. NMFS responded by email back to EPA that these studies were important as they represent some of the higher concentrations and it was critical NMFS evaluate the study design.

On February 21, 2012, the court in *Northwest Coalition for Alternatives to Pesticides v. NMFS*, Civ. No. 07-1791 (W.D. WA) granted NMFS' request, agreed to by the plaintiffs, to extend the schedule for completion of consultation established in the July 30, 2008 settlement agreement. The deadline for this Biological Opinion was extended until June 30, 2012.

On February 21, 2012, NMFS notified EPA and Valent that NMFS was granted an extension to June 30, 2012, for the thiobencarb final Biological Opinion.

On February 23, 2012, EPA sent NMFS an email confirming that Bolero Ultramax (a thiobencarb formulation) is the same as Bolero 15G.

On February 27, 2012, NMFS staff working on this Opinion flew to Sacramento to gather information about rice growing, water monitoring, rice regulation and over-sight, and Sacramento River salmonids. NMFS gave a webinar presentation to California Department of Pesticide Regulation staff, Valent representatives, and others on the process used to develop NMFS Opinions on EPA FIFRA actions.

On February 28, 2012, NMFS met with a representative of the California Rice Commission (CRC). The CRC took NMFS staff on a tour of the rice areas including the Yolo Bypass, four of the five water monitoring stations where we observed how water samples are taken, and the Sacramento weir bypass. The CRC also arranged a meeting with a rice farmer who dry seeded rice and used the liquid formulation of thiobencarb.

On February 29, 2012, EPA sent to NMFS (via email) the two studies requested in NMFS February 13,14, and 15, 2012, email.

On February 29, 2012, NMFS met again with the CRC who arranged to meet with a second rice farmer that wet seeded rice and used the granular formulation of thiobencarb. Later in the morning, NMFS met with California Department of Water Resources (CDWR) to gather information on salmon use on the Yolo Bypass floodplain. In the afternoon, NMFS met with California Department of Pesticide Regulation (CDPR) to gather information on rice pesticide use enforcement issues and rice pesticide modeling. NMFS also met with CDPR Endangered Species Division to learn more about California's County Bulletins.

On March 1, 2012, NMFS met with U.S. Geological Survey (USGS) staff to gather information on water monitoring, results from recent studies related to thiobencarb use, and to discuss future data needs. Later in the day, NMFS met with CDWR and EPA Region 9 staff to discuss a new pesticide loading model developed for the Sacramento and San Joaquin rivers and Bay Delta.

On March 2, 2012, NMFS met again with USGS and California Department of Fish and Game (CDFG) staff to discuss salmonids' temporal and spatial use of the Sacramento River.

On March 14, 2012, Valent sent NMFS (via email) examples of state, regional, and county permit requirements for thiobencarb use in California. Seven documents were included in this email.

- Recommended Permit Conditions for Rice Pesticides (Rice Pesticide Program), obtained from Sutter County but written by the California Department of Pesticide Regulation and the Central Valley Regional Water Quality Control Board for all rice thiobencarb use in California
- Sutter County Rice Permit Conditions for Water Holding Requirements (Rice Water Holding Conditions) 2011
- Butte County 2011 Rice Permit Conditions for Bolero/Abolish
- Yolo County Rice Pesticide Drift Control Requirements
- Yolo County Rice Water Seepage Management
- Yolo County 2008 Rice Pesticide Permit Conditions Cover Letter
- Yolo County 2008 Rice Pesticide Permit Conditions

On March 15, 2012, NMFS sent an email to Valent and EPA seeking clarification on the hold times for Abolish 8 EC liquid formulation of thiobencarb. NMFS also requested updated labels for these products. Valent responded (via email) by providing the California SLN label for Abolish 8 EC. Valent confirmed the water hold time.

On March 15, 2012, Valent provided NMFS (via email) relative use proportions of active ingredient thiobencarb used in California as Abolish 8 EC (liquid) and Bolero UltraMax (granular). Valent informed NMFS the ratio has been running approximately 30 percent thiobencarb as Abolish 8 EC and 70 percent thiobencarb as Bolero UltraMax.

On April 11, 2012, NMFS sent (via email) a draft portion of the *Environmental Baseline* section of the Opinion to be CRC to review for accuracy. The CRC reviewed the section and returned it later this same day with suggested edits and additions. The CRC also provided additional information on rice water seepage management.

On May 9, 2012, NMFS sent (via email) a draft of the Reasonable and Prudent Measures and associated Appendix to EPA for review prior to a joint meeting to discuss on May 10, 2012.

On May 10, 2012, NMFS met with EPA to discuss the draft Reasonable and Prudent Measures and corresponding Appendix.

On May 11, 2012, NMFS, provided a draft of this Opinion to EPA.

On May 22, 2012, NMFS met with EPA and Valent to discuss the draft Opinion. At this meeting, NMFS describe the process used to reach our conclusions. Valent then asked some questions and provided several constructive comments.

On June 1, 2012, EPA and NMFS received via email formal comments from Valent on the draft Opinion released on May 11, 2012. The comments reflected those made during the May 22, 2012 meeting.

On June 6, 2012, NMFS received via fax CBI information from EPA on Valent's thiobencarb formulations.

On June 11, 2012, NMFS retrieved from EPA's Docket (EPA-HQ-OPP-2008-0654) comments on the draft Opinion from the California Rice Commission, and Valent comments previously received.

On June 13, 2012, NMFS received via fax additional CBI information from EPA on Valent's thiobencarb formulations.

On June 14, 2012, EPA and NMFS received comments from California's Department of Fish and Game on the draft Opinion released May 11, 2012.

On June 15, 2012, EPA and NMFS received comments from California's Department of Pesticide Regulation on the draft Opinion released May 11, 2012.

On June 18, 2012, NMFS received via email partial comments from EPA on the draft Opinion released on May 11, 2012. The comments were on the draft RPMs. EPA and NMFS had a follow up phone call to discuss EPA's issues with the draft RPM number one and related terms and condions. EPA and NMFS agreed to discuss the issue further.

On June 21, 2012, NMFS and EPA met again to discuss again the draft RPMs. NMFS agreed to go through the CDPR permit requirements and incorporate into the RPM and the terms and conditions the specific requirements from the CDPR permit requirements and recommended best management practices intended to reduce thiobencarb loading to salmon waters.

On June 22, 2012, NMFS provided EPA via email an amended RPM based on discussion on June 18 and June 21.

On June 25, 2012, EPA sent an email to inform NMFS that they intended to share the draft RPMs with CDPR.

On June 25, 2012, EPA, NMFS, and CDPR met via teleconference to discuss the RPMs. EPA had some clarifying questions of CDPR about the CDPR regulatory program and their enforcement.

On June 27, 2012, EPA and NMFS received a letter from Valent via email regarding draft RPMs.

2.1. Species Addressed in the BE

When an action agency concludes its action will not affect any listed species or critical habitat, section 7 consultation is not necessary (USFWS, & NMFS 1998). If an action agency concludes its action may affect but is not likely to adversely affect (NLAA) listed species, it seeks concurrence from NMFS or USFWS on the conclusion. When an action may adversely affect listed species or designated critical habitat, NMFS or USFWS conducts a formal consultation to determine whether that action is likely to jeopardize listed species or destroy or adversely modify critical habitat and issues a biological opinion describing their determinations.

EPA's BE considered the effects of pesticides containing thiobencarb and molinate to 26 species of listed Pacific salmonids and their designated critical habitat (EPA undated Memorandum – Effects Determinations for Molinate and Thiobencarb for Pacific Anadromous Salmonids; August 1, 2002). Three listed species, Central Valley spring-run Chinook, Sacramento River winter-run Chinook, and Central Valley steelhead were determined to be may affect, but not likely to adversely affect. All other ESUs and DPSs were determined to be no effect. Two species, Lower Columbia River coho and Puget Sound steelhead, were not considered in the BE.

NMFS did not concur with any of EPA's "NLAA" determinations for thiobencarb and determined that thiobencarb may adversely affect some ESUs or DPSs. Generally, once NMFS enters into formal consultation it considers impacts to all species and critical habitat.. In this Opinion, NMFS analyzed the impacts to all ESUs/DPSs of Pacific salmonids present in the action area, including those salmonid species identified by EPA as being unaffected, and including the two species of salmonids listed after EPA provided its BEs to NMFS. EPA's effect determinations are summarized in Table 1.

Table 1. EPA's effects determinations for thiobencarb and molinate on Pacific salmon.

Species	ESU or DPS	Herbicide	
		Thiobencarb	Molinate
Chinook	Central Valley spring-run Chinook	NLAA	NLAA
	Sacramento River winter-run Chinook	NLAA	NLAA
Steelhead	Central Valley	NLAA	NLAA
	Puget Sound	Not evaluated	Not Evaluated
Coho	Lower Columbia River	Not evaluated	Not evaluated
All other listed species ESUs and DPSs		No effect	No effect

3. Description of the Proposed Action

The proposed action encompasses EPA's registration of the uses (as described by product labels) of all pesticides containing thiobencarb. The purpose of the proposed action is to provide tools for pest control that do not cause unreasonable adverse effects to the environment throughout the U.S. and its affiliated territories. Pursuant to FIFRA, before a pesticide product may be sold or distributed in the U.S. it must be exempted or registered with a label identifying approved uses by EPA's Office of Pesticide Programs (OPP). Once registered, a pesticide may not legally be used unless the use is consistent with directions on its approved label (http:www.epa.gov/pesticides/regulating/registering/index.htm). EPA authorization of pesticide uses are categorized as FIFRA sections 3 (new product registrations), 4 (re-registrations and special review), 18 (emergency use), or 24(c) Special Local Needs (SLN).

On April 7, 2004, EPA published the final cancelation notice for molinate (69 FR 18368). Sale of end-use products containing molinate was prohibited after June 30, 2008, and all uses of those products were prohibited following August 31, 2009. As there are no remaining legal uses of molinate, there is no need for consultation and we will not provide an analysis of its effects to listed salmonids.

EPA's pesticide registration process involves an examination of the ingredients of a pesticide, the site or crop on which it will be used, the amount, frequency and timing of its use, and its storage and disposal practices. Pesticide products may include a.i.s and other ingredients, such as adjuvants, and surfactants (described in greater detail below). The EPA evaluates the pesticide to ensure that it will not have unreasonable adverse effects on humans, the environment, and non-target species. An unreasonable adverse effect on the environment is defined in FIFRA as, "(1) any unreasonable risk to man or the environment, taking into account the economic, social, and environmental costs and benefits of the use of the pesticide, or (2) a human dietary risk from residues that result from a use of a pesticide in or on any food inconsistent with the standard under section 408 of the FFDCA (21 U.S.C. §346a)" 7 U.S.C. 136(b).

After registering a pesticide, EPA retains discretionary involvement and control over such registration. EPA must periodically review the registration to ensure compliance with FIFRA and other federal laws (7 U.S.C. §136d). A pesticide registration can be canceled whenever "a pesticide or its labeling or other material does not comply with the provisions of FIFRA or, when used in accordance with widespread and commonly recognized practice, generally causes unreasonable adverse effects on the environment."

On December 12, 2007, EPA, NMFS, and FWS agreed that the federal action for EPA's FIFRA registration actions will be defined as the "authorization for use or uses described in labeling of a pesticide product containing a particular pesticide ingredient." In order to ensure that EPA's action will not jeopardize listed species or destroy or adversely modify critical habitat, NMFS' analysis necessarily encompasses the impacts to listed Pacific salmonid ESUs/DPSs of all uses authorized by EPA, regardless of whether those uses have historically occurred.

Pesticide Labels. For this consultation, EPA's proposed action encompasses all approved product labels containing the a.i. thiobencarb; its degradates, metabolites, and formulations, including other ingredients within the formulations; adjuvants; and tank mixtures. These activities comprise the stressors of the action (Figure 1). The BE assessing the impacts to listed salmon and steelhead indicates that the subject a.i. is labeled for use on rice only (EPA 2002c). EPA provided no labels indicating any other use in California, Washington, Oregon or Idaho. This partial consultation differs substantially from other consultations NMFS has completed or will complete on pesticide registrations because thiobencarb is labeled for use on one crop only, and that crop is grown in identified counties in one area in California. It is unlikely rice production will expand beyond the current area because rice production needs certain environmental factors that are not widely found on the west coast. Three of the four states, Washington, Oregon and Idaho, currently prohibit the use of thiobencarb, and there is no known rice production in those states. In addition, California has imposed restrictions on thiobencarb use and conducts substantial monitoring to ensure that those restrictions are followed and effective. NMFS was therefore able to focus this consultation on the impacts of thiobencarb in the area where rice is grown, and evaluate the effects to three ESU/DPS present in that area and their designated critical habitat.[5] Because there is no authorized use elsewhere in the four western states, none of the other of the 28 ESU/DPS of Pacific salmon or their critical habitat will be exposed to the effects of the action. NMFS will not consider these 25 ESU/DPS or any critical habitat designated for them further in this opinion.

[5] EPA does not control where rice is grown or state authorization or restrictions on thiobencarb use. However, in the July 19, 2011 meeting with NMFS and the applicant Valent, EPA has agreed to reinitiation triggers based on changes in any state's authorization of thiobencarb for use on rice or expansion of rice production beyond its current location in California that would affect additional listed salmon and their critical habitat. Changes in the federal FIFRA labels that would authorize use in these four states on crops or use sites other than rice growing would also trigger reinitiation.

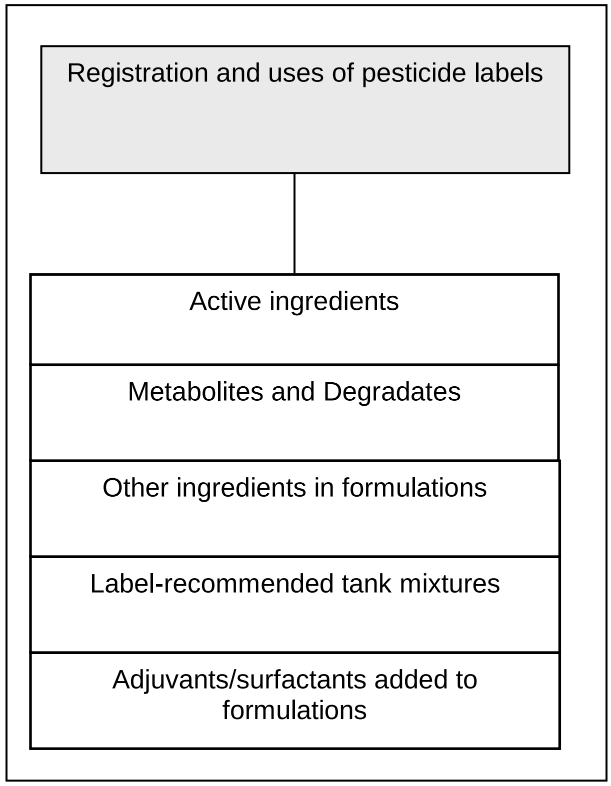


Figure 1. Stressors of the Action

Active and Other Ingredients. Thiobencarb is the a.i. that kills or otherwise negatively affects targeted organisms (listed on the label). However, pesticide products that contain this a.i. also contain inert ingredients. Inert ingredients are ingredients that EPA defines as not "pesticidally" active. EPA also refers to inert ingredients as "other ingredients." The specific identification of the compounds that make up the inert fraction of a pesticide is not required on the label. However, this does not necessarily imply that inert ingredients are non-toxic, non-flammable, or otherwise non-reactive. EPA authorizes the use of chemical adjuvants to make pesticide products more efficacious. An adjuvant aides the operation or improves the effectiveness of a pesticide. Examples include wetting agents, spreaders, emulsifiers, dispersing agents, solvents, solubilizers, stickers, and surfactants. A surfactant is a substance that reduces surface tension of a system, allowing oil-based and water-based substances to mix more readily. A common group of non-ionic surfactants is the alkylphenol polyethoxylates (APEs), which may be used in pesticides or pesticide tank mixes, and also are used in many common household products. Nonylphenol (NP), one of the APEs, has been linked to endocrine-disrupting effects in aquatic animals.

Formulations. Pesticide products come in a variety of solid and liquid formulations. Examples of formulation types include dusts, dry flowables, emulsifiable concentrates, granulars, solutions, soluble powders, ultra-low volume concentrates, water-soluble bags, powders, and baits. The formulation type can have implications for product efficacy and exposure to humans and other non-target organisms.

Tank Mix. A tank mix is a combination by the user of two or more pesticide formulations as well as any adjuvants or surfactants added to the same tank prior to application. Typically, formulations are combined to reduce the number of spray operations or to obtain better pest control than if the individual products were applied alone. The compatibility section of a label may advise on tank mixes known to be incompatible or provide specific mixing instructions for use with compatible mixes. Labels may also recommend specific tank mixes. Pursuant to FIFRA, EPA has the discretion to prohibit tank mixtures. Applicators are permitted to include any combination of pesticides in a tank mix as long as each pesticide in the mixture is permitted for use on the application site and the label does not explicitly prohibit the mix.

Pesticide Registration. The Pesticide Registration Improvement Act (PRIA) of 2003 became effective on March 23, 2004. The PRIA directed EPA to complete REDs for pesticides with food uses/tolerances by August 3, 2006, and to complete REDs for all remaining non-food pesticides by October 3, 2008. The goal of the reregistration program is to mitigate risks associated with the use of older pesticides while preserving their benefits. Pesticides that meet today's scientific and regulatory standards may be declared "eligible" for reregistration. The eligibility for continued registration may be contingent on label modifications to mitigate risk and can include phase-out and cancellation of uses and pesticide products. The terms of EPA's regulatory decisions are summarized in the RED (EPA 1997b).

Registrants can submit applications for the registration of new products and new uses following reregistration of an active ingredient. Several types of products are registered, including the pure (or nearly pure) active ingredient, often referred to as technical grade active ingredient (TGAI), technical, or technical product. This is generally used in manufacturing and testing, and not applied directly to crops or other use sites. Products that are applied to crops, either on their own or in conjunction with other products or surfactants in tank mixes are called end-use products (EUPs). Sometimes companies will also register the pesticide in a manufacturing formulation, intended for sale to another registrant who then includes it into a separately registered EUP. Manufacturing formulations are not intended for application directly to use sites. EPA may also cancel product registrations. With exceptions[6], EPA allows the use of canceled products, and products that do not reflect RED label mitigation requirements, until those products have been exhausted. Labels that reflect current EPA mitigation requirements are referred to as "active labels." Products that do not reflect current label requirements are referred to as "existing stocks." EPA's action includes all authorizations for use of pesticide products including use of existing stocks, and active labels, of products containing thiobencarb for the duration of the proposed action.

Duration of the Proposed Action. EPA's goal for reassessing currently registered pesticide a.i.s is every 15 years. Given EPA's timeframe for pesticide registration reviews, NMFS' evaluation of the affects of the proposed action on listed Pacific salmonids is also 15 years.

Interrelated and Interdependent Activities. No interrelated and interdependent activities are associated with the proposed action.

---

[6] For example, EPA placed an end date on authorized use of molinate.

Registration Information of Pesticide a.i.s under Consultation. As discussed above, the proposed action encompasses EPA's registration of the uses (as described by product labels) of all pesticides containing thiobencarb. EPA provided copies of all active product labels for this active ingredient. The following descriptions represent information acquired from review of these labels as well as information conveyed in the EPA BEs, REDs, and other documents.

3.1.1. Thiobencarb

Thiobencarb is a thiocarbamate. Thiobencarb is the common name for S-((4-chlorophenyl)methyl)diethylcarbamothioate (CAS 28249-77-6). Thiobencarb is a systemic, preemergence herbicide that acts by inhibiting shoots of emerging seedlings. It is used to control grasses, sedge and broadleaf weeds in food crops such as rice (nationwide rice represents 95% of use), lettuce, celery, and endive. Thiobencarb was first registered for use on rice in 1982. In 1991, thiobencarb was issued regional tolerances for use on celery, endives, and lettuce in the State of Florida; however, thiobencarb is not authorized for use on these crops in Washington, Oregon, Idaho or California. Currently there are eight products containing thiobencarb registered for use under Section 3 of FIFRA. They consist of one technical (manufacturing use) product containing 97% a.i., emulsifiable concentrate end-use products containing 84% a.i., and granular end-use products containing 10 to 15% a.i. RiceBeaux®, which contains a mixture of thiobencarb and propanil is not registered for use in California (EPA Reg. No. 71085-30). Other trade names for thiobencarb products currently used in the U.S. are Bolero and Abolish. There is one Special Local Needs (SLN) product registered for use in California (EPA Reg. No. 59639-79, EPA SLN CA-930003) under Section 24(c) of FIFRA (EPA 1997a). This SLN product is aerial spray applied for water seeded rice, and is to be applied to non-flooded fields only. While thiobencarb is registered for use nationwide, California is the only state within the range of listed Pacific salmon that has approved thiobencarb for use (rice only). As noted above, Oregon, Idaho, and Washington have not approved the use of thiobencarb.

3.1.1.1. *Usage Information*

Reported annual use of thiobencarb in California declined from over one million lbs in 2000 to approximately 300 thousand lbs per year during the most recent surveys, 2006-2009 (cite most recent report CDPR 2009). Use of thiobencarb products within the state is reflective of the distribution of rice, the only registered use site for thiobencarb. Roughly 95 percent of the state's rice acreage is in the Sacramento Valley, with most of the remaining rice grown in northern and central San Joaquin Valley (CDPR 2009 report). Although rice acreage and thiobencarb use has remained relatively stable in California in recent years, differences have been observed among the rice producing counties in both the amount of thiobencarb used, and the selection in thiobencarb products (CDPR 2010a). As reported by California Department of Pesticide Regulation, from 1980 to 2010, thiobencarb has been typically applied at or near the maximum application rate of 4 lbs/A (Appendix 3).

3.1.1.2. *Use Sites Authorized*

Agricultural. Use of thiobencarb within the distribution of listed Pacific salmonids is limited to rice fields. Among the four states where listed salmon and steelhead occur, California is the only state that has approved the use of thiobencarb. According to the registrants, there are no foreseeable plans to extend the use of thiobencarb in the Pacific Northwest States (July 19, 2011 meeting between EPA, NMFS, and the applicant).

Developed. Thiobencarb is not registered for residential, commercial, or industrial uses.

Forestry. Thiobencarb is not registered for forestry uses.

Aquatic. Thiobencarb is not registered for aquatic uses, other than rice production in ponded fields.

Other. Thiobencarb is not registered for other uses such as rights-of-way.

Registered Formulation Types. Thiobencarb enduse products are typically formulated as liquid or granular. None of the products registered for use in California have more than one a.i. in the formulation.

3.1.1.3. *Methods and Rates of Application.*

Methods. Thiobencarb may be applied via ground and aerial applications. Application methods include broadcast spray, granular applicator, high pressure sprayer, and dilute high volume spray. The timing of application of recommended application is either late preemergence (e.g., 5 to 9 days after planting of the rice), or early post emergence of weeds (e.g., rice leaf stage 2-3). Several products recommend use in combination with propanil for the control of specific weeds.

Application Rates. Active labels within the action area allow a maximum single and seasonal application rate of up to 4 lbs thiobencarb/A to rice (Table 2). Current labels allow for a single application, or they do not specify a limit on the number of applications.

Table 2. Thiobencarb use patterns in the action area.

Use(s)	Use Site	Land Use category	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
Rice	Crop	Agriculture	4	NS	4	NS	Aerial and ground broadcast	59639-79[1]
Rice	Crop	Agriculture	3.5	NS	4	NS	Aerial and ground broadcast	59639-112

Notes:

[1] EPA label for Abolish 8 EC formulation specifies not to release flood water within 14 days of application. Additionally, this product can be applied to non-flooded fields only.

[2] EPA label for Bolero Ultramax granular flake formulation specifies not to release flood water within 30 days of application.

NS = not specified

3.1.1.4. *Metabolites and Degradates.*

Thiobencarb has two major degradates (defined by EPA as those representing 10% or more of the applied radiation of the parent test substance) identified. These are 4-chlorobenzoic acid (56% of applied radiation at 30 days) and 4-chlorobenzaldehyde (29.4% at 14 days), both in a sensitized aquatic photodegradation study. Both of these degradates are expected to be soluble and mobile in water. Also, because of their simple molecular structure, both should be subject to further degradation by metabolism (EPA 2009b).

4. Action Area

The action area is defined as all areas to be affected directly or indirectly by the federal action and not merely the immediate area involved in the action (50 CFR §402.02). Given EPA's nationwide authorization of these pesticides, the action area would encompass the entire U.S. and its territories. This geographic area would include all listed species and designated critical habitat under NMFS jurisdiction.

In this instance, as a result of the 2002 order in Washington Toxics Coalition v. EPA, EPA initiated consultation on its authorization of 37 pesticide a.i.s and their effects on listed Pacific salmonids under NMFS' jurisdiction and associated designated critical habitat in the states of California, Idaho, Oregon, and Washington. However, in this consultation with EPA, NMFS is determining the effect of EPA's proposed reauthorization of thiobencarb. Thiobencarb is only authorized for use on rice. In this case, rice is only grown in California's Central Valley (CV) and nowhere else where Pacific salmon are listed. Within the CV there are three listed Pacific anadromous salmonids: the CV steelhead, Sacramento River winter-run Chinook salmon, and CV spring-run Chinook salmon. Consequently, for this consultation, the action area consists of the entire range of these three listed salmonids and their designated critical habitat in California's Central Valley (Figure 2).[7] The action area encompasses the following aquatic components of the central valley: all freshwater, estuarine, marsh, swamps, and nearshore marine surface waters.

NMFS' analysis focuses on the effects of EPA's action on listed Pacific salmonids in the California Central Valley. It includes the effects of thiobencarb on CV steelhead, Sacramento River winter-run Chinook and CV spring-run Chinook salmon and their designated critical habitats.

[7] As noted above, EPA will reinitiate consultation if Washington, Oregon or Idaho authorize use of thiobencarb or if the rice growing in California expands beyond its current area. If this occurs, the action area for a reinitiated consultation would most likely be larger.

EPA's consultation with NMFS remains incomplete until it analyzes the effects of its authorization of pesticide product labels with this compound for all remaining threatened and endangered species and all designated critical habitat under NMFS' jurisdiction. EPA must ensure its action does not jeopardize the continued existence or result in the destruction or adverse modification of critical habitat for other listed species and

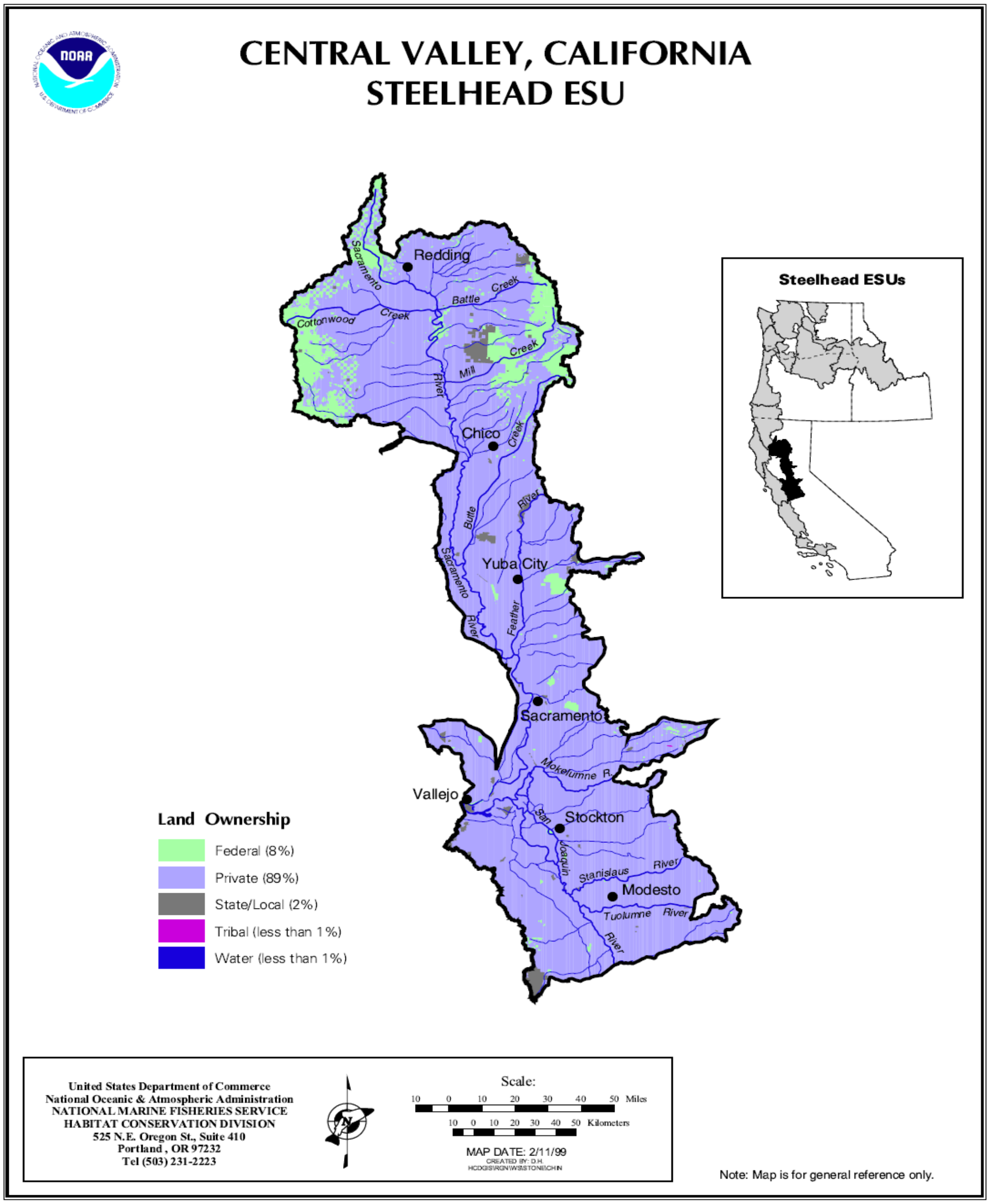


Figure 2. The Central Valley steelhead ESU encompasses the Sacramento River winter-run Chinook, and the Central Valley spring-run Chinook ESUs. Rice is only grown within these boundaries, consequently, the CV steelhead ESU comprises the area of interest for this consultation.

designated critical habitat under NMFS' jurisdiction throughout the U.S. and its territories.

5. Approach to the Assessment

NMFS uses a series of steps to assess the effects of federal actions on endangered and threatened species and designated critical habitat. The first step of our analysis identifies those physical, chemical, or biotic aspects of proposed actions that are likely to have individual, interactive, or cumulative direct and indirect effects on the environment (we use the term "potential stressors" for these aspects of an action). As part of this step, we identify the spatial extent of any potential stressors and recognize that the spatial extent of those stressors may change with time. The spatial extent of these stressors is the "action area" for a consultation.

The second step of our analyses identifies the listed resources (endangered and threatened species and designated critical habitat) that are likely to occur in the same space and at the same time as these potential stressors. If we conclude that such co-occurrence is likely, we then try to estimate the nature of co-occurrence (these represent our *Exposure Analyses*). In the exposure analysis, we try to identify life stages and life histories of the individuals that are likely to be exposed to an action's effects and the populations or subpopulations those individuals represent. Spatial analyses are used to overlay each species range with land types or crop types that pesticides are used on including agriculture, urban/residential, forested, and right of ways, to evaluate co-occurrence of pesticides and salmonids.

Once we identify which listed resources are likely to be exposed to potential stressors associated with an action and the nature of that exposure, in the third step of our analysis we examine the scientific and commercial data available to determine whether and how those listed resources are likely to respond given their exposure (these represent our *Response Analyses*). We integrate the exposure and response analyses within the *Risk*

Characterization section to assess the risk to listed individuals and their habitat from the stressors of the action.

In the *Risk Characterization Section*, we also determine whether population level effects are anticipated (these analyses are conducted within the risk characterization phase). NMFS' analysis is ultimately a qualitative assessment that draws on a variety of quantitative and qualitative tools and measures to address risk to listed resources.

In the final steps of our analyses, we establish the risks posed to listed species and to designated critical habitat. This part of the analysis is found within the *Integration and Synthesis section*.

Our jeopardy determinations for listed species must be based on an action's effects on the continued existence of threatened or endangered species as those "species" have been listed, which can include true biological species, subspecies, or distinct population segments of vertebrate species. Because the continued existence of listed species depends on the fate of the populations that comprise them, the viability (that is, the probability of extinction or probability of persistence) of listed species depends on the viability of the populations that comprise the species. Similarly, the continued existence of populations are determined by the fate of the individuals that comprise them; populations grow or decline as the individuals that comprise the population live, die, grow, mature, migrate, and reproduce (or fail to do so).

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

We evaluate risks to listed individuals by determining whether an individual's "fitness" defined as changes in an individual's growth, survival, annual reproductive success, or lifetime reproductive success is reduced. In particular, we examine the scientific and commercial data available to determine if an individual's probable response to an action's effect on the environment (which we identify in our *Response Analyses*) is likely to have consequences for the individual's fitness.

Reductions in abundance, reproduction rates, or growth rates (or increased variance in one or more of these rates) of the populations those individuals represent is a *necessary* condition for reductions in a population's viability, which is itself a *necessary* condition for reductions in a species' viability. On the other hand, when listed plants or animals exposed to an action's effects are *not* expected to experience reductions in fitness, we would not expect that action to have adverse consequences on the viability of the population those individuals represent or the species those populations comprise ((Anderson et al. 2006), (Mills and Beatty 1979), (Stearns 1982)). If we conclude that individuals of listed species are *not* likely to experience reductions in their fitness, we would conclude our assessment because an action that is not likely to affect the fitness of individuals is not likely to jeopardize the continued existence of listed species.

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

We conduct a separate analysis on species' designated critical habitat. The analysis focuses on reductions in the quality, quantity, or availability of primary constituent elements (PCEs) from exposure to the stressors of the action. Since chemicals are the stressors of the action for this Opinion, PCEs potentially affected are freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, estuarine areas, and nearshore marine areas. The PCE attributes of prey availability, water quality, and natural cover are the primary assessment endpoints addressed when evaluating the effects of an herbicide registration on designated critical habitat. Information evaluated for effects to prey include prey survival, prey growth, prey drift, prey reproduction, abundance of prey, health of invertebrate aquatic communities, and recovery of aquatic communities following pesticide exposure. Information evaluated for degradation of water quality include measured and anticipated exposure concentrations leading to toxic responses within aquatic organisms (including salmonids, their prey, and primary producers that support riparian areas and inwater natural cover) as well as instances of water bodies not meeting local, state, or federal water quality standards and criteria.

5.1. Evidence Available for the Consultation

We search, compile and use a variety of resources to conduct our analyses including:

- EPA's BEs, REDs, IREDS, other documents developed by EPA
- Peer-reviewed literature
- Gray literature
- Books
- Available pesticide labels
- Any correspondence (with EPA or others)
- Available monitoring data and other local, county, and state information
- Pesticide registrant generated data
- Online toxicity databases (PAN, EXTOXNET, ECOTOX, USGS, NPIC)
- Pesticide exposure models run by NMFS and EPA
- Information and data provided by the registrants identified as applicants
- Comments on the draft Opinion from EPA, applicants, and others
- Incident reports

Collectively, this information provides the basis for our determination as to whether and to what degree listed resources under our jurisdiction are likely to be exposed to EPA's action and whether and to what degree the EPA can ensure that its authorization of pesticides is not likely to jeopardize the continued existence of threatened and endangered species or is not likely to result in the destruction or adverse modification of designated critical habitat.

5.2. Application of Approach in this Consultation

For this consultation, we adapt our general approach to incorporate elements of EPA's ecological risk assessment (ERA) framework (EPA 1998).

Figure 3 shows the overall framework used in this Opinion. This risk assessment framework organizes the available information in three phases: problem formulation, analysis of exposure and response, and risk characterization (EPA 1998). We adapted the EPA framework to address ESA-specific considerations. The NMFS framework follows a process for organizing, evaluating, and synthesizing the available information on listed resources and the stressors of the action. We separately evaluate the risk to listed species and the risk to designated critical habitat from the stressors of the action (See *Effects of the Proposed Action to Threatened and Endangered Pacific Salmonids* and *Effects of the Proposed Action to Designated Critical habitat*). Below, we briefly describe the problem formulation phase used to evaluate risk of thiobencarb products.

5.2.1. Problem Formulation

Problem formulation includes conceptual models based on our initial evaluation of the relationships between stressors of the action (pesticides and other identified chemical stressors) and receptors (listed species and habitat). Unlike OPP's pesticide ERAs[8], which begin with the use, fate, and toxicity properties of thiobencarb, and evaluate risk based on a small number of standard toxicity test organisms exposed to thiobencarb, NMFS begins with the species' range and life history to determine relevant assessment endpoints, identifies if those endpoints are likely to be affected by the stressors of the

[8] Which may be referred to as ERAs, BEs (Biological Evaluations) or pesticide risk assessments in various locations throughout this document.

action, and seeks data with which to evaluate those effects. In brief, we employ a species-centric approach, rather than a chemical-centric approach. Assessment endpoints and measures may vary by life stage and are presented in Table 3. Many of the relevant assessment endpoints and measures are not ones typically considered or used in EPA's registration of pesticide active ingredients.

Table 3. Salmonid life stage and habitat assessment endpoints and measures.

Salmonid Life Stage	Assessment Endpoint	Assessment Measure
	Individual fitness	**Measures of changes in individual fitness**
Egg	Development	Size, hatching success, morphological deformities
	Survival	Viability (percent survival)
Alevin (yolk-sac fry)	Respiration	Gas exchange, respiration rate
	Swimming: predator avoidance and/or site fidelity	Swimming speed, orientation, burst speed, predator avoidance assays
	Yolk-sac utilization, growth rate, size at first feeding	Rate of yolk absorption, growth weight and length
	Development	Morphology, histology
	Survival	LC_{50}, (dose-response slope), percent dead at a given concentration
Fry, juvenile, smolt	First exogenous feeding (fry)–post yolk-sac absorption	Time to first feeding, starvation
	Survival	LC_{50}, (dose-response slope). Percent dead at a given concentration
	Growth	Stomach contents, weight, length, starvation, prey capture rates
	Feeding	Stomach contents, weight, length, starvation, prey capture rates
	Swimming: predator avoidance behavior, migration, use of shelter	Swimming speed, orientation, burst swimming speed, predator avoidance assays, swimming rate, downstream migration rate, fish monitoring, bioassays

Salmonid Life Stage	Assessment Endpoint	Assessment Measure
	Individual fitness	**Measures of changes in individual fitness**
	Olfaction: kin recognition, predator avoidance, imprinting, feeding	Electro-olfactogram (EOG) measurements, behavioral assays
	Smoltification	Na/K ATPase activity, sea water challenge tests
Returning adult	Development	Length, weight, malformations
	Survival	LC_{50}, (dose-response slope). Percent dead at a given concentration
	Feeding	Prey consumption rates, stomach contents, length and weight
	Swimming: predator avoidance, migration, spawning, feeding	Behavioral assays, numbers of adult returns, numbers of eggs fertilized or redds, stomach contents
	Sexual development	Histological assessment of ovaries/testis, measurements of intersex
	Olfaction: predator avoidance, homing, spawning	Electro-olfactogram (EOG) measurements, behavioral assays
Habitat	In-stream: Aquatic primary producers, salmonid prey abundance, dissolved oxygen and pH, natural cover for salmonids	Growth inhibition bioassays (EC_{25} or EC_{50}), prey survival (EC_{50}); field measured community metrics direct measurement
	Riparian zone: Riparian zone vegetation, natural cover for salmonids, sedimentation, temperature	Growth inhibition (EC_{25} or EC_{50}), salmonid monitoring (field) direct measurements

These assessment endpoints consider effects on all life stages of the salmon (direct effects), as well as effects on plants and prey items (indirect effects). Based on the assessment endpoints, we evaluate the following risk hypotheses for the species.

5.2.1.1. *Species Risk Hypotheses*

1. Exposure to thiobencarb and other chemical stressors of the action via drift or runoff is sufficient to:
    - a. kill salmonids from direct exposure;
    - b. reduce salmonid survival through impacts to growth;
    - c. reduce reproduction;
    - d. impair swimming;
    - e. impair respiration
    - f. reduce salmonids growth through impacts on the availability and quantity of prey.
2. Exposure to the thiobencarb via drift or runoff is sufficient to:
    - a. reduce numbers of aquatic primary producers, thereby affecting salmonid prey communities, salmonids and salmonids instream cover;
    - b. reduce riparian vegetation to such an extent that stream temperatures are elevated, erosion increases, and reductions in natural cover results through reduced inputs of woody debris and vegetation.
3. Exposure to degradates of thiobencarb will cause adverse effects to salmonids and their habitat.
4. Exposure to adjuvants, tank mixes and other chemicals within pesticide products containing thiobencarb will cause adverse effects to salmonids and their habitats.
5. Exposure to other pesticides present in the action area will act in combination with thiobencarb to increase effects to salmonids and their habitats.
6. Exposure to elevated temperatures will enhance the toxicity of thiobencarb.

5.2.2. Designated Critical Habitat

When designated critical habitat for the species is identified, primary constituent elements (PCEs) of that habitat are also identified in Table 4. To determine potential effects to designated critical habitat, we evaluate the effects of the action by first looking at whether PCEs of critical habitat are potentially affected by the stressors of the action.

Effects to PCEs include changes to the functional condition of salmonid habitat caused by the action in the action area. Properly functioning salmonid PCEs are important to the conservation of the ESU/DPS. NMFS convened Critical Habitat Analytical Review Teams (CHARTs) that have ranked the conservation value of particular watersheds within the designated critical habitat of most Pacific salmonids as high, medium , or low based on their review of the PCEs. The stressors of the action for this Opinion are chemicals introduced into the environment by application of pesticide products containing thiobencarb. Key PCEs potentially affected are freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, and estuarine areas. Based on the action we do not anticipate nearshore marine areas or offshore marine areas to be exposed because thiobencarbs use is limited to the Central Valley of California.

Table 4. Essential physical and biological features of PCEs in salmonid critical habitat designations.

Site	Essential Physical and Biological features	Species Life Stage and Functional Developmental Response
Freshwater Spawning	Water quality, water quantity, and substrate	Spawning, incubation larval development
Freshwater rearing	Water quantity and floodplain connectivity	Juvenile growth and mobility
	Water quality and forage	Juvenile growth and development
	Natural cover[a]	Juvenile mobility and survival
Freshwater migration	Free of obstructions, water quality and quantity, and natural cover[a]	Juvenile and adult mobility and survival
	forage	Juvenile growth and development
Estuarine areas	Free of obstruction, water quality and quantity, and salinity	Juvenile and adult physiological transitions between salt and freshwater
	Natural cover[a] and forage[b] and water quantity	Growth and maturation
Nearshore Marine areas	Free of obstruction, water quality and quantity, natural cover[a] and forage[b]	Growth and maturation, survival
Offshore marine areas	Water quality and forage[b]	Growth and maturation

[a] Natural cover examples include shade, large wood, riparian systems, log jams, beaver dams, aquatic vegetation, large rocks and boulders, side channels, and undercut banks.

[b] Forage includes aquatic and terrestrial invertebrates and fish and shellfish species that support growth and maturation of salmonids.

Based on the PCEs and life stage potentially affected Table 4, we developed risk hypotheses for critical habitat. Properly functioning salmonid PCEs are important to the conservation of the ESU/DPS. The stressors of the action for this Opinion are chemicals introduced into the environment by application of pesticide products.

5.2.3. Critical Habitat Risk Hypotheses

1. Exposure to the stressors of the action is sufficient to degrade water quality, natural cover, and/or reduce prey availability in freshwater rearing and migrating areas;
2. Exposure to the stressors of the action is sufficient to degrade riparian areas adjacent to rearing and migration corridors;
3. Exposure to the stressors of the action is sufficient to degrade water quality, natural cover, and/or reduce prey availability in estuarine areas;

5.2.4. Evaluating Exposure and Response

As part of the problem formulation phase, we consider the toxic mode and mechanism of action of thiobencarb to provide insight into potential consequences following exposure. Identification of the mode and mechanism of action allows us to identify other chemicals that might co-occur and affect the response (*i.e.,* identify potential toxic mixtures in the environment). We consider authorized thiobencarb use sites to determine spatial overlap between its use on rice and the species and its designated critical habitat. We consider fate properties of thiobencarb to determine its persistence in aquatic systems. Conceptual diagrams are shown in
Figure 3 and Figure 4.

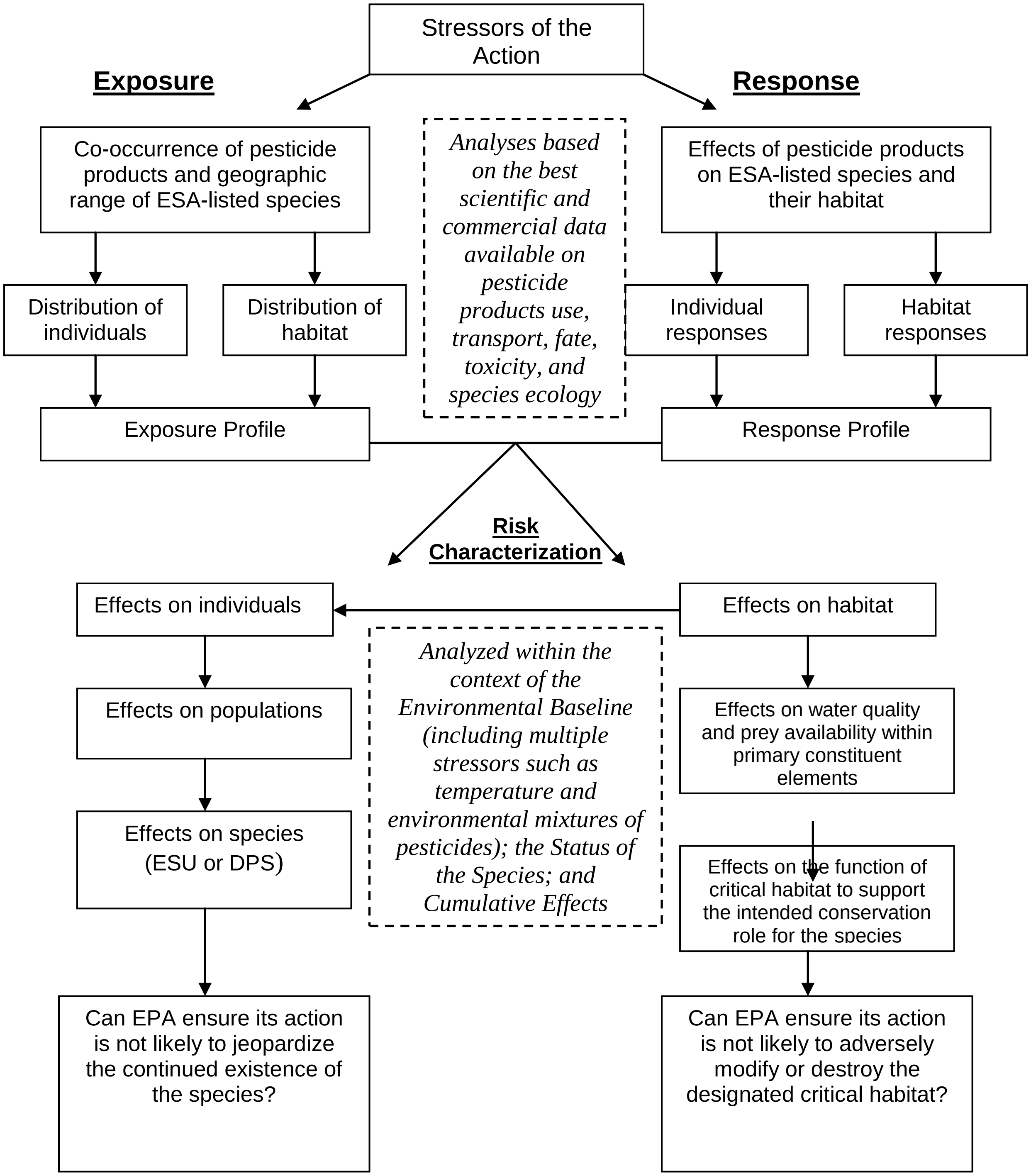


Figure 3. Conceptual framework for assessing risks of EPA's action to ESA listed resources.

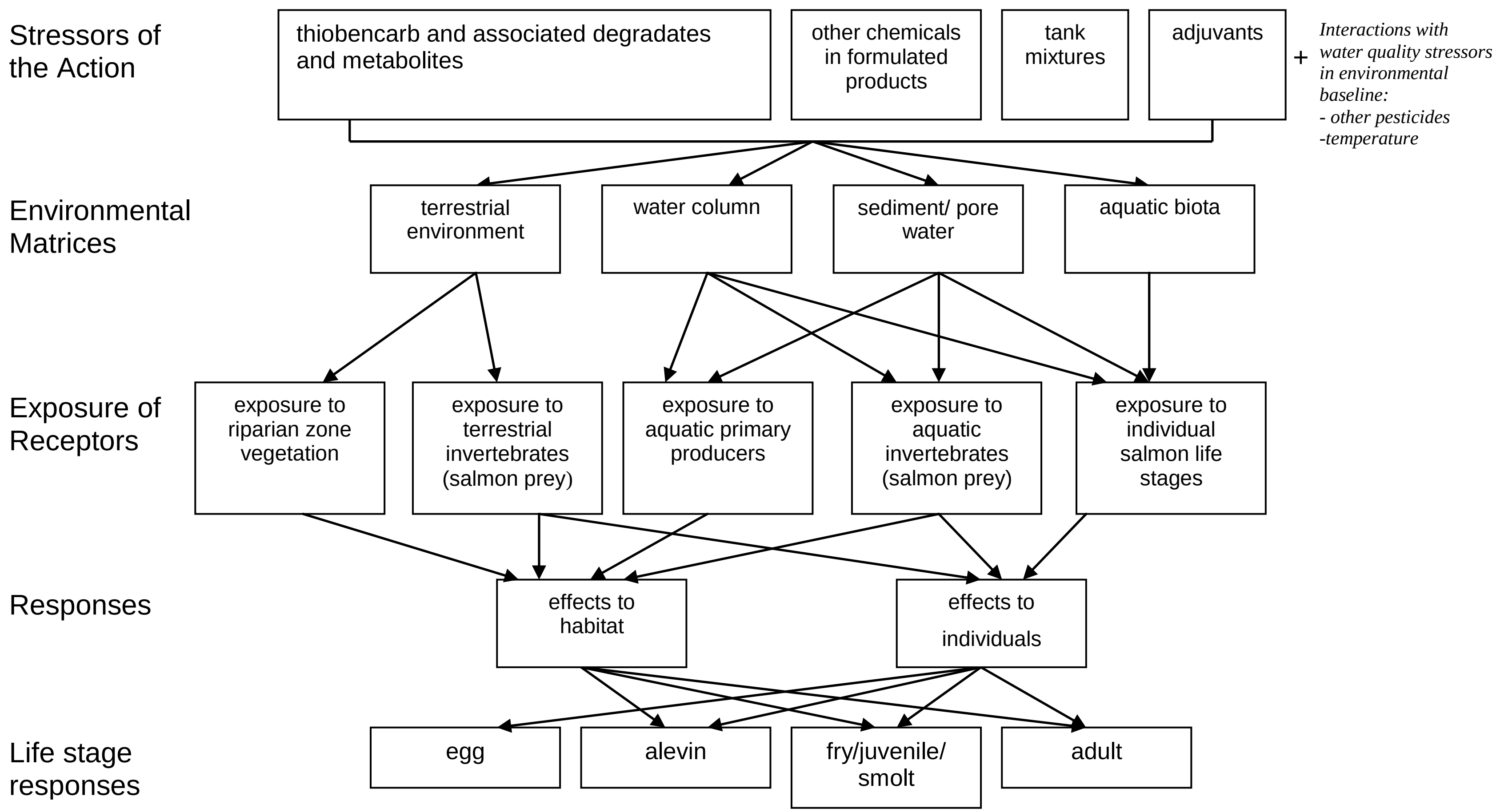


Figure 4. Exposure pathways for stressors of the action, and general response of Pacific salmonids and habitat.

5.2.5. Analysis Plan

5.2.5.1. *Status of the Species*

In this section, we present information regarding each of the ESUs and DPSs considered in this Opinion. We discuss life history, population abundance and trends and overall viability of the species. This provides part of the context in which we evaluate the effect of the proposed action.

5.2.5.2. *Environmental Baseline*

In this section we discuss all stressors affecting salmon populations including natural predators, events and disease; and anthropogenic effects such as pollution and habitat modification. This also provides part of the context in which we evaluate the effect of the proposed action.

5.2.5.3. *Effects of the Proposed Action to Threatened and Endangered Pacific Salmonids*

In the *Exposure* section we discuss life histories of the various species which may make them more or less likely to be exposed to stressors of the actions. Then we evaluate measured and estimated environmental concentrations of the stressors from various sources. In this section we also evaluate spatial and temporal co-occurrences of rice growing and salmon habitat. The *Response* section details information for the assessment endpoints identified in the problem formulation. In the *Risk Characterization* sections for listed species and designated critical habitat, we integrate the exposure and response information and evaluate the risk hypotheses. *Risk Characterization* may also include population-level analyses to determine if effects on individual fitness are sufficiently large to affect population parameters.

5.2.5.4. *Integration and Synthesis*

We begin Integration and Synthesis with a summary of risk as described/identified in the *Risk Characterization* associated with each of the a.i.s. In separate sections for listed species and critical habitat, we combine these risk conclusions regarding the effects of the proposed action with information in the *Status of the Species, Environmental Baseline* and *Cumulative Effects* (those effects of future State or private activities, not involving Federal activities, that are reasonably certain to occur within the action area), to determine potential effects on populations and species.

5.2.5.5. *Conclusion*

Based on the potential effects for each species, we determine if the proposed action is likely to jeopardize the species or cause destruction or adversely modify of destroy designated critical habitat.

5.3. Other Considerations

In this Opinion, we evaluated lines of evidence constructed as species-specific risk hypotheses to ensure relevant endpoints were addressed. The analysis weighs each line of evidence by evaluating the best commercial and scientific data available that pertain to a given risk hypothesis. Overall, the analysis is a qualitative approach that uses some quantitative tools to provide examples of potential risks to listed salmonids and their habitat. Multiple methods and tools currently exist for addressing contaminant-induced risk to the environment. Hazard-based assessments, probabilistic risk assessment techniques, combinations of the two, and deterministic approaches such as screening level assessments have been applied to questions of risk related to human health and the environment.

We also evaluated a recent assessment tool to quantify spatial and temporal loadings of pesticides in the Central Valley (Hoogeweg et al. 2011). We found that much of the information on species, pesticide fate, monitoring data, and rice culture was similar to the information we used in this assessment.

In recent pesticide risk assessments, probabilistic techniques have been used to evaluate the probability of exceeding a “toxic” threshold for aquatic organisms by combining pesticide monitoring data with species sensitivity distributions (Giddings 2009, Geisy et al. 1999). There is utility in information generated by probabilistic approaches if supported by robust data.

NMFS considered the use of probabilistic risk assessment techniques for addressing risk of thiobencarb use at population and species (ESU and DPS) scales for the stressors of the action.

However, we encountered significant limitations in available data that suggested the information was not sufficient to define exposure and/or response probabilities necessary to determine the probability of risk to thiobencarb. Probabilistic techniques were not used in the Opinion due to issues with data collection, paucity of data, non-normal distributions of data, and quality assurance and quality control. For example, it was not deemed appropriate to pair the salmonid prey responses with exposure probabilities based on monitoring results given the limitations of that data set discussed in the *Effects of the Proposed Action*.

To evaluate population consequences associated with potential lethality from thiobencarb exposure in salmon, NMFS selected the lowest reported salmonid LC_{50} from the available information to ensure risk was not underestimated. When we consider the data limitations coupled with the inherent complexity of EPA's proposed action in California's Central Valley, we find that probabilistic assessments at population and species scales introduce an unquantifiable amount of uncertainty that undermines confidence in derived risk estimates. These same studies do not factor the status of the species, baseline conditions of the environment or anticipated cumulative effects into their assessment. At this time, the best available data do not support such an analysis and conclusions from such an analysis would be highly speculative.

6. Status of Listed Resources

The purpose of this section is to characterize the condition of the three salmonid species[9] under consultation relative to their likelihood of viability and to describe the conservation role and function of their respective critical habitats. NMFS has determined that the following species and critical habitat designations may occur in the action area for EPA's registration of thiobencarb containing products (Table 5). More detailed information on the status of these species and critical habitat are found in a number of published documents including recent recovery plans, status reviews, stock assessment reports, and technical memorandums. Many are available on the Internet at http://www.nmfs.noaa.gov/pr/species/.

Table 5. Listed Species and Critical Habitat in the Action Area.

Common Name (Evolutionarily Significant Unit, or Distinct Population Segment)	***Scientific Name***	**Status**
Chinook (Sacramento River winter-run)	*Oncorhynchus tshawytscha*	Endangered
Chinook (California Central Valley spring-run)		Threatened
Steelhead (California Central Valley)	*Oncorhynchus mykiss*	Threatened

The following narratives summarize the biology and ecology of threatened and endangered Pacific salmonids that are relevant to EPA's proposed action. This includes a description of the timing and duration of each life stage such as adult river entry, spawning, egg incubation, freshwater rearing, smolt outmigration, and ocean migration. These summaries provide a foundation for NMFS' evaluation of the effects of the proposed action on listed salmonids. We also highlight information related to the viability of salmonid populations and the primary constituent elements (PCEs) of designated critical habitat.

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

6.1. Species Status

The status of an ESU or DPS is determined by the degree that it (1) maintains sufficient genetic and phenotypic diversity to ensure continued fitness in the face of environmental change, (2) maintains spatial distribution of populations so that not all populations would be affected by a catastrophic event, and (3) maintains sufficient connectivity among populations within the ESU or DPS to maintain long-term demographic and evolutionary processes (ICTRT 2007, McElhany et al. 2000, Spence et al. 2008). We describe the current condition of the spatial structure and major life histories within the ESUs or DPSs. In order to maintain a spatial distribution and diversity that support a viable ESU or DPS, a species must maintain multiple viable populations that are sustainable in the long-term in the face of environmental variability.

Before assessing population viability, we first identify the historic and current populations that constitute a species. How NMFS defines a population and its function are found in McElhany *et al. (2000)* and in Bjorkstedt *et al.(2005)*, NMFS' Pacific salmon Technical Recovery Teams (TRTs) have identified historic populations within ESUs/DPSs. These historical populations have been categorized based on their distribution and demographic role (*i.e.*, functionally independent, potentially independent, or dependent). Functionally independent (independent) populations were sufficiently large to be viable in isolation, (*i.e.*, a negligible extinction risk). Potentially independent populations were potentially viable in isolation, but were likely influenced by immigrants from adjacent populations. Dependent populations were unlikely to persist over a 100-year time period in isolation. However, immigration from other nearby populations reduced the extinction risk for dependent populations. The historical conditions of the populations for each ESU/DPS serve as a point of reference for evaluating the current viability of populations[10] and the status of the species. The current viability is used as the base condition from which the effects of the proposed action on individuals are evaluated to determine whether these effects are likely to increase the probability of extinction of the populations those individuals represent.

[10] The TRTs did not propose that historical conditions are the criteria or benchmark for evaluating population or ESU viability (extinction risk).

In our *Approach to the Assessment* section, NMFS introduced the VSP concept and its four criteria. We restate that a VSP is an independent population (a population of which extinction probability is not substantially affected by exchanges of individuals with other populations) with a negligible risk of extinction, over a 100-year period, when threats from random catastrophic events, local environmental variation, demographic variation, and genetic diversity changes are taken into account (McElhany et al. 2000). The four factors defining a viable population are a population's: (1) spatial structure; (2) abundance; (3) annual growth rate, including trends and variability of annual growth rates; and (4) diversity (McElhany et al. 2000).

A population's tendency to increase in abundance (i.e., lambda (λ) > 1.000) and its variation in annual population growth defines a viable population (McElhany et al. 2000, Morris and Doak 2002). A negative long-term trend in average annual population growth rate (i.e., λ < 1.000) will eventually result in extinction. Further, a weak positive long-term growth rate will increase the risk of extinction as it maintains a small population at low abundances over a longer time frame. A large variation in the growth rates also increases the likelihood of extinction (Lande 1993, Morris and Doak 2002).

Thus, in our status reviews of each listed salmonid species, we provide information on population abundance and annual growth rate of extant populations. We use the median annual population growth rate (denoted as lambda, λ) from available time series of abundance for independent populations (Good et al. 2005). Several publications provide a detailed description of the calculation of lambda (Good et al. 2005, McClure et al. 2003). The lambda values for salmonid populations of concern in this Opinion are summarized in Table 6.

Table 6. Species and population annual growth rates (Good et al., 2005)

ESU	Population	λ– H=0	95% CI – lower	95% CI -upper
CV spring-run Chinook	Butte Creek Deer Creek Mill Creek	1.300 1.170 1.190	1.060 1.040 1.000	1.600 1.350 1.470
Sac. River winter-run Chinook	Sacramento River	0.970	0.870	1.090
California CV steelhead	Sacramento River	0.950	0.900	1.020

6.2. Conservation Role of Critical Habitat for the Species

The action area for this consultation contains designated critical habitat. Critical habitat is defined as the specific areas within the geographical area occupied by the species, on which are found those physical or biological features that are essential to the conservation of the species, and which may require special management considerations or protection. Critical habitat can also include specific areas outside the geographical area occupied by the species that are determined by the Secretary to be essential for the conservation of the species (ESA of 1973, as amended, section 3(5)(A)).

The primary purpose in evaluating the status of critical habitat is to identify for each ESU or DPS the function of the critical habitat to support the intended conservation role for each species. Such information is important for an adverse modification analysis as it establishes the context for evaluating whether the proposed action results in negative changes in the function and role of the critical habitat for species conservation. NMFS bases its critical habitat analysis on the areas of the critical habitat that are affected by the proposed action and the area's physical or biological features that are essential to the conservation of a given species, and not on how individuals of the species will respond to changes in habitat quantity and quality.

In evaluating the status of designated critical habitat, we consider the current quantity, quality, and distribution of those primary constituent elements or PCEs that are essential to the conservation of the species [50 CFR 424.12(b)]. NMFS has identified PCEs of critical habitat for each life stage (*e.g.*, migration, spawning, rearing, and estuary) common for each species. To fully understand the conservation role of these habitats, specific physical and biological habitat attributes (*e.g.*, water temperature, water quality, forage, etc.) were identified for each life stage. Specifically, during all freshwater life stages, salmonids require cool water that is free of contaminants. During the juvenile life stage, salmonids also require stream habitat that provides excess forage (*i.e.*, prey abundance). Besides potential toxicity, water free of contaminants is important as contaminants can disrupt normal behavior necessary for successful migration, spawning, and juvenile rearing. Sufficient forage is necessary for juveniles to maintain growth that reduces freshwater predation mortality, increases overwintering success, initiates smoltification, and increases ocean survival. A description of the past, ongoing, and continuing activities that threaten the functional condition of PCEs and their attributes are described in the *Environmental Baseline* section of this Opinion.

NMFS has identified six common PCEs for 7 California listed Chinook salmon (including the Sacramento winter-run and Central Valley spring-run) and steelhead (including the Central Valley DPS) (70 FR 52488, Sept. 2, 2005), 12 ESUs of Oregon, Washington, and Idaho salmon (chum, sockeye, Chinook) and steelhead (70 FR 52630, Sept. 2, 2005), and for the Oregon Coast coho salmon (73 FR 7816, Feb. 11, 2008). They are:

(1) Freshwater spawning sites with water quantity and quality, and suitable substrate size as attributes necessary to support spawning, incubation and larval development;

(2) Freshwater rearing sites with the following attributes: (i) Water quantity and floodplain connectivity to form and maintain physical habitat conditions and support juvenile growth and mobility; (ii) Water quality and forage supporting juvenile development; and (iii) Natural cover such as shade, submerged and overhanging large wood, log jams and beaver dams, aquatic vegetation, large rocks and boulders, side channels, and undercut banks.

(3) Freshwater migration corridors free of obstruction and excessive predation with water quantity and quality conditions and natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, side channels, and undercut banks supporting juvenile and adult mobility and survival.

(4) Estuarine areas free of obstruction and excessive predation with:
(i) Water quality, water quantity, and salinity conditions supporting juvenile and adult physiological transitions between fresh- and saltwater; (ii) Natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, side channels; and (iii) Juvenile and adult forage, including aquatic invertebrates and fishes, supporting growth and maturation.

(5) Nearshore marine areas free of obstruction and excessive predation with:
(i) Water quality and quantity conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation; and (ii) Natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, and side channels.

(6) Offshore marine areas with water quality conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation.

In this section, we also identify the conservation values of watersheds located within the critical habitat designated for a species. If the effects on PCEs are important at the watershed scale, then the conservation value for the watershed is used to assess the conservation role of that watershed in the context of range wide critical habitat. The conservation value of a particular watershed was determined by Critical Habitat Analytical Review Teams (CHARTs). These teams considered the presence of PCEs within each occupied area of a watershed and the activities that potentially affect the PCEs, and assigned conservation values for watersheds within designated critical habitat.

Each watershed was scored as low, moderate, or high conservation value. High value watersheds/areas have a high likelihood of promoting species conservation, while low value watersheds/areas are less important for species conservation. Scores were based on: (1) a comparison of current quantity of PCEs within a watershed relative to other watersheds and probable historic quantity of PCEs within the watershed; (2) existing quality of PCEs in watersheds; (3) the likelihood of achieving PCE potential in a watershed; (4) the PCEs' support of rare genetic or life history characteristics or rare/important habitat types in the watershed; (5) consideration of the PCEs' support of variable-sized populations relative to other watersheds and the probable historical levels in the watershed; and (6) considerations of the PCE support of spawning or rearing of varying numbers of populations.

6.3. Chinook Salmon

6.3.1.1. *Description of the Species*

Chinook salmon are the largest of the Pacific salmon and historically ranged from the Ventura River in California to Point Hope, Alaska in North America, and in northeastern Asia from Hokkaido, Japan to the Anadyr River in Russia (Healey 1991). Chinook salmon prefer streams that are deeper and larger than those used by other Pacific salmon species. We discuss the distribution, life history, status, and critical habitat of the two species of endangered and threatened Chinook salmon separately.

Chinook salmon are generally described as one of two races, within which there is substantial variation (Healey 1991, Groot and Margolis 1991). One race, the “stream-type,” resides in fresh water for a year or more following emergence from gravel nests. Juveniles migrate to sea as yearlings. Stream-type Chinook salmon normally returns in late winter and early spring (spring-run) as immature adults and reside in deep pools during summer before spawning in fall. The other race, the “ocean-type,” migrate to the ocean within their first year (sub-yearlings) and usually return as full mature adults in fall (fall-run). Fall-run adults spawn soon after river entry.

The timing of return to fresh water, and ultimately spawning, often provides a temporal isolating mechanism for populations with different life histories. Return timing is often related to spawning location. Thus, differences in the timing of spawning migration also serve as a geographic isolating mechanism. Fall-run Chinook salmon generally spawn in the mainstem of larger rivers and are less dependent on flow, although early autumn rains and a drop in water temperature often provide cues for movements to spawning areas. Spring-run Chinook salmon take advantage of high flows from snowmelt to access the upper reaches of rivers.

Successful incubation depends on several factors including dissolved oxygen (DO) levels, temperature, substrate size, amount of fine sediment, and water velocity. Chinook salmon egg incubation time is highly correlated with water temperature (McCullough 1999). Spawning sites have larger gravel and more water flow up through the gravel than the sites used by other Pacific salmon. Maximum survival of incubating eggs and the pre-emergent alevins occurs at water temperatures between about 5.5° and 13.5°C. Development time is influenced by degree days with fertilization to emergence taking up to 325 days at 2°C and about 50 days at 16°C (McCullough 1999). Fry emergence commonly begins in December and continues into mid-April (Leidy 1984). When emerging from the redd, fry move through the interstitial spaces in the redd substrate to escape the gravel. However, a high content of fines and sand in the redd substrate can severely hinder fry emergence and cause high mortality (Bjornn and Reiser 1991). Optimal temperatures for both Chinook salmon fry and fingerlings range from 12° to 14°C (Boles 1988). Temperatures above 15°C increase the risk of diseases and lower the tolerance to other stressors (McCullough 1999). At about 19°C, Chinook salmon cease to eat. In the laboratory, 50% mortality during a 24 hour period is observed at 24° to 25°C (Brett 1952, Hanson 1997) the exact lethal temperature being somewhat dependent on the temperature that the fish has been acclimated to.

Chinook salmon alevins, as is the case for other salmonids, rely on yolk for nutrition until the onset of active feeding. It is important that the young start feeding at the proper time since failure to start feeding can retard growth and lead to behavioral or developmental problems that reduce survival. In Chinook salmon, alevins may start feeding immediately upon emergence even if they have not yet absorbed all of the egg yolk (Linley 2001). During freshwater residence, Chinook salmon juveniles feed in the water column and from the water surface. Food items include a variety of small terrestrial and aquatic insects and aquatic crustaceans; the prey species of juveniles depend on availability (habitat and months), prey size distribution, and the size of the fish (Koehler et al. 2006, Rondorf et al. 1990). The coarse bottom substrate found in faster flowing riverine habitats supports drift of larger aquatic insects such as caddisflies (*Trichoptera*), mayflies (*Ephemeroptera*), stoneflies (*Plecoptera*), and other benthic organisms when they are present in the water column during high flow events. These taxa, when present, are important food items in terms of biomass for Chinook salmon juveniles. Terrestrial insects and midges (*Diptera*: *Chironmidae*) often dominate the diet in slower moving water with finer bottom substrate such as floodplains like the Yolo Bypass, off-channel ponds, sloughs, and in lakes/reservoirs (Miller and Simenstad 1997, Rondorf et al. 1990, Sommer et al. 2001, Tabor et al. 2006). In addition, copepods and daphnia may make up a high proportion of the diet in ponds, reservoirs and lakes, and in the mainstems of large rivers (Koehler et al. 2006, Rondorf et al. 1990, Sommer et al. 2001). At periods, swarming terrestrial insects such as ants can make up a substantial portion of the diet of Chinook salmon rearing in floodplains, ponds and reservoirs (Rondorf et al. 1990). In estuaries, scuds, mysids, and gammarid amphipods may be major prey (Miller and Simenstad 1997).

Studies of stream habitat use show that there are velocity thresholds for rearing fry and juveniles, that fish move to faster and deeper water as they grow, and that fish use substrate and cover as refuge from high velocities (Chapman and Bjornn 1969, Everest and Chapman 1972, Johnson et al. 1992). In the mainstem of large rivers and in lakes, fry and juveniles rear along the river margins and in nearshore areas that are less than one meter deep and have low lateral bank slopes (Sergeant and Beauchamp 2006, Tiffan et al. 2006). Juveniles tend to avoid the elevated water velocities found in the thalweg of river channels. As they grow larger, their habitat preferences change; juveniles move away from stream margins and begin to use deeper water (Everest and Chapman 1972, Tabor et al. 2006). When the river channel is greater than 9- to 10-ft in depth, juvenile salmon tend to inhabit the surface waters (Healey 1982).

Chinook salmon fry may also move into non-natal tributaries (*i.e.,* streams other than those where they incubated) to rear (Limm and Marchetti 2009, Teel et al. 2009). In the Sacramento River, California, fry and juveniles move into seasonally inundated floodplains (Yolo Bypass) and off-channel water bodies to rear as they move downstream (Limm and Marchetti 2009, Sommer et al. 2001, Teel et al. 2009). However, Chinook salmon use of floodplain and off-channel habitat depend on availability of these habitats, the life history of the race, time of year, flow, and temperatures. Up to a certain limit, distribution in floodplain habitat is positively correlated with water temperatures (Limm and Marchetti 2009, Sommer et al. 2001, Teel et al. 2009). Floodplain wetlands and off-channel habitat also often have higher prey densities (Sommer et al. 2001). Several studies have shown that fry rearing on large floodplains experience a higher growth rate, and possibly higher survival, than fry remaining in the main channel (Jeffres et al. 2008, Limm and Marchetti 2003, Sommer et al. 2001). The increased growth rate is likely caused by the higher water temperatures as well as the higher prey densities in these habitats. Having sufficient growth during the juvenile stage is critical as some studies indicate that size at smolting influence survival during the first year in the ocean. As flow decreases and water temperature increases in summer, juveniles move out of the inundated floodplain habitat or succumb to lethal temperatures and stranding.

Many Chinook salmon populations use the estuary intensively for rearing, and a downstream movement of large numbers of fry is typical for many populations (Reimers 1973, Sazaki 1966, Thorpe 1994). Estuaries can provide a productive environment and additional growth, refuge from predators, and a transition to marine waters; availability of unmodified estuaries is correlated with difference between rivers in survival of hatchery reared fish from smolt to maturity (Magnusson and Hilborn 2003). Ocean-type Chinook salmon migrate downstream as fry immediately after emerging from spawning beds (Healey 1991). These smaller fry and sub-yearlings extensively use shallow water habitat and sloughs within the estuary to rear to the smolt stage (Fresh et al. 2005). Yearling juveniles of the river-type life history enter the estuaries at the smolting stage; they usually spend less time in estuaries and use deeper water than fry or sub-yearlings (Fresh et al. 2005).

Upon entering the marine environment, immature Chinook salmon maintain close proximity to nearshore areas. The highest ocean mortality of immature Chinook salmon occurs during the first year after entering the ocean. Expected survival during this period depends both on the condition of the fish such as size and the physical conditions of the marine environment. Ocean condition such as coastal upwelling and atmospheric condition such as El Niño have a significant influence on returning run size. Because of the annual variability in ocean and climatic conditions, the stock-recruitment relationship in Chinook salmon is weak.

Immature Chinook salmon of the ocean- and river-type may have different dispersal and migration patterns during their first marine year (Healey 1991). The larger stream-type immature fish disappear from the surface waters of the Strait of Georgia in early summer. In contrast, during their first ocean year, ocean-type fish are abundant in the sheltered surface waters and estuaries of the Strait of Georgia and the Puget Sound from July through November and some continue to be present throughout winter. Estuaries provide the only shelter along the open coasts of Washington, Oregon, and California; in these areas, ocean-type fry remain longer in their native estuaries. After ocean entry, immature Chinook salmon may move into large estuaries and bays as they migrate along the coast. Chinook salmon remain at sea for one to six years (more commonly two to four years), with the exception of a small proportion of yearling males (called jack salmon) which mature in fresh water or return after two or three months in salt water.

6.3.1.2. *Status and Trends*

Chinook salmon face natural threats from flooding, changes in ocean productivity, and predation. Chinook salmon have declined from over harvests, loss of genetic integrity by mixing with hatchery reared fish, reductions in their distribution by migration barriers such as dams, mortality and loss in water diversions, loss of rearing habitat from gravel mining, degradation of riparian habitat, and modified stream function and reduced water quality from land use practices (logging, agriculture, and urbanization).

Climate change also poses significant hazards to the survival and recovery of salmonids. They included elevated water temperature, earlier spring runoff and lower summer flows, and winter flooding.

6.3.2. Central Valley Spring-run Chinook Salmon

The Sacramento River has sole distinction among the salmon-producing rivers of western North America of supporting four runs of Chinook salmon – spring, fall, late-fall, and winter runs (Yoshiyama et al. 2001). The Central Valley (CV) Spring-run Chinook salmon includes all naturally spawned populations of spring-run Chinook salmon in the Sacramento River, California, and its tributaries (Figure 5). The Feather River Hatchery spring-run Chinook salmon is included in this ESU. This artificially propagated population is no more divergent relative to the local natural populations than would be expected between closely related populations within this ESU. Table 9 identifies populations within the CV Spring-run Chinook salmon ESU, their abundances, and hatchery input.

Historically, spring-run occupied the upper and middle reaches (1,000 to 6,000 feet in elevation) of the San Joaquin, American, Yuba, Feather, Sacramento, McCloud, and Pit rivers, with smaller populations in most tributaries with sufficient habitat for over-summering adults (Yoshiyama et al. 1998).

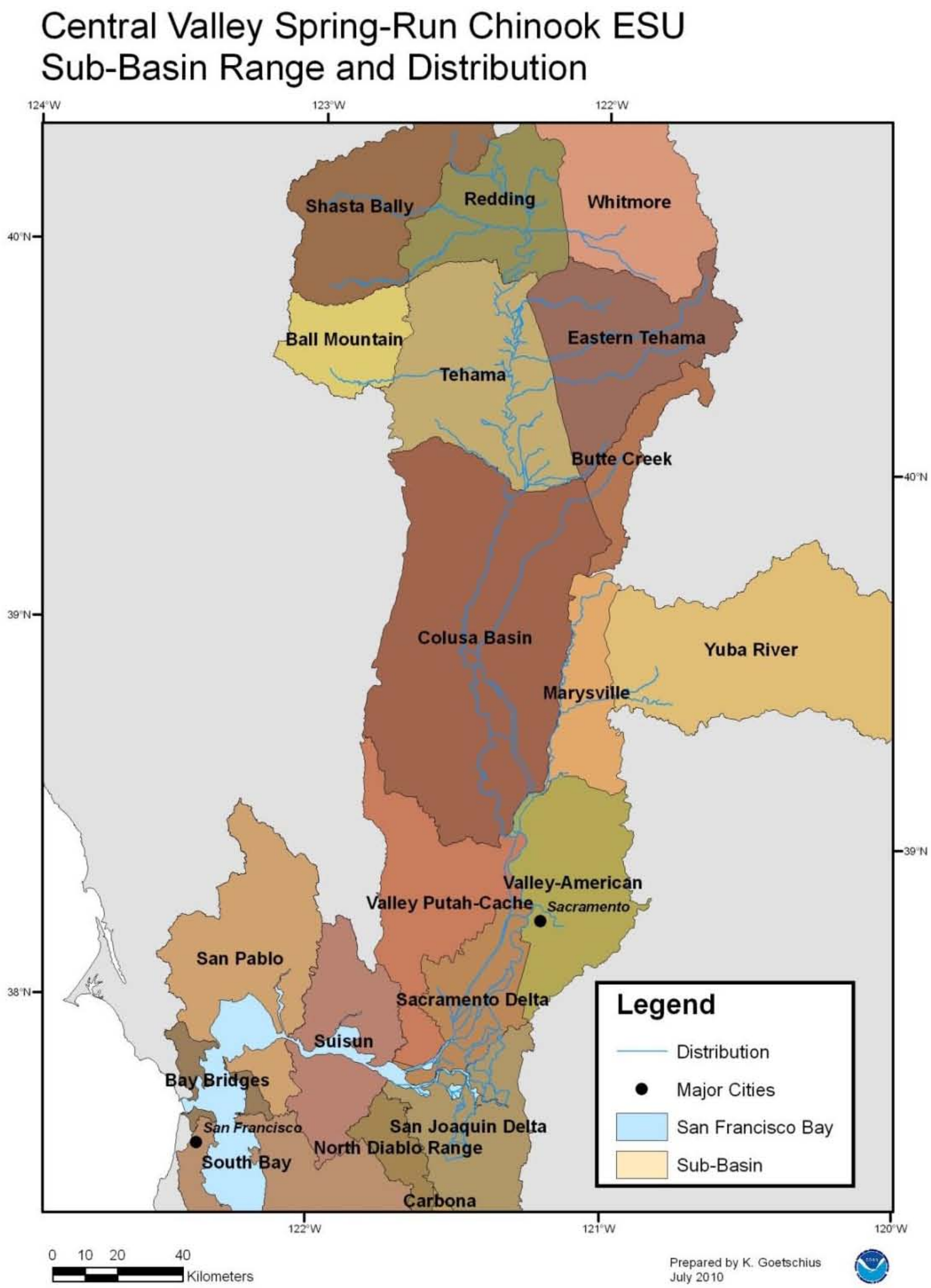


Figure 5. Central Valley Spring-run Chinook salmon distribution.

6.3.2.1. *Life History*

The CV spring-run exhibit a stream-type life history. CV Spring-run Chinook salmon are high-elevation mainstem spawners that migrate as 3-5 year old adults into the Sacramento River from March to May (Yoshiyama et al. 2001) during peak snow-melt flows. Typically, spring-run Chinook salmon spawn higher in the watershed than other stocks. They over-summer in cool temperature pools before migrating out beginning in late August (Lindley et al. 2004b), or early September (Marcotte 1984b) to spawn. Adult spring-run leave the ocean to begin their upstream migration in late January and early February (CDFG 1998b) and enter the Sacramento River between March and September, primarily in May and June (Table 7). When they enter fresh water, spring Chinook are immature. They typically utilize mid- to high-elevation streams that provide appropriate temperatures and sufficient flow, cover, and pool depth to allow over-summering while conserving energy and allowing their gonadal tissue to mature (Yoshiyama et al. 1998). In Deer and Mill creeks, spawning occurs from late August to mid-October. Embryos hatch following a three to six month incubation period (Marcotte 1984b), and the alevins (sac-fry) remain in the gravel for another 2-3 weeks. Once their yolk sac is absorbed, juveniles emerge between November and March to immediately begin feeding (Moyle 2002). Marcotte (1984b) reported observations that juvenile spring-run Chinook in Deer and Mill creeks, during most years, spend 8 to 9 months in the higher elevation streams, where they feed on drift insects (Table 7). Newly emerged fry tend to school in calm, shallow water near shoreline areas (Marcotte 1984b). As they grow larger, juvenile salmon shift to faster deeper water (Groot and Margolis 1991). The fry growing season is typically April to September with growth rate varying depending on stream productivity and water temperature (Spence et al. 1996a, Groot and Margolis 1991). Adult Chinook salmon require cool fresh water while they mature over the summer. Adult upstream migration may be blocked by temperatures above 21ºC (McCullough 1999). Juvenile spring-run emigration in the Sacramento River is highly variable and they may migrate either as soon as they emerge from the gravel or as yearlings. The majority of spring-run fry emerging in the tributaries migrate downstream from December through February during high flows. Juvenile CV Spring-run Chinook salmon have been observed rearing in the lower reaches of non-natal tributaries and intermittent streams in the Sacramento Valley during the winter months. Peak fry/sub-yearling movements are observed farther downstream in lower

Sacramento River (Knights Landing) and the Delta during March and April. Up to 25% of juveniles may remain in the tributaries to rear and outmigrate as yearlings the next fall, normally starting in December.

Table 7. Temporal occurrence of adult and juvenile CV Spring-run Chinook salmon in the Sacramento River. Darker shades indicate months of greatest relative abundance. Note: Yearling rear in their natal streams through the first summer following their birth. Downstream migration generally occurs the following fall and winter. Young of year migrate during the first spring after they hatch.

Adult migration[a]												
Location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Sac. River basin[a,b]												
Up Sac. R. mainstem[c]												
Mill Creek[d]												
Dear Creek[d]												
Butte Creek[d]												
Adult Holding[b]												
Adult Spawning[c]												
Juvenile migration[d]												
Location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Sac. R. Tribs[e]												
Upper Butte Ck.[f]												
Mill, Deer, Butte Cks.[d]												
Sac. R. at RBDD[c]												
Sac. R. at KL[g,h]												
North Delta[i]												
Relative Abundance			= High				= Medium				= Low	

Sources: [a]Yoshiyama et al. (1998); [b]Moyle (2002); [c]Myers et al. (1998); [d]Lindley et al. (2007) and Marcotte (1984b); [e]CDFG (1998b) [f]McReynolds et al. (2005) and Ward et al. (2002, 2003); [g]Snider and Titus (2000b); [h]Bajjaliya and Vincik (2008b); [i]USFWS(2010a).

Studies by California's Department of Fish and Game have shown Juvenile CV Spring-run Chinook had migrated past their sample sight just downstream of Knights Landing by the end of April (Bajjaliya and Vincik 2008b). Spring-run juvenile Chinook salmon were caught in screw traps below Knights Landing from December through April. Bajjaliya and Vincik (2008b) found peak abundance of CV Spring-run Chinook during this sampling period was late December into early January. Another peak of juvenile Chinook showed up later in the spring which they determined to be fall-run based on coded-wire tags and size distribution of fall-run released from the Colman National Fish Hatchery (Bajjaliya and Vincik 2008b). However, estimating the race of juvenile Chinook in the field by size and date of capture has some uncertainty. Criteria for determining race was developed by the CDFG in 1992 (Fisher 1992) as a weekly model of juvenile salmonid growth. This was modified to a daily criterion by the California Department of Water Resources (Greene 1992). It is currently the only tool used by several salmon monitoring programs with the Central Valley to determine race of juvenile Chinook in the field. Until markers for genetic differentiation of races are developed, the race determinations reported cannot be considered definitive (USFWS 2012b). Some fish caught in the traps below Knights Landing later in the spring identified by biologists in the field as fall-run, could in fact have some of the tail end of the spring-run population intermixed.

Juvenile CV Spring-run Chinook are likely to be lingering in the lower river and Sacramento-San Joaquin Delta into June before entering into San Francisco Bay (USFWS 2012b, USFWS 2010a). Sampling conducted in the North Delta of the Sacramento River from 1993 – 2009 showed the abundance of CV Spring-run juvenile Chinook (based on catch per unit effort) in most years peaked either in February or March (Table 7). A combination of Spring- and Fall-run Chinook continue to migrate through the North Delta through June (USFWS 2012b, USFWS 2010a).

6.3.2.2. *Status and Trends*

NMFS originally listed CV Spring-run Chinook salmon as threatened on September 16, 1999 (64 FR 50393), and reaffirmed their threatened status on June 28, 2005 (70 FR 37160) and again on August 15, 2011. Historically, spring-run Chinook salmon were predominant throughout the Sacramento and San Joaquin River drainages. All runs within the San Joaquin River basin are now extirpated. The CV Technical Recovery Team (TRT) delineated 18 independent populations of CV spring-run Chinook, along with a number of smaller populations, and four geographically separated groups (Lindley et al. 2004b). Of these 18 populations, only three are extant on the upper Sacramento River and they represent only the Northern Sierra Nevada geographic group.

Naturally spawning populations of CV Spring-run Chinook salmon currently are restricted to accessible reaches of the upper mainstem Sacramento River and its tributaries Butte, Deer, and Mill Creeks. Limited spawning occurs in the basins of smaller tributaries (CDFG 1998b, Lindley et al. 2004b). Using data through 2005 and the criteria in Table 8, Lindley et al. (2007) found that the Mill Creek, Deer Creek, and Butte Creek populations were at or near low risk of extinction. However, the ESU as a whole, could not be considered viable because there were no extant populations in the three other diversity groups. In addition, Mill, Deer, and Butte creeks are close together, decreasing the independence of their extinction risks due to catastrophic disturbance (Williams et al. 2011).

Table 8. Criteria for assessing the level of risk of extinction for populations of Pacific salmonids in the Central Valley of California. Overall risk is determined by the highest risk score for any category.

Risk of Extinction			
Criterion	High	Moderate	Low
Extinction Risk from PVA	>20 % within 20 years	>5% within 100 years	<5% within 100 years
	or any ONE of:	or any ONE of:	or any ONE of:
Population size[a]	$N_e \leq 50$	$50 < N_e \leq 500$	$N_e > 500$
	-or-	-or-	-or-
	$N \leq 250$	$250 < N \leq 2500$	$N > 2500$
Population decline	Precipitous decline[b]	Chronic decline or depression[c]	No decline apparent or probable
Catastrophe, rate and effect[d]	Order of magnitude decline within one generation	Smaller but significant decline[e]	Not apparent
Hatchery influence	High	Moderate	Low

[a] Census size N can be used if direct estimates of effective size N_e are not available, assuming $N_e / N = 0.2$.
[b] Decline within last two generations to annual run size ≤ 500 spawners, or run size > 500 but declining at ≥ 10% per year over the past 10 years. Historically small but stable population not included.
[c] Run size has declined to ≤ 500, but now stable.
[d] Catastrophes occurring within the last 10 years.
[e] Decline < 90% but biologically significant.

Table 9. Central Valley Spring-run Chinook salmon--preliminary population structure, historic and most recent natural production, spawner abundance, and hatchery contributions (Good et al. 2005, USFWS and Reclamation 2007, Williams et al. 2011).

Population	Historic Natural Production (1967 – 1991)	Most Recent Natural Production[1] (2000 – 2006)	Most Recent Spawner Abundance[2] (2000- 2006)	Hatchery Abundance Contributions
Butte Creek	1,000	6,516 – 19,809	4,118 – 10,625	< 1%[3]
Deer Creek	3,300	1,387 – 3,461	637 – 2,759	Unknown
Mill Creek	2,200	1,184 – 26,190	544 – 1594	Unknown
Sacramento River	29,000	0 – 1,134	0 – 394	Unknown
Total	Estimated historic abundance: ~700,000 for all populations	11,403 – 26,190	5,370 – 14,044	Unknown

[1] Includes catches
[2] *i.e.*, escapement
[3] (Williams et al. 2011)

The Central Valley drainage supported spring-run Chinook salmon runs as large as 700,000 fish between the late 1880s and the 1940s (Brown et al. 1994). Before construction of Friant Dam, nearly 50,000 adults were counted in the San Joaquin River alone (Fry 1961).

Median natural production of spring-run Chinook salmon from 1970 to 1989 was 30,220 fish. In the 1990s, the population experienced a substantial production failure with an estimated natural production ranging between 3,863 and 7,806 fish (with the exception of 1995 which had a natural production of an estimated 35,640 adults) during the years between 1991 and 1997 (USFWS and Reclamation 2007). Numbers of naturally produced fish increased significantly in 1998 to an estimated 48,755 adults and estimated natural production has remained above 10,000 fish since then (USFWS and Reclamation 2007).

In previous Opinions, NMFS reported that the Sacramento River trends and lambda show a long- and short-term negative trend and negative population growth (Good et al. 2005). NMFS also reported the median production of Sacramento River tributary populations increased from a low of 4,248 with only one year exceeding 10,000 fish before 1998 to a combined natural production of more than 10,000 spring-run Chinook in all years after 1998 (data from (USFWS and Reclamation 2007)). It was then stated that time series data for Mill, Deer, Butte, and Big Chico Creeks spring-run Chinook salmon (updated through 2006) showed that all three tributary spring-run Chinook populations have long-and short-term lambdas >1; indicating population growth (Good et al. 2005). Finally we reported that although the populations are small, CV spring-run Chinook salmon have some of the highest population growth rates in the Central Valley.

Currently Williams et al. (2011) report that the escapement of CV spring-run Chinook salmon to various areas of the Central Valley, with few exceptions, have declined over the past 10 years, in particular since 2006. The recent declines in abundance place Mill and Deer Creek populations in the high risk category due to their rate of decline, and in the case of Deer Creek, also the level of escapement. Butte Creek continues to satisfy the criteria for low extinction risk, although the rate of decline is close to triggering the population decline criterion for high risk (Williams et al. 2011). Overall, the recent declines have been significant but not severe enough to qualify as a catastrophe under the criteria of Lindley et al (2007). On a positive note, spring-run Chinook salmon appear to be repopulating Battle Creek, home to an historical independent population in the Basalt and Porous Lava diversity group that was extirpated for many decades. This population has increased in abundance to levels that would qualify it for a moderate extinction risk score (Williams et al. 2011). Similarly, the spring-run Chinook salmon population in Clear Creek has been increasing, although Lindley et al. (2004b) classified this population as a dependent population, and thus is not expected to exceed the low-risk population size threshold of 2500 fish.

In order to determine the current likelihood of the spring-run ESU becoming viable, we used the historical population structure of spring-run presented in Lindley *et al*. (2007) and McElhany et al.'s (2000) description of viable salmonid populations (VSP) for evaluating populations. While McElhany *et al.* (2000) introduced and described the concept of VSP, Lindley *et al*. (2007) applied the concept to the spring-run ESU. Lindley *et al*. (2004b) identified 26 historical populations within the spring-run ESU; 19 were independent populations, and 7 were dependent populations. Of the 19 independent populations of spring-run that occurred historically, only three remain, in Deer, Mill, and Butte creeks. Extant dependent populations occur in Battle, Antelope, Big Chico, Clear, Beegum, and Thomes creeks, as well as in the Yuba River, the Feather River below Oroville Dam, and in the mainstem Sacramento River below Keswick Dam.

Table 8 provides various quantitative criteria to evaluate the risk of extinction. The following provides the evaluation of the likelihood of the threatened spring-run ESU becoming viable based on the VSP parameters of population size, population growth rate, spatial structure, and diversity.

6.3.2.3. *Population Size*

As discussed above, Spring-run Chinook numbers declined drastically in the mid to late 1980s before stabilizing at very low levels in the early to mid 1990s. Since the late 1990s, there does not appear to be a trend in basin-wide abundance, having fluctuated from approximately 25,000 fish in 1999 to slightly more than 10,000 fish in 2008. Abundance is generally dominated by the Butte Creek population. Other independent and dependent populations are smaller. The cohort replacement rate behaved similarly, falling below 1.0 in the 3 of the previous 4 years, in parallel with the reduced escapement numbers. The 5-year moving average cohort replacement rate, however, has remained above 1.0 since 1995.

6.3.2.4. *Population Growth Rate*

Cohort replacement rates are indications of whether a cohort is replacing itself in the next generation. As mentioned in the previous subsection, the cohort replacement rate since the late 1990s has fluctuated, and does not appear to have a pattern. Since the cohort replacement rate is a reflection of population growth rate, there does not appear to be an increasing or decreasing trend. The 5-year moving average of population estimate indicated an increasing population trend since the mid-1990s until very recently (2006), at which point the population has decreased in two consecutive years. Good et al., (2005)
report lambda (λ) values slightly above replacement (1.000) for each of the three main spawning areas (Table 6).

6.3.2.5. *Spatial Structure*

As stated above, Lindley *et al*. (2007) indicated that of the 19 independent populations of spring-run that occurred historically, only three (Butte, Mill, and Deer creeks) remain, and their current distribution makes the spring-run ESU vulnerable to catastrophic disturbance. Butte, Mill, and Deer Creeks all occur in the same biogeographic region (diversity group), whereas historically, independent spring-run populations were distributed throughout the CV among at least three diversity groups (*i.e*., basalt and porous lava, northern Sierra Nevada, and southern Sierra Nevada). In addition, dependent spring-run populations historically persisted in the Northwestern California diversity group (Lindley et al. 2004b). Currently, there are dependent populations of spring-run in the Big Chico, Antelope, Clear, Thomes, Battle, and Beegum creeks, and in the Sacramento, Feather, and Yuba rivers. The extant Feather River and mainstem Sacramento River populations probably do not represent historical populations (Lindley et al. 2007).

6.3.2.6. *Diversity*

Diversity, both genetic and behavioral, provides a species the ability to react to sudden environmental changes. As a species' abundance decreases, and spatial structure of the ESU is reduced, a species has less flexibility to respond to changes in the environment. Spring-run have been entirely extirpated from the basalt and porous lava region and the southern Sierra Nevada region. The only viable and independent populations (*i.e*., Mill, Deer, and Butte creeks) of spring-run are limited to the northern Sierra Nevada region, and a few ephemeral or dependent populations are found in the Northwestern California region. A single catastrophe, for example, the eruption of Mount Lassen, a large wildland fire at the headwaters of Mill, Deer, and Butte creeks, or a drought, poses a significant threat to the extinction risk of the ESU that otherwise would not be there if the ESU's spatial structure and diversity were greater. Spring-run do reserve some genetic and behavioral variation in that in any given year, at least two cohorts are in the marine environment, and therefore, not exposed to the same environmental stressors as their freshwater cohorts.

Spring-run Chinook produced at the FRFH are part of the spring-run ESU (June 28, 2005, 70 FR 37160), and they compromise the genetic diversity of naturally-spawned spring-run. More than 523,000 FRFH spring-run fry were planted at the base of Whiskeytown Dam during the 3-year period 1991−1993 (CDFG 1998b). The fact that these hatchery fish behave more like fall-run (spawn later than spring-run in Deer, Mill, and Butte creeks), likely increases introgression of the spring- and fall- runs, and reduces diversity.

Until recently NMFS was unaware of any current reports of hatchery-origin fish spawning in the higher elevation areas of Butte, Deer, or Mill creeks utilized by spring-run Chinook. However, in 2010, McReynolds, CDFG, reported 10 coded-wire tags of Feather River spring Chinook Salmon were recovered from a sample of 1,113 carcasses in the upper reach of Butte Creek (Williams et al. 2011). As 100% of Feather River hatchery spring-run Chinook are marked and tagged, this translates into slightly less than 1% of the Butte Creek returns (Table 9**)** being compromised with hatchery strays. This is well below the 10% allowable stray rate for out-of-diversity-group-origin fish within one generation as described in Williams et al. (2011).

6.3.2.7. *Summary of Current Viability*

As reported in Williams et al. (2011), the status of CV spring-run Chinook salmon has probably deteriorated on balance since the 2005 status review (Good et al. 2005), and Lindley et al.'s (2007) assessment, with two of the three extant populations slipping from low or moderate extinction risk to high extinction risk. Butte Creek and Deer Creek spring-run are at low risk of extinction, satisfying both the population viability analysis (PVA) and other viability criteria. However, continued documentation of hatchery strays to Butte Creek could put it on the verge of being moved to the high risk category. Mill Creek is at moderate extinction risk according to the PVA, but appear to satisfy the other viability criteria for low-risk status (Lindley et al. 2007). As a whole, the spring-run fail the representation and redundancy rule for ESU viability. The current distribution of independent populations has been severely constricted to only one of their former geographic diversity groups. Therefore, the spring-run ESU is at moderate risk of extinction in 100 years.

6.3.2.8. *Critical Habitat*

NMFS designated critical habitat for this species on September 2, 2005 (70 FR 52488).
The critical habitat boundary includes the Sacramento River and several tributaries from the Big Chico tributary with Sacramento River upstream to Shasta Dam (Table 10).

There are 38 occupied HSA watersheds within the freshwater and estuarine range of this ESU. As shown in Figure 6, seven watersheds received a low rating, 3 received a medium rating, and 27 received a high rating of conservation value to the ESU (NMFS 2005). Four of these HSA watersheds comprise portions of the San Francisco-San Pablo-Suisun Bay estuarine complex which provides rearing and migratory habitat for this ESU.

The current condition of PCEs of the CV Spring-run Chinook salmon critical habitat indicates that PCEs are not currently functioning or are degraded; their conditions are likely to maintain a low population abundance across the ESU. Spawning and rearing PCEs are degraded by high water temperature caused by the loss of access to historic spawning areas in the upper watersheds which maintained cool and clean water throughout the summer. The rearing PCE is degraded by floodplain habitat being disconnected from the mainstem of larger rivers throughout the Sacramento River watershed, thereby reducing effective foraging. Migration PCE is degraded by lack of natural cover along the migration corridors. Juvenile migration is obstructed by water diversions along Sacramento River and by two large state and federal water-export facilities in the Sacramento-San Joaquin Delta.

Table 10. CV Spring-run Chinook salmon CALWATER HSA watersheds with conservation values.

HUC 4 Subbasin	HUC 5 Watershed conservation Value (CV)					
	High CV	PCE(s) [1]	Medium CV	PCE(s) [1]	Low CV	PCE(s) [1]
San Francisco Bay	San Francisco Bay	Estuary PCEs	0	0	1	Estuary PCEs
Suisun Bay	Suisun Bay	1	0	0	0	
Tehama	1	(1, 2, 3)	1	(1, 2, 3)	0	
Whitmore	1	(1, 2, 3)	0		2	(1, 2, 3)
Redding	2	(1, 2, 3)	0		0	
Eastern Tehama	4	(1, 2, 3)	0		0	
Sacramento Delta	1	(2, 3, 1)	0		0	
Valley Putah-Cache	1	(1, 2, 3)	0		0	
Marysville	3	(1, 2, 3)	0		0	
Yuba River	2	(1, 2, 3)	1	(1, 2, 3)	1	(1, 2, 3)
Valley-American	2	(1, 2, 3)	0		0	
Colusa Basin	4	(1, 2, 3)	0		0	
Butte Creek	1	(1, 2, 3)	0		0	
Ball Mountain	0		0		1	(1, 2, 3)
Shasta Bally	3	(1, 2, 3)	0		1	(1, 2, 3)
North Diablo Range	0		1	(1, 2, 3)	0	
San Joaquin Delta	0		0		1	(1, 2, 3)
Total	28		3		7	

1 Numbers in parenthesis refers to the dominant (in river miles) PCE(s) within the HUC 5 watersheds. PCE 1 is spawning and rearing, 2 is rearing and migration, and 3 is migration and presence. PCEs with < means that the number of river miles of the PCE is much less than river miles of the other PCE.

Contaminants from agriculture and urban areas have degraded rearing and migration PCEs to the extent that they have lost their functions necessary to serve their intended role to conserve the species. Water quality impairments in the designated critical habitat of this ESU include inputs from fertilizers, insecticides, fungicides, herbicides, surfactants, heavy metals, petroleum products, animal and human sewage, sediment in the form of turbidity, and other anthropogenic pollutants. Pollutants enter the surface waters and riverine sediments as contaminated stormwater runoff, aerial drift and deposition, and via point source discharges. Some contaminants such as mercury and pentachlorophenol enter the aquatic food web after reaching water and may be concentrated or even biomagnified in salmon tissue.

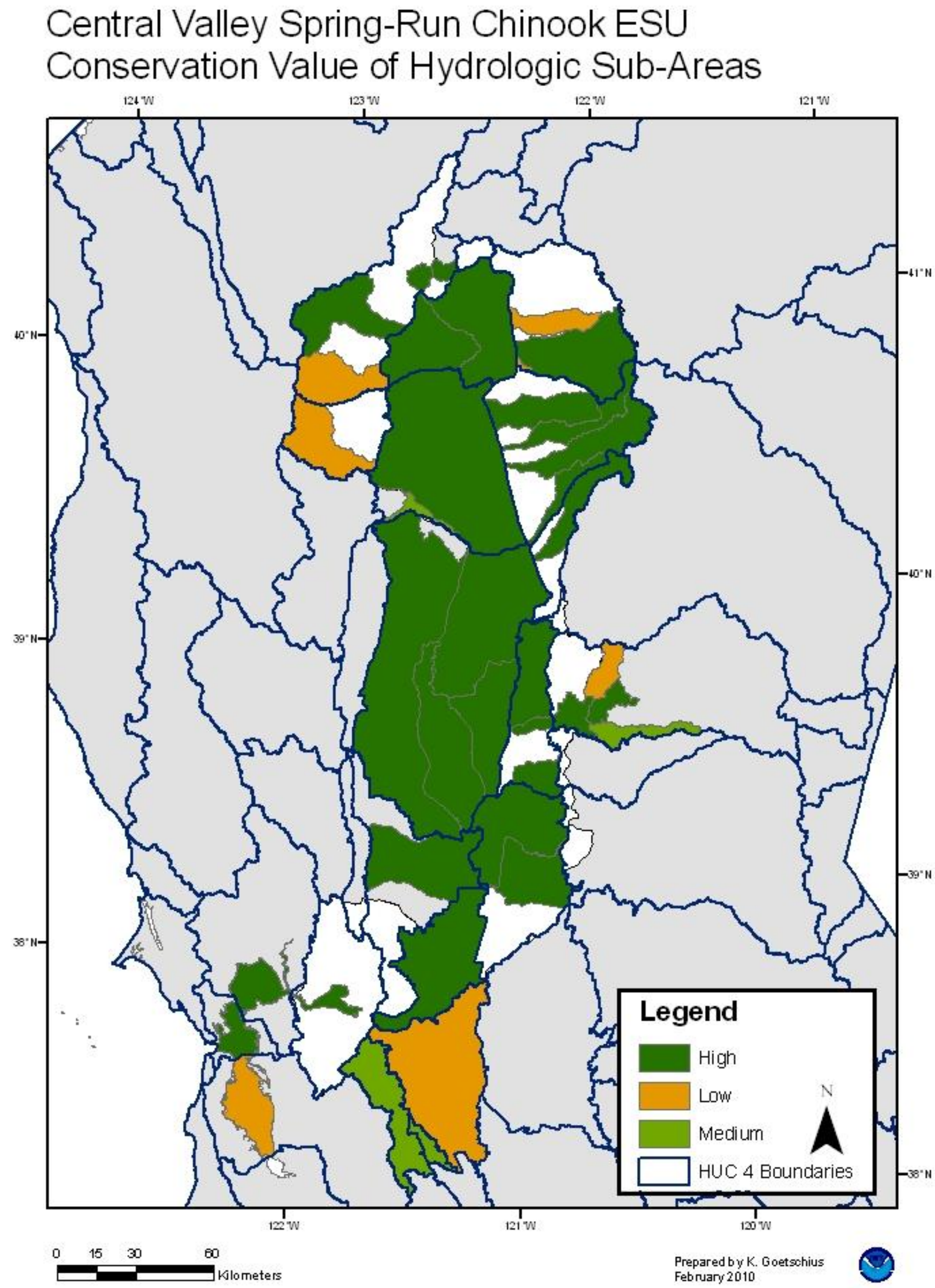


Figure 6. Central Valley Spring-run Chinook salmon Conservation Values per Sub-Area.

6.3.3. Sacramento River Winter-run Chinook Salmon

The ESU includes all winter-run Chinook salmon entering and using the Sacramento River system in the Central Valley, California. The ESU boundary extends from the Carquinez Strait by the City of Vallejo and

Benicia upstream to the Sacramento River, including all its tributaries, to below Keswick Dam (Figure 7). The ESU now consists of a single spawning population.

6.3.3.1. *Life History*

The distribution of winter-run spawning and most rearing is limited to the upper Sacramento River and its tributaries, where spring-fed streams provide cold water throughout the summer, allowing for spawning, egg incubation, and rearing during the mid-summer period (Yoshiyama et al. 1998). The headwaters of the McCloud, Pit, and Little Sacramento rivers, and Hat and Battle creeks, historically provided clean, loose gravel; cold, well-oxygenated water; and optimal stream flow in riffle habitats for spawning and incubation. These areas also provided the cold, productive waters for egg and fry development and survival, and juvenile rearing over the summer.

The upper Sacramento River is the only spawning area used by winter-run, although occasional strays have been reported in Battle and Clear Creek. In recent years, the majority of winter-run (i.e., > 50 percent since 2007) spawn in the area from Keswick Dam downstream to the ACID Dam (approximately 5 miles). Keswick Dam re-regulates flow from Shasta Dam and mixes it with water diverted from the Trinity River through the Spring Creek tunnel. When the gates are down at RBDD, or flashboards in at the ACID Dam, access to the upper Sacramento River basin, including tributaries, can only be achieved through the RBDD and ACID Dam fish ladders.

The winter-run Chinook salmon have characteristics of both stream- and ocean-type races (Healey 1991). Adults enter fresh water in winter or early spring but delay spawning most years until May and June. During the 4 to 6 week period when alevins remain in the gravel, they utilize their yolk-sac to nourish their bodies. As their yolk-sac is depleted, fry begin to emerge from the gravel to begin exogenous feeding in their natal stream. Fry typically range from 25 mm to 40 mm at this stage (Groot and Margolis 1991, Healey 1991). Upon emergence, fry swim or are displaced downstream (Healey 1991). The post-emergent fry disperse to the margins of their natal stream, seeking out shallow waters with slower currents, finer sediments, and bank cover such as overhanging and submerged vegetation, root wads, and fallen woody debris, and begin feeding on zooplankton, small insects, and other micro-crustaceans. Some fry may take up residence in their natal stream for several weeks to a year or more, while others are displaced downstream by the stream’s current. Once started downstream, fry may continue downstream to the estuary and rear there, or may take up residence in river reaches farther downstream for a period of time ranging from weeks to a year (Healey 1991). Fry seek nearshore habitats containing riparian vegetation and associated substrates important for providing aquatic and terrestrial invertebrates, predator avoidance, and slower velocities for resting. The benefits of shallow water habitats for salmonid rearing have been found to be more productive than the main river channels, supporting higher growth rates, partially due to higher prey consumption rates, as well as favorable environmental temperatures (Sommer et al. 2001). Fry emerge from the gravel in late June to early July and continue through October (Fisher 1994). As stated above, some offspring move downstream as fry (ocean-type) while others rear in the upper Sacramento River and move down as smolt (river-type). Monitoring programs have shown that young winter-run Chinook salmon fry start migrating to sea and may migrate past the Red Bluff Diversion Dam (RBDD) as early as mid-July with a peak movement over the RBDD in September (Vogel and Marine 1991, NMFS 1997, Bajjaliya and Vincik 2008b, 2010a, USFWS and Reclamation 2007). Normally fry have passed the RBDD by October, while smolts may pass over the RBDD until March (Table 11). Juvenile winter-runs occur in the Delta beginning in October, primarily from November through February, but can continue until early May. Winter-run juveniles remain in the Delta until they are from 5 to 10 months of age, and then begin emigrating to the ocean as early as November and continue through May (Fisher 1994, Myers et al. 1998). As with all runs, the timing of migration each year may vary somewhat due to changes in river flows, dam operations, and type of water year. The winter-run race matures between two and six years of age with the majority returning as three-year olds.

Table 11. Temporal occurrence of adult and juvenile winter-run Chinook salmon in the Sacramento River. Darker shades indicate months of greatest relative abundance.

Adult migration												
Location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Sac. River basin[a]												
Upper Sac. R. mainstem[b]												
Juvenile migration												
Location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Sac. River at RBDD[c]												
Sac. River at KL[d]												
Lower Sac. River[e]												
North Delta[e]												
RBDD=Red Bluff Diversion Dam KL = Knights Landing												
Relative Abundance:	= High			= Medium				= Low				

[a] Yoshiyama et al. (1998); [b]Moyle (2002); [c] Martin et al. (2001); [d]Snider and Titus (2000b) and Bajjaliya and Vincik (2008b); [e] USFWS (2010a, 2012b)

6.3.3.2. *Status and Trends*

NMFS listed Sacramento River winter-run Chinook salmon as endangered on January 4, 1994 (59 FR 440), and reaffirmed their endangered status on June 28, 2005 (70 FR 37160). The winter-run Chinook salmon spawned and reared in the upper Sacramento River and its tributaries (Slater 1963b, Yoshiyama et al. 1998). Today the Shasta Dam eliminates access to the historic spawning habitat. Cold water releases from Shasta Dam have also created conditions suitable for winter-run spawning and rearing in a 60- to 100-mile long portion of the Sacramento River downstream of the dam. However, spawning only occurs from Keswick Dam to approximately 44 miles downstream (Yoshiyama et al. 1998). As a result, the Sacramento River winter-run Chinook salmon has been reduced to a single spawning population confined to a portion of the main stem Sacramento River.

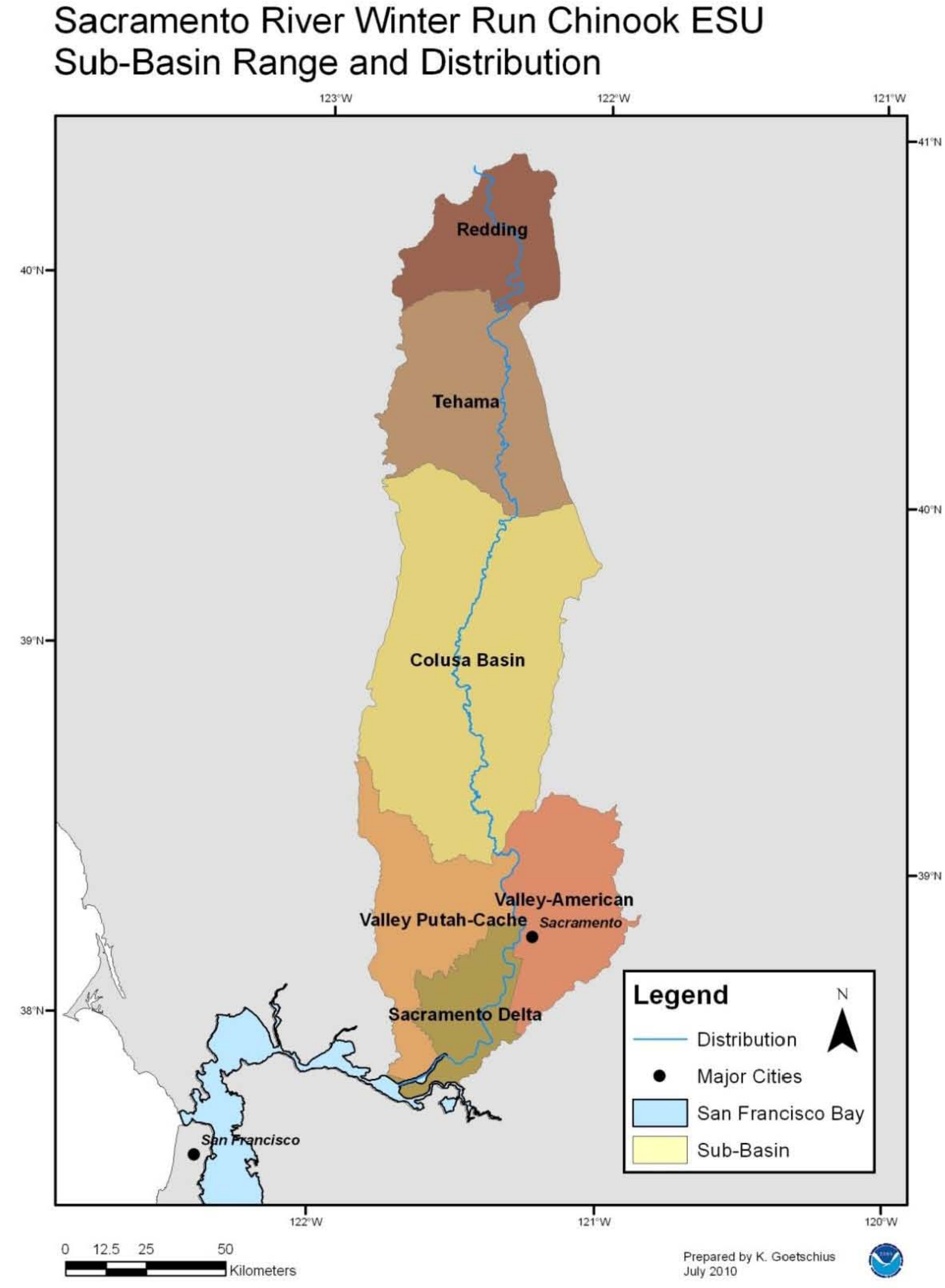


Figure 7. Sacramento River Winter-run Chinook salmon distribution.

Winter-runs may have been as large as 200,000 fish based upon commercial fishery records from the 1870s (Fisher 1994). During the first three years of operation of the counting facility at the RBDD (From 1967 to 1969), an average of 86,500 winter-run Chinook salmon were counted with a high of over 230,000 adults (CDFG 2008, Good et al. 2005). A rapid decline occurred from 1969 to 1979 after completion of the RBDD. Over the next 20 years, the population eventually reached a low point of only 186 adults in 1994. At that point, winter-run was at a high risk of extinction, as defined in the guideline for recovery of CV salmonids (Lindley et al. 2007). If not for a very successful captive broodstock program, construction of a temperature control device on Shasta Dam, changing operational procedures at the RBDD (keeping the gates up for most of the year), and restrictions in ocean harvest, the population would have likely failed to exist in the wild.

The population grew rapidly from the early 1990s to mid-2005. Mean run size increased from 1,363 before 2000 with all runs estimated to less than 10,000 fish to an average run of 8,470 adults between 2000 and 2006 with two runs estimated to more than 10,000 fish (USFWS and Reclamation 2007). However, the natural produced winter-run Chinook salmon plunged in 2007 and 2008, with 4,461 adults estimated for 2007 and a preliminary estimate between of 2,600-2,850 adults for 2008 (USFWS 2008).

The Sacramento River winter-run Chinook salmon is expected to have lost some genetic diversity through bottleneck effects in the late 1980s and early 1990s. Hatchery releases may also have affected population genetics. The loss of natural spawning habitat and hydrological conditions has further removed the natural evolutionary processes that maintained the unique winter-run life history.

The status of winter-run is typical of most endangered species populations, that is, a sharp downward decline followed by years of low abundance (Figure 8). Lindley et al. (2004b) identified four historical populations within the winter-run ESU, all independent populations, defined as those sufficiently large to be historically viable in isolation and whose demographics and extinction risk were minimally influenced by immigrants from adjacent populations (McElhany et al. 2000). All four independent populations, however, are extinct in their historical spawning ranges. Three (Little Sacramento; Pit, Fall, Hat; and McCloud River) are blocked by the impassable Keswick and Shasta dams (Lindley et al. 2004b), and Battle Creek independent population is no longer self-sustaining (Lindley et al. 2007). Lindley et al. (2007) provided various quantitative criteria to evaluate the risk of extinction (Table 8). A population must meet all the low-risk thresholds to be considered viable. The following provides the evaluation of the likelihood of winter-run becoming viable based on the

VSP parameters of size, population growth rate, spatial structure, and diversity. These specific parameters are important to evaluate because they are predictors of extinction risk, and the parameters reflect general biological and ecological processes that are critical to the growth and survival of salmon (McElhany et al. 2000).

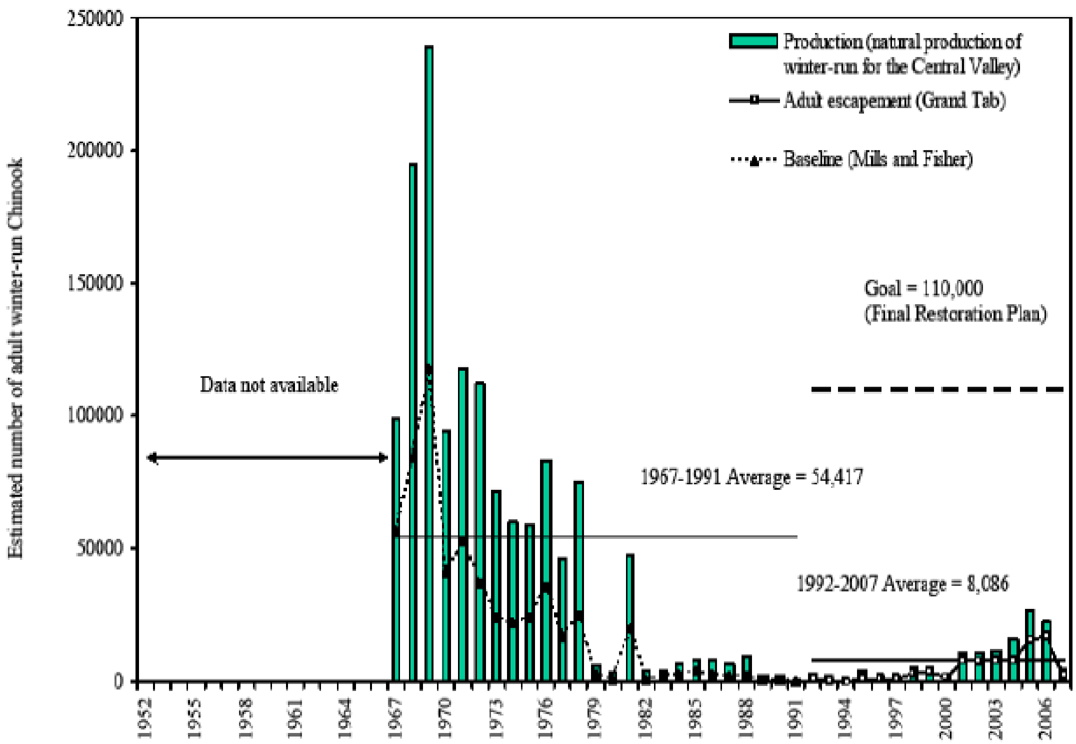


Figure 8. Estimated yearly adult natural production and in-river adult escapement of winter-run from 1967 - 2007 based on RBDD ladder counts.

In a recent Biological Opinion on the long-term operations of the Central Valley Project and State Water Project (NMFS 2009a), NMFS concluded for all four VSP parameters that winter-run Chinook salmon are at a high risk of extinction. These parameters are summarized below.

6.3.3.3. *Population Size*

Information about population size provides an indication of the type of extinction risk that a population faces. For instance, smaller populations are at a greater risk of extinction than large populations because the processes that affect populations operate differently in small populations than in large populations (McElhany et al. 2000). One risk of low population sizes is depensation. Depensation occurs when populations are reduced to very low densities and per capita growth rates decrease as a result of a variety of mechanisms [e.g., failure to find mates and therefore reduced probability of fertilization, failure to saturate predator populations (Liermann and Hilborn 2001). The winter-run population as represented by the 5-year moving average from adult escapement was following an increasing trend from the mid-1990s until 2006. In 2007, the winter-run population declined precipitously. Low adult escapement was repeated in 2008. Likewise, the 5-year moving average cohort replacement rate was relatively stable since the late 1990s, with each cohort approximately doubling in size. However, the cohort replacement of 6.08 in 2003 buffered the effect of the significant decline in the cohort replacement rate of 0.32 in 2007. In consideration of the almost 7-fold decrease in population in 2007, and the dry years that have followed, NMFS concludes that the winter-run are at high risk of extinction based on population size.

6.3.3.4. *Population Growth Rate*

The productivity of a population (*i.e.,* production over the entire life cycle) can reflect conditions (*e.g.,* environmental conditions) that influence the dynamics of a population and determine abundance. In turn, the productivity of a population allows an understanding of the performance of a population across the landscape and habitats in which it exists and its response to those habitats (McElhany et al. 2000). In general, declining productivity equates to declining population abundance. McElhany *et al.*(2000) suggested a population's natural productivity should be sufficient to maintain its abundance above the viable level (a stable or increasing population growth rate). This guideline seems reasonable in the absence of numeric abundance targets. Winter-run Chinook salmon have declined substantially from historic levels. The one remaining population of winter-run on the mainstem Sacramento River is also the entire current ESU. Good *et al.* (2005) reported the medium population growth rate at slightly under 1.000 (Table 6). Although the population growth rate (indicated by the cohort replacement rate λ) increased since the late 1990s, it drastically decreased in 2007 and 2008, indicating that the population is not replacing itself, and is at a high risk of extinction in the foreseeable future.

6.3.3.5. *Spatial Structure*

In general, there is less information available on how spatial processes relate to salmonid viability than there is for the other VSP parameters (McElhany et al. 2000). Understanding the spatial structure of a population is important because the population structure can affect evolutionary processes and, therefore, alter the ability of a population to adapt to spatial or temporal changes in the species' environment (McElhany et al. 2000). The spatial structure of Sacramento River winter-run Chinook salmon resembles that of a panmictic population, where there are no subpopulations, and every mature male is equally likely to mate with every other mature female. The four historical independent populations of winter-run have been reduced to one population, resulting in a significant reduction in their spatial diversity. An ESU comprised of one population is not viable because it is unlikely to be able to adapt to significant environmental changes. A single catastrophe (*e.g.*, volcanic eruption of Lassen Peak, prolonged drought which depletes the cold water pool at Lake Shasta, or some related failure to manage cold water storage, spill of toxic materials, or a disease outbreak) could extirpate the entire winter-run ESU if its effects persisted for 3 or more years. The majority of winter-run return to spawn in 3 years, so a single catastrophe with effects that persist for at least 3 years would affect all of the winter-run cohorts. Therefore, NMFS concludes that winter-run are at a high risk of extinction based on spatial structure.

6.3.3.6. *Diversity*

Diversity, both genetic and behavioral, is critical to success in a changing environment. Salmonids express variation in a suite of traits, such as anadromy, morphology, fecundity, run timing, spawn timing, juvenile behavior, age at smolting, age at maturity, egg size, developmental rate, ocean distribution patterns, male and female spawning behavior, and physiology and molecular genetic characteristics. The more diverse these traits (or the more these traits are not restricted), the more adaptable a population is, and the more likely that individuals, and therefore the species, would survive and reproduce in the face of environmental variation (McElhany et al. 2000). However, when this diversity is reduced due to loss of entire life history strategies or to loss of habitat used by fish exhibiting variation in life history traits, the species is in all probability less able to survive and reproduce given environmental variation.

The primary factor affecting the diversity of winter-run is the limited area of spawning habitat available on the main stem Sacramento River downstream of Keswick Dam. This specific and narrow spawning habitat limits the flexibility and variation in spawning locations for winter-run to tolerate environmental variation. For example, a catastrophe on the main stem Sacramento River could affect the entire population, and therefore, ESU. However, with the majority of spawners being 3 years old, winter-run do reserve some genetic and behavioral variation in that in any given year, two cohorts are in the marine environment, and therefore, not exposed to the same environmental stressors as their freshwater cohorts. Although the Livingston Stone

National Fish Hatchery (LSNFH) is characterized as one of the best examples of a conservation hatchery operated to maximize genetic diversity and minimize domestication of the offspring produced in the hatchery, it still faces some of the same diversity issues as other hatcheries in reducing the diversity of the naturally-spawning population. Therefore, Lindley *et al*. (2000) characterizes hatchery influence as a looming concern with regard to diversity. Even with a small contribution of hatchery fish to the natural spawning population, hatchery contributions could compromise the long term viability and extinction risk of winter-run. NMFS concludes that the current diversity in this ESU is much reduced compared to historic levels, and that winter-run are at a high risk of extinction based on the diversity VSP parameter.

6.3.3.7. *Critical Habitat*

NMFS designated critical habitat for this species on June 16, 1993 (58 FR 33212). The designated critical habitat for winter-run includes the Sacramento River from Keswick Dam (RM 302 to Chipps Island (RM 0) at the westward margin of the 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 Bidge; and all waters of San Francisco Estuary to the Golden Gate Bridge north of the San Francisco/Oakland Bay Bridge. In the Sacramento River, critical habitat includes the river water column, river bottom, and adjacent riparian zone (limited to those areas above a streambank that provide cover and shade to the nearshore aquatic areas) used by fry and juveniles for rearing. In the area westward of Chipps Island, critical habitat includes the estuarine water column and essential foraging habitat and food resources used by winter-run as part of their juvenile emigration or adult spawning migration.

In designating critical habitat, NMFS considers the following requirements of the species: (1) space for individual and population growth, and for normal behavior; (2) food, water, air, light, minerals, or other nutritional or physiological requirements; (3) cover or shelter; (4) sites for breeding, reproduction, or rearing offspring; and generally, (5) habitats that are protected from disturbance or are representative of the historic geographical and ecological distributions of a species [see 50 CFR 424.12(b)]. In addition to these factors, NMFS also focuses on the known physical and biological features (essential features) within the designated area that are essential to the conservation of the species and that may require special management considerations or protection. These essential features may include, but are not limited to, spawning sites, food resources, water quality and quantity, and riparian vegetation and other natural cover.

NMFS identified specific water temperature criteria, minimum instream flow criteria, and water quality standards as essential physical features (PCEs) of the ESU's habitat for species conservation. In addition, biological features vital for the Sacramento River winter-run Chinook salmon include unimpeded adult upstream migration routes, spawning habitat, egg incubation and fry emergence areas, rearing areas for juveniles, and unimpeded downstream migration routes for juveniles.

The designated critical habitat of this ESU has not been evaluated for the conservation value of individual subbasins or river sections. However, since spawning, rearing, and migration of the winter-run race is restricted to the mainstem of the Sacramento River, the entire Sacramento River is considered of high conservation value. The Delta is similarly considered of high conservation value for rearing and migration.

As there is overlap in designated critical habitat for both the Sacramento River winter-run Chinook salmon and the spring-run Chinook salmon, the conditions of PCEs for both ESUs are similar. The current condition of PCEs for the Sacramento River winter-run Chinook salmon indicates that they are not currently functioning or are degraded. Their conditions are likely to maintain low population abundances across the ESU. Spawning and rearing PCEs are especially degraded by high water temperature caused by the loss of access to historic spawning areas in the upper watersheds where water maintain lower temperatures. The rearing PCE is further degraded by floodplain habitat disconnected (with the exception of the Yolo Bypass floodplain) from the mainstems of larger rivers throughout the Sacramento River watershed. The migration PCE is also degraded by the lack of natural cover along the migration corridors. Rearing and migration PCEs are further affected by pollutants entering the surface waters and riverine sediments as contaminated stormwater runoff, aerial drift and deposition, and via point source discharges. Juvenile migration is obstructed by water diversions along Sacramento River and by two large state and federal water-export facilities in the Sacramento-San Joaquin Delta.

6.4. California Central Valley Steelhead

The California Central Valley (CCV) steelhead DPS includes all naturally spawned steelhead populations below natural and manmade impassable barriers in the Sacramento and San Joaquin Rivers and their tributaries,

excluding steelhead from San Francisco and San Pablo Bays and their tributaries, as well as two artificial propagation programs: the Coleman NFH, and Feather River Hatchery steelhead hatchery programs (
Figure 9).

6.4.1.1. *Life History*

Steelhead can be divided into two life history types, summer-run steelhead and winter-run steelhead, based on their state of sexual maturity at the time of river entry and the duration of their spawning migration, stream-maturing and ocean-maturing. Only winter steelhead are currently found in Central Valley rivers and streams (McEwan and Jackson 1996a). At present, summer steelhead are found only in northern California coast drainages, mostly in tributaries of the Eel, Klamath, and Trinity River systems (McEwan and Jackson 1996a).

The CCV steelhead winter steelhead have the longest freshwater migration of any population of winter steelhead. CCV steelhead generally leave the ocean from August through April (Busby et al. 1996), and spawn from December through April, with peaks from January through March, in small streams and tributaries where cool, well oxygenated water is available year-round (Hallock et al. 1961, McEwan and Jackson 1996a). Most spawning habitat for steelhead in the Central Valley is located in areas directly downstream of dams containing suitable environmental conditions for spawning and incubation. The length of time it takes for eggs to hatch depends mostly on water temperature. Hatching of steelhead eggs in hatcheries takes about 30 days at 51°F. Fry emerge from the gravel usually about 4 to 6 weeks after hatching, but factors such as redd depth, gravel size, siltation, and temperature can affect emergence timing (Shapovalov and Taft 1954). Newly emerged fry move to the shallow, protected areas associated with the stream margin (McEwan and Jackson 1996a). Steelhead rearing during the summer occurs primarily in higher velocity areas in pools, although young of the year also are abundant in glides and riffles. Both spawning areas and migratory corridors comprise rearing habitat for juveniles, which feed and grow before and during their outmigration. Non-natal, intermittent tributaries, and inundated floodplains also may be used for juvenile rearing. On flood years when the Yolo Bypass (floodplain to the Sacramento River) is inundated, steelhead find exceptional rearing and foraging opportunities (Sommer et al. 2001). Steelhead migratory corridors are downstream of the spawning areas and include the lower mainstems of the Sacramento and San Joaquin rivers and the Delta.

Productive steelhead habitat is characterized by complexity, primarily in the form of large and small woody debris. Cover is an important habitat component for juvenile steelhead both as velocity refugia and as a means of avoiding predation (Meehan and Bjornn 1991).

Juvenile steelhead emigrate episodically from natal streams during fall, winter, and spring high flows (Table 12). Emigrating CV steelhead use the lower reaches of the Sacramento River and the Delta for rearing and as a migration corridor to the ocean. Juvenile CV steelhead feed mostly on drifting aquatic organisms and terrestrial insects and will also take active bottom invertebrates (Moyle 2002).

Hallock *et al*. (1961) found that juvenile steelhead in the Sacramento River basin migrate downstream during most months of the year, but the peak period of emigration occurred in March and April, with a much smaller peak in the fall. Emigrating CCV steelhead use the lower reaches of the Sacramento River and the Delta for rearing and as a migration corridor to the ocean. Some juvenile steelhead may use tidal marsh areas, non-tidal freshwater marshes, and other shallow water areas in the Delta as rearing areas for short periods prior to their final emigration to the sea (Hallock et al. 1961). Nobriga and Cadrett (2003) also have verified these temporal findings based on analysis of captures at Chipps Island, Suisun Bay.

Table 12. Temporal occurrence of adult and juvenile CCV steelhead in the Central Valley. Darker shades indicate months of greatest relative abundance.

Adult migration												
Location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Sac. River basin[a,c]												
Sac. R. RBDD[b,c]												
Mill, Deer creeks[d]												
Sac. R. Freemont Weir[e]												
San Joaquin River[f]												
Juvenile migration												
Location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Sac. River at RBDD [a,b]												
Sac. River at KL[b,g,h]												
Chipps Island[i]												
Mossdale[g]												
Woodbridge Dam[g]												
Sac. R. at Hood[j]												

RBDD=Red Bluff Diversion Dam KL = Knights Landing																	
Relative Abundance:		= High						= Medium						= Low			

[a] Hallock et al. (1961); [b]McEwan (2001b); [c] USFWS (unpublished (1995) data); [d] CDFG (1995); [e] Bailey (1954); [f] CDFG Steelhead Report Card Data; [g] CDFG (unpublished data); [h] Snider and Titus (2000b); [i] Nobriga and Cadrett (2003);[j] Schaffter (1980, 1997).

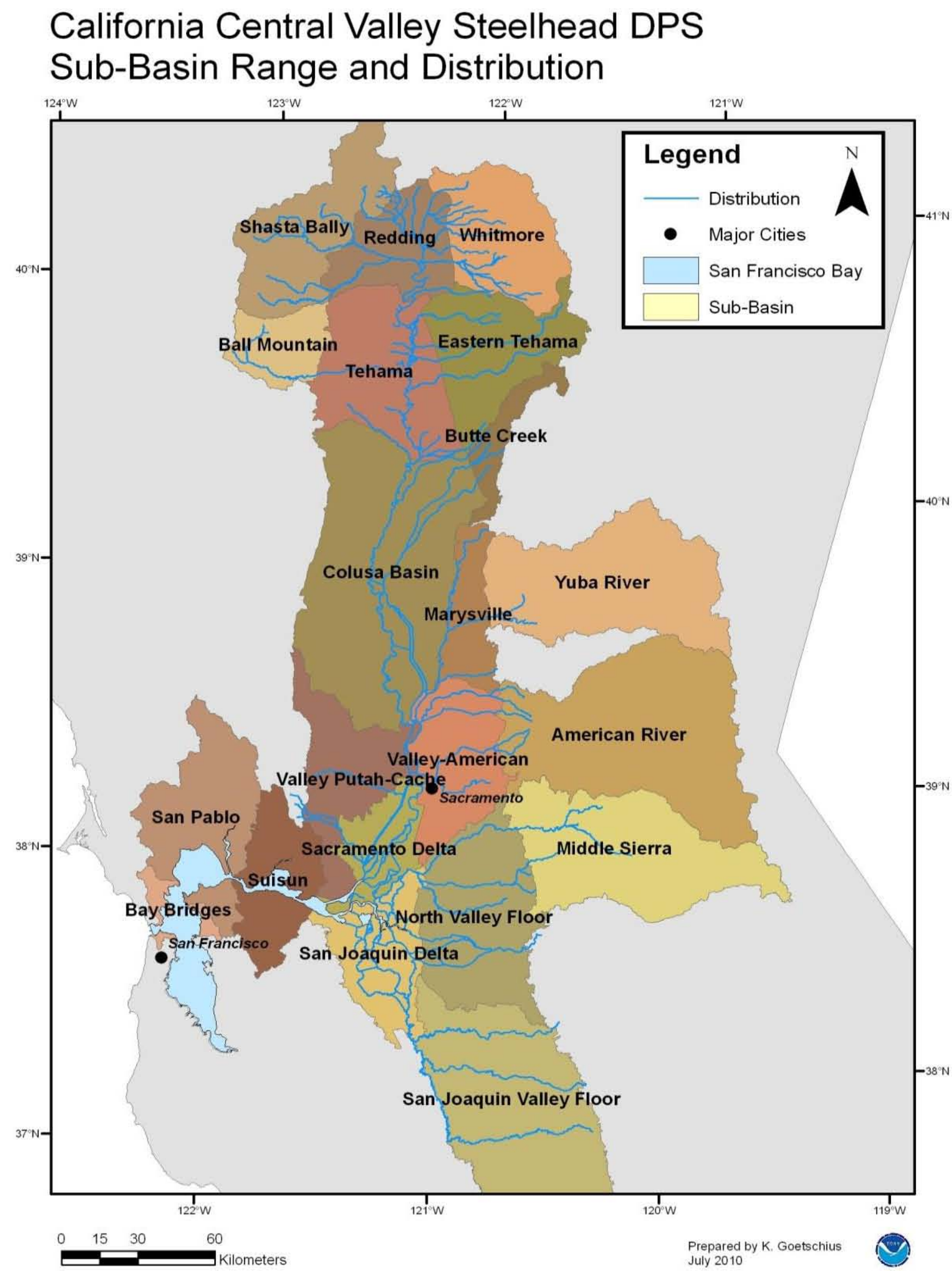


Figure 9. CCV steelhead distribution.

6.4.1.2. *Status and Trends*

NMFS originally listed CCV steelhead as threatened on March 19, 1998, and reaffirmed their threatened status on January 5, 2006 (71 FR 834). The CCV steelhead DPS may have consisted of 81 historical and independent populations (Lindley et al. 2006b). Spatial structure and patchiness strongly influenced suitable habitats being isolated due largely to high summer temperatures on the valley floor.

Over the past 30 years, the naturally-spawned steelhead populations in the upper Sacramento River have declined substantially. Hallock *et al.*(1961) estimated an average of 20,540 adult steelhead through the 1960s in the Sacramento River, upstream of the Feather River. Steelhead counts at the RBDD declined from an average of approximately 8,000 for the period of 1967 to 1977, to an average of approximately 2,000 through the early 1990s, with an estimated total annual run size for the entire Sacramento-San Joaquin system, based on RBDD counts, to be no more than 10,000 adults (McEwan and Jackson 1996a, McEwan 2001b). Steelhead escapement surveys at RBDD ended in 1993 due to changes in dam operations.

Nobriga and Cadrett (2003) compared CWT and untagged (wild) steelhead smolt catch ratios at Chipps Island trawl from 1998 through 2001 to estimate that about 100,000 to 300,000 steelhead juveniles are produced naturally each year in the Central Valley. Good *et al.* (2005) made the following conclusion based on the Chipps Island data:

> "If we make the fairly generous assumptions (in the sense of generating large estimates of spawners) that average fecundity is 5,000 eggs per female, 1 percent of eggs survive to reach Chipps Island, and 181,000 smolts are produced (the 1998-2000 average), about 3,628 female steelhead spawn naturally in the entire Central Valley. This can be compared with McEwan's (2001b) estimate of 1 million to 2 million spawners before 1850, and 40,000 spawners in the 1960s."

The species’ present distribution has been greatly reduced with about 80% of historic habitat lost behind dams and about 38% of habitat patches that supported independent populations are no longer accessible to steelhead (Lindley et al. 2006b). Existing wild steelhead stocks in the Central Valley are mostly confined to the upper Sacramento River and its tributaries, including Antelope, Deer, and Mill Creeks and the Yuba River. Populations may exist in Big Chico and Butte Creeks. A few wild steelhead are produced in the American and

Feather Rivers (Good et al. 2005, McEwan and Jackson 1996a). Steelhead have also been observed in Clear Creek and Stanislaus River (Good et al. 2005, Demko and Cramer 2000). Until recently, steelhead were considered extirpated from the San Joaquin River system. Recent monitoring has detected small self-sustaining populations of steelhead in the Stanislaus, Mokelumne, Calaveras, and other streams previously thought to be void of steelhead (Good et al. 2005). In 2004, a total of 12 steelhead smolts were collected in monitoring trawls at the Mossdale station in the lower San Joaquin River (CDFG unpublished data).

Historic CCV steelhead run size may have approached one to two million adults annually (McEwan 2001b). By the early 1960s, the steelhead run size had declined to about 40,000 adults (McEwan 2001b). Steelhead were counted at the Red Bluff Diversion Dam (RBDD) up until 1993. Counts at the dam declined from an average of 11,187 for the period of 1967 to 1977, to an average of approximately 2,000 through the early 1990s. An estimated total annual run size for the entire Sacramento-San Joaquin system was no more than 10,000 adults during the early 1990s (McEwan and Jackson 1996a, McEwan 2001b). Based on catch ratios at Chipps Island in the Delta and using some generous assumptions regarding survival, the average number of CV steelhead females spawning naturally in the entire Central Valley during the years 1980 to 2000 was estimated at about 3,600 (Good et al. 2005).

CCV steelhead lack annual monitoring data for calculating trends and lambda. However, the RBDD counts and redd counts up to 1993 and later sporadic data show that the DPS has had a significant long-term downward trend in abundance (NMFS 2009a).

The CCV steelhead distribution ranged over a wide variety of environmental conditions and likely contained biologically significant amounts of spatially structured genetic diversity (Lindley et al. 2006b). Thus, the loss of populations and reduction in abundances have reduced the large diversity that existed within the DPS. The genetic diversity of the majority of CCV steelhead spawning runs is also compromised by hatchery-origin fish. In order to determine the current likelihood of the CV steelhead DPS becoming viable, we used the historical population structure of CV steelhead presented in Lindley *et al*. (2006b, 2007) and the concept of VSP for evaluating populations described by McElhany *et al.* (2000), and applied by Lindley et al (2007) to the CV steelhead DPS.

Table 8 provides various quantitative criteria to evaluate the risk of extinction. The following provides the evaluation of the likelihood of the threatened CV steelhead DPS becoming viable based on the VSP parameters of population size, population growth rate, spatial structure, and diversity.

6.4.1.3. *Population Size*

Estimated natural CV steelhead escapement in the upper Sacramento River has declined substantially from 1967 through 1993. There is still a nearly complete lack of steelhead monitoring in the Central Valley (Good et al. 2005), and therefore, data are lacking regarding a definitive population size for CV steelhead. However, the little data that exist indicate that the CV steelhead population continues to decline (Good et al. 2005).

6.4.1.4. *Population Growth Rate*

CV steelhead has shown a pattern of a negative growth rate since the late 1960s. Good *et al*. (2005) provided no indication that this trend has changed since the last CV steelhead population census in 1993. The λ value for CV steelhead is less than 1.000 (Table 6; (Good et al. 2005).

6.4.1.5. *Spatial Structure*

Lindley *et al*. (2006b) identified 81 historical and independent populations within the CV steelhead DPS. These populations form 8 clusters, or diversity groups, based on the similarity of the habitats they occupied for spawning and rearing. About 80 percent of the habitat that was historically available to CV steelhead is now behind impassable dams, and 38 percent of the populations have lost all of their habitats. Although much of the habitat has been blocked by impassable dams, or degraded, small populations of CV steelhead are still found throughout habitat available in the Sacramento River and many of the tributaries, and some of the tributaries to the San Joaquin River.

6.4.1.6. *Diversity*

Diversity, both genetic and behavioral, provides a species the opportunity to respond to sudden environmental changes. CV steelhead naturally experience the most diverse life history strategies of the listed Central Valley anadromous salmonid species. In addition to being iteroparous, they reside in freshwater for 2-4 years before emigrating to the ocean. However, as the species' abundance decreases, and spatial structure of the DPS is reduced, it has less flexibility to respond to changes in the environment. CV steelhead abundance and growth rate continue to decline, largely the result of a significant reduction in the diversity of habitats available to CV steelhead (Lindley et al. 2006b). The genetic diversity of CV steelhead is also compromised by hatchery-origin fish, which likely comprise the majority of the natural spawning run, placing the natural populations at high risk of extinction (Lindley et al. 2007). Consistent with the life history strategy of winter-run and spring-run Chinook salmon, some genetic and behavioral variation is conserved in that in any given year, there are additional cohorts in the marine environment, and therefore, not exposed to the same environmental stressors as their freshwater cohorts.

6.4.1.7. *Critical Habitat*

NMFS designated critical habitat for CCV steelhead on September 2, 2005 (70 FR 52488). Critical habitat includes stream reaches such as those of the Sacramento, Feather, and Yuba Rivers, and Deer, Mill, Battle, and Antelope creeks in the Sacramento River basin; the lower San Joaquin River to the confluence with the Merced River, including its tributaries, and the waterways of the Delta (Figure 10). The total area of critical habitat includes about 2,300 miles of stream habitat and about 250 square miles of estuarine habitat in the San Francisco-San Pablo-Suisan Bay estuarine complex.

There are 67 occupied HAS watersheds within the freshwater and estuarine range of this DPS. Twelve watersheds received a low rating, 18 received a medium rating, and 37 received a high rating of conservation value to the ESU (NMFS 2005). Four of these HSA watersheds comprise portions of the San Francisco-San Pablo-Suisun Bay estuarine complex which provides rearing and migratory habitat for this ESU.

Table 13. CCV spring-run Chinook salmon CALWATER HSA watersheds with conservation values.

HUC 4 Subbasin	HUC 5 Watershed conservation Value (CV)					
	High CV	PCE(s) [1]	Medium CV	PCE(s) [1]	Low CV	PCE(s) [1]
San Francisco Bay	1	2	0		0	
South Bay	0		0		1	2
San Pablo	1	2	0		0	
Suisun Bay	1	2	0		0	
Tehama	1	1, 2, 3	1	1, 2, 3	0	
Whitmore	3	1, 2, 3	2	1, 2, 3	2	1, 2, 3
Redding	2	1, 2, 3	0		0	
Eastern Tehama	4	1, 2, 3	1	1, 2, 3	1	1, 2, 3

HUC 4 Subbasin	HUC 5 Watershed conservation Value (CV)					
	High CV	PCE(s) [1]	Medium CV	PCE(s) [1]	Low CV	PCE(s) [1]
Sacramento Delta	1	1, 2, 3	0		0	
Valley Putah-Cache	0		2	1, 2, 3	0	
American River	0		1	1, 2, 3	0	
Marysville	2	1, 2, 3	1	1, 2, 3	0	
Yuba River	2	1, 2, 3	0		2	1, 2, 3
Valley-American	2	1, 2, 3	0		0	
Colusa Basin	4	1, 2, 3	0		0	
Butte Creek	1	1, 2, 3	1	1, 2, 3	1	1, 2, 3
Ball Mountain	1	1, 2, 3	0		0	
Shasta Bally	2	1, 2, 3	3	1, 2, 3	0	
North Valley Floor	1	1, 2, 3	1	1, 2, 3	1	1, 2, 3
Middle Sierra	0		0		4	1, 2, 3
Upper Calaveras	1	1, 2, 3	0		0	
Stanislaus River	1	1, 2, 3	0		0	
San Joaquin Valley Floor	4	1, 2, 3	3	1, 2, 3	0	
Delta-Mendota Canal	1	1, 2, 3	1	1, 2, 3	0	
North Diablo Range	0		1		0	
San Joaquin Delta	1	1, 2, 3	0		0	
Total	37		18		12	

[1] Numbers in parenthesis refers to the dominant (in river miles) PCE(s) within the HUC 5 watersheds. PCE 1 is spawning and rearing, 2 is rearing and migration, and 3 is migration and presence. PCEs with < means that the number of river miles of the PCE is much less than river miles of the other PCE.

The current condition of CCV steelhead critical habitat is degraded, and does not provide the conservation value necessary for species recovery (Table 13). In addition, the Sacramento-San Joaquin River Delta, as part of CCV steelhead designated critical habitat, provides very little function necessary for juvenile CCV steelhead rearing and physiological transition to salt water.

The spawning PCE is subject to variations in flows and temperatures, particularly over the summer months. Some complex, productive habitats with floodplains remain in the system and flood bypasses (e.g., Yolo and Sutter bypasses). However, the rearing PCE is degraded by the channelized, leveed, and riprapped river reaches and sloughs that are common in the Sacramento-San Joaquin system and which typically have low habitat complexity, low abundance of food organisms, and offer little protection from either fish or avian predators. Stream channels commonly have elevated temperatures.

The current conditions of migration corridors are substantially degraded. Both migration and rearing PCEs are affected by dense urbanization and agriculture along the mainstems and in the Delta which contribute to reduced water quality by introducing several contaminants. In the Sacramento River, the migration corridor for both juveniles and adults is obstructed by the RBDD gates which are down from May 15 through September 15. The migration PCE is also obstructed by complex channel configuration making it more difficult for CCV steelhead to migrate successfully to the western Delta and the ocean. In addition, the state and federal government pumps and associated fish facilities change flows in the Delta which impede and obstruct for a functioning migration corridor that enhance migration. The estuarine PCE, which is present in the Delta, is affected by contaminants from agricultural and urban runoff and release of wastewater treatment plants effluent.

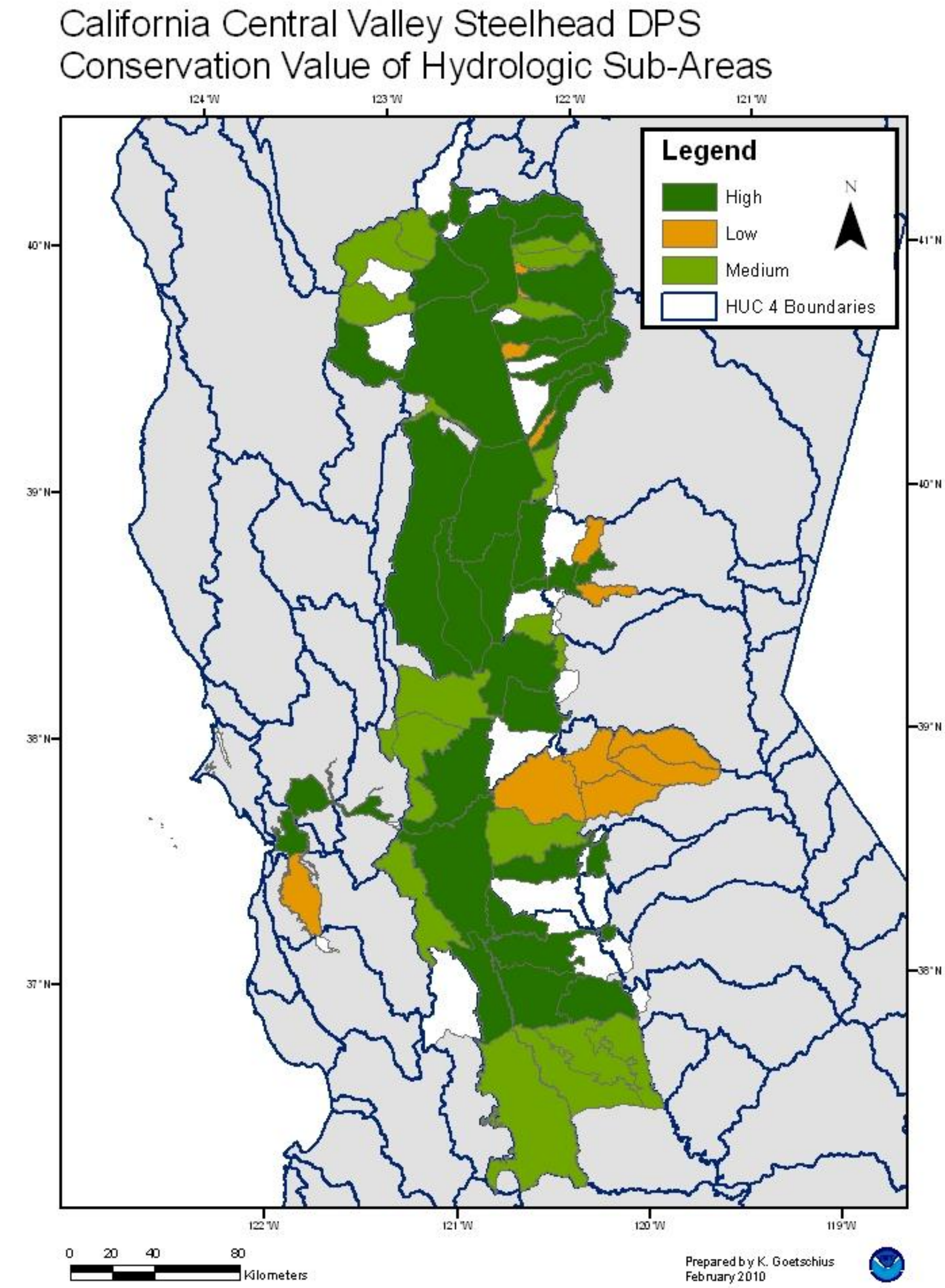


Figure 10. California Central Valley Steelhead Conservation Value per Sub-area.

6.4.1.8. *The Yolo Bypass floodplain habitat*

Sommer et al. (2001, 2004, 2005b) have documented the importance of the Yolo Bypass as beneficial rearing habitat for juvenile salmonids relative to the riverine habitat in the near-by Sacramento River. Indeed, the 24,000-ha Yolo Bypass floodplain is the primary floodplain of the Sacramento River, and of the combined Sacramento-San Joaquin delta. When inundated the Yolo provides significant rearing habitat for all species of Chinook, for steelhead and other fishes. Sommer et al. (2001) estimated that complete inundation of the Yolo Bypass creates a wetted area approximately 10 times larger than the adjacent reach of the Sacramento River, and is equivalent to a doubling of the wetted area of the entire delta portion of the San Francisco Estuary. Much of the floodplain habitat consists of broad shoals composed of soil and vegetation that are typical of the low-velocity conditions selected by young salmon (Everest and Chapman 1972, Healey 1982). Sommer et al. (2001) demonstrated Chinook salmon rearing in the Yolo Bypass floodplain have higher apparent growth rates than those that remain in the Sacramento River channel. The Yolo has higher temperatures and a more diverse and abundant food supply (Sommer et al. 2001).

The Yolo Bypass floods an average of every other year, typically under high flow periods in winter and spring. The Yolo Bypass has a complex hydrology, with inundation possible from several different sources. The floodplain typically has a peak inundation period during January – March but can flood as early as October and as late as June. The primary input to the Yolo Bypass is through the Fremont Weir in the north, which conveys floodwaters from the Sacramento and Feather rivers. During major storm events, additional water enters from the east via the Sacramento Weir, adding flow from the American and Sacramento rivers. Flow also enters the Yolo Bypass from several small streams on its western margin, including Knights Landing, Ridge Cut, Cache Creek, and Putah Creek. Hydraulic residence times are typically longer in the Yolo Bypass than in the Sacramento River (Sommer et al. 2004).

Floodwaters recede from the northern and western portions of the Bypass along relatively even elevation gradients of 0.09 percent west – east and 0.01 percent north – south into a perennial channel on the eastern edge (the “toe-drain”). The receding waters then join the Sacramento River near Rio Vista. The majority of the Yolo Bypass is managed for wildlife in a mosaic that includes riparian, wetland, upland, and perennial pond habitats; however a dominant land use is agriculture, of which rice growing is a significant component.

During typical flood events between January and March, all three listed salmonid species would benefit from the enhanced rearing and growing conditions (as described in Sommer et al. (2001)) the Yolo Bypass provides. Juvenile spring- and winter-run Chinook would be passing through in relatively high abundance in January and medium to low abundance February to March (Table 7 and Table 11). Steelhead would be passing through in medium abundance January – February but would be in relatively high abundance during March (Table 12). Sommer et al. (2001)found salmon in all regions of the floodplain and on all substrate types. Juvenile Chinook were in a high percentage of the samples in each region of the floodplain: (1) Fremont Weir (100 percent, n = 13 samples), (2) Cache Creek Sinks (50 percent, n = 16 samples), (3) Yolo Bypass Wildlife Area (77 percent, n = 22 samples),

7. Environmental Baseline

By regulation, environmental baselines for Opinions include the past and present impacts of all state, federal or private actions and other human activities in the action area, the anticipated impacts of all proposed federal projects in the action area that have already undergone formal or early section 7 consultation, and the impact of state or private actions which are contemporaneous with the consultation in process (50 CFR §402.02). The environmental baseline for this Opinion includes a general description of the natural and anthropogenic factors influencing the current status of the three listed Pacific salmonids and the environment within the action area.

Our summary of the environmental baseline complements the information provided in the *Status of Listed Resources* section of this Opinion, and provides the background necessary to understand information presented in the *Effects of the Proposed Action*, and *Cumulative Effects* sections of this Opinion. We then evaluate the consequences of these activities in combination with the environmental baseline to determine the likelihood of jeopardy or adverse modification of designated critical habitat.

The proposed action under consultation is focused geographically on the aquatic ecosystems in California's Central Valley. Accordingly, the environmental baseline for this consultation focuses on the general status and trends of the aquatic ecosystem there and the consequences of that status for the listed salmonids under NMFS' jurisdiction. We describe the principal natural phenomena affecting all listed Pacific salmonids under NMFS jurisdiction in the action area. The action area for the proposed action encompasses the entire freshwater range or a large portion of the freshwater range of the listed fish species and their proposed or designated critical habitat in this consultation. Therefore, we refer the reader to the *Status of the Species* section for general information on the species' biology, ecology, status, and population trends at the species scale.

We further describe anthropogenic factors through the predominant land and water uses within the region. Background information on pesticides in the aquatic environment is also provided. This context illustrates how the physical and chemical health of regional waters and the impact of human activities have contributed to the current status of listed resources in the action area.

Much of the freshwater habitat overlaps geographically between Sacramento River winter-run Chinook, CV spring-run Chinook, and CV steelhead. Therefore, most of the baseline factors responsible for their current statuses are similar. Therefore, each of the following factors applies to Sacramento River winter-run Chinook, CV spring-run Chinook, and CV steelhead unless specified.

7.1. Natural Mortality Factors

Available data indicate high natural mortality rates for salmonids, especially in the open ocean/marine environment. According to Bradford et al. (1997), salmonid mortality rates range from 90 to 99%, depending on the species, the size at ocean entry, and the length of time spent in the ocean. Predation, inter- and intraspecific competition, food availability, smolt quality and health, and physical ocean conditions likely influence the survival of salmon in the marine environment (Brodeur et al. 2004, Bradford et al. 1997). In general, in freshwater rearing habitats, the natural mortality rate averages about 70% for all salmonid species (Bradford et al. 1997). For example, past studies in the Pacific Northwest suggest that the average freshwater survival rate (from egg to smolt) is 2 to 3% throughout the region (Bradford et al. 1997, Marshall and Britton 1990). A number of suspected causes contributing to natural mortality include parasites and/or disease, predation, water temperature, low water flow, wildland fire, and oceanographic features and climatic variability.

7.1.1. Parasites and/or Disease

Most young fish are highly susceptible to disease during the first two months of life. The cumulative mortality in young animals can reach 90 to 95%. Although fish disease organisms occur naturally in the water, native fish have co-evolved with them. Fish can carry these diseases at less than lethal levels (Walker and Foott 1993, Kier Associates 1991, Foott et al. 2003). However, disease outbreaks may occur when water quality is diminished and fish are stressed from crowding and diminished flows (Spence et al. 1996b, Guillen 2003). Young salmonid species may become stressed and lose their resistance in higher temperatures (Spence et al. 1996b). Consequently, diseased fish become more susceptible to predation and are less able to perform essential functions, such as feeding, swimming, and defending territories (McCullough 1999). Examples of parasites and disease for salmonids include whirling disease, infectious hematopoietic necrosis (IHN), sea-lice (*Lepeophtheirus salmonis*), *Henneguya salminicola*, *Ichthyopthirius multifiliis* or Ich, and Columnaris *(Flavobacterium columnare*).

Whirling disease is a parasitic infection caused by the microscopic parasite *Myxobolus cerebrali*. Infected fish continually swim in circular motions and eventually expire from exhaustion. The disease occurs in the wild and in hatcheries and results in losses to fry and fingerling salmonids, especially rainbow trout. The disease is transmitted by infected fish and fish parts and birds.

IHN is a viral disease in many wild and farmed salmonid stocks in the Pacific Northwest. This disease affects rainbow/steelhead trout, cutthroat trout (*Salmo clarki*), brown trout (*Salmo trutta*), Atlantic salmon (*Salmo salar*), and Pacific salmon including Chinook, sockeye, chum, and coho salmon. The virus is triggered by low water temperatures and is shed in the feces, urine, sexual fluids, and external mucus of salmonids. Transmission is mainly from fish to fish, primarily by direct contact and through the water.

Additionally, ich (a protozoan) and Columnaris (a bacterium) are two common fish diseases that were implicated in the massive kill of adult salmon in the Lower Klamath River in September 2002 (Guillen 2003, CDFG 2003).

7.1.2. Predation

Salmonids are exposed to high rates of natural predation, during freshwater rearing and migration stages, as well as during ocean migration. Salmon along the U.S. west coast are prey for marine mammals, birds, sharks, and other fishes. Concentrations of juvenile salmon in the coastal zone experience high rates of predation. In the Pacific Northwest, the increasing size of tern, seal, and sea lion populations may have reduced the survival of some salmon ESUs/DPSs.

7.1.2.1. *Marine Mammal Predation*

Marine mammals are known to attack and eat salmonids. Harbor seals (*Phoca vitulina*), California sea lions (*Zalophus californianus*), and killer whales (*Orcinus orca*) prey on juvenile or adult salmon. Killer whales have a strong preference for Chinook salmon (up to 78% of identified prey) during late spring to fall (Hanson et al. 2005, Hard et al. 1992, Ford and Ellis 2006). Generally, harbor seals do not feed on salmonids as frequently as California sea lions (Pearcy 1997). California sea lions from the Ballard Locks in Seattle, Washington have been estimated to consume about 40% of the steelhead runs since 1985/1986 (Gustafson et al. 1997). In the Columbia River, salmonids may contribute substantially to sea lion diet at specific times and locations (Pearcy 1997). Spring Chinook salmon and steelhead are subject to pinniped predation when they return to the estuary as adults (NMFS 2006). Adult Chinook salmon in the Columbia River immediately downstream of Bonneville

Dam have also experienced increased predation by California sea lions. In recent years, sea lion predation of adult Lower Columbia River winter steelhead in the Bonneville tailrace has increased. This prompted ongoing actions to reduce predation effects. They include the exclusion, hazing, and in some cases, lethal take of marine mammals near Bonneville Dam.

7.1.2.2. *Avian Predation*

Large numbers of fry and juveniles are eaten by birds such as mergansers (*Mergus* spp.), common murre (*Uria aalage*), gulls (*Larus* spp.), and belted kingfishers (*Megaceryle alcyon*). Avian predators of adult salmonids include bald eagles (*Haliaeetus leucocephalus*) and osprey (*Pandion haliaetus*) (Pearcy 1997). Caspian terns (*Sterna caspia*) and cormorants (*Phalacrocorax* spp.) also take significant numbers of juvenile or adult salmon. Stream-type juveniles, especially yearling smolts from spring-run populations, are vulnerable to bird predation in the estuary. This vulnerability is due to salmonid use of the deeper, less turbid water over the channel, which is located near habitat preferred by piscivorous birds (Binelli et al. 2005). For example, research shows that subyearlings from the LCR Chinook salmon ESU are also subject to tern predation. This may be due to the long estuarine residence time of the LCR Chinook salmon (Ryan et al. 2006). Caspian terns and cormorants may be responsible for the mortality of up to 6% of the outmigrating stream-type juveniles in the Columbia River basin (Roby et al. 2006, Collis 2007).

Antolos *et al.* (2005) quantified predation on juvenile salmonids by Caspian terns nesting on Crescent Island in the mid-Columbia reach. Between 1,000 and 1,300 adult terns were associated with the colony during 2000 and 2001, respectively. These birds consumed about 465,000 juvenile salmonids in the first and approximately 679,000 salmonids in the second year. However, caspian tern predation in the estuary was reduced from a total of 13,790,000 smolts to 8,201,000 smolts after relocation of the colony from Rice to East Sand Island in 1999. Based on PIT-tag recoveries at the colony, these were primarily steelhead for Upper Columbia River stocks. Less than 0.1% of the inriver migrating yearling Chinook salmon from the Snake River and less than 1% of the yearling Chinook salmon from the Upper Columbia were consumed. PIT-tagged coho smolts (originating above Bonneville Dam) were second only to steelhead in predation rates at the East Sand Island colony in 2007 (Roby et al. 2008). There are few quantitative data on avian predation rates on Snake River sockeye salmon. Based on the above, avian predators are assumed to have a minimal effect on the long-term survival of Pacific salmon (NMFS 2008b).

7.1.2.3. *Fish Predation*

Predation is a threat to salmonids within this ESU/DPS, especially in the lower Feather River, the Sacramento River, and in the Delta where there are high densities of non-native species such as striped bass (*Morone saxatilis*), largemouth bass (*Micropterus salmoides*), and sunfishes (*Centrarchidae* spp.) and native fish species (e.g., pikeminnows (*Ptychocheilus oregonensis*)) that prey on outmigrating juveniles (NMFS 2011a).

The primary fish predators in estuaries are probably adult salmonids or juvenile salmonids which emigrate at older and larger sizes than others. They include steelhead smolts preying on young Chinook smolts. Outside estuaries, many large non-salmonid populations reside just offshore and may consume large numbers of smolts. These fishes include Pacific hake (*Merluccius productus*), Pacific mackerel (*Scomber japonicus*), lingcod (*Ophiodon elongates*), spiny dogfish (*Squalus acanthias*), various rock fish, and lamprey (Pearcy 1992, Beamish and Neville 1995, Beamish et al. 1992).

7.1.3. Wildland Fire

Wildland fires that are allowed to burn naturally in riparian or upland areas may benefit or harm aquatic species, depending on the degree of departure from natural fire regimes. Although most fires are small in size, large size fires increase the chances of adverse effects on aquatic species. Large fires that burn near the shores of streams and rivers can have biologically significant short-term effects. They include increased water temperatures, ash, nutrients, pH, sediment, toxic chemicals, and large woody debris (Rinne 2004, Buchwalter et al. 2004). Nevertheless, fire is also one of the dominant habitat-forming processes in mountain streams (Bisson et al. 2003). As a result, many large fires burning near streams can result in fish kills with the survivors actively moving downstream to avoid poor water quality conditions (Rinne 2004, Greswell 1999). The patchy, mosaic pattern burned by fires provides a refuge for those fish and invertebrates that leave a burning area or simply spares some fish that were in a different location at the time of the fire (USFS 2000). Small fires or fires that burn entirely in upland areas also cause ash to enter rivers and increase smoke in the atmosphere, contributing to ammonia concentrations in rivers as the smoke adsorbs into the water (Greswell 1999).

The presence of ash also has indirect effects on aquatic species depending on the amount of ash entry into the water. All ESA-listed salmonids rely on macroinvertebrates as a food source for at least a portion of their life histories. When small amounts of ash enter the water, there are usually no noticeable changes to the macroinvertebrate community or the water quality (Bowman and Minshall 2000). When significant amounts of ash are deposited into rivers, the macroinvertebrate community density and composition may be moderately to drastically reduced for a full year with long-term effects lasting 10 years or more (Buchwalter et al. 2003, Buchwalter et al. 2004, Minshall et al. 2001). Larger fires can also indirectly affect fish by altering water quality. Ash and smoke contribute to elevated ammonium, nitrate, phosphorous, potassium, and pH, which can remain elevated for up to four months after forest fires (Buchwalter et al. 2003).

7.1.4. Oceanographic Features, Climatic Variability and Climate Change

Oceanographic features of the action area may influence prey availability and habitat for Pacific salmonids. These features comprise climate regimes which may suffer regime shifts due to climate changes or other unknown influences. The action area includes important spawning and rearing grounds and physical and biological features essential to the conservation of listed Pacific salmonids - *i.e.*, water quality, prey, and passage conditions. These Pacific oceanographic conditions, climatic variability, and climate change may affect salmonids in the action area.

There is evidence that Pacific salmon abundance may have fluctuated for centuries as a consequence of dynamic oceanographic conditions (Beamish and Bouillon 1993, Beamish et al. 2009, Finney et al. 2002). Sediment cores reconstructed for 2,200-year records have shown that Northeastern Pacific fish stocks have historically been regulated by these climate regimes (Finney et al. 2002). The long-term pattern of the Aleutian Low pressure system has corresponded to the trends in salmon catch, to copepod production, and to other climate indices, indicating that climate and the marine environment may play an important role in salmon production. Pacific salmon abundance and corresponding worldwide catches tend to be large during naturally-occurring periods of strong Aleutian low pressure causing stormier winters and upwelling, positive Pacific Decadal Oscillation (PDO), and an above average Pacific circulation index (Beamish et al. 2009). A trend of an increasing Aleutian Low pressure indicates high pink and chum salmon production and low production of coho and Chinook salmon (Beamish et al. 2009). The abundance and distribution of salmon and zooplankton also relate to shifts in North Pacific atmosphere and ocean climate (Francis and Hare 1994).

Over the past century, regime shifts have occurred as a result of the North Pacific's natural climate regime. Reversals in the prevailing polarity of the PDO occurred around 1925, 1947, 1977, and 1989 (Hare and Mantua. 2000, Mantua et al. 1997). The reversals in 1947 and 1977 correspond to dramatic shifts in salmon production regimes in the North Pacific Ocean (Mantua et al. 1997). During the pre-1977 climate regime, the productivity of salmon populations from the Snake River exceeded expectations (residuals were positive) when values of the

PDO were negative (Levin 2003). During the post-1977 regime when ocean productivity was generally lower (residuals were negative), the PDO was negative (Levin 2003).

A smaller, less pervasive regime shift occurred in 1989 (Hare and Mantua. 2000). Beamish *et al.*(2000) analyzed this shift and found a decrease in marine survival of coho salmon in Puget Sound and off the coast of California to Washington. Trends in coho salmon survival were linked over the southern area of their distribution in the Northeast Pacific to a common climatic event. The Aleutian Low Pressure Index and the April flows from the Fraser River also changed abruptly about this time (Beamish et al. 2000).

The Intergovernmental Panel on Climate Change (IPCC) has high confidence that some hydrological systems have been affected through increased runoff and earlier spring peak discharge in glacier- and snow-fed rivers and through effects on thermal structure and water quality of warming rivers and lakes (IPCC 2007). Oceanographic models project a weakening of the thermohaline circulation resulting in a reduction of heat transport into high latitudes of Europe, an increase in the mass of the Antarctic ice sheet, and a decrease in the Greenland ice sheet (IPCC 2001). These changes, coupled with increased acidification of ocean waters, are expected to have substantial effects on marine and hydrological productivity and food webs, including populations of salmon and other salmonid prey (Hard et al. 1992).

Carbon dioxide emissions are also predicted to have major environmental impacts along the west coast of North America during the 21st century and beyond (Climate Impacts Group (CIG) 2004, IPCC 2001). Eleven of the past 12 years (1995 - 2006) rank among the 12 warmest years in the instrumental record of global surface temperature since 1850 (IPCC 2007). The IPCC predicts that, for the next two decades, a warming of about 0.2ºC per decade will occur for a range of predicted carbon dioxide emissions scenarios (IPCC 2007). This warming trend continues in both water and air. Global average sea level has risen since 1961 at an average rate of 1.8 mm/year and since 1993 at 3.1 mm/year, with contributions from thermal expansion, melting glaciers and ice caps, and the polar ice sheets (IPCC 2007).

Poor environmental conditions for salmon survival and growth may be more prevalent with projected warming increases. Increasing climate temperatures can influence smolt development which is limited by time and temperature (McCormick et al. 2009). Food availability and water temperature may affect proper maturation and smoltification and feeding behavior (Mangel 1994). Climate change may also have profound effects on seawater entry and marine performance of anadromous fish, including increased salinity intrusion in estuaries due to higher sea levels, as well as a projected decrease of seawater pH (Orr et al. 2005). There is evidence that Chinook salmon survival in the Pacific during climate anomalies and El Nino events changes as a result of a shift from predation- to competition-based mortality in response to declines in predator and prey abundances and increases in pink salmon abundance (Ruggerone and Goetz 2004). If climate change leads to an overall decrease in the availability of food, then returning fish will likely be smaller (Mangel 1994). Finally, future climatic warming could lead to alterations of river temperature regimes, which could further reduce available fish habitat (Yates et al. 2008).

Although the impacts of global climate change are less clear in the ocean environment, early modeling efforts suggest that increased temperatures will likely increase ocean stratification. This stratification coincides with relatively poor ocean habitat for most Pacific Northwest salmon populations (Climate Impacts Group (CIG) 2004, IPCC 2001).

We expect changing weather and oceanographic conditions may affect prey availability, temperature and water flow in habitat conditions, and growth for all 28 ESUs/DPSs. Consequently, we expect the long-term survival and reproductive success for listed salmonids to be greatly affected by global climate change.

In addition to changes in hydrological regimes that will affect salmon, climate change will affect agriculture as rainfall and temperature patterns shift. Some crops currently well-suited for particular regions may instead be grown in alternate locations. Agricultural pest pressures are also likely to change over time. Both the shifts in crop location and pest pressure are likely to change pesticide use patterns.

7.2. Anthropogenic Mortality Factors

In this section we address anthropogenic threats in the geographic regions across the action area. Land use activities associated with logging, road construction, urban development, mining, agriculture, and recreation have significantly altered fish habitat quantity and quality. Impacts associated with these activities include: (1) alteration of migration corridors and streambank and channel morphology; (2) alteration of ambient stream temperatures; (3) degradation of water quality; (4) elimination or degradation of spawning and rearing habitat; (5) fragmentation of available habitats: and (6) removal or impairment of riparian vegetation – resulting in increased water temperatures and streambank erosion.

In the following discussion we provide information on habitat access, water development, land use, water quality, and hatchery operations. This is followed with a discussion on pesticide detections in the aquatic environment and highlights their background levels from past and ongoing anthropogenic activities. This information is pertinent to EPA's proposed registration of thiobencarb in the U.S. and its territories. Thiobencarb has been in use for multiple decades, it has documented presence in our nation's rivers, and thus over the years have contributing effects to the environmental baseline. In this section, we cover California's special requirements for thiobencarb use in this state. As water temperature plays such a strong role in salmonid distribution, we also provide a general discussion of anthropogenic temperature impacts. Next, we discuss the health of riparian systems and floodplain connectivity, as this habitat is vital to salmonid survival. Finally, we provide a brief overview of the results of section 7 consultations relevant to this analysis.

7.2.1. Habitat Blockages

Hydropower, flood control, and water supply dams have permanently blocked or hindered salmonid access to historical spawning and rearing grounds. Yoshiyama *et al.* (1996)calculated that roughly 2,000 linear miles of salmon habitat was actually available before dam construction and mining, and concluded that 82 percent is not accessible today. The percentage of habitat loss for steelhead is presumable greater, because steelhead were more extensively distributed upstream than Chinook salmon. See Figure 15 for location of dams within ESU/DPSs in California.

As a result of migrational barriers, winter-run, spring-run, and steelhead populations have been confined to lower elevation mainstems that historically only were used for migration and rearing. Population abundances have declined in these streams due to decreased quantity, quality, and spatial distribution of spawning and rearing habitat (Lindley et al. 2009). Higher temperatures at these lower elevations during late-summer and fall are also a major stressor to adult and juvenile salmonids. According to Lindley *et al.*(2004a), of the four independent populations of winter-run that occurred historically, only one mixed stock of winter-run remains below Keswick Dam.

Similarly, of the 19 independent populations of spring-run that occurred historically, only three independent populations remain in Deer, Mill, and Butte Creeks. Dependent populations of spring-run continue to occur in Big Chico, Antelope, Clear, Thomes, and Beegum creeks and the Yuba River, but rely on the extant independent populations for their continued survival. CV steelhead historically had at least 81 independent populations based on Lindley *et al.*'s (2006a) analysis of potential habitat in the Central Valley. However, due to dam construction, access to 38 percent of all spawning habitat has been lost, as well as access to 80 percent of the historically available habitat.

Juvenile downstream migration patterns have been altered by the presence of dams. Juvenile winter-run, and spring-run on the mainstem Sacramento River, arrive at any given location downstream of Keswick Dam earlier than historical, since they are hatched much further downstream and have less distance to travel. Therefore, in order to smolt at the same size and time as historical, they must rear longer within the Sacramento River. However, as will be discussed below, the mainstem Sacramento River is not conducive to the necessary habitat features that provide suitable rearing habitat for listed anadromous fish species, especially for an extended duration of time.

The California Department of Water Resources (DWR) Suisun Marsh Salinity Control Gates (SMSCG) located in Montezuma Slough were installed in 1988, and are operated with gates and flashboards to decrease the salinity levels of managed wetlands in Suisun Marsh. The SMSCG have delayed or blocked passage of adult Chinook salmon migrating upstream (Edwards et al. 1996, Tillman et al. 1996, CDWR 2002). As a result of the SMSCG fish passage study and a term and condition in NMFS' 2004 CVP/SWP operations Opinion, the boat lock has remained open since the 2001-2002 control season (CVP/SWP operations BA), and adult fish passage has improved.

The Red Bluff Diversion Dam (RBDD) impedes adult salmonid passage throughout its May 15 through September 15 "gates in" period. Although there are fish ladders at the right and left banks, and a temporary ladder in the middle of the dam, they are not very efficient at passing fish. The range of effects resulting from delays in upstream migration at RBDD include delayed, but eventually successful spawning, to prespawn mortality and the complete loss of spawning potential in that fraction of the population (NMFS 2009a).

7.2.2. Water Development

The diversion and storage of natural flows by dams and diversion structures on Central Valley waterways have depleted streamflows and altered the natural cycles by which juvenile and adult salmonids base their migrations. As much as 60 percent of the natural historical inflow to Central Valley watersheds and the Delta have been diverted for human uses. Depleted flows have contributed to higher temperatures, lower dissolved oxygen

(DO) levels, and decreased recruitment of gravel and large woody debris (LWD). More uniform flows year round have resulted in diminished natural channel formation, altered food web processes, and slower regeneration of riparian vegetation. These stable flow patterns have reduced bedload movement (Mount 1995, Associates 2001), caused spawning gravels to become embedded, and decreased channel widths due to channel incision, all of which has decreased the available spawning and rearing habitat below dams. The storage of unimpeded runoff in these large reservoirs also has altered the normal hydrograph for the Sacramento and San Joaquin River watersheds. Rather than seeing peak flows in these river systems following winter rain events (Sacramento River) or spring snow melt (San Joaquin River), the current hydrology has truncated peaks with a prolonged period of elevated flows (compared to historical levels) continuing into the summer dry season.

Water withdrawals, for agricultural and municipal purposes, have reduced river flows and increased water temperatures during the critical summer months, and in some cases, have been of a sufficient magnitude to result in reverse flows in the lower San Joaquin River (Reynolds et al. 1993). Direct relationships exist between water temperature, water flow, and juvenile salmonid survival (Brandes and McLain 2001). Elevated water temperatures in the Sacramento River have limited the survival of young salmon in those waters. Juvenile fall-run survival in the Sacramento River is also directly related to June streamflow and June and July Delta outflow (Dettman et al. 1987).

Water diversions for irrigated agriculture, municipal and industrial use, and managed wetlands are found throughout the Central Valley. Thousands of small and medium-size water diversions exist along the Sacramento River, San Joaquin River, and their tributaries. Although efforts have been made in recent years to screen some of these diversions, many remain unscreened. Depending on the size, location, and season of operation, these unscreened diversions entrain and kill many life stages of aquatic species, including juvenile salmonids. For example, as of 1997, 98.5 percent of the 3,356 diversions included in a Central Valley database were either unscreened or screened insufficiently to prevent fish entrainment (Herren and Kawasaki 2001). Most of the 370 water diversions operating in Suisun Marsh are unscreened (Herren and Kawasaki 2001). Outmigrant juvenile salmonids in the Delta have been subjected to adverse environmental conditions created by water export operations at the Central Valley Project (CVP) and State Water Project (SWP) facilities.

Specifically, juvenile salmonid survival has been reduced by: (1) water diversion from the mainstem Sacramento River into the Central Delta via the Delta Cross Channel (DCC); (2) upstream or reverse flows of water in the lower San Joaquin River and southern Delta waterways; (3) entrainment at the Central Valley Project and the State Water Project (CVP & SWP) export facilities and associated problems at the Clifton Court Forebay; and (4) as discussed above, increased exposure to introduced, non-native predators such as striped bass (*Morone saxatilis*), largemouth bass (*Micropterus salmoides*), and sunfishes (*Centrarchidae* spp.) within the waterways of the Delta while moving through the Delta under the influence of CVP/SWP pumping.

The Anderson-Cottonwood Irrigation District (ACID) Dam operates a diversion dam across the Sacramento River located 5 miles downstream from Keswick Dam. ACID is one of the 3 largest diversions on the Sacramento River and has senior water rights of 128 thousand acre feet (TAF) of water since 1916 for irrigation along the west side of the Sacramento River. The installation and removal of the diversion dam flashboards requires close coordination between the Bureau of Reclamation and ACID. The diversion dam is operated from April through October. Substantial reductions in Keswick releases to install or remove the flashboards have resulted in dewatered redds, stranded juveniles, and higher water temperatures. Based on generalized run timing (Table 14), the diversion dam operations could impact winter- and spring-run Chinook. Redd dewatering would most likely affect spring-run Chinook in October; however, the reductions in flows are usually short-term, lasting less than 8 hours. Such short-term reductions in flows may cause some mortality of incubating eggs and loss of stranded juveniles. Reductions in Keswick releases are limited to 15 percent in a 24-hour period and 2.5 percent in any 1 hour. Past operations have shown that the most significant reductions occur during wet years when Shasta releases are higher than 10,000 cubic feet per second (cfs). Average April releases from Keswick are 6,000 to 7,000 cfs. The likelihood of a flow fluctuation occurring (when Shasta storage > 4.5 MAF in April) is 17 percent, or 14 out of the 82-year historical record. During wet years, flows released from Shasta Dam are typically higher than in drier water year types. The amount of flow that needs to be reduced to get to safe operating levels for the installation of the flashboards at the ACID dam is therefore greater and the wetted area reduction downstream of Keswick Dam is thus greater. The likelihood of an October reduction in flows that could dewater redds is even lower, since average releases are 6,000 cfs in all water year types.

Table 14. Generalized life-history timing for listed salmonids in the upper Sacramento River.

Species	Adult Immigration	Adult Holding	Typical Spawning	Egg Incubation	Juvenile Rearing	Juvenile Emigration
Winter-run	Nov-Jul	Jan-May	Apr-Jul	Apr-Oct	Jun-Mar	Jun-Mar
Spring-run	Mar-Sep	May-Sep	Aug-Oct	Aug-Mar	Year round	Oct-May
Steelhead	Oct-May	Sep-Dec	Dec-Apr	Dec-May	Year round	Jan-Jul

The ACID diversion dam was improved in 2001 with the addition of new fish ladders and fish screens around the diversion. Since upstream passage was improved a substantial shift in winter-run spawning has occurred. In recent years, more than half of the winter-run redds have typically been observed above the ACID diversion dam. This makes flow fluctuations more a concern since such a large proportion of the run is spawning so close to Keswick Dam.

The Red Bluff Diversion Dam (RBDD) is owned and operated by the Bureau of Reclamation. The Tehama-Colusa Canal Authority (TCCA) operates the Corning Canal and Tehama-Colusa Canal, which divert up to 328 TAF from the Sacramento River. RBDD is located 59 miles downstream of Keswick Dam. It blocks or delays adult salmonid migrating upstream to various degrees, depending on run timing. Based on various studies (Vogel et al. 1988, Hallock 1989, CDFG 1998a), problems in salmonid passage at RBDD provide a well-documented example of a diversion facility impairing salmon migration. A portion of the winter-run adults encounter the gates down and are forced to use the fish ladders. There are 3 fish ladders on RBDD, one on each side and one temporary ladder in the middle of the dam. The RBDD fish ladders are not efficient at passing adult salmonids due to the inability of salmon to find the entrances. Water released from RBDD flows through a small opening under 11 gates across the river, causing turbulent flows that confuse fish and keep them from finding the ladders. The fish ladders are not designed to allow enough water through them to attract adult salmonids towards them. Previous studies (Vogel 2008, USFWS 2001) have shown that salmon can be delayed up to 20 days in passing the dam. These delays can reduce the fitness of adults that expend their energy reserves fighting the flows beneath the gates, and increase the chance of prespawn mortality. Run timing is critical to salmon, as it is what distinguishes one race from another. Delays of a week or even days in passage likely prevents some spring-run adults (those that encounter gates down in May and June) from entering tributaries above RBDD that dry up or warm up in the spring (*e.g.*, Cottonwood Creek, Cow Creek). These delays have the potential of preventing these fish from accessing summer holding pools in the upper areas of the creeks. Delays could allow for the onset of temperature related diseases that could affect spawning success or cause pre-spawn mortalities.

7.2.3. Water Conveyance and Flood Control

The development of the water conveyance system in the Delta has resulted in the construction of armored, rip-rapped levees on more than 1,100 miles of channels and diversions to increase channel elevations and flow capacity of the channels (Mount 1995). Levee development in the Central Valley affects spawning habitat, freshwater rearing habitat, freshwater migration corridors, and estuarine habitat PCEs. As Mount (1995) indicates, there is an "underlying, fundamental conflict inherent in this channelization." Natural rivers strive to achieve dynamic equilibrium to handle a watershed's supply of discharge and sediment (Mount 1995). The construction of levees disrupts the natural processes of the river, resulting in a multitude of habitat-related effects, including isolation of the watershed's natural floodplain behind the levee from the active river channel and its fluctuating hydrology.

Many of these levees use angular rock (riprap) to armor the bank from erosive forces. The effects of channelization, and rip-rapping, include the alteration of river hydraulics and cover along the bank as a result of changes in bank configuration and structural features. These changes affect the quantity and quality of nearshore habitat for juvenile salmonids and have been thoroughly studied (USFWS 2000, Garland et al. 2002, Schmetterling et al. 2001). Simple slopes protected with rock revetment generally create nearshore hydraulic conditions characterized by greater depths and faster, more homogeneous water velocities than occur along natural banks. Higher water velocities typically inhibit deposition and retention of sediment and woody debris. These changes generally reduce the range of habitat conditions typically found along natural shorelines, especially by eliminating the shallow, slow-velocity river margins used by juvenile fish as refuge and to escape from fast currents, deep water, and predators (USFWS 2000).

Prior to the 1970s, there was so much debris resulting from poor logging practices that many streams were completely clogged and were thought to have been total barriers to fish migration. As a result, in the 1960s and early 1970s it was common practice among fishery management agencies to remove woody debris thought to be a barrier to fish migration (NMFS 1996b). However, it is now recognized that too much LWD was removed from the streams resulting in a loss of salmonid habitat and it is thought that the large scale removal of woody debris prior to 1980 had major, long-term negative effects on rearing habitats for salmonids in northern California (NMFS 1996b). Areas that were subjected to this removal of LWD are still limited in the recovery of salmonid stocks; this limitation could be expected to persist for 50 to 100 years following removal of the debris.

Large quantities of downed trees are a functionally important component of many streams (NMFS 1996b). LWD influences stream morphology by affecting channel pattern, position, and geometry, as well as pool formation (Keller and Swanson 1979, Bilby 1984, Robison and Beschta 1990). Reduction of wood in the stream channel, either from past or present activities, generally reduces pool quantity and quality, alters stream shading which can affect water temperature regimes and nutrient input, and can eliminate critical stream habitat needed for both vertebrate and invertebrate populations. Removal of vegetation also can destabilize marginally stable slopes by increasing the subsurface water load, lowering root strength, and altering water flow patterns in the slope.

In addition, the armoring and revetment of stream banks tends to narrow rivers, reducing the amount of habitat per unit channel length (Sweeney et al. 2004). As a result of river narrowing, benthic habitat decreases and the number of macroinvertebrates, such as stoneflies and mayflies, per unit channel length decreases, affecting salmonid food supply.

7.2.4. Land Use Activities

Table 15 summarizes general land use categories in the CV by percent and provides population densities. Figure 11 depicts general land use in California. In the pages that follow, Figure 11 is zoomed in to just the Central Valley to show the overlay of the CV spring-run Chinook ESU (Figure 12), the Sacramento River winter-run Chinook ESU (
Figure 13), and CV steelhead (Figure 14). Land use activities continue to have large impacts on salmonid habitat in the Central Valley watershed. Until about 150 years ago, the Sacramento River was bordered by up to 500,000 acres of riparian forest, with bands of vegetation extending outward for 4 or 5 miles (CRA 1989). Starting with the gold rush, these vast riparian forests were cleared for building materials, fuel, and to clear land for farms on the raised natural levee banks. The degradation and fragmentation of riparian habitat continued with extensive flood control and bank protection projects, together with the conversion of the fertile riparian lands to agriculture outside of the natural levee belt. The dominant land use in the CV is agriculture. By 1979, riparian habitat along the Sacramento River diminished to 11,000 to 12,000 acres, or about 2 percent of historic levels (McGill 1987). The clearing of the riparian forests removed a vital source of snags and driftwood in the Sacramento and San Joaquin River basins. This has reduced the volume of LWD input needed to form and maintain stream habitat that salmon depend on in their various life stages. In addition to this loss of LWD sources, removal of snags and obstructions from the active river channel for navigational safety has further reduced the presence of LWD in the Sacramento and San Joaquin Rivers, as well as the Delta.

Table 15. Land uses and population density in the Sacramento and San Joaquin watersheds (Carter and Resh 2005).

Watershed	Land Use Categories (Percent)				Density (people/mi^2)
	Agriculture	Forest	Urban	Other	

Sacramento River	15	49	2	30 grass & shrub	61
San Joaquin River	30	27	2	36 grass & shrub	76

Increased sedimentation resulting from agricultural and urban practices within the Central Valley is one of the primary causes of salmonid habitat degradation (NMFS 1996a). Sedimentation can adversely affect salmonids during all freshwater life stages by: clogging or abrading gill surfaces, adhering to eggs, hampering fry emergence (Phillips and Campbell 1961), burying eggs or alevins, scouring and filling in pools and riffles, reducing primary productivity and photosynthesis activity (Cordone and Kelley 1961), and affecting intergravel permeability and DO levels. Excessive sedimentation over time can cause substrates to become embedded, which reduces successful salmonid spawning and egg and fry survival (Waters 1995).

Land use categories are presented in Table 16 below. Land use activities associated with road construction, urban development, logging, mining, agriculture, and recreation have significantly altered fish habitat quantity and quality through the alteration of streambank and channel morphology; alteration of ambient water temperatures; degradation of water quality; elimination of spawning and rearing habitat; fragmentation of available habitats; elimination of downstream recruitment of LWD; and removal of riparian vegetation, resulting in increased streambank erosion (Meehan 1991). Urban stormwater and agricultural runoff are contaminated with pesticides, petroleum products, sediment, *etc.* Agricultural practices in the Central Valley have eliminated large trees and logs and other woody debris that would otherwise be recruited into the stream channel (NMFS 1998). Rice cultivation accounts for approximately half of all cultivated crops in the Sacramento Valley, and a quarter of all cultivated crops after combining the San Joaquin Valley with the Sacramento (Table 16).

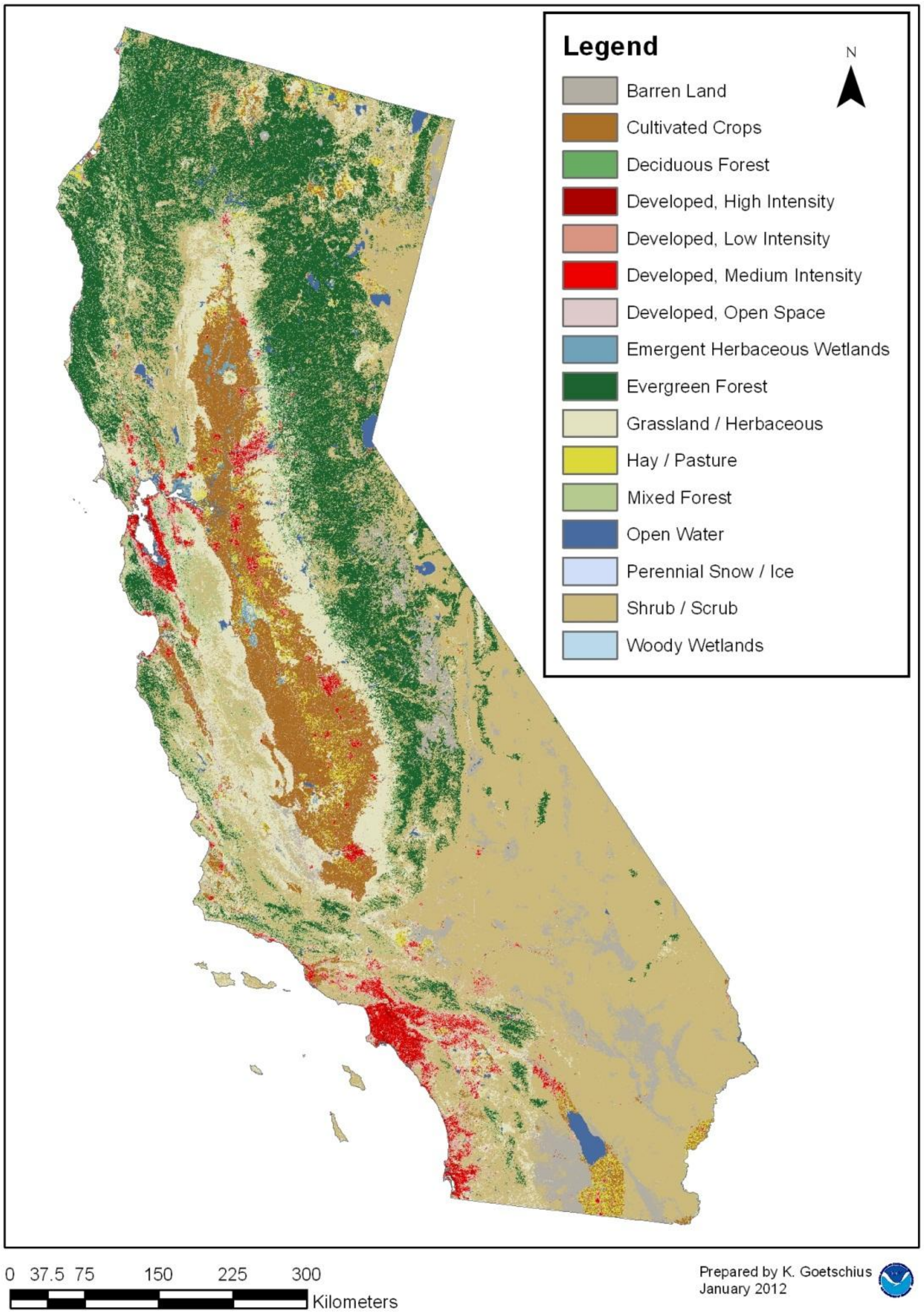


Figure 11. Landuse in California (National Land Cover Database 2006).

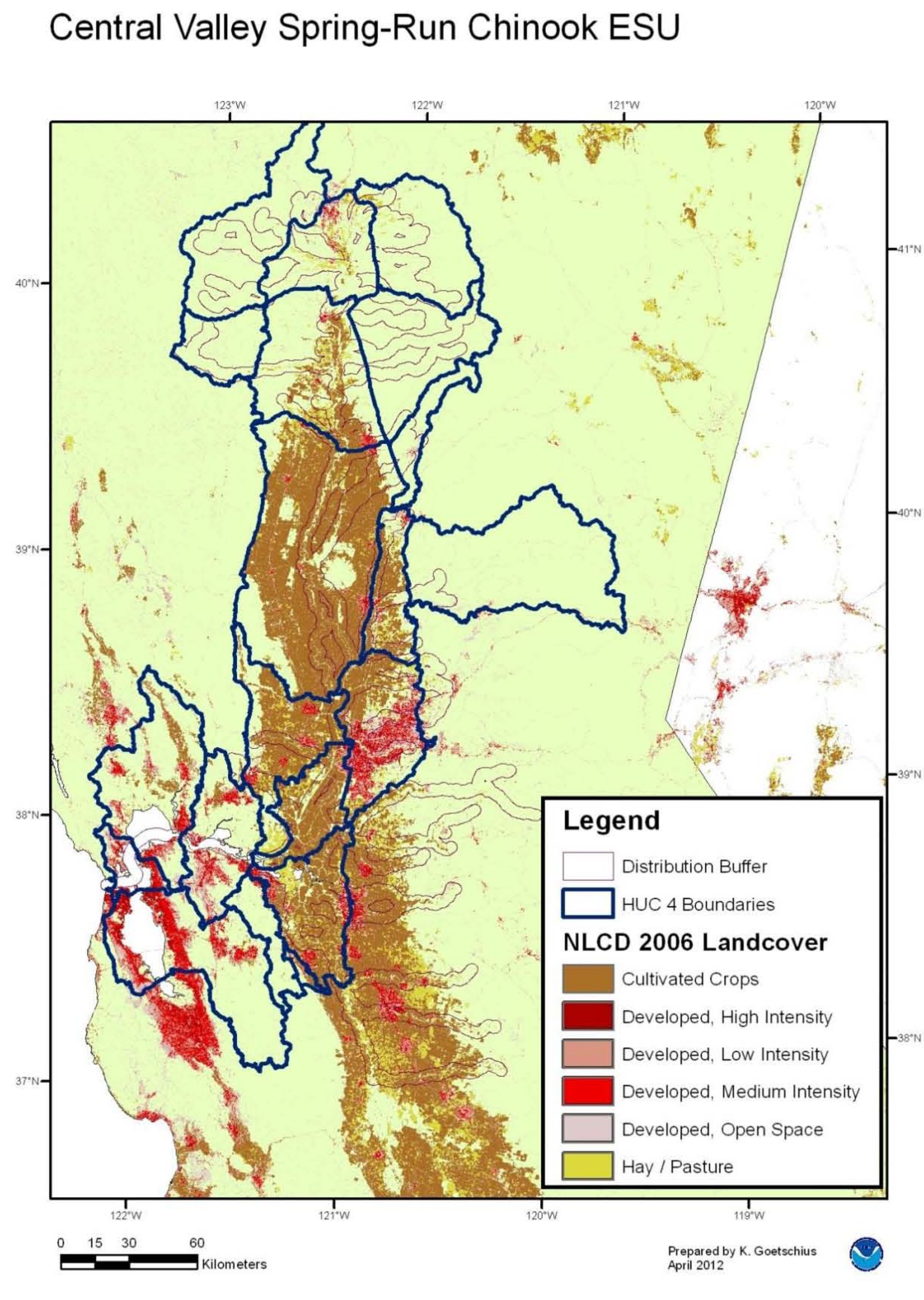


Figure 12. CV spring-run Chinook ESU with associated land-use patterns.

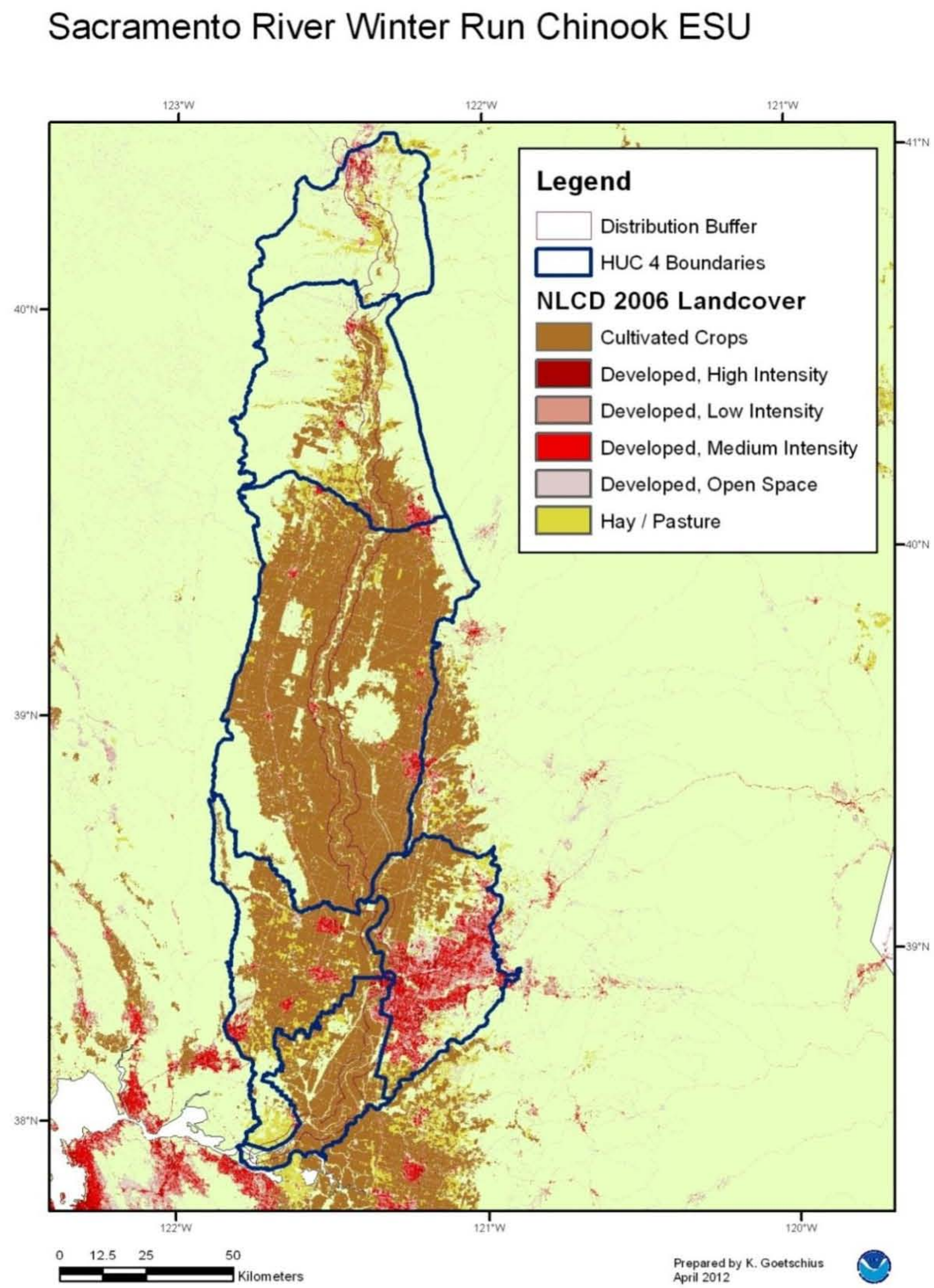


Figure 13. Sacramento River winter-run Chinook ESU with associated land use patterns.

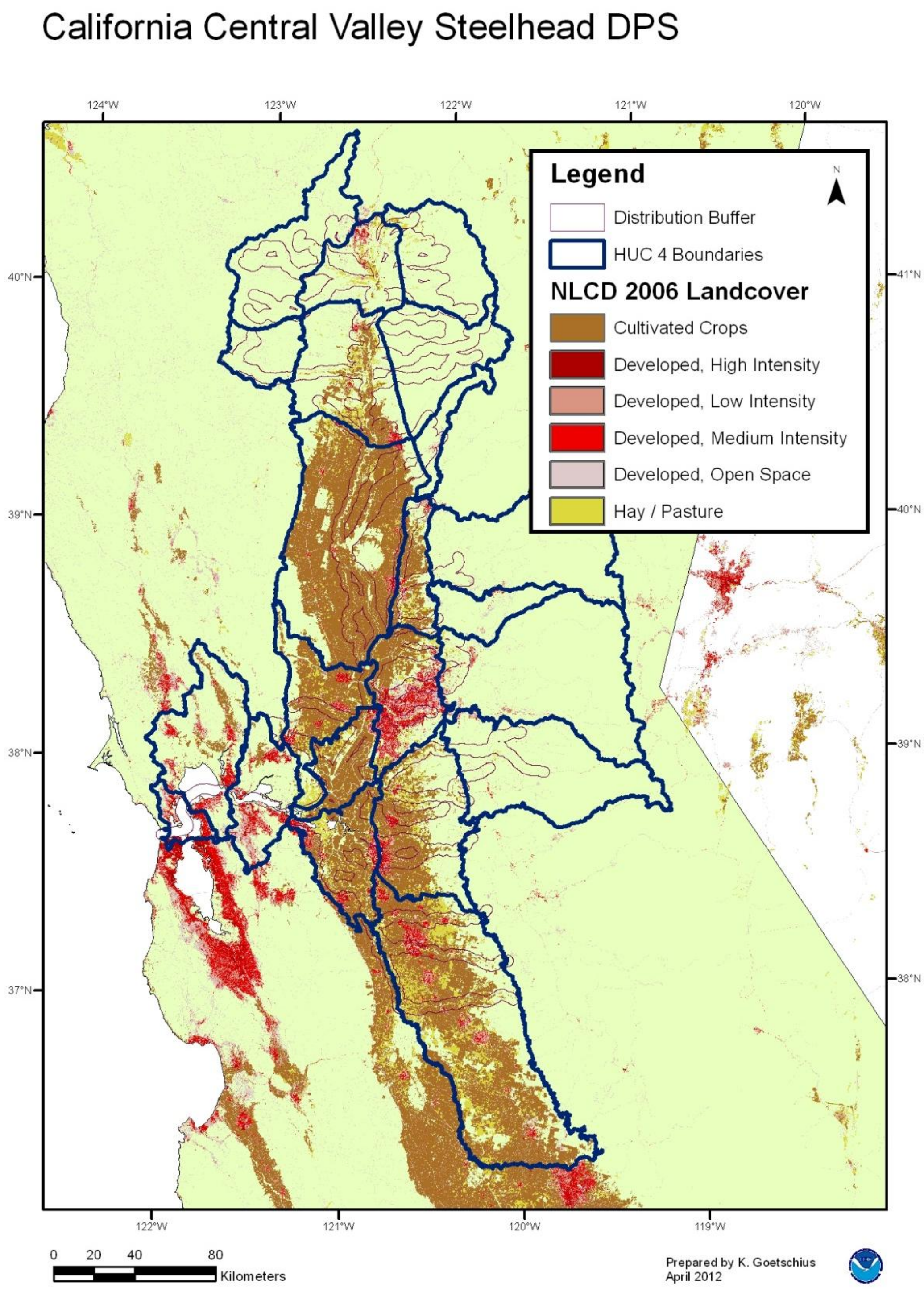


Figure 14. CV steelhead DPS with associated land use patterns.

Since the 1850s, wetlands reclamation for urban and agricultural development has caused the cumulative loss of 79 and 94 percent of the tidal marsh habitat in the Delta downstream and upstream of Chipps Island, respectively (Conomos et al. 1985, Nichols et al. 1986, Wright and Phillips 1988, Monroe et al. 1992, Project 1999). Prior to 1850, approximately 1400 km2 of freshwater marsh surrounded the confluence of the Sacramento and San Joaquin Rivers, and another 800 km2 of saltwater marsh fringed San Francisco Bay's margins. Of the original 2,200 km2 of tidally influenced marsh, only about 125 km2 of undiked marsh remains today. In Suisun Marsh, saltwater intrusion and land subsidence gradually has led to the decline of agricultural production.

Presently, Suisun Marsh consists largely of tidal sloughs and managed wetlands for duck clubs, which first were established in the 1870s in western Suisun Marsh (Goals Project 1999). Even more extensive losses of wetland marshes occurred in the Sacramento and San Joaquin River Basins. Little of the extensive tracts of wetland marshes that existed prior to 1850 along the valley's river systems and within the natural flood basins exist today. Most has been "reclaimed" for agricultural purposes, leaving only small remnant patches.

Table 16. Area of land use categories within the range of listed salmonids in the Central Valley (km2). Total area for each category is given in bold. Land cover was determined via the National Land Cover Database 2006, developed by the Multi-Resolution Land Characteristics (MRLC) Consortium, comprised of nine federal agencies (USGS, EPA, USFS, NOAA, NASA, BLM, NPS, NRCS, and USFWS). Land cover class definitions are available at: http://www.mrlc.gov/nlcd_definitions.php . Rice information was obtained from the California Rice Commission (2010 values) and from the Cal. DPR pesticide use reporting data base (2010 county reports).

Land Cover		Chinook Salmon	Steelhead
Sub Category	code	Central Valley Spring-run Sacramento Winter-run	Central Valley
Water		**367**	**422**
Open Water	11	346	422
Perennial Snow/ice	12	0	0
Developed Land		**2,755**	**3,534**
Open Space	21	1,174	1,472
Low Intensity	22	635	792
Medium Intensity	23	616	837
High Intensity	24	153	211
Barren Land	31	178	222
Undeveloped Land		**15,063**	**19,138**
Deciduous Forest	41	657	744
Evergreen Forest	42	3,707	3,942
Mixed Forest	43	476	583
Shrub/Scrub	52	3,245	3,786
Herbaceous	71	6,261	9,396
Woody Wetlands	90	189	245
Emergent Wetlands	95	527	431
Agriculture		**5,796**	**10,507**
Hay/Pasture	81	754	1,640
Cultivated Crops	82	5,043	8,867
	(Rice)	(2,344)	(2,353)
Total (including open water)		23,982	33,601
Total (without open water)		23,615	33,179

Dredging of river channels to enhance inland maritime trade and to provide raw material for levee construction has significantly and detrimentally altered the natural hydrology and function of the river systems in the Central Valley. Starting in the mid-1800s, the Corps and private consortiums began straightening river channels and artificially deepening them to enhance shipping commerce. This has led to declines in the natural meandering of river channels and the formation of pool and riffle segments. The deepening of channels beyond their natural depth also has led to a significant alteration in the transport of bedload in the riverine system as well as the local flow velocity in the channel (Mount 1995). The Sacramento Flood Control Project at the turn of the nineteenth century ushered in the start of large scale Corps actions in the Delta and along the rivers of California for reclamation and flood control. The creation of levees and the deep shipping channels reduced the natural tendency of the San Joaquin and Sacramento Rivers to create floodplains along their banks with seasonal inundations during the wet winter season and the spring snow melt periods. These annual inundations provided necessary habitat for rearing and foraging of juvenile native fish that evolved with this flooding process. The armored rip-rapped levee banks and active maintenance actions of Reclamation Districts precluded the establishment of ecologically important riparian vegetation, introduction of valuable LWD from these riparian corridors, and the productive intertidal mudflats characteristic of the undisturbed Delta habitat.

Urban stormwater and agricultural runoff may be contaminated with pesticides, oil, grease, heavy metals, polycyclic aromatic hydrocarbons (PAHs), and other organics and nutrients (CRWQCB-CVR 1998) that can destroy aquatic life necessary for salmonid survival (NMFS 1996a, NMFS 1996b). Point source (PS) and non-point source (NPS) pollution occurs at almost every point that urbanization activity influences the watershed. Impervious surfaces (*i.e.*, concrete, asphalt, and buildings) reduce water infiltration and increase runoff, thus creating greater flood hazard (NMFS 1996a, NMFS 1996b). Flood control and land drainage schemes may increase the flood risk downstream by concentrating runoff. A flashy discharge pattern results in increased bank erosion with subsequent loss of riparian vegetation, undercut banks and stream channel widening. In addition to the PS and NPS inputs from urban runoff, juvenile salmonids are exposed to increased water temperatures as a result of thermal inputs from municipal, industrial, and agricultural discharges.

Past mining activities routinely resulted in the removal of spawning gravels from streams, the straightening and channelization of the stream corridor from dredging activities, and the leaching of toxic effluents into streams from mining operations. Many of the effects of past mining operations continue to impact salmonid habitat today. Current mining practices include suction dredging (sand and gravel mining), placer mining, lode mining and gravel mining. Present day mining practices are typically less intrusive than historic operations (hydraulic mining); however, adverse impacts to salmonid habitat still occur as a result of present-day mining activities. Sand and gravel are used for a large variety of construction activities including base material and asphalt, road bedding, drain rock for leach fields, and aggregate mix for concrete to construct buildings and highways. Most aggregate is derived principally from pits in active floodplains, pits in inactive river terrace deposits, or directly from the active channel. Other sources include hard rock quarries and mining from deposits within reservoirs. Extraction sites located along or in active floodplains present particular problems for anadromous salmonids. Physical alteration of the stream channel may result in the destruction of existing riparian vegetation and the reduction of available area for seedling establishment. Loss of vegetation impacts riparian and aquatic habitat by causing a loss of the temperature moderating effects of shade and cover, and habitat diversity. Extensive degradation may induce a decline in the alluvial water table, as the banks are effectively drained to a lowered level, affecting riparian vegetation and water supply (NMFS 1996b). Altering the natural channel configuration will reduce salmonid habitat diversity by creating a wide, shallow channel lacking in the pools and cover necessary for all life stages of anadromous salmonids. In addition, waste products resulting from past and present mining activities, include cyanide (an agent used to extract gold from ore), copper, zinc, cadmium, mercury, asbestos, nickel, chromium, and lead.

Juvenile salmonids are exposed to increased water temperatures in the Delta during the late spring and summer due to the loss of riparian shading, and by thermal inputs from municipal, industrial, and agricultural discharges. Studies by DWR on water quality in the Delta over the last 30 years show a steady decline in the food sources available for juvenile salmonids and an increase in the clarity of the water due to a reduction in phytoplankton and zooplankton. These conditions have contributed to increased mortality of juvenile Chinook salmon and steelhead as they move through the Delta. The following are excerpts from Lindley *et al*. (2009):

> "The long-standing and ongoing degradation of freshwater and estuarine habitats and the subsequent heavy reliance on hatchery production were also likely contributors to the collapse of the [fall-run] stock. Degradation and simplification of freshwater and estuary habitats over a century and a half of development have changed the Central Valley Chinook salmon complex from a highly diverse collection of numerous wild populations to one dominated by fall Chinook salmon from four large hatcheries."
>
> "In conclusion, the development of the Sacramento-San Joaquin watershed has greatly simplified and truncated the once-diverse habitats that historically supported a highly diverse assemblage of populations. The life history diversity of this historical assemblage would have buffered the overall abundance of Chinook salmon in the Central Valley under varying climate conditions."

7.2.5. Water Quality

The water quality of the Central Valley has been negatively impacted over the last 150 years. Increased water temperatures, decreased DO levels, and increased turbidity and contaminant loads have degraded the quality of the aquatic habitat for the rearing and migration of salmonids. Some common pollutants include effluent from wastewater treatment plants and chemical discharges such as dioxin from San Francisco bay petroleum refineries (McEwan and Jackson 1996b). In addition, agricultural drain water, another possible source of contaminants, can contribute up to 30 percent of the total inflow into the Sacramento River during the low-flow period of a dry year (NMFS 2009c).

7.2.6. Pesticides

The Regional Board, in its 1998 Clean Water Act §303(d) list characterized the Delta as an impaired waterbody having elevated levels of chlorpyrifos, dichlorodiphenyltrichlor (*i.e.* DDT), diazinon, electrical conductivity, Group A pesticides [aldrin, dieldrin, chlordane, endrin, heptachlor, heptachlor epoxide, hexachlorocyclohexanes (including lindane), endosulfan and toxaphene], mercury, low DO, organic enrichment, and unknown toxicities (CRWQCB-CVR 2010). Figure 16 shows numerous reaches where streams and rivers within the Central Valley are listed as impaired due to pesticides.

In general, water degradation or contamination can lead to either acute toxicity, resulting in death when concentrations are sufficiently elevated, or more typically, when concentrations are lower, to chronic or sublethal effects that reduce the physical health of the organism, and lessens its survival over an extended period of time. Mortality may become a secondary effect due to compromised physiology or behavioral changes that lessen the organism's ability to carry out its normal activities. For example, increased levels of heavy metals are detrimental to the health of an organism because they interfere with metabolic functions by inhibiting key enzyme activity in metabolic pathways, decrease neurological function, degrade cardiovascular output, and act as mutagens, teratogens or carcinogens in exposed organisms (Rand et al. 1995, Goyer 1996). For listed species, these effects may occur directly to the listed fish or to its prey base, which reduces the forage base available to the listed species. In the aquatic environment, most anthropogenic chemicals and waste materials, including toxic organic and inorganic chemicals eventually accumulate in sediment (Ingersoll 1995). Direct exposure to contaminated sediments may cause deleterious effects to listed salmonids. This may occur if a fish swims through a plume of the resuspended sediments or rests on contaminated substrate and absorbs the toxic compounds through one of several routes: dermal contact, ingestion, or uptake across the gills. Elevated contaminant levels may be found in localized "hot spots" where discharge occurs or where river currents deposit sediment loads. Sediment contaminant levels can thus be significantly higher than the overlying water column concentrations (EPA 1994). However, the more likely route of exposure to salmonids is through the food chain, when the fish feed on organisms that are contaminated with toxic compounds. Prey species become contaminated either by feeding on the detritus associated with the sediments or dwelling in the sediment itself. Therefore, the degree of exposure to the salmonids depends on their trophic level and the amount of contaminated forage base they consume. Response of salmonids to contaminated sediments is similar to water borne exposures once the contaminant has entered the body of the fish (Heath 1995).

7.2.6.1. *National Pollutant Discharge Elimination System*

Pollution originating from a discrete location such as a pipe discharge or wastewater treatment outfall is known as a point source. Point sources of pollution require a National Pollutant Discharge Elimination System (NPDES) permit. These permits are issued for, among other things, aquaculture, concentrated animal feeding operations, industrial wastewater treatment plants, biosolids (sewer/sludge), pre-treatment and stormwater overflows. The EPA administers the NPDES permit program and states certify that NPDES permit holders comply with state water quality standards. Nonpoint source discharges do not originate from discrete points; thus, nonpoint sources are difficult to identify and quantify. Examples of nonpoint source pollution include, but are not limited to, urban runoff from impervious surfaces, areas of fertilizer and pesticide application, sedimentation, and manure.

According to EPA's database of NPDES permits, about 243 NPDES individual permits are co-located with listed Pacific salmonids in California. See Figure 15 for NPDES permits located within listed salmonid ESUs/DPSs in California.

On November 27, 2006, EPA issued a final rule which exempted pesticides from the NPDES permit process, provided that application was approved under FIFRA. On January 7, 2009, the Sixth Circuit Court of Appeals vacated this rule (National Cotton Council v. EPA, 553 F.3d 927 (6th Cir. 2009)). The result of the vacature, according to the Sixth Circuit, is that "discharges of pesticide pollutants are subject to the NPDES permitting program" under the CWA. In response, EPA has developed a Pesticide General Permit through the NPDES permitting program to regulate such discharges. The permit has recently gone through Section 7(a)(2) consultation (NMFS 2011b) and was issued on October 31, 2011. The Pesticides General Permit generally authorizes discharge of pesticides into waterways, but provides for additional review of discharges of pesticides to waters containing NMFS listed resources.

7.2.7. Baseline Water Temperature - Clean Water Act

Elevated temperature is considered a pollutant in most states with approved Water Quality Standards under the federal Clean Water Act (CWA) of 1972. Under the authority of the CWA, states periodically prepare a list of all surface waters in the state for which beneficial uses - such as drinking, recreation, aquatic habitat, and industrial use – are impaired by pollutants. This process is in accordance with section 303(d) of the CWA. Estuaries, lakes, and streams listed under 303(d) are those that are considered impaired or threatened by pollution. They are water quality limited, do not meet state surface water quality standards, and are not expected to improve within the next two years.

Each state has separate and different 303(d) listing criteria and processes. Generally a water body is listed separately for each standard it exceeds, so it may appear on the list more than once. If a water body is not on the 303(d) list, it is not necessarily contaminant-free; rather it may not have been tested. Therefore, the 303(d) list is a minimum list for the each state regarding polluted water bodies by parameter (Figure 16).

After states develop their lists of impaired waters, they are required to prioritize and submit their lists to EPA for review and approval. Each state establishes a priority ranking for such waters, considering the severity of the pollution and the uses to be made of such waters. States are expected to identify high priority waters targeted for Total Maximum Daily Load (TMDL) development for specific pollutants within two years of the 303(d) listing process. TMDLs once developed are considered during permitting processes.

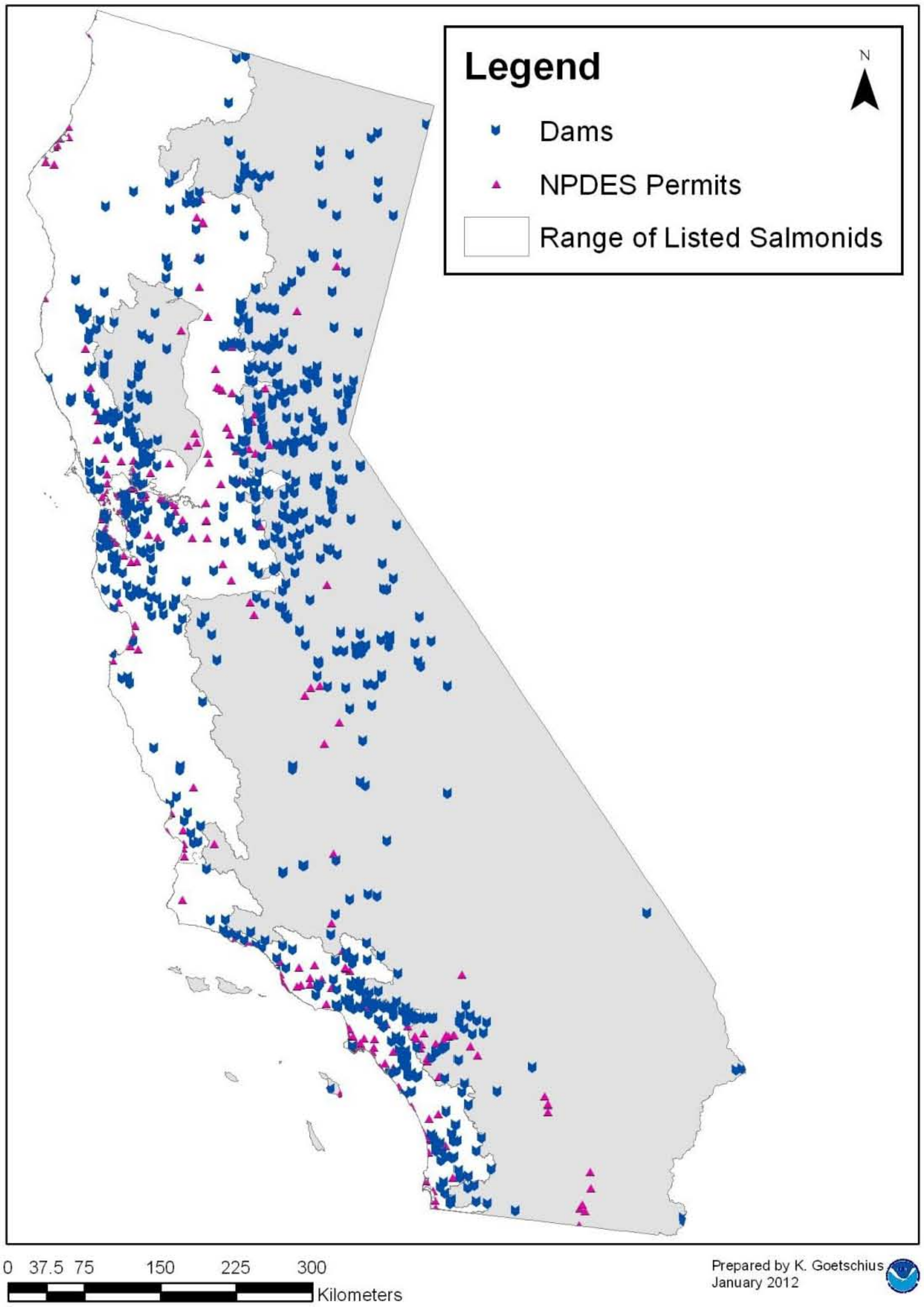


Figure 15. California dams and NPDES permit sites.

Temperature is significant for the health of aquatic life. Water temperatures affect the distribution, health, and survival of native cold-blooded salmonids in the Pacific Northwest. These fish will experience adverse health effects when exposed to temperatures outside their optimal range. For listed Pacific salmonids, water temperature

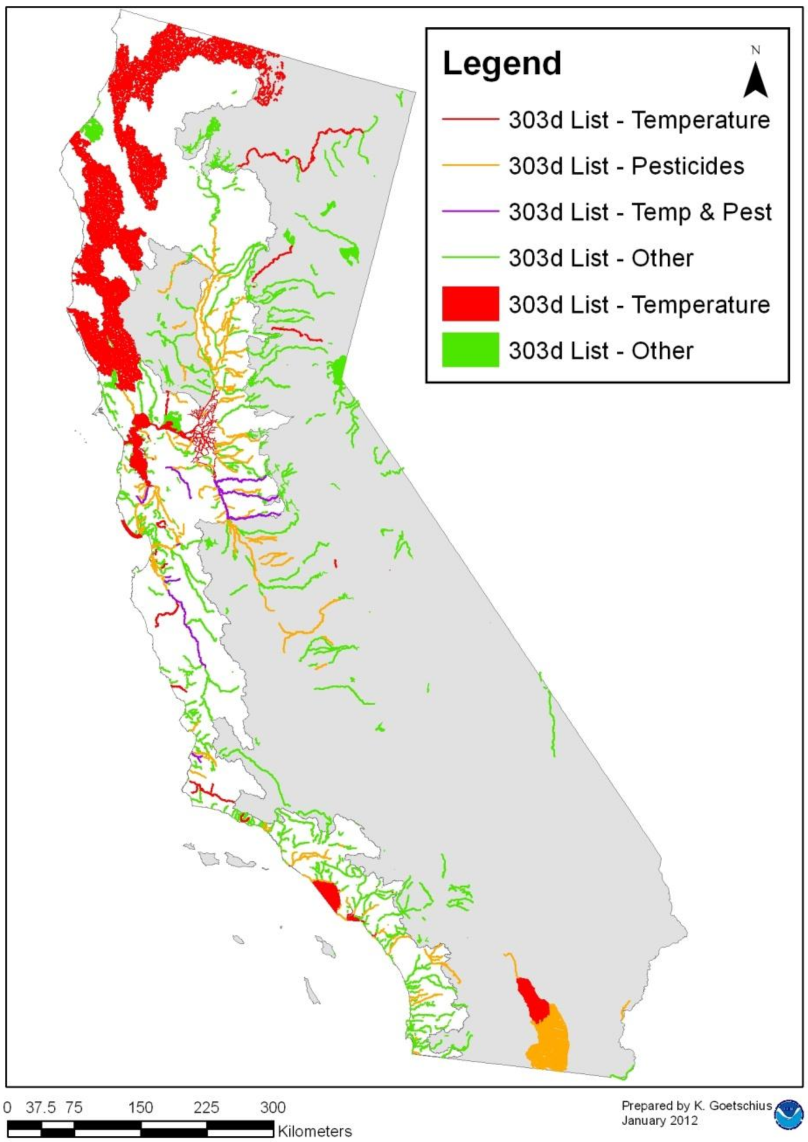


Figure 16. California 303(d) list: water bodies and stream segments included in the 2010 Integrated Report.

tolerance varies between species and life stages. Optimal temperatures for rearing salmonids range from 10ºC to 16ºC. In general, the increased exposure to stressful water temperatures and the reduction of suitable habitat caused by drought conditions reduce the abundance of salmon. Warm temperatures can reduce fecundity, reduce egg survival, retard growth of fry and smolts, reduce rearing densities, increase susceptibility to disease, decrease the ability of young salmon and trout to compete with other species for food, and to avoid predation (Spence et al. 1996b, McCullough 1999). Migrating adult salmonids and upstream migration can be delayed by excessively warm stream temperatures. Excessive stream temperatures may also negatively affect incubating and rearing salmonids (Gregory and Bisson 1997).

Sublethal temperatures (above 24ºC) could be detrimental to salmon by increasing susceptibility to disease (Colgrove and Wood 1966) or elevating metabolic demand (Brett 1995). Substantial research demonstrates that many fish diseases become more virulent at temperatures over 15.6ºC (McCullough 1999). Due to the sensitivity of salmonids to temperature, states have established lower temperature thresholds for salmonid habitat as part of their water quality standards. A water body is listed for temperature on the 303(d) list if the 7-day average of the daily maximum temperatures (7-DADMax) exceeds the temperature threshold.

Water bodies that are not designated salmonid habitat are also listed if they have a one-day maximum over a given background temperature. Using publicly available Geographic Information System (GIS) layers, we determined the number of km on the 303(d) list for exceeding temperature thresholds within the boundaries of each ESU/DPS (Table 17). Because the 303(d) list is limited to the subset of rivers tested, the chart values should be regarded as lower-end estimates.

Table 17. Kilometers of river, stream, and estuary included in state 303(d) lists due to temperature. Data from most recent GIS layers available from CA 2010

Species	ESU or DPS	303(d) listed water (Km)
Chinook	Sacramento River Winter-run	29.9
	Central Valley Spring-run	29.9
Steelhead	Central Valley	367.8

CA 2010 (California EPA TMDL Program 2011)

While some ESU/DPS ranges do not contain any 303(d) rivers listed for temperature, others show considerable overlap. These comparisons demonstrate the relative significance of elevated temperature among ESUs/DPSs. Increased water temperature may result from wastewater discharge, decreased water flow, minimal shading by riparian areas, and climatic variation.

7.2.8. Hatchery Operations and Practices

Five hatcheries currently produce Chinook salmon in the Central Valley, and four of these also produce steelhead. Releasing large numbers of hatchery fish can pose a threat to wild Chinook salmon and steelhead stocks through genetic impacts, competition for food and other resources between hatchery and wild fish, predation of hatchery fish on wild fish, and increased fishing pressure on wild stocks as a result of hatchery production (Waples 1991). The genetic impacts of artificial propagation programs in the Central Valley are primarily caused by straying of hatchery fish and the subsequent interbreeding of hatchery fish with wild fish. In the Central Valley, practices such as transferring eggs between hatcheries and trucking smolts to distant sites for release contribute to elevated straying levels (DOI 1999). For example, Nimbus Hatchery on the American River rears Eel River steelhead stock and releases these fish in the Sacramento River basin. One of the recommendations in the Joint Hatchery Review Report (NMFS/CDFG 2001) was to identify and designate new sources of steelhead brood stock to replace the current Eel River origin brood stock.

Hatchery practices as well as spatial and temporal overlaps of habitat use and spawning activity between spring- and fall-run fish have led to the hybridization and homogenization of some subpopulations (CDFG 1998a). As early as the 1960s, Slater (1963a) observed that spring-run and early fall-run were competing for spawning sites in the Sacramento River below Keswick Dam, and speculated that the two runs may have hybridized. Spring-run from the FRFH have been documented as straying throughout the Central Valley for many years (CDFG 1998a), and in many cases have been recovered from the spawning grounds of fall-run, an indication that FRFH spring-run may exhibit fall-run life history characteristics. Although the degree of hybridization has not been comprehensively determined, it is clear that the populations of spring-run spawning in the Feather River and counted at RBDD contain hybridized fish.

The management of hatcheries, such as Nimbus Fish Hatchery and FRFH, can directly impact spring-run and steelhead populations by over-saturating the natural carrying capacity of the limited habitat available below dams. In the case of the Feather River, significant redd superimposition occurs in-river due to hatchery overproduction and the inability to physically separate spring-run and fall-run adults. This concurrent spawning has led to hybridization between the spring-run and fall-run in the Feather River. At Nimbus Hatchery, operating Folsom Dam to meet temperature requirements for returning hatchery fall-run often limits the amount of water available for steelhead spawning and rearing the rest of the year.

The increase in Central Valley hatchery production has reversed the composition of the steelhead population, from 88 percent naturally-produced fish in the 1950s (McEwan 2001a) to an estimated 23 to 37 percent naturally-produced fish currently (Nobriga and Cadrett 2003). The increase in hatchery steelhead production proportionate to the wild population has reduced the viability of the wild steelhead populations, increased the use of out-of-basin stocks for hatchery production, and increased straying (NMFS/CDFG

2001). Thus, the ability of natural populations to successfully reproduce and continue their genetic integrity likely has been diminished.

The relatively low number of spawners needed to sustain a hatchery population can result in high harvest-to-escapements ratios in waters where fishing regulations are set according to hatchery population. This can lead to over-exploitation and reduction in the size of wild populations existing in the same system as hatchery populations due to incidental bycatch (McEwan 2001a).

Hatcheries also can have some positive effects on salmonid populations. Winter-run produced in the LSNFH are considered part of the winter-run ESU. Spring-run produced in the FRFH are considered part of the spring-run ESU. Artificial propagation has been shown to be effective in bolstering the numbers of naturally spawning fish in the short term under specific scenarios. Artificial propagation programs can also aid in conserving genetic resources and guarding against catastrophic loss of naturally spawned populations at critically low abundance levels, as was the case with the winter-run population during the 1990s. However, relative abundance is only one component of a viable salmonid population.

7.3. Baseline Habitat Condition

As noted above in the discussion on land use, and in the *Status of the Species* section, the riparian zones for much of the ESUs/DPSs are degraded. Riparian zones are the areas of land adjacent to rivers and streams. These systems serve as the interface between the aquatic and terrestrial environments. Riparian vegetation is characterized by emergent aquatic plants and species that thrive on close proximity to water, such as willows. This vegetation maintains a healthy river system by reducing erosion, stabilizing main channels, and providing shade. Leaf litter that enters the river becomes an important source of nutrients for invertebrates (Bisson and Bilby 2001). Riparian zones are also the major source of large woody debris (LWD). When trees fall and enter the water, they become an important part of the ecosystem. The LWD alters the flow, creating the pools of slower moving water preferred by salmon (Bilby et al. 2001). While not necessary for pool formation, LWD is associated with around 80% of pools in northern California, Washington, and the Idaho pan-handle (Bilby and Bisson 2001).

Bilby and Bisson (2001) discuss several studies that associate increased LWD with increased pools, and both pools and LWD with salmonid productivity. Their review also includes documented decreases in salmonid productivity following the removal of LWD. Other benefits of LWD include deeper pools, increased sediment retention, and channel stabilization.

Floodplains are relatively flat areas adjacent to larger streams and rivers (e.g., the Yolo Bypass). They allow for the lateral movement of the main channel and provide storage for floodwaters during periods of high flow. Water stored in the floodplain (i.e., hyporheic flow) is later released during periods of low flow. This process ensures adequate flows for salmonids during the summer months, and reduces the possibility of high-energy flood events destroying salmonid redds (Smith 2005).

Periodic flooding of these areas creates habitat used by salmonids. Thus, floodplain areas vary in depth and widths and may be intermittent or seasonal. Storms also wash sediment and LWD into the main stem river, often resulting in blockages. These blockages may force the water to take an alternate path and result in the formation of side channels and sloughs (Benda et al. 2001). Side channels and sloughs are important spawning and rearing habitat for salmonids. The degree to which these off-channel habitats are linked to the main channel via surface water connections is referred to as connectivity (PNERC 2002). As river height increases with heavier flows, more side channels form and connectivity increases. Juvenile salmonids migrate to and rear in these channels for a certain period of time before swimming out to the open sea.

Healthy riparian habitat and floodplain connectivity are vital for supporting a salmonid population. Chinook salmon and steelhead have life history strategies that rely on floodplains during their juvenile life stages. Sommer et al. (2001) was able to show that the Yolo Bypass, floodplain to the Sacramento River, provides better rearing and migration habitat for juvenile Chinook than adjacent river channels. During a 1998 and 1999 study, salmon increased in size substantially faster in the seasonally inundated floodplain than in the river.

Once floodplain areas have been disturbed, it can take decades for their recovery (Smith 2005). Consequently, most land use practices cause some degree of impairment. Development leads to construction of levees and dikes, which isolate the main stem river from the floodplain. Agricultural development and grazing in riparian areas also significantly change the landscape. Riparian areas managed for logging, or logged in the past, are often impaired by a change in species composition. Most areas in the northwest were historically dominated by conifers. Logging results in recruitment of deciduous trees, decreasing the quality of LWD in the rivers. Deciduous trees have smaller diameters than conifers; they decompose faster and are more likely to be displaced (Smith 2005).

Without a properly functioning riparian zone, salmonids contend with a number of limiting factors. They face reductions in quantity and quality of both off-channel and pool habitats. Also, when seasonal flows are not moderated, both higher and lower flow conditions exist. Higher flows can displace fish and destroy redds, while lower flows cut off access to parts of their habitat. Finally, decreased vegetation limits the available shade and cover, exposing individuals to higher temperatures and increased predation.

7.4. Baseline Pesticide Detections in Aquatic Environments

In the environmental baseline, we address pesticide detections reported as part of the U.S. Geological Survey (USGS) National Water-Quality Assessment Program's (NAWQA) national assessment (Gilliom et al. 2006b). We chose this approach because the NAWQA studies present the same level of analysis for each area. Further, given the lack of uniform reporting standards, we are unable to present a comprehensive basin-specific analysis of detections from other sources.

According to Gilliom *et al.* (2006b), the distributions of the most prevalent pesticides in streams and ground water correlate with land use patterns and associated present or past pesticide use. When pesticides are released into the environment, they frequently end up as contaminants in aquatic environments. Depending on their physical properties some are rapidly transformed via chemical, photochemical, and biologically mediated reactions into other compounds, known as degradates. These degradates may become as prevalent as the parent pesticides depending on their rate of formation and their relative persistence.

In the *Exposure* section of the *Effects of the Proposed Action* we present a more comprehensive discussion of available monitoring data from the NAWQA program, state databases maintained by California, and other targeted monitoring studies.

7.4.1.1. *National Water-Quality Assessment Program*

From 1992 - 2001, the USGS sampled water from 186 stream sites within 51 study units; bed-sediment samples from 1,052 stream sites, and fish from 700 stream sites across the continental U.S. Concentrations of pesticides were detected in streams and groundwater within most areas sampled with substantial agricultural or urban land uses. NAWQA results further detected at least one pesticide or degradate more than 90% of the time in water, in more than 80% in fish samples, and greater than 50% of bed-sediment samples from streams in watersheds with agricultural, urban, and mixed land use (Gilliom et al. 2006b).

Twenty-four pesticides and one degradate were each detected in over 10% of streams in agricultural, urban, or mixed land use areas. These 25 compounds include 11 agriculture-use herbicides and the atrazine degradate deethylatrazine; 7 urban-use herbicides; and 6 insecticides used in both agricultural and urban areas. Two of the herbicides used primarily in urban areas are 2,4-D and diuron. Both herbicides were detected roughly 12% of the time in agricultural streams and between 20% and 25% of the time in urban streams. In a previous Opinion, NMFS assessed the effects of these two herbicides on salmonids (http://www.nmfs.noaa.gov/pr/pdfs/consultations/pesticide_opinion4.pdf). Five of the insecticides were carbaryl, carbofuran, chlorpyrifos, diazinon, and malathion. NMFS assessed the effects of these five insecticides on listed salmonids in its 2008 and

2009 Opinions (NMFS 2008c, NMFS 2009e).

Another dimension of pesticides and their degradates in the aquatic environment is their simultaneous occurrence as mixtures (Gilliom et al. 2006b). Mixtures result from the use of different pesticides for multiple purposes within a watershed or groundwater recharge area. Pesticides generally occur more often in natural waterbodies as mixtures than as individual compounds. Mixtures of pesticides were detected more often in streams than in ground water and at relatively similar frequencies in streams draining areas of agricultural, urban, and mixed land use. More than 90% of the time, water from streams in these developed land use settings had detections of two or more pesticides or degradates. About 70% and 20% of the time, streams had five or more and ten or more pesticides or degradates, respectively (Gilliom et al. 2006b). Fish exposed to multiple pesticides at once may also experience additive and synergistic effects. If the effects on a biological endpoint from concurrent exposure to multiple pesticides can be predicted by adding the potency of the pesticides involved, the effects are said to be additive. If, however, the response to a mixture leads to a greater than expected effect on the endpoint, and the pesticides within the mixture enhance the toxicity of one another, the effects are characterized as synergistic. These effects are of particular concern when the pesticides share a mode of action. NAWQA analysis of all detections indicates that more than 6,000 unique mixtures of 5 pesticides were detected in agricultural streams (Gilliom et al. 2006b). The number of unique mixtures varied with land use.

More than half of all agricultural streams sampled and more than three-quarters of all urban streams had concentrations of pesticides in water that exceeded one or more benchmarks for aquatic life. Aquatic life criteria are EPA water-quality guidelines for protection of aquatic life. Exceedance of an aquatic life benchmark level indicates a strong probability that aquatic species are being adversely affected. However, aquatic species may also be affected at levels below criteria. In agricultural streams, most concentrations that exceeded an aquatic life benchmark involved chlorpyrifos (21%), azinphos methyl (19%), atrazine (18%), *p,p'*-DDE (16%), and alachlor (15%) (Gilliom et al. 2006b). Finally, organochlorine pesticides that were discontinued 15 to 30 years ago still exceeded benchmarks for aquatic life and fish-eating wildlife in bed sediment or fish tissue samples from many streams.

7.5. Baseline Pesticide Consultations

NMFS has consulted with EPA on the registration of several pesticides (Table 18). NMFS (NMFS 2008a) determined that current use of chlorpyrifos, diazinon, and malathion is likely to jeopardize the continued existence of 27 listed salmonid ESUs/DPSs – including CV spring-run Chinook, Sacramento winter-run Chinook, and CCV steelhead (CV listed species). NMFS (NMFS 2009b) further determined that current use of carbaryl and carbofuran is likely to jeopardize the continued existence of 22 ESUs/DPSs – including the CV listed species; and the current use of methomyl is likely to jeopardize the continued existence of 18 ESUs/DPSs of listed salmonids – including the CV listed species. NMFS also published conclusions regarding the registration of 12 different a.i.s (NMFS 2010b). NMFS concluded that pesticide products containing azinphos methyl, disulfoton, fenamiphos, methamidophos, or methyl parathion are not likely to jeopardize the continuing existence of any listed Pacific Salmon or destroy or adversely modify designated critical habitat. NMFS also concluded that the effects of products containing bensulide, dimethoate, ethoprop, methidathion, naled, phorate, or phosmet are likely to jeopardize the continued existence of some listed Pacific Salmonids and to destroy or adversely modify designated habitat of some listed salmonids – including the CV listed species. In June 2011, NMFS issued a biological opinion on the effects of four herbicides and two fungicides (Table 18). NMFS concluded that products containing 2,4-D are likely to jeopardize the existence of all listed salmonids, and adversely modify or destroy the critical habitat of some ESU / DPSs. Products containing chlorothalonil or diuron were also likely to adversely modify or destroy critical habitat – including CV listed species but not likely to jeopardize listed salmonids. NMFS also concluded that products containing captan, linuron, or triclopyr BEE do not jeopardize the continued existence of any ESUs/DPSs of listed Pacific salmonids or adversely modify designated critical habitat (NMFS 2011) (Table 18). In May 2012, NMFS issued a biological opinion on the effects of three herbicides, oryzalin, pendimethalin, and trifluralin concluding registration of all three herbicides are likely to jeopardize listed species in the Central Valley and destroy or adversely modify their designated critical habitat (Table 18).

Table 18. Consultations with EPA's proposed FIFRA actions. Past Opinion's jeopardy (J) and adverse modification (AM) determinations pertaining to California Central Valley ESA listed Pacific salmonids.

Opinion number	Pesticide	ESU or DPS		
		CV Spring-run Chinook	Sac R. Winter-run Chinook	CCV Steelhead
1	Chlorpyrifos	J, AM	J, AM	J, AM
	Diazinon	J, AM	J, AM	J, AM
	Malathion	J, AM	J, AM	J, AM
2	Carbaryl	J, AM	J, AM	J, AM
	Carbofuran	J, AM	J, AM	J, AM
	Methomyl	J, AM	J, AM	J, AM
3	Dimethoate	J, AM	J, AM	J, AM
	Naled	J, AM	J, AM	J, AM
	Phosmet	J, AM	J, AM	J, AM
	Ethroprop	No J, No AM	J, AM	No J, No AM
	Phorate	J, AM	J, AM	J, AM
	Methidathion	J, AM	J, AM	J, AM
	Bensulide	J, AM	J, AM	No J, No AM
4	2,4-D	J, AM	J, AM	J, AM
	Triclopyr BEE	No J, No AM	No J, No AM	No J, No AM
	Diuron	AM, No J	AM, No J	AM, No J
	Linuron	No J, No AM	No J, No AM	No J, No AM
	Captan	No J, No AM	No J, No AM	No J, No AM
	Chlorthalonil	AM	AM	AM
5	Oryzalin	J, AM	J, AM	J, AM
	Pendimethalin	J, AM	J, AM	J, AM
	Trifluralin	J, AM	J, AM	J, AM

7.6. Rice Production and the Use of Thiobencarb in California

Rice growing is of interest because of its overlap with the distribution of listed salmonids. Figure 18 shows this overlap between rice and CV spring-run Chinook, Figure 19 shows the overlap with Sacramento River winter-run Chinook, and Figure 20 shows the rice overlap with CV steelhead. California is the nation's second largest rice producing state, with 2011 production totaling 4 ½ billion pounds on over one-half million acres (USDA 2012). Current rice production is mainly in nine Sacramento Valley counties (Butte, Colusa, Glenn, Placer, Sacramento, Sutter, Tehama, Yolo, and Yuba) as displayed in Figure 17 (CRC 2011). Rice is also farmed in counties outside the Sacramento Valley; however currently the acreages in the San Joaquin Valley are generally small (CRC 2011). Currently, more than 95 percent of the state's rice crop is grown within 100 miles of the city of Sacramento (CRC 2011).

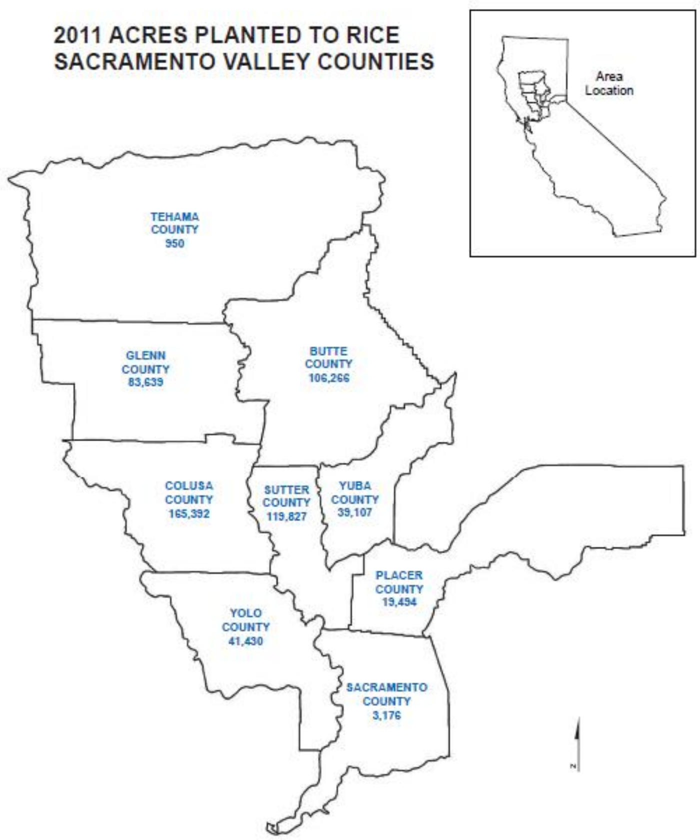


Figure 17. 2011 Sacramento Valley Rice Acres by County (CRC 2011).

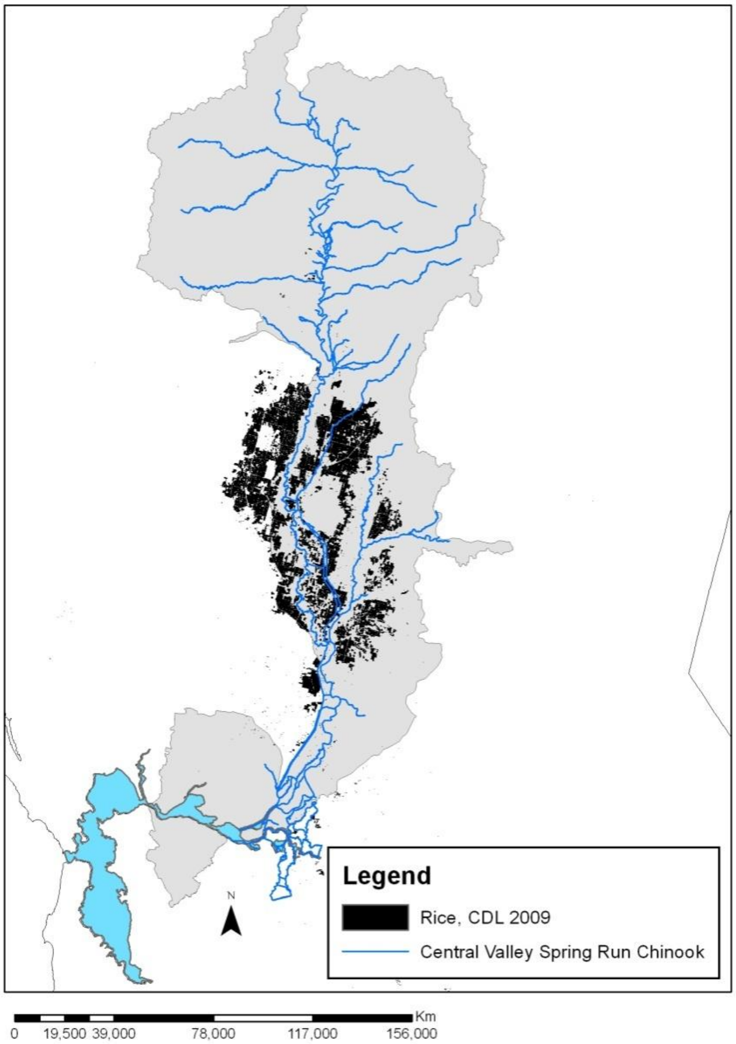


Figure 18. Central Valley rice growing overlay with listed CV spring-run Chinook ESU.

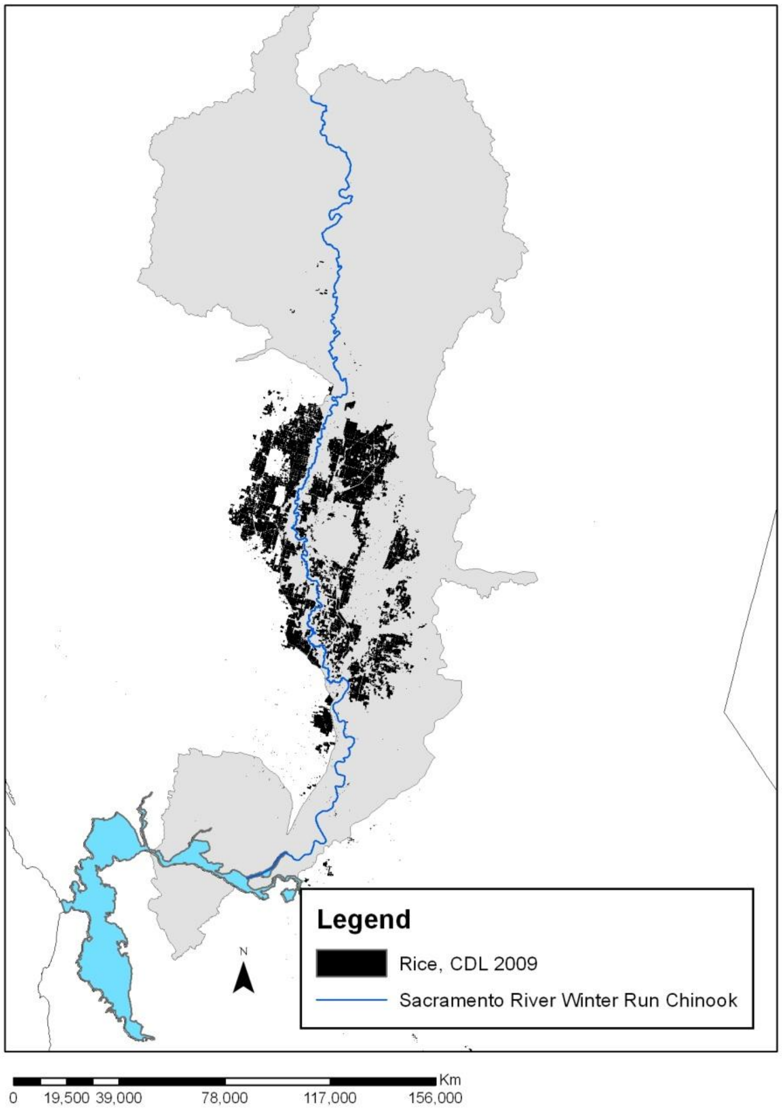


Figure 19. Central Valley rice growing overlay with Sacramento River winter-run Chinook ESU.

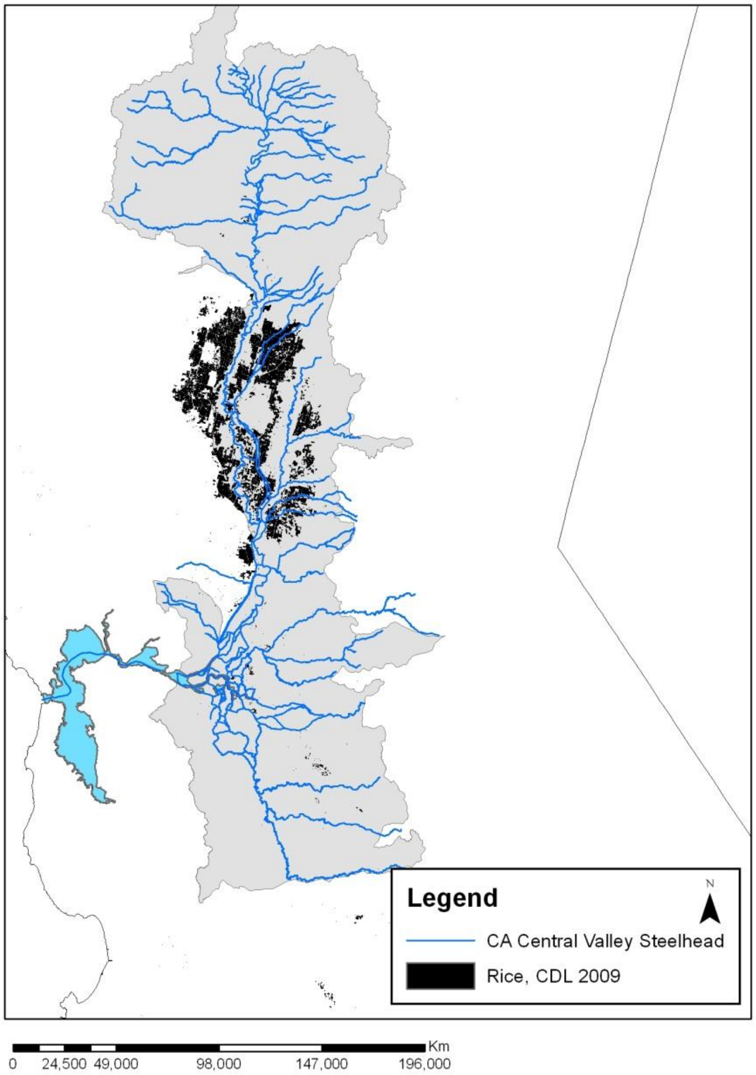


Figure 20. Central Valley rice growing overlay with CV steelhead DPS (note change in scale from previous figures).

Most California rice is produced by direct seeding into standing water (wet seeded), and a continuous flood is maintained for most of the season. Limited acreage is dry seeded, which also uses permanent flood after stand establishment. Seasonal rainfall and weather conditions influence rice planting and rice pesticide application. On a typical year, fields are planted from mid-April to the end of May, and field drainage occurs during August and September. Thiobencarb is typically applied in May (CRC 2011).

Reported annual use of thiobencarb in California declined from over one million lbs in 2000 to approximately 300 thousand lbs per year during the most recent surveys, 2006-2009 (CDPR 2010a). Use of thiobencarb products within the state is reflective of the distribution of rice, the only registered use site for thiobencarb. Although rice acreage and thiobencarb use has remained relatively stable in California in recent years, differences have been observed among the rice producing counties in both the amount of thiobencarb used, and the selection of thiobencarb products (CDPR 2010a). As reported by California Department of Pesticide Regulation, from 1980 to 2010, active ingredient thiobencarb has been typically applied at or near the maximum application rate of 4 lbs/A. Thiobencarb has liquid and granular (dry) formulations. The common liquid formulation in California is Abolish 8 EC. The most common granular formulation used in California is Bolero 15 G with a transition to Bolero UltraMax beginning in 2008 (both are 15 percent thiobencarb products). According to the CDPR Pesticide Use Report (PUR), the relative proportions of active ingredient thiobencarb used in California has been running approximately 25 percent as Abolish 8 EC and 75 percent as Bolero UltraMax (PUR database, January 26, 2012).

California has set a drinking water standard known as a maximum contaminant level (MCL) for thiobencarb at 1.0 µg/L as a nuisance for taste at the intakes for Sacramento and West Sacramento. California's Department of Pesticide Regulation (DPR), in cooperation with the Central Valley Regional Water Quality Control Board (CVRWQCB), developed recommended permit conditions to meet this water quality objective for thiobencarb. Under the CVRWQCB *Water Quality Control Plan (Basin Plan) for the Sacramento River and San Joaquin River Basins*, the performance goal in the agricultural drains is 1.5 µg/L. The CVRWQCB approved resolution can be reviewed at:

www.waterboards.ca.gov/centralvalley/board_decisions/adopted_orders/.

The rice industry, via the California Rice Commission (CRC), is responsible for water monitoring, annual reporting to the CVRWQCB, and coordinating the participation of all program stakeholders. The rice industry is ultimately responsible for meeting the water quality objectives. DPR as a co-regulator with the water boards continues to use their authority to regulate the sales and use of pesticides to address water quality issues involving pesticides. DPR works with CVRWQCB, the CRC, and the rice industry to address all rice pesticide issues and have developed criteria for water management, measures to minimize drift, and set minimum educational requirements for growers.[11]

7.6.1. Water Management

Rice growing requires water to be held on fields for extended periods of time. Water must be held on the fields after thiobencarb is applied. The duration period varies depending on formulation. According to the labels EPA is considering for re-registration, liquid formulations (e.g., Abolish) require a 14-day hold period after application. Granular formulations (e.g., Bolero) specify field water is not to be released within 30 days of application. However, DPR has additional hold time restrictions that are presented in Table 19 (CDPR 2011).

Table 19. California DPR thiobencarb water management requirements summary.

Water must be held for the indicated number of 24-hour periods on site or containment before release to State Waters.	Bolero 15-G	Bolero UltraMax	Abolish 8EC
	Hold	Hold	Hold
Single Field$_{(e)}$	30	30	19
Single field Southern areas only $_{(a)}$.	19	19	
Single permitted release into tailwater recovery system or pond onto fallow field [Except Southern area $_{(a)}$].	14$_{(b)}$	14$_{(c)}$	14$_{(b)}$
Multi-growers & district release onto closed recirculating systems.	6	6	6

11 With Resolution No. R5-2010-9001, the CRWQCB - Central Valley Region approved management practices for thiobencarb, that include the formation of a Storm Event Work Group, increased monitoring of thiobencarb, increased focus on seepage, restricting the use of thiobencarb near rivers, and increased education efforts including CRC-hosted preseason mandatory stewardship meetings.

Multi-growers & district release onto closed recirculating systems in Southern area(a).	6	6	
Release from closed recirculating system.	19	19	19
Release into area that discharge negligible amount into perennial streams	19	19	6(d)
Emergency Release of tailwater	19	19	19
Commissioner verifies the hydrologic isolation of the fields	6	6	6

a – Sacramento/San Joaquin Valley defined as: South of the line defined by Roads E10 and 116 in Yolo County and the American River in Sacramento County.
b – Thiobencarb permit condition allows Bolereo 15G label hold period of 14 days.
c – Thiobencarb permit condition allows Bolero UltraMax label hold period of 14 days.
d – See hydrologic isolation fields.
e – When drainage begins after 30 day hold, discharge must not exceed two inches of water over a drain box weir for seven additional days. Unregulated discharges from these fields may then begin after 37 days.

Currently Bolero UltraMax is primarily used on water seeded fields. Abolish 8EC is primarily used on dry-seeded fields or fields that must be drained prior to application.

Seepage through berms is a concern and sometimes an issue where treated fields can release thiobencarb earlier than the prescribed holding times. County agricultural commissioners (CACs) monitor for seepage when inspecting for water-holding compliance. Any visible seepage moving offsite during the water-holding period draining into waters of the State is considered an early release and is a violation of California's water-holding requirements. Over a recent 5 year span (2006-2010), on average CACs inspected 773 fields each year. On average 1.8 enforcement actions were taken; CRC annual reports may be found at: http://www.waterboards.ca.gov/centralvalley/water_issues/irrigated_lands/monitoring_plans_reports_reviews/monitoring_report_reviews/coalitions/california_rice_commision/.

7.6.1.1. *Water Monitoring – the Rice Pesticide Program*

The Rice Pesticide Program (RPP) is a longstanding watershed effort whereby rice growers follow Regional Water Board-approved management practices contained in use permits obtained from CACs when applying rice pesticides. The RPP includes monitoring of Sacramento Valley agricultural drains and the Sacramento River by the CRC (Figure 21 and Table 20). The Cities of Sacramento and West Sacramento also conduct monitoring at their drinking water intakes on the Sacramento River for thiobencarb. Monitoring is conducted twice weekly during the peak discharge period for six weeks from May until early June.

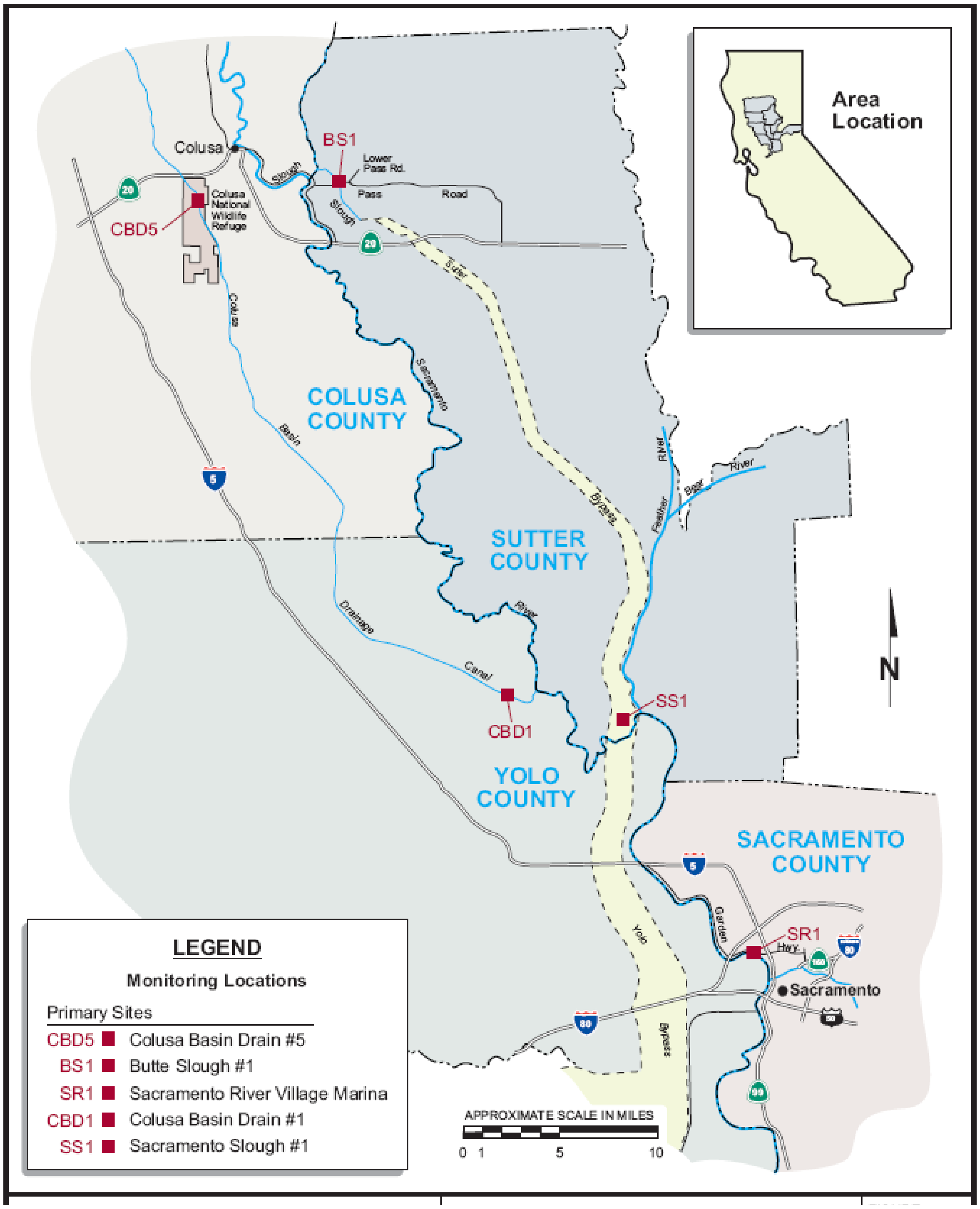


Figure 21. Rice Pesticide Program (RPP) Monitoring Sites

Table 20. RPP Monitoring Sites (CRWQCB-CVR 2005)

Abbreviation	Name (County Location)	Type
CBD5	Colusa Basin Drain (CBD) at Hwy 20 (Colusa County)	Ag drain
CBD1	CBD at Road 99E (Yolo County)	Ag drain
BS1	Butte Slough at Lower Pass Rd (Sutter County)	Ag drain
SS1	Sacramento Slough at DWR gauging station (Sutter County)	Ag drain
SR1	Sacramento River at Village Marina (Sacramento County)	River
Municipal Intake Sites		
SSR	City of Sacramento Intake, Sacramento River 0.3 Km downstream of the American River (Sacramento County)	River
WSR	City of West Sacramento Intake at Bryte Bend (Yolo County)	River

The CRC maintains close communication with the CACs during the thiobencarb use season. California spring weather conditions can result in occasional storms, which cause concern during the thiobencarb water-holding period. The CRC implemented a Storm Event Work Group to facilitate communication with the CACs, DPR, water boards, registrants, city stakeholders and rice growers during storm events, and to determine an increase in thiobencarb monitoring based on information gathered from the group.

CACs also monitor compliance with prescribed holding times (Table 19). Over the same five-year period, CACs conducted 4,569 inspections and issued 11 civil penalties for non-compliance (average 2.2 penalties per year).

7.6.2. Drift Minimization

California has stricter standards for minimizing drift from aerial applications then those required on EPA approved labels. These are summarized below:

- No aerial applications shall be made or continued within ½ mile of the Sacramento or Feather Rivers in the Sacramento Valley rice growing counties of Butte, Colusa, Glenn, Placer, Sacramento, Sutter, Tehama, Yolo, and Yuba unless there is a continuous positive airflow away from the river.
- In the Sacramento Valley rice growing counties of Butte, Colusa, Glenn, Placer,

Sacramento, Sutter, Tehama, Yolo, and Yuba, no aerial application shall be made or continued within ½ mile of the Sacramento or Feather Rivers when the wind speed exceeds 7 miles per hour.

- In Sacramento and Yolo Counties, no aerial applications shall be made or continued within ¼ mile of the Sacramento River unless they are made under the direct supervision of the county agricultural commissioner's representative.
- In Sacramento and Yolo Counties, the maximum acres treated by air each day within ¼ mile of the Sacramento River shall not exceed 33 percent of the average acres treated per day by air within this area in each county during 2002.

In addition to the above, DPR provides focused oversight inspections of thiobencarb aerial applications to monitor drift mitigation requirements. From 2006-2010, there were 144 inspections. During these inspections, six compliance violations were issued (CRC Annual Reports).

7.6.3. Applicator Education

The CVRQWCB resolution approving the use of thiobencarb under the Rice Pesticide Program, requires the CRC to host annual thiobencarb mandatory stewardship meetings. The pre-season meetings, in collaboration with Valent, the CACs, DPR and the CVRWQCB, provides stewardship to the permit applicant and/or his/her authorized representative. In addition, the CRC extends stewardship to the pest control advisors (PCAs) and pilots through collaboration with the California Aerial Applicators Association (CAAA). The permit applicant must have a certification of completion, issued by the CRC, in order to obtain a restricted materials permit from the CAC.

Thiobencarb is a restricted material in California requiring a permit is issued by the CAC in the county the rice is grown. The DPR restricted materials permit conditions require growers to add thiobencarb to the permit on an annual basis. Under the restricted materials program, the grower, or his/her representative must file a Notice of Intent (NOI) within 24-hours prior to commencing the use of a pesticide requiring a permit. The CACs retain the right to deny permits and thiobencarb NOIs if the field conditions are not adequate for a thiobencarb application.

All applications require a written recommendation from a licensed PCA prior to the submittal of the NOI. The permit is also required for the use and sale of the restricted material through a licensed dealer, or business. A licensed private applicator, licensed pilot, or qualified applicator must apply the restricted material as authorized by DPR.

The CDPR publishes voluntary interim measures for mitigating the potential impacts of pesticide usage to listed species. These measures are available online as county bulletins (http://www.cdpr.ca.gov/docs/endspec/colist.htm). Measures from these county bulletins that apply to thiobencarb use in salmonid habitat are:

- Do not use in currently occupied habitat except as specified in Habitat Descriptors, in organized habitat recovery programs, or for selective control of exotic plants.

- For sprayable or dust formulations: when the air is calm or moving away from habitat, commence applications on the side nearest the habitat and proceed away from the habitat. When air currents are moving toward habitat, do not make applications within 200 yards by air or 40 yards by ground upwind from occupied habitat. The CAC may reduce or waive buffer zones following a site inspection, if there is an adequate hedgerow, windbreak, riparian corridor or other physical barrier that substantially reduces the probability of drift.

8. Effects of the Proposed Action to Threatened and Endangered Pacific Salmonids

The analysis includes three primary components: exposure, response, and risk characterization. We analyze exposure and response, and integrate the two in the risk characterization phase where we address support for risk hypotheses. These risk hypotheses are predicated on effects to salmonids. Designated critical habitat is analyzed separately and predicated on effects to salmonid-supporting habitats (see *Effects of the Proposed Action to Designated Critical Habitat* and *Integration and Synthesis for Designated Critical Habitat*).

8.1. Exposure Analysis

In this section, we identify and evaluate potential exposure of salmonids to the stressors of the action (Figure 22). We begin by presenting general life history information of vulnerable life stages of Pacific salmon and steelhead. Next, we discuss the physical and chemical properties of thiobencarb and its degradation products that influence exposure of listed species and designated critical habitat to these stressors of the action. We then evaluate co-occurrence of salmon habitat with the stressors of the action by comparing the distribution of sites authorized for thiobencarb use by product labeling to the distribution of each species and their designated critical habitat.

To further characterize exposure where co-occurrence exists, we summarize EPA exposure estimates presented in BEs, present additional exposure estimates for shallow floodplain habitats utilized by salmonids, and summarize the available water quality monitoring data. Finally, we conclude with a summary of anticipated ranges of exposure when pesticide use is proximate to salmon habitats, and characterize the uncertainty contained in this analysis. Because the ESA section 7 consultation process is intended to insure that the agency action is not likely to jeopardize listed species or destroy or adversely modify critical habitat, NMFS considers a variety of exposure scenarios in addition to those presented in EPA's BEs. These scenarios provide thiobencarb estimates for the range of habitats used by listed salmonids.

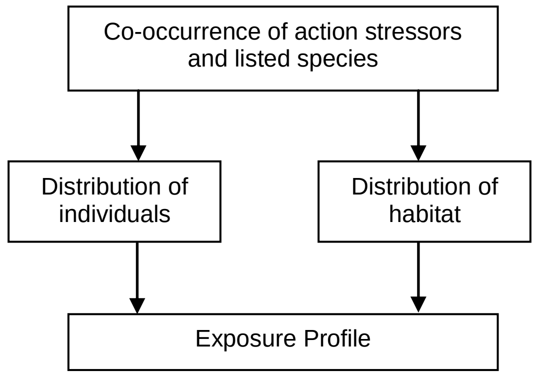


Figure 22. Exposure analysis

8.1.1. Threatened and Endangered Pacific Salmonids use of Aquatic Habitats

Within the *Status* section we discussed salmonid lifecycles, life histories, and the use and significance of aquatic habitats. Listed salmonids occupy a variety of aquatic habitats that range from shallow, low-flow freshwaters to open reaches of the Pacific Ocean. All listed Pacific salmonid species use freshwater, estuarine, and marine habitats at some point during their life. The temporal and spatial use of habitats by salmonids depends on the species and the individuals' life history and life stage as well as environmental factors such as river flows.

In this section we describe the habitats used by the three species of listed salmonids that occur in California's Central Valley rice growing region: Central Valley Spring-run Chinook salmon, Sacramento River Winter-run Chinook salmon, and California Central Valley Steelhead. General life history descriptions illustrating the use of aquatic habitats by Chinook salmon and steelhead are provided below in Table 21. Additionally, we describe species temporal use of aquatic habitats of the three species within California's Central Valley to determine potential spatial and temporal overlap with thiobencarb.

Table 21. General life histories of Pacific salmonids which utilize habitat that overlaps with thiobencarb spatial use patterns.

Species (number of listed ESUs or DPSs)	General Life History Descriptions		
	Spawning Migration	Spawning Habitat	Juvenile Rearing and Migration
Chinook (2)	Mature adults (usually three to five years old) enter rivers (spring through fall, depending on run). Adults migrate and spawn in river reaches extending from above the tidewater inland hundreds of miles from the Pacific. Migrating adults typically follow the thalweg. Chinook salmon migrate and spawn in four distinct runs (spring, fall, summer, and winter). Chinook salmon are semelparous[1].	Generally spawn in the middle and upper reaches of main stem rivers and larger tributary streams.	The alevin life stage primarily resides just below the gravel surface until they approach or reach the fry stage. Immediately after leaving the gravel, fry distribute to habitats that provide refuge from fast currents and predators. Juveniles exhibit two general life history types: Ocean-type fish migrate to sea in their first year, usually within six months of hatching. Ocean-type juveniles may rear in the estuary for extended periods. Stream-type fish migrate to the sea in the spring of their second year.
Steelhead (1)	Mature adults (typically three to five years old) may enter rivers any month of the year, and spawn in late winter or spring. Migrating adults typically follow the thalweg. Steelhead are iteroparous[2].	Usually spawn in fine gravel in a riffle above a pool.	The alevin life stage primarily resides just below the gravel surface until they approach or reach the fry stage. Immediately after leaving the gravel, swim-up fry usually inhabit shallow water along banks of stream or aquatic habitats on streams margins. Steelhead rear in a wide variety of freshwater habitats, generally for two to three years, but up to six or seven years is possible. They smolt and migrate to sea in the spring.

1 spawn only once
2 may spawn more than once

Freshwater, estuarine, and marine near-shore habitats are areas subject to pesticide loading from runoff and drift given their proximity to pesticide application sites. Small streams and many floodplain habitats are more susceptible to higher pesticide concentrations than other aquatic habitats used by salmon because their physical characteristics provide less dilution and dissipation. Examples of floodplain habitats include alcoves, channel edge sloughs, overflow channels, backwaters, terrace tributaries, off-channel dredge ponds, off-channel ponds, and braids (Anderson 1999, Beechie and Bolton 1999, Swift III 1979). Though floodplain habitats typically vary in surface area, volume, and flow, they are frequently shallow, low to no-flow systems protected from a river's, or a stream's, primary flow.

Rearing and migrating juvenile salmonids use floodplain habitats extensively (Beechie and Bolton 1999, Beechie et al. 2005, Caffrey 1996, Henning et al. 2006, Montgomery 1999, Morley et al. 2005, Opperman and Merenlender 2004, Roni 2002). Diverse, abundant communities of invertebrates also populate floodplain habitats and many species represent an important food source for salmon. The presence of abundant food resources is partially responsible for juvenile salmonids reliance on these habitats. Both stream-type juvenile Chinook salmon and steelhead use floodplain habitats for extended durations (several months).

8.1.2. Chemical Exposure Pathways to Salmonids Habitats

Pesticides can contaminate surface waters via runoff, erosion, leaching, spray drift from application at terrestrial sites or direct application to aquatic habitats, and atmospheric deposition. Aquatic habitats can be contaminated by pesticides applied to terrestrial target sites through several pathways and by direct application to surface waters for control of plants, mosquitoes, and other aquatic pests. For example, spray-applied pesticides may result in off-target deposition of droplets at the time of application. The likelihood of spray drift to an aquatic habitat is determined by the application method, the proximity to the habitat, and meteorological conditions at the time of application.

Surface water contamination is also influenced by the environmental fate properties of the chemical. For example, secondary drift or vapor drift is dependent on a chemical's volatility and refers to the redistribution of pesticides from plant and soil surfaces through volatilization and subsequent atmospheric deposition. Runoff and leaching, the horizontal and vertical movement of pesticides with rainwater or irrigation water, are influenced by chemical-specific properties that determine the compound's persistence and mobility in soil and water. Standardized tests are typically used to characterize mobility (*e.g.,* solubility, K_d and K_{oc}) and persistence under different environmental conditions (*e.g.,* hydrolysis, photolysis, and metabolism half-lives in aerobic and anaerobic environments). Below we present environmental fate properties of thiobencarb to characterize the relative importance of different exposure pathways in terms of the potential for the active ingredient and its degradates to contaminate salmonid bearing habitats and designated critical habitats.

8.1.3. Summary of Chemical Fate of Thiobencarb

Thiobencarb can be applied to rice either before or after rice fields are flooded. After a specified holding period, rice irrigation water can be discharged to salmonid habitats via a system of drains within the Central Valley of California. Salmonids may be exposed to thiobencarb when present in the water column where the chemicals cross gill surfaces during respiration, or where fish sensory systems come in direct contact with contaminated water (*i.e.,* olfactory sensory neurons). Other routes may contribute to overall exposure including incidental ingestion of the chemical in sediment or ingestion of the chemical in food items. Below we summarize chemical fate properties of thiobencarb reported by EPA. Where discrepancies existed between EPA assessments, we deferred to the more recent document.

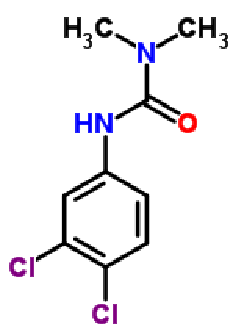
8.1.3.1. 

Figure 23. Chemical structure of thiobencarb

Thiobencarb (Figure 23) is a thiocarbamate herbicide relatively stable to abiotic and biotic degradation in water (Table 24). However, it is not expected to persist in surface water given its affinity for soil and sediment. Two field studies reported median partitioning of thiobencarb residues of 5.6:1 and 6.6:1 for the soil:water ratio (EPA 2009b). Aqueous photolysis occurs slowly in surface waters with a half-life of 190 d. However, it may occur more quickly in the presence of photo-sensitizers based on its degradation in acetone (half-life of 12 d). The photodegradation rate of thiobencarb in the presence of relevant photo-sensitizers, such as humic substances that occur in surface waters, is uncertain. Degradation in soils also occurs slowly, as evidenced by soil photolysis half-lives of 168-280 d and aerobic soil metabolism half-lives of 27-58 d. The K_{oc}'s of 384-1438 indicate thiobencarb is moderately to slightly mobile in the soil. The vapor pressure indicated that thiobencarb has an intermediate volatility from dry surfaces and the Henry's Law Constant suggests it will not volatilize from water (EPA 2009b). Thiobencarb has been found in trace concentrations in air and precipitation and is expected to travel up to 5 km from the site of application. However, thiobencarb has an atmospheric degradation half-life of 0.421 days suggesting a limited potential for long-range transport (EPA 2009b). Potential transport mechanisms include primary and secondary spray drift, surface water runoff, and rice paddy discharge and seepage. Thiobencarb moderately accumulates in fish tissue but depuration occurs rapidly with 93-95% of residues eliminated in three days (EPA 2009b).

Table 22. Environmental fate characteristics of thiobencarb[1].

Parameter	Value
Water solubility	30 mg/L at 20 °C
Vapor pressure	1.476×10^{-6} - 2.2×10^{-5} mm Hg
Henry's law constant	2.49×10^{-7} atm m^3 mol^{-1}
Octanol/Water partition coefficient	Log K_{ow} = 1.3 – 3.42
Hydrolysis ($t_{½}$) pH 5, pH 7, & pH 9	Stable
Aqueous photolysis ($t_{½}$)	190 d ; 12 d in acetone
Soil photolysis ($t_{½}$)	168 - 280 d
Aerobic soil metabolism ($t_{½}$)	27 - 58 d
Anaerobic soil metabolism ($t_{½}$)	Stable
Aerobic aquatic metabolism ($t_{½}$)	Stable
Anaerobic aquatic metabolism ($t_{½}$)	Stable
Soil partition coefficient	K_{oc} = 384-1435 L/kg_{soil}
Fish Bioconcentration Factor (BCF)	128x (edible) 639x (non-edible) 411x (whole fish)

1- (EPA 2009b)

8.1.3.2.

8.1.3.3. *Degradates of thiobencarb*

The molecular structure of a pesticide may be modified by biotic (*e.g.,* microbial metabolism) or abiotic (*e.g.,* photolysis and hydrolysis) processes. The products of these processes typically have different toxicities, environmental fate characteristics, and risks compared to the parent pesticide. EPA indicated that the main transformation products of thiobencarb found in laboratory photolysis and metabolism studies were 4-chlorobenzoic acid, 4-chlorobenzaldehyde, and carbon dioxide (EPA 2009b). Two additional microbial metabolites were identified as thiobencarb transformation products monitored in field dissipation studies (Table 23).

Table 23. Degradates of thiobencarb (EPA 2009b).

Transformation product	Percent of applied parent
4-chlorobenzoic acid	56[1] 5[2]
4-chlorobenzaldehyde	29.4[3]
Carbon dioxide	23 - 77[2]
Thiobencarb sulfoxide	Not specified[4]
4-chlorobenzylmethylsulfone	Not specified[4]

1- Aqueous photolysis
2- Soil metabolism (aerobic or anaerobic conditions not specified)
3- Photolysis (aqueous or soil matrix not specified)
4- Aquatic field dissipation studies

Existing laboratory studies provided insufficient information to characterize aquatic exposure. Field dissipation studies are relatively more informative than laboratory studies because they provide information on the concentrations and persistence of thiobencarb and two metabolites in flooded rice paddies (Table 24).

Table 24. Concentration of thiobencarb and metabolites reported in field dissipation studies

Study	Results
Field dissipation of Bolero 8EC in rice (Lai 1991)(MRID#42003404) Port Barre, Louisiana field site Aerial application of liquid formulation at 4 lbs a.i./acre to dry-seeded rice A "flush irrigation" was conducted 3 days post application. The plots were permanently flooded to 4.5 inches at 7 days post-application	**Parent Thiobencarb** **Mean concentration (µg/L)** Day 7: 8.2 Day 8: 11.9 **Day 10: 13.0 (highest mean concentration 3 days post flooding)** Day 12: 10.3 Day 28: 0.58 **Thiobencarb sulfoxide** Highest mean concentration 8.9 µg/L, Day 8 (1 day post-flood) **4-chlorobenzylmethylsulfone** Highest mean concentration 5.2 µg/L, Day 12

Study	Results
	(5 days post-flood) **Dissipation half-life:** Parent thiobencarb 5.8 d Thiobencarb sulfoxide 3.4 d 4-chlorobenzylmethylsulfone 6.0 d
Aquatic field dissipation of Bolero 10G in rice (Ho 1990) (MRID#43404005) Nelson, California field site Aerial application of granular formulation to flooded rice at a rate of 4 lbs a.i./acre Flood water maintained for 6 days then released to fallow check where it was held for 8 days then re-circulated	**Parent Thiobencarb** **Mean concentration µg/L** Day 0: 267 Day 1: 252 Day 2: 341 **Day 3: 438** Day 5: 159 ----------- drained----------- Day 14: 44 Day 33: 3 Day 92: 1 **Peak mean concentration 438 µg/L observed 3-d post application** **Thiobencarb sulfoxide** **Day 3 highest mean concentration: 22 µg/L** **4-chlorobenzylmethylsulfone** **Day 10 highest mean concentration: 8.3 µg/L** **Dissipation half-life:** Parent thiobencarb 8.7 d Thiobencarb sulfoxide 2.8 d 4-chlorobenzylmethylsulfone 10.4 d
Fate of thiobencarb and molinate in rice fields (Ross and Sava 1986) Aerial application of granular formulation	**Parent Thiobencarb (No degradates measured)** **Mean concentration µg/L** Day 0: 79 Day 2: 567 **Day 4: 576**

Study	Results
to flooded rice at a rate of 4 lbs a.i./acre Water depth held at 10 inches ± 2 inches for 6 day holding period, then all water drained from field. Water depth was then maintained at 7 inches ± 2 inches with inflow/outflow during post-holding	Day 6: 515 ----------- drained----------- Day 8: 367 Day 16: 56 Day 32: 8 **Peak mean concentration 576 µg/L observed 4-d post application** **Dissipation half-life > 6 days** Thiobencarb did not decline significantly during the 6-d holding period

8.1.4. Exposure of salmonid habitats to the stressors of the action

8.1.4.1. *Co-occurrence associated with thiobencarb use.*

We evaluated co-occurrence of listed salmonids with stressors of the action by comparing the spatial and temporal distribution of salmon with potential use of thiobencarb based on label specifications. To evaluate the areal extent of application sites near salmon-bearing waters, we used a GIS overlay containing land use classifications and salmon distributions[12] to determine overlap with the three California Central Valley ESUs/DPSs (Table 16, Figure 11). Based on the 2006 National Land Cover Database, the spatial coverage of cropland accounts for approximately 21-26% of the total area within the freshwater distribution of the three species (Table 25). Because cropping patterns may change over time, the land use classification for cropland was used for the initial evaluation of registered uses. Statistics from CDPR indicate that more than a half million acres have been planted annually in rice in the Sacramento Valley since 2006 (CDPR 2011). Rice is the dominant crop in the Sacramento Valley and accounts for more than 6% of the total area with the distribution of the California Central Valley Steelhead and more than 8% of the total areas within the distribution of spring-run and winter-run Chinook.

[12] http://www.nwr.noaa.gov/ESA-Salmon-Listings/Salmon-Populations/Maps/

Table 25. Co-occurrence of listed Pacific salmonids with potential application of pesticides to use sites within the salmonids' freshwater distribution.

ESU / DPS	Spatial coverage of cropland within species freshwater distribution	Temporal overlap of ESU and labeled use of Thiobencarb
Central Valley Spring-Run Chinook Salmon	21.0%	Yes
Sacramento River Winter-Run Chinook	21.0%	Yes
California Central Valley Steelhead	26.4%	Yes

There is temporal overlap of approved labeled uses with species presence in freshwater habitats because most of the listed Pacific salmonids in California occur in freshwater year-round in some life stage, and thiobencarb labels place no restrictions on the timing of application. Historically thiobencarb has been applied most frequently in the months of May and June (Figure 24). A considerable number of applications (20 or more during the month in a given year) have occurred as early as April and as late as July (Figure 24). Application of thiobencarb during other months has been extremely rare (average 0 - <1 application/month). Peak aquatic concentrations associated with the drift pathway are expected to occur on the day of application. Exposure associated with thiobencarb discharge from treated fields is expected to occur after the 14 and 30 day holding requirements have been met for liquid and granular formulations, respectively. However, exposure associated with discharge from treated fields may also occur earlier due to thiobencarb seepage through berms or checks or through emergency releases due to flooding or salinity.

After reviewing thiobencarb's label authorized use and historical use in California, we expect designated critical habitats of the listed Central Valley salmonids to be exposed to thiobencarb from April – July, and particularly during May and June (Figure 24). The application and discharge period for thiobencarb overlaps with the general timing of freshwater residence of several salmonid life stages (Table 26). For example, the peak May – June application period overlaps with adult freshwater migration and juvenile freshwater rearing and migration in all three Central Valley species. Sacramento River winter-run Chinook are also actively spawning, and are present in freshwater as eggs, alevin, and fry during the application and discharge period (April-July). While Central Valley steelhead are likely to have completed spawning activities prior to applications in May and June, eggs, alevin and fry are all present in freshwater habitats during this timeframe. From these patterns, we expect all three species and their designated critical habitat to be exposed to thiobencarb and other ingredients in thiobencarb products. .

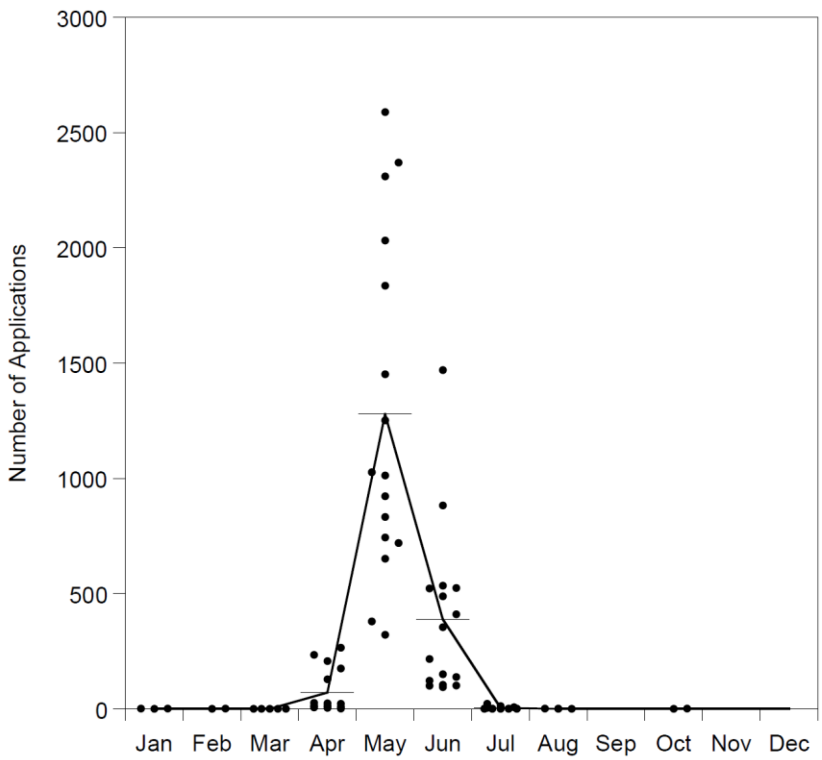


Figure 24. Number of thiobencarb applications in California by month from 1995-2010 (source: CDPR PUR database). Points in figure represent the number of application for a year. Months with zero applications are not plotted. The solid horizontal line represents the monthly mean number of applications over all of the years.

Table 26. Generalized run-timing of ESA-listed Central Valley Pacific Salmonids by life stage.

ESU/DPS	Life Stage	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Central Valley Spring-run Chinook	Adult prespawn migration												
	Spawning												
	Incubation												
	Alevin to fry emergence												
	Rearing and migration												
Sacramento River Winter-run Chinook	Adult prespawn migration												
	Spawning												
	Incubation												
	Alevin to fry emergence												
	Rearing and migration												
Central Valley Steelhead	Adult prespawn migration												
	Spawning												
	Incubation												
	Alevin to fry emergence												
	Rearing and migration												

8.1.4.2. *Species-specific temporal and spatial considerations*

Because thiobencarb is registered for use on only one crop that has a relatively localized and predictable distribution, we were able to take a more in-depth approach to assess temporal and spatial co-occurrence than was possible in previous NMFS Opinions on EPA registration actions. This approach assumes that within the distribution of listed species, (1) rice production will continue to be confined to the Central Valley of California, and (2) thiobencarb will not be approved for other uses within the species range. Reinitiation of formal consultation will be required if there are changes in these use patterns or Washington, Oregon, or Idaho authorize use of thiobencarb (see *Reinitiation Notice* below). We expect co-occurrence of thiobencarb use with several different life-history stages of each listed species (Figure 24, Table 26). To understand the extent of potential spatial and temporal overlap between thiobencarb use (April –

July) and species presence we discuss each species' adult and juvenile migration through rice growing areas. Figure 25 shows the distribution of rice in the Sacramento and upper San Joaquin Valleys according to the 2010 NASS Crop Data Layer.

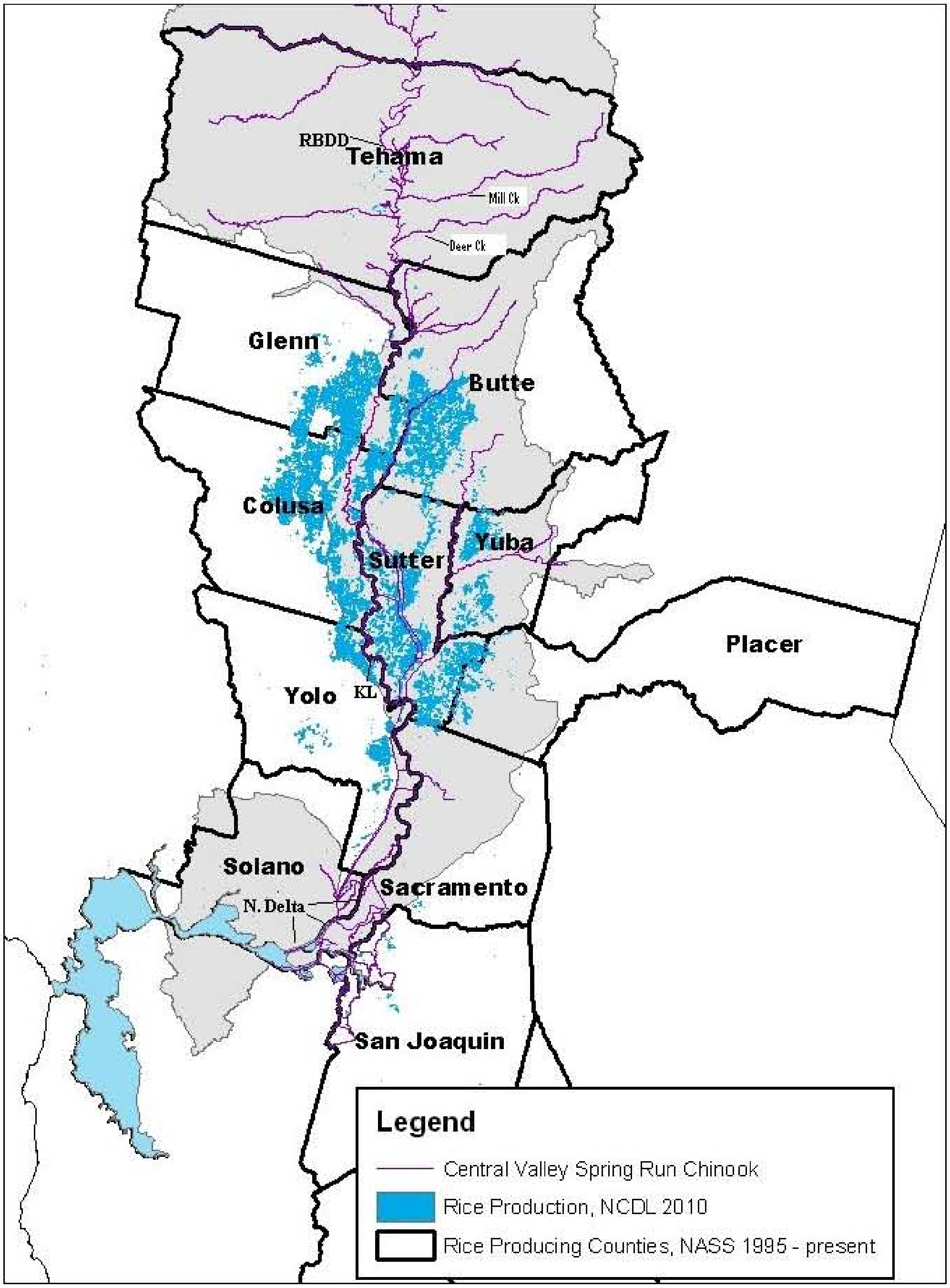


Figure 25. Distribution of rice in the Sacramento Valley based on the 2010 NASS Crop Data Layer. North Delta, Knights Landing (KL) and Red Bluff Diversion Dam (RBDD) fish sampling sites and distribution of the Central Valley Spring Run Chinnok are also shown.

8.1.4.3. *Abundance of the Three Species in CA Central Valley*

The relative abundance of the listed salmonids at various sampling stations is reported in Table 27, Table 29, and Table 31. We scored relative abundance as "low," "medium," or "high" at the different sampling locations over two-week intervals qualitatively by comparing capture rates at sample stations to overall abundance of each species observed at that station during the year. For example, several thousand spring-run Chinook are monitored annually at the Knights Landing (KL) station (Snider and Titus 2000a, Bajjaliya and Vincik 2008a). The relative abundance ratings in Table 27 of low, medium, and high correspond to capture rates in the 10's, 100's, and 1,000s, respectively, for that species. We used this information to determine the likely extent of co-occurrence of the three species with thiobencarb (Table 28, Table 30, and Table 32). This qualitative analysis is based on two primary assumptions: first, the spatial pattern of current rice growing area as defined by the 2010 NASS Crop Data Layer is a reasonable surrogate for future thiobencarb use sites, and, second, that the timing of thiobencarb applications and release of water from thiobencarb treated fields will remain relatively comparable for the next 15 years. We find support for these assumptions from the conditions needed for successful rice growing in California's CV including temperature, water, and soil. These necessary conditions are not met by other geographic areas within CA, therefore, the geographical expansion outside of the Central Valley is unlikely (Hill et al. 1992, California 1993)

Using fish collection data from CDFG and USFWS, we evaluated typical species presence at sampling stations on the Sacramento River (at Red Bluff Diversion Dam (RBDD) and Knights Landing (KL)), in the North Delta of the Sacramento River (at Chipps Island and Hood), on the San Joaquin River (at Mossdale), and on the Mokelumne River (at Woodbridge dam) (Figure 25, (USFWS 2010a, USFWS 2012a, Marcotte 1984a, Bajjaliya and Vincik 2008a, Snider and Titus 2000a). We used these monitoring locations to help us understand the movement of the species upstream as adults and downstream as juveniles as they entered and left current rice growing areas. The current distribution of rice in California is predominately located in the Sacramento Valley downstream of the RBDD and upstream of the North Delta sampling stations. The KL sampling station is within current rice growing areas. Steelhead monitored at Mossdale and

Woodbridge stations also represent occurrence in rice growing areas, although rice is grown to a lesser extent in these locations compared to the KL area on the Sacramento River.

8.1.5. Analysis of Species Presence in Rice Growing Areas

8.1.5.1. *Co-occurrence of Central Valley spring-run Chinook salmon with thiobencarb*

It is likely that adult Central Valley (CV) spring-run Chinook will be exposed to thiobencarb as they travel through rice growing areas while migrating up-stream to cooler, higher elevation waters because their abundance is high in rice growing areas when thiobencarb is most frequently applied (Table 7). Since peak aquatic concentrations associated with the drift pathway are expected to occur on the day thiobencarb is applied, individuals may be exposed multiple times during their upstream migration (*i.e.*, they may be exposed to thiobencarb at multiple locations given the distribution of potential application sites along their migration corridor). Individuals could also be exposed during run-off events involving normal discharges, unintended seepage from the fields, or from approved emergency discharges. The occurrence of seepage has been relatively infrequent, occurring in less than 0.5% of rice fields inspected over a five year period (Table 43).

Adult holding and spawning mostly occur at higher elevations in the upper main stem Sacramento River tributaries of Tehama and Butte counties (*e.g.*, Mill Creek and Deer Creek, (Figure 26), while rice is grown primarily on the valley floor (Lindley et al. 2004a, Yoshiyama et al. 1998, Marcotte 1984a). In the main stem, most spawning is above RBDD; in Mill Creek spawning extends from above the Little Mill Creek confluence upstream to about one mile above the Highway 36 bridge; in Deer Creek most spawn from about the Ponderosa Way bridge upstream to upper Deer Creek falls; in Butte Creek, most spawning takes place about 40 miles upstream from the mouth. Some spring-run fish may spawn in the main stem below RBDD but estimates are not available and these fish are often considered fall-run Chinook (CDFG 1995, Marcotte 1984a).

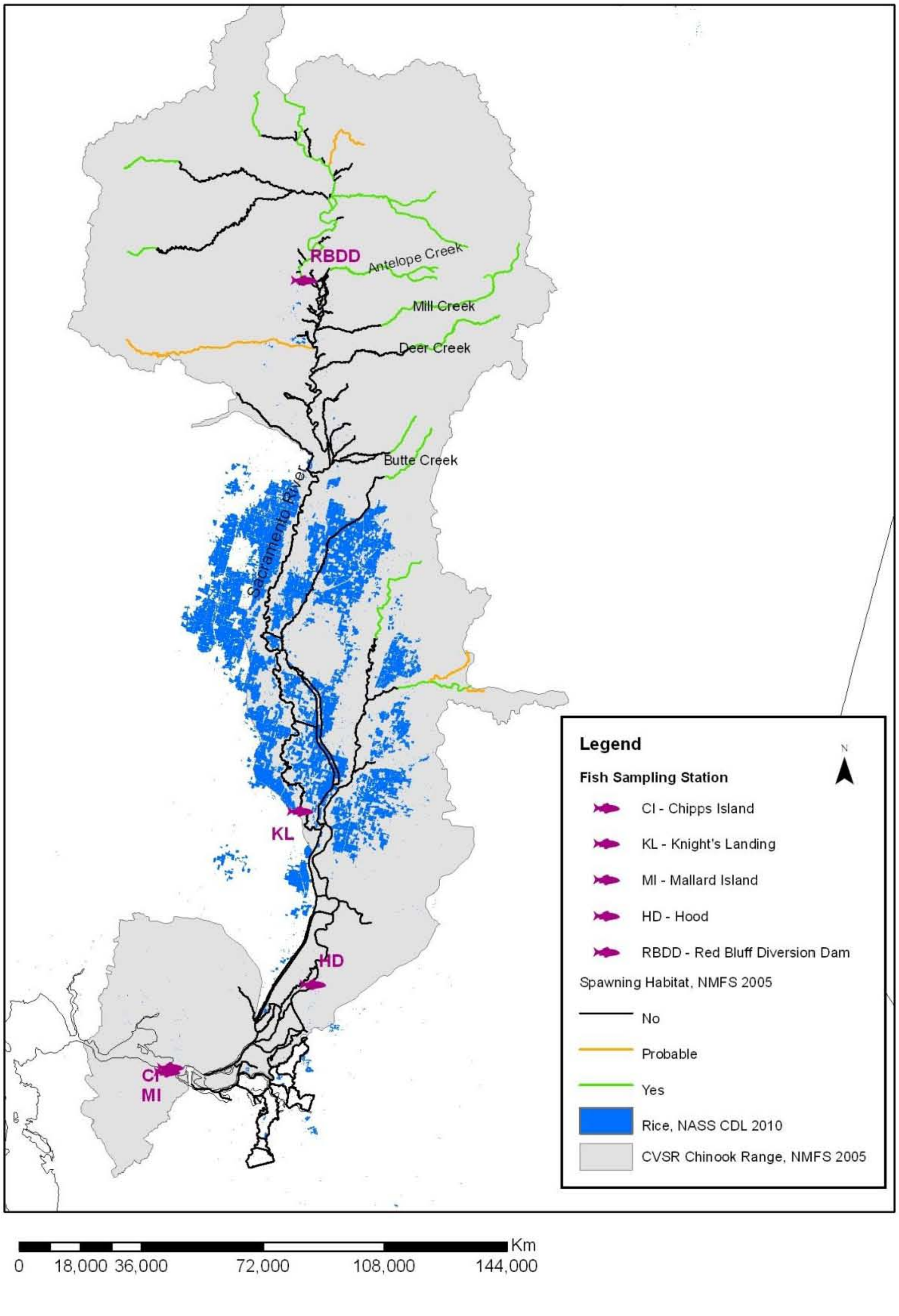


Figure 26. CV spring-run Chinook probable and known spawning areas in relation to 2010 rice growing areas.

(Good et al. 2005). Spawning also occurs in the Feather River, but it begins in August and ends in October. Overall, we do not anticipate spawning, egg incubation, and fry emergence to co-occur with thiobencarb runoff in rice growing areas.

Exposure to significant concentrations of thiobencarb via primary drift is also not anticipated given what we understand about the timing and proximity of spawning to expected thiobencarb use sites (Table 28). A drift event during application of thiobencarb liquid formulations could lead to deposition into spawning areas, but we expect such deposition to be limited and more of an exception than the rule. Furthermore, should primary drift occur, it would likely happen prior to the onset of spawning.

After fry emergence, most juveniles rear in their natal streams (Mill, Deer, and Butte creeks) upstream of the rice growing areas (Lindley et al. 2004a, Marcotte 1984a). Some fry (young of the year) may begin migration downstream post emergence from the gravel, but this typically occurs in the fall (Lindley et al. 2004a). Based on fish monitoring programs in the Sacramento (Snider and Titus 2000a, Bajjaliya and Vincik 2008a, Marcotte 1984a, USFWS 2010b), juvenile migration out of natal streams and through rice growing areas is initiated in late summer and largely complete by May, when the majority of thiobencarb is typically applied (Figure 24 and Table 27). Juveniles are found below Knights Landing (KL in Figure 25 and Table 27) and in the North Delta by April (Figure 25 and Table 27). Some juveniles do occupy waters in rice growing areas into early May. We expect few juveniles to occupy water within, or adjacent to, rice areas in June. Exposure from drift is expected to be limited as juvenile abundance in close proximity to rice fields is expected to be low during applications (Table 27 and Table 28).

Juveniles are abundant in the San Francisco Bay estuary (*i.e.*, in the North Delta; downstream of rice growing areas) during thiobencarb application and discharge periods. We expect exposure in the estuarine habitats will be at concentrations that are low compared to concentrations that are found in salmon habitats located closer to field discharge. For example, the highest concentration reported by Kuivila and Jennings (2007) in eastern Suisun Bay, at Mallard Island (approximately one-half mile from Chipps Island), was 66 ng/L, 1-2 orders of magnitude lower than values frequently detected at sample stations located near rice cultivation (summary of *Rice Pesticide Monitoring Program* reported below).

The Yolo Bypass is a historical floodplain of the Sacramento River that is now utilized to manage flood waters and produce a variety of crops. It is seasonally inundated with water to varying degrees and is recognized as an important rearing area for salmonids in the Sacramento Valley. During high water years, when the Yolo Bypass floods, juveniles are expected to occupy the Yolo and linger there until flood waters recede (Sommer et al. 2001, Sommer et al. 2005a). Rice growing does occur in the Yolo Bypass, but is delayed until after flood waters recede. Therefore, during flood years when juveniles have access to the Yolo Bypass, co-occurrence with thiobencarb use is unlikely.

Table 27. Temporal occurrence of adult and juvenile CV spring-run Chinook salmon in the Sacramento River. Darker shades indicate months of greatest relative abundance in each location along the river.

Adult upstream migration												
Location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Sac. River basin												
Up Sac. R. mainstem												
Juvenile rearing and downstream migration												
Location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Sac. R. at RBDD[1]												
Sac. R. at KL[2]												
North Delta[3]												
Thiobencarb use/release												
Relative Abundance		= High				= Medium					= Low	

[1] upstream of current rice growing areas; [2] within current rice growing areas; [3] downstream of current rice growing areas.

Table 28. CV spring-run Chinook relative abundance in rice areas (Figure 25) during thiobencarb use periods April - July.

Life Stage	None Expected	Low	Medium	High
Adult Migration				X
Spawning	X			
Early rearing		X		
Juvenile migration		X		

8.1.5.2. *Co-occurrence of Sacramento River winter-run Chinook with thiobencarb*

Typically, adult Sacramento River winter-run Chinook salmon enter fresh water in winter or early spring. The adults migrate through rice growing areas of the Sacramento River Basin en route to spawning grounds in medium abundance during the time when thiobencarb is usually applied and discharged. Spawning usually occurs between May and June, the peak period of thiobencarb applications. Moderate numbers of adults moving up-river to reach spawning grounds are likely exposed to thiobencarb by drift or runoff from thiobencarb applications (Table 30). As discussed in the *Status of Listed Resources* section, winter-run Chinook spawning is now restricted within roughly 44 miles of the main-stem upper Sacramento River immediately below Keswick Dam (Figure 27). A majority of the spawning (greater than 50 percent) is between Keswick Dam and the ACID Dam approximately 5 miles away. All of the spawning is located upstream of current RBDD, above the areas where rice is currently cultivated (Figure 25 and Figure 27). Consequently, we do not expect spawning, egg incubation, and fry emergence to co-occur with thiobencarb based on the current location of spawning areas.

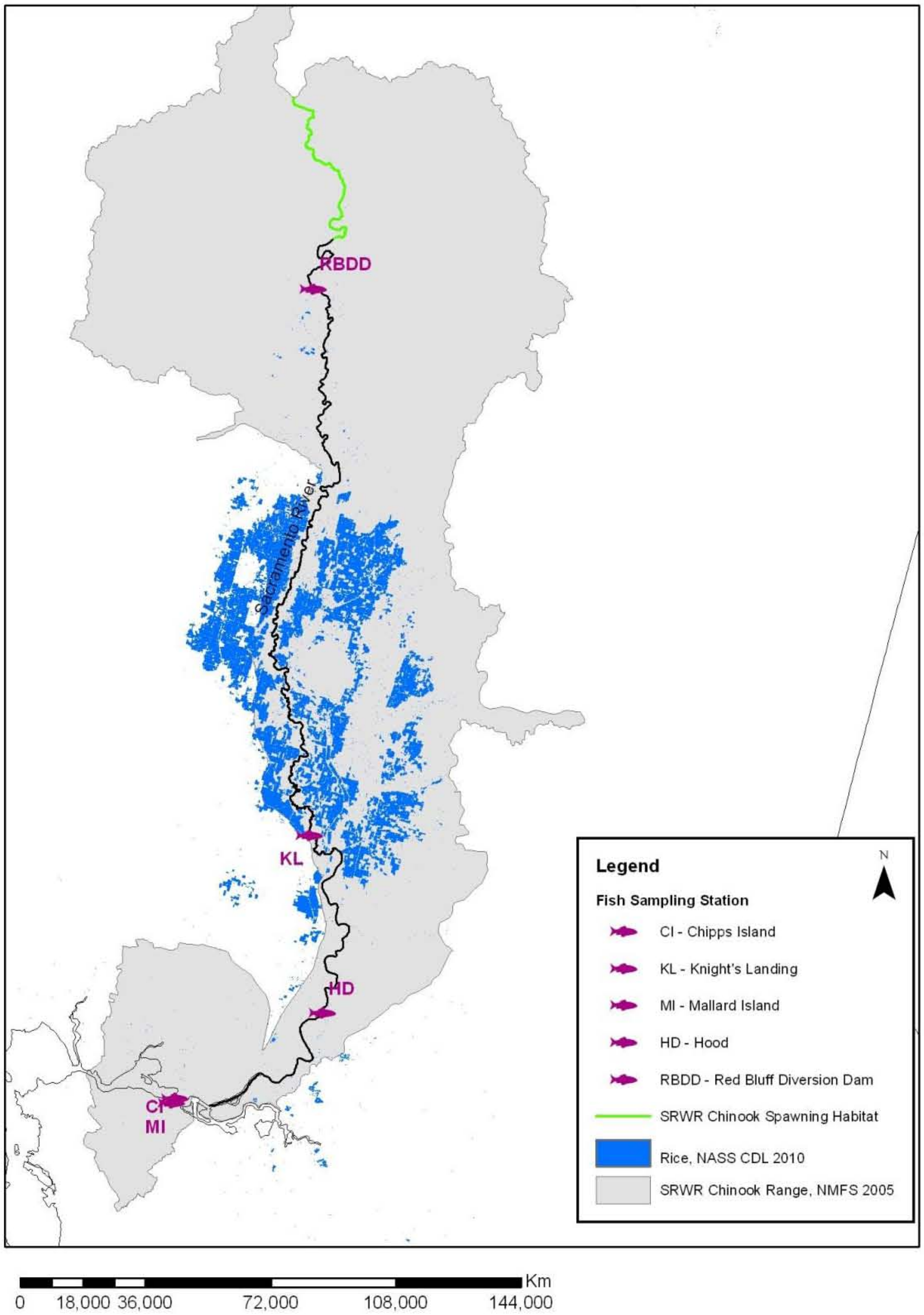


Figure 27. Sacramento River winter-run Chinook spawning areas relative to rice growing areas. Spawning is confined to the 44 river miles immediately downstream of Keswick Dam (Yoshiyama et al. 1998). In California, Rice is primarily cultivated in the Sacramento Valley downstream of the Red Bluff Diversion Dam (RBDD).

Upon emergence, most fry take up residence in their natal stream for several weeks to a year or more, while others are displaced downstream by the stream's current. Early hatched fry that do move down soon after emergence, instead of rearing in their natal waters, may be exposed to thiobencarb. Monitoring studies suggest juveniles are either absent, or in low abundance, in rice growing areas during the months of April – July when thiobencarb exposure is most likely (Table 29). Relative abundance estimates suggest most fish will have passed through the lower end of the rice growing area (*i.e.*, the lower Sacramento River monitoring stations) prior to May. Early hatched fry start arriving at the Red Bluff Diversion Dam (RBDD) in low relative abundance in July. While the RBDD is above rice growing areas (Figure 25), some of these fish could migrate down into waters that may receive late discharge waters off rice fields following a 30-day hold. However, according to fish monitoring studies, most juvenile winter-run Chinook migrate through the rice growing areas during the period when thiobencarb is not being applied, August – April (Snider and Titus 2000a, Bajjaliya and Vincik 2008a, Marcotte 1984a, USFWS 2010b). This suggests exposure from drift and runoff into habitats in close proximity to rice fields would occur to a relatively low number of individuals (Table 30).

As with the spring-run Chinook, the Yolo Bypass and North Delta represent important rearing areas for winter-run Chinook. During high water years, when the Yolo Bypass floods early, juveniles may linger in floodplains habitats until flood waters recede. However, rice growing in the Yolo Bypass is delayed until after flood waters recede. Therefore, during flood years when juveniles have access to the Yolo Bypass, co-occurrence with thiobencarb use is unlikely. Juveniles rear in high abundance in the North Delta during the periods of thiobencarb application and discharge of thiobencarb-treated fields. Although these habitats are downstream of rice growing areas, they are generally not in close proximity to rice fields. Exposure to thiobencarb is expected to be significantly less than predicted by drift and runoff models for water bodies in close proximity to rice fields.

Table 29. Temporal occurrence of adult and juvenile winter-run Chinook salmon in the Sacramento River. Darker shades indicate months of greatest relative abundance in each location along the river.

Adult migration												
Location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Sac. River basin												
Upper Sacramento River												
Juvenile migration												
Location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Sac. River at RBDD[1]												
Sac. River at KL[2]												
Lower Sac. River[3]												
North Delta[3]												
Thiobencarb use/release												
RBDD=Red Bluff Diversion Dam KL = Knights Landing												
Relative Abundance:	= High			= Medium				= Low				

[1] upstream of current rice growing areas; [2] within current rice growing areas; [3] downstream of current rice growing areas.

Table 30. Sacramento River winter-run Chinook relative abundance in rice areas (Figure 25) during thiobencarb use periods April – July.

Life Stage	None expected	Low	Medium	High
Adult migration			X	
Spawning	X			
Early rearing		X		
Juvenile migration		X		

8.1.5.3. *Co-occurrence of Central Valley steelhead with thiobencarb*

Generally, CV steelhead that utilize the Sacramento River Basin enter freshwater from late June to April, but some individuals are present in freshwater year-round. Steelhead that utilize the San Joaquin and its tributaries generally begin to enter the river in July. Relatively few pre-spawn adults migrate through rice production areas (Figure 24) at the time thiobencarb is applied or when held water is released from rice fields post application (Table 32 and Table 31). Some co-occurrence is likely, but the number of individuals exposed during adult migrations is expected to be relatively low as most of the pre-spawning adults would be higher in the watershed (above RBDD) during the application and discharge period (April-July; Table 32). Unlike salmon, steelhead can spawn several times before they die and therefore spawned adults moving down river as kelts could be exposed to thiobencarb via mechanisms described above. We do not have data indicating relative abundance of down migrant kelts.

CV Steelhead have the longest freshwater migration of any population of winter steelhead. Available information for natural populations of steelhead reveals considerable overlap in migration and spawn timing between populations of the same run type. Peak spawning is January through March, and takes place mostly high in the watersheds of the Sacramento, Mokelumne, and possibly the San Joaquin rivers (Figure 28). Most spawning habitat for CV steelhead is located downstream of dams in areas containing suitable environmental conditions (*e.g.*, cold water, suitable gravels and stream flow) for spawning and incubation. We expect little, if any, spawning, egg incubation, and fry emergence to co-occur with thiobencarb use in rice production based on the current locations of known spawning areas (Figure 28). Furthermore, the primary thiobencarb application window (May and June) does not overlap with peak spawning.

As with adults, juveniles are present in the river year-round. After fry emergence, most juveniles rear in their natal streams at elevations above rice growing areas. Steelhead rearing takes place primarily in higher velocity areas of pools, although they are also abundant in glides and riffles. Juvenile steelhead emigrate episodically from natal streams during fall, winter, and spring high flows. Their relative abundance is low to moderate in rice growing areas during the thiobencarb application and discharge period (Table 32 and Table 31). Relative abundance peaks at the RBDD from August into November and at the Knights Landing screw traps in March. By March/April the bulk of the steelhead run has entered the estuary and individuals are present in monitoring data at Chipps Island (lower North Delta). There are many late arrivals from the natal spawning grounds that would migrate through rice growing areas (e.g. Knights Landing) well into May and perhaps early June (*e.g.*, lower Sacramento River near Hood). However, fish monitoring data suggest that the majority of fish would have migrated through the rice growing areas prior to the thiobencarb application period.

While most of the rice grown in the Central Valley is in the lower Sacramento River Basin, some rice is grown in the lower San Joaquin River and Mokelumne River watersheds. Fish monitoring on the San Joaquin (Mossdale) and Mokelumne (Woodbridge dam) indicate steelhead, in general, are present in these waters at very low abundance. Juveniles are present in moderate relative abundance in rice growing areas of both systems during times when risk of exposure to thiobencarb is highest. Therefore, while we expect thiobencarb exposure to juveniles rearing in the San Joaquin and Mokelumne Rivers, we expect the exposure to be limited given the relatively low abundance of both fish and rice in these areas (Table 31). Overall, drift and runoff into occupied steelhead rearing and migration habitats are expected infrequently. We expect low juvenile steelhead abundance in the Sacramento River and medium abundance in the San Joaquin River when thiobencarb is applied and rice holding waters are released (Table 32).

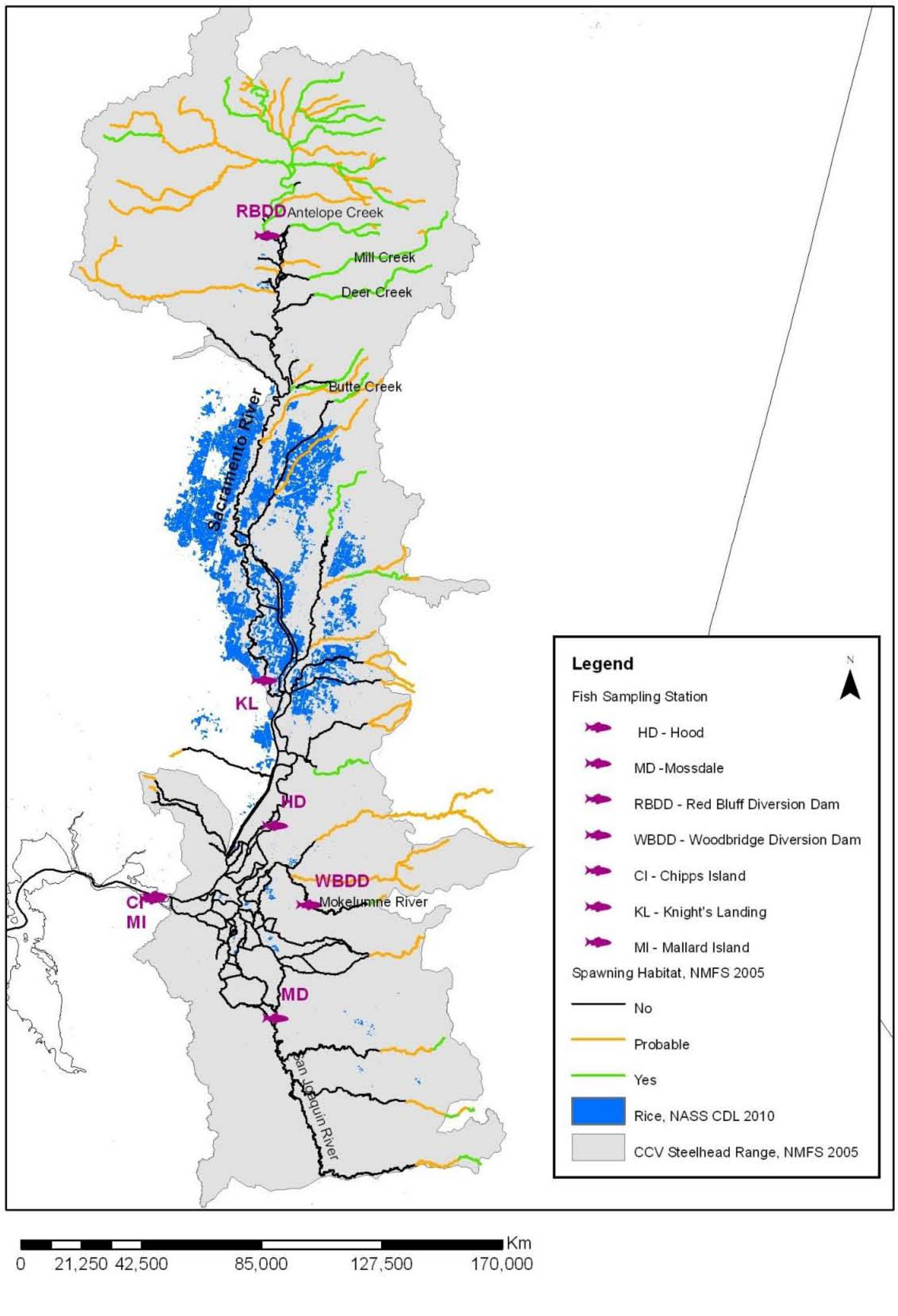


Figure 28. CV steelhead probable and known spawning areas in relation to 2010 rice growing areas.

Table 31. Temporal occurrence of adult and juvenile CV steelhead in the Central Valley. Darker shades indicate months of greatest relative abundance in each location.

Adult migration												
Location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Sac. River basin												
San Joaquin River												
Juvenile migration												
Location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Sac. River at RBDD[1]												
Sac. River at KL[2]												
Sac. R. at Hood[3]												
Chipps Island[3]												
Mossdale[2]												
Woodbridge Dam[2]												
Thiobencarb use/release												
RBDD=Red Bluff Diversion Dam KL = Knights Landing												
Relative Abundance:		= High			= Medium				= Low			

[1] upstream of current rice growing areas; [2] within current rice growing areas; [3] downstream of current rice growing areas.

Table 32. CV steelhead relative abundance in rice areas (Figure 25) during thiobencarb use periods April - July.

Life Stage	None expected	Low	Medium	High
Adult migration		X		
Spawning		X		
Early rearing		X		
Juvenile migration – San Joaquin & Mokelumne rivers			X	
Juvenile migration – Sacramento R.		X		

8.2. Modeling: Estimates of exposure to thiobencarb

8.2.1. EPA exposure estimates

EPA did not utilize modeling to estimate potential exposure of listed Pacific salmonids to thiobencarb or other stressors of the action (EPA 2002a). However, EPA used the Rice Model and the AgDrift model to estimate thiobencarb exposure to California red-legged frog and Delta smelt (EPA 2009b). Below, we discuss the utility of these estimates for evaluating exposure to listed Pacific salmonids.

8.2.2. Utility of EPA-derived EECs for defining exposure to Pacific salmonid habitats

As described in the *Approach to the Assessment* section, our exposure analysis begins at the organism (individual) level of biological organization. We consider the life stage and life histories of the individuals likely to be exposed. This scale of assessment is essential as adverse effects to individuals may result in population-level consequences, particularly for populations of extremely low abundance (*i.e.*, threatened and endangered species). Characterization of impacts to an individual's fitness is necessary to assess potential impacts to populations, and ultimately to the species. To assess risk to individuals, we must consider the range in concentrations to which individuals of the population may be exposed. The highest concentrations in aquatic habitats are typically associated with direct application to water, or off-target deposition of pesticides into shallow habitats in close proximity to the target site. Pacific salmonids utilize a variety of aquatic habitats (Table 21). All three listed salmonid species within CA's Central Valley use shallow, low flow habitats at some point in their life cycle.

EPA Rice Model. EPA estimated the peak concentrations of thiobencarb within a rice paddy to be 2018 µg/L using the Rice Model. This model calculates peak concentration based solely on the dilution volume of the paddy and an assumption of instantaneous partitioning of the pesticide between the rice field water and sediment. The simulation included direct application of thiobencarb at the labeled application rate (4 lbs a.i./A) to a 0.1 meter-deep rice field. This depth is consistent with the cultivation of rice. A first-order decay rate of 0.1252/day was then applied to the peak concentration to estimate concentrations within the rice field at various times post application, as well as time-weighted-average concentrations. This decay rate is a field dissipation rate and its use resulted in "double counting" of partitioning to sediment and potentially an underestimation of aquatic concentrations.

Although we do not expect listed Pacific salmonids to use flooded rice fields, salmonids may be exposed to thiobencarb that is discharged to their habitats or manages to seep from rice fields into drainage canals and ultimately into salmonid habitats. Therefore, water concentration estimates in flooded rice fields, particularly at discharge, can be used as an upper bound for runoff concentrations for salmon and their designated critical habitat. FIFRA labels currently require treated flood waters to be held a minimum of 14 d before discharge (EPA Reg. No. 59639-79). Some labels specify not to drain herbicide-treated fields for a minimum of 30 days after application (EPA Reg. No. 59639-112). Using the rice model and assuming a 5.5 day half-life for thiobencarb in water, EPA estimated discharge concentrations of 350 µg/L and 47 µg/L after 14 d and 30 d holding periods, respectively. The actual discharge concentrations are expected to vary somewhat depending on site-specific dissipation and may be either greater or less than predicted. Three aquatic dissipation studies were available for thiobencarb. The authors of these studies report half-lives of 5.8 d, > 6 d, and 8.7 d, slightly longer than the dissipation rate assumed by EPA modeling (Table 24). Concentrations observed in waters receiving discharge will be reduced to different degrees depending on flow rate and dilution capacity of the receiving water.

EPA AgDrift model estimates. EPA used the AgDrift model to determine distance to listed species habitats required to prevent adverse impacts to listed species through exposure caused by primary drift of the liquid formulation of thiobencarb. Model simulations assumed EPA default inputs and modeled drift to a 2-meter deep farm pond (EPA 2009b). The physical characteristics of EPA's farm pond provide a reasonable representation of some of the habitats used by salmonids. NMFS used AgDrift to estimate thiobencarb concentrations in other important habitats, such as floodplain habitats with lower dilution capacity (*NMFS exposure estimates for flood habitats*).

EPA estimates of exposure to other action stressors. EPA did not provide model estimates for other ingredients, tank mixtures components, or degradates of thiobencarb. Only one thiobencarb product contains multiple a.i.s, and is formulated with propanil (EPA registration #71085-30). This end-use product is currently not registered for use in California. However, several thiobencarb labels recommend tank mixture applications with propanil or other herbicides to improve product efficacy. Estimates of exposure to identified metabolites and degradates were also absent (Table 23). The missing estimates of exposure to these stressors introduce substantial uncertainty to the exposure analysis.

NMFS conclusions on utility of available EPA model estimates The Rice Model simulation paired with field dissipation calculations provide a reasonable estimate of parent thiobencarb concentrations that may be discharged into habitats where listed salmonids reside. AgDrift modeling was useful in identifying that primary drift of spray applied thiobencarb represents a transportation pathway predicted to exceed EPA's Endangered Species Levels of Concern for fish and their habitat (Page 111 *in* (EPA 2009b). NMFS provides additional modeling estimates below to characterize the potential range of exposure in salmonid habitats and to address exposure to other stressors of the action.

8.2.3. NMFS exposure estimates for floodplain habitats

The "farm pond" scenario utilized by EPA is representative of some habitats used by listed salmonids. However, other habitats may be more or less susceptible to higher pesticide concentrations given their physical characteristics. For example, small streams and some floodplain habitats represent examples of habitats used by salmonids that can have a lower capacity to dilute pesticide inputs than the farm pond. NMFS derived estimates for a vulnerable floodplain habitat to capture the potential range of thiobencarb exposure in listed salmonids.

8.2.3.1. *Application of thiobencarb to fields adjacent to salmonid habitat*

Drift during application is a transport mechanism that can result in significant deposition of pesticides in aquatic habitats immediately adjacent to treated fields, including shallow floodplain habitats where juvenile salmonids rear and shelter. We derived exposure estimates for floodplain habitats using the AgDrift model to estimate downwind deposition from pesticide drift (Teske 2001). AgDrift is a field-scale model that evaluates pesticide drift. The drift estimates derived represent average projected drift. Although AgDrift reasonably predicts drift, drift is highly variable and is influenced by site-specific conditions and application equipment (Bird et al. 2002).

Our model inputs incorporated application requirements from thiobencarb labels (Table 33). Some thiobencarb product labels include a set-back to aquatic habitats. For example, Bolero 8 EC prohibits product application within 1000 feet of the endangered fat pocketbook pearly mussel (EPA Registration No. 59639-79). Additionally, under the Environmental Hazards section of the label, application of the product is not permitted south of the Intercoastal Waterway in Louisiana or within two miles from the shorelines of Matagorda Bay or Galveston Bay in Texas. Existing labels contain no setbacks to salmonid habitats. The label instructs applicators to apply liquid formulations of thiobencarb at up to 4 lbs of thiobencarb per acre with aerial or ground application equipment. Historic use data for California suggest the maximum labeled rate is the predominant rate used (Appendix 3). Our simulations assumed an aquatic habitat that was 2 m wide and of variable depths (0.1 – 2 m). These dimensions are consistent with some of the smaller, and potentially more vulnerable, floodplain habitats and small streams used by salmonids.

Table 33. Estimated average initial thiobencarb concentrations in a floodplain habitat that is 2m wide and of variable depths using AgDrift 2.0.05.

Application method	Model inputs	Simulation: Rate in lbs a.i./A	Habitat Depth (m)	Average Initial Concentration in Surface Water (µg/L)
Ground	Tier 1 ground, Low ground boom spray, ASAE fine to medium/coarse distribution, 50th percentile estimate	4	0.1 0.5 1 2	736 147 74 37
Aerial	Tier 1 aerial spray, ASAE medium to coarse droplet distribution	4	0.1 0.5 1 2	1,910 382 191 96

We also used AgDrift to estimate potential exposure of riparian vegetation to drift of spray-applied thiobencarb. The simulations evaluated application of thiobencarb at the 4 lbs a.i./A labeled rate. The terrestrial assessment tool in AgDrift was used to identify point-deposition at various distances downwind from the target treatment site (Table 34). Predicted deposition of thiobencarb decreased as distance from the target site increased.

Table 34. AgDrift Estimates for point deposition of thiobencarb applied at the 4 lbs a.i./A rate.

Application method	Distance (ft) from target spray area	Estimated drift of spray-applied thiobencarb	
		lbs a.i./A	Percent of applied
Ground Tier 1 ground, Low ground boom spray, ASAE fine to medium/coarse distribution, 50th percentile estimate	1 10 100 1000*	1.19 0.09 0.010 0.001	29.68 2.24 0.24 0.03
Aerial Tier 1 aerial spray, ASAE medium to coarse droplet distribution	1 10 100 1000*	1.91 1.21 0.22 0.02	47.76 30.34 5.61 0.55

*Simulation runs at buffer of 997 feet due to model limitation.

8.2.4. NMFS exposure estimates for pesticide mixtures

Co-application of pesticides increases the likelihood of exposure to pesticide mixtures. Several thiobencarb labels recommend application with other pesticides to improve weed management (*e.g.*, propanil, glyphosate, and paraquat; EPA Reg. No. 59639-79, 63588-6, 63588-14). Thiobencarb labels also make general recommendations for applications of other pesticides, such as avoiding applications of other ingredients that share a common mode of action to manage weed resistance (e.g., EPA Reg. No. 59639-112, 63588-14). We reviewed the available pesticide use data for California to determine the pesticides that are most commonly co-applied with thiobencarb. Co-application refers to situations where more than one active ingredient is applied to an agricultural field on the same day. This might be accomplished through tank mixture applications or separate applications to the same field (e.g. treatment of weeds on levees surrounding the rice). Over the last decade, thiobencarb has been co-applied with other pesticides 23% of the time (Appendix 4). During that timeframe there have been more than 150 unique pesticide combinations involving the application of thiobencarb and at least one other pesticide applied to the same field on the same day (Appendix 4).

It is not feasible to estimate exposure to all of the possible pesticide mixtures that are allowed to be used. Below we present exposure estimates associated with applications of thiobencarb, propanil, and copper sulfate (Table 35). According to labels, both propanil and copper sulfate products can legally be applied with thiobencarb. Propanil is the active ingredient most commonly recommended for co-application on thiobencarb labels. The Bolero® 8 EC label (EPA Reg. No. 59639-79), recommends applying thiobencarb at a rate of 2 – 2.5 lbs a.i./A in combination with propanil. Propanil may be applied to rice at rates of up to 6 lbs a.i./A (EPA Reg. No. 71085-2). Copper sulfate has been the most frequent pesticide co-applied with thiobencarb since 1999. Although copper sulfate products are not specifically recommended for tank mixture on thiobencarb product labels, these mixtures are authorized under FIFRA unless specifically prohibited on the product label. Copper can be applied to rice at rates up to 15 lbs a.i./A (EPA Reg. No. 56576-1).

Table 35. Estimated average initial pesticide concentration in a floodplain habitat that is 2m wide and 0.1m deep using AgDrift 2.0.05.

Application method	Model inputs[1]	Simulation: Rate in lbs a.i./A	Buffer (feet)	Average Initial Concentration in Surface Water (µg/L)
Ground	Tier 1 ground, Low ground boom spray, ASAE fine to medium/coarse distribution, 50th percentile estimate	2 (Thiobencarb) 6 (Propanil) 15 (Copper sulfate)	0 0 0	368 1,103 2,759
Aerial	Tier 1 aerial spray, ASAE medium to coarse droplet distribution	2 (Thiobencarb) 6 (Propanil) 15 (Copper sulfate)	0 0 0	956 2,940 7,170

8.3. Monitoring Data: Measured Concentrations of Parent Compounds in Surface Waters

We evaluated data from three sources: USGS' NAWQA database, a state database maintained by California Department of Pesticide Regulation, and monitoring studies that target specific applications of thiobencarb and monitor thiobencarb concentrations on or adjacent to the treated rice fields. Information provided by the two databases includes ambient monitoring data with sampling stations distributed across a range of land uses. They also include studies that investigate water quality impacts associated with specific pesticide uses, such as the use of pesticides for rice production. We also reviewed monitoring studies that target field-scale operations investigating thiobencarb concentrations in discharge from treated fields and in aquatic habitats immediately adjacent to treated fields. These studies were obtained from a variety of sources and include both published scientific literature and gray literature. In the following section we describe study design considerations for assessing the utility of monitoring data for evaluating exposure of pesticides to salmon.

8.3.1. Monitoring data considerations

Surface water monitoring can provide useful information regarding real-time exposure and the occurrence of environmental mixtures. A primary consideration in evaluating monitoring data is whether the study design is sufficient to address exposure in a qualitative, quantitative, or probabilistic manner. The available monitoring studies were conducted under a variety of protocols and for varying purposes. General water quality monitoring conducted in larger streams and rivers frequently does not capture "peak" concentrations because it is not correlated with applications and/or storm events following those applications and not all habitat types are sampled. This is one of the reasons NMFS did not use available monitoring data for probabilistic modeling (*i.e.*, it likely does not contain the complete range of possible concentrations).

Of the monitoring programs discussed, none were specifically designed to evaluate potential exposure of listed salmonids to pesticides in surface waters. Common aspects that limit the utility of the available monitoring data as accurate depictions of exposure within listed salmonid habitats include: (1) protocols were not designed to capture peak concentrations or durations of exposure in habitats occupied by listed species; (2) limited utility as a surrogate for other non-sampled surface waters; and (3) lack of representativeness of current and future pesticide uses and conditions.

Protocols not designed to capture peak exposure. None of the available monitoring studies were designed to evaluate peak exposure of listed salmonids in the Central Valley (or anywhere else salmonids reside) to thiobencarb in surface waters. The NAWQA program provides a considerable dataset that is useful for evaluating trends in water quality (Hirsch et al. 1988). The NAWQA design does not result in an unbiased representation of surface waters, which limits the ability to make statistical extrapolations to waters not sampled. Sampling by NAWQA and other studies was frequently not conducted in coordination with specific applications of thiobencarb at the field scale. Similarly, sampling was not designed with consideration of salmon distributions or to target salmonid habitats most likely to contain the greatest concentrations of pesticides. Given the relatively rapid dissipation of pesticides in flowing water habitats, it is not surprising that pesticide concentrations from these datasets were generally much lower than those predicted by modeling efforts and those that monitored targeted pesticide applications at the field scale.

Limited applicability to other locations. Pesticide runoff and drift are influenced by a variety of site-specific variables such as meteorological conditions, soil type, slope, and physical barriers to runoff and drift. Additionally, surface water variables such as volume, flow, and pH influence both initial concentrations and persistence of pesticides in aquatic habitats. Finally, cropping patterns and pesticide use have high spatial variability. Given these and other site-specific factors, caution should be used when extrapolating monitoring data to other sites.

Representativeness of current and future uses. Pesticide use varies annually depending on regulatory changes, market forces, cropping patterns, and pest pressure. Pesticide use patterns change annually and may result in either increases or decreases in use of pesticide products for specific uses. Since 1980, the acreage treated with thiobencarb has ranged between a low of 1,973 acres in 1980, and a high of 252,506 acres in 2000 (Figure 29). Prediction of future use of pesticides is complicated by a number of factors including climate change that may affect agriculture uses and pest pressures. Additionally, regulatory changes specifying how thiobencarb can be used have changed over the years (*e.g.*, changes in holding time requirements for thiobencarb-treated rice fields likely reduced thiobencarb concentrations in receiving waters). Such changes add to the difficulty of predicting future water quality conditions from historic monitoring data.

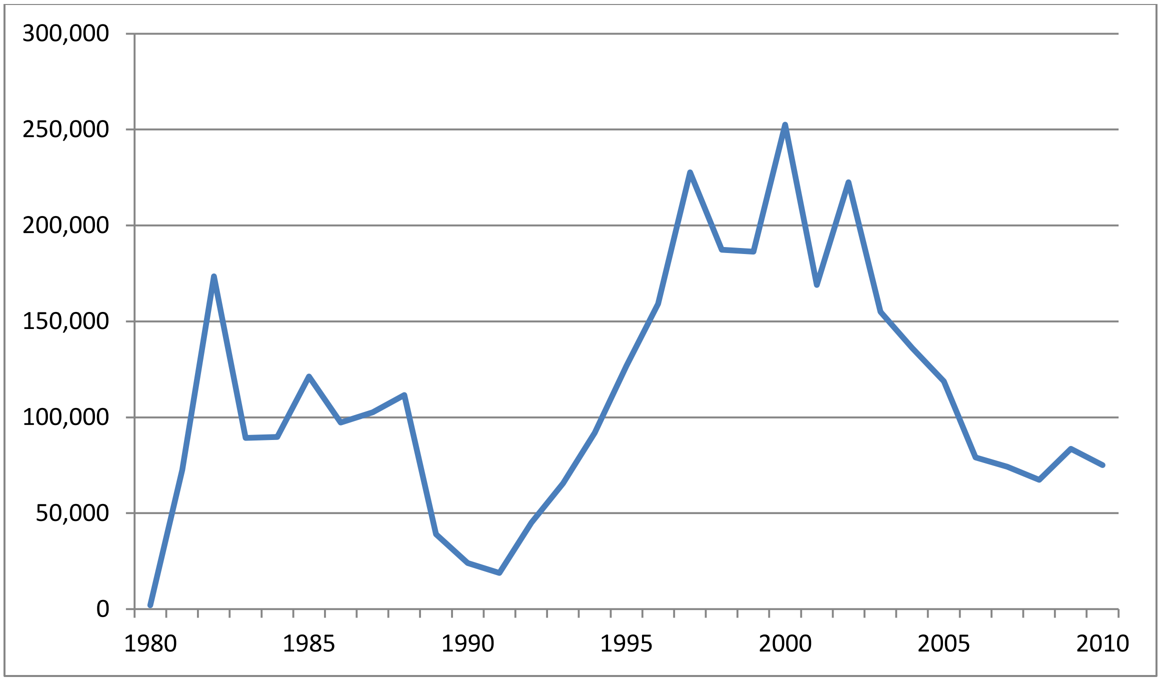


Figure 29. Acres treated with thiobencarb in California. Data from the California Pesticide Use Reporting Database 1980 – 2010.

8.3.2. USGS NAWQA Data

We obtained updated data from the USGS NAWQA database to evaluate the occurrence of thiobencarb in surface waters monitored in California from 1992-2011. The database query resulted in approximately two thousand surface water samples obtained from 74 unique locations in which thiobencarb was an analyte (Figure 30).

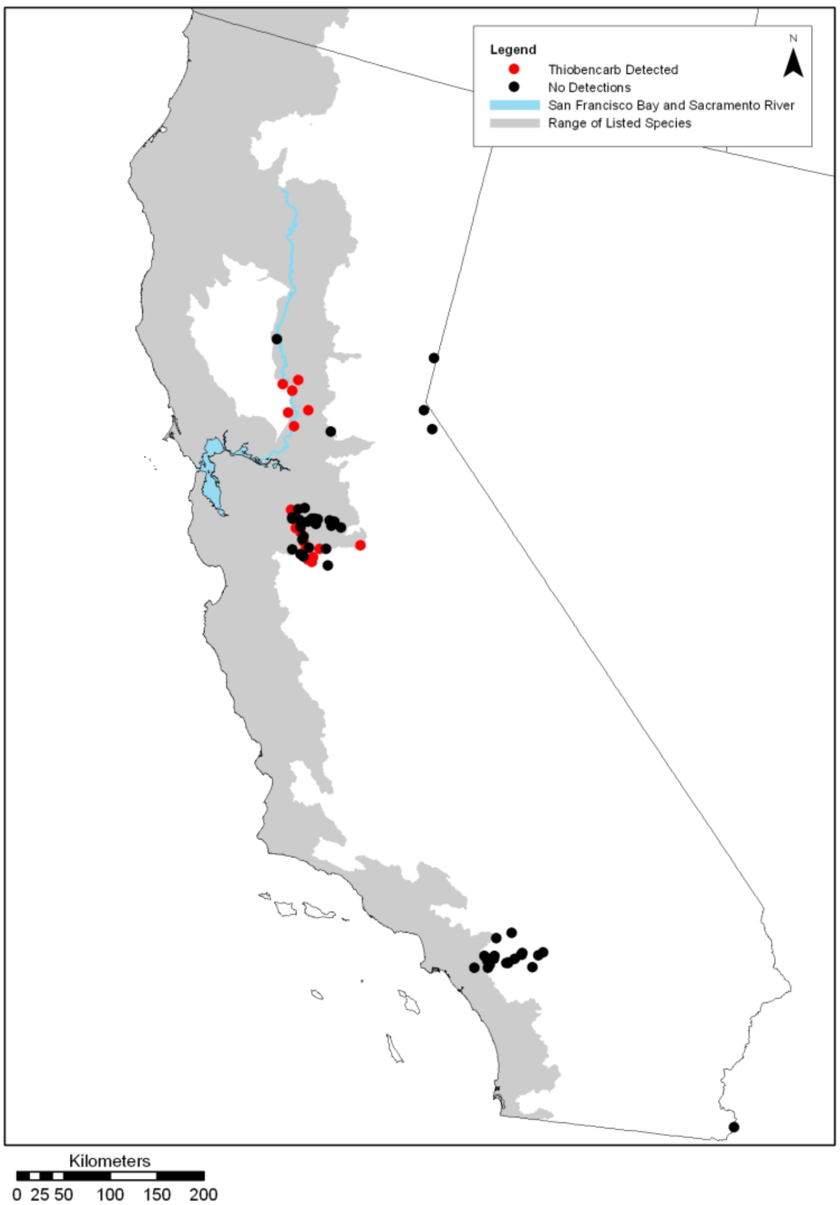


Figure 30. Distribution of NAWQA monitoring sites that have sampled for the presence of thiobencarb relative to the range of threatened and endangered Pacific salmonids in California.

Many of the locations sampled for thiobencarb do not contain listed salmonids. Of the 74 locations sampled, five were within the distribution of Central Valley spring-run Chinook and Sacramento River winter-run Chinook. Forty sites located within the distribution of the California Central Valley steelhead have been sampled since 1992 (Table 36). Many of the sample locations were outside of the rice growing region of California including the Sacramento Valley and the north to central San Joaquin Valley (Hill et al. 1992).

Table 36. Number of NAWQA sample sites within the freshwater distribution of listed Pacific salmonids in the Central Valley of California, as determined through GIS analysis

ESU / DPS	Kilometers of Stream Inhabited	Sample Sites in Freshwater Habitat (Sites with Detections)
Central Valley Spring – Run Chinook	2,212.94	5 (4)
Sacramento River Winter – Run Chinook	546.84	5 (4)
California Central Valley Steelhead	4,273.66	40 (13)

Sampling effort varied considerably among the sample locations. Approximately 54% of the sites screened for thiobencarb were sampled five or fewer times during the span of 19 years. A small number of sites accounted for the majority of the data; four of the 74 sites accounted for more than 50% of the sampling data for thiobencarb. Eighty-six percent of the data was collected from 18 sites, including 4 sites that fell outside the distribution of listed salmonids. Overall, the sampling effort does not correspond well with the distribution of listed salmon or the co-occurrence of listed salmon and rice growing areas. Consequently, we do not expect the data set to be representative of exposure distributions for listed salmonids.

The USGS monitoring program does not generally coordinate sampling efforts with specific pesticide applications or runoff events, detected concentrations are likely to be lower than actual peak concentrations that occur in surface waters proximate to application sites immediately following drift or runoff events. Summary information for quantifiable concentrations of thiobencarb addressed in this Opinion is presented in Table 37. Non-detects are reported as less than (“<”) the laboratory reporting level (LRL) for that sample. Other than total number of samples (n), summary statistics were calculated on samples not designated as (“<”). The LRL ranges reported were estimated based on “<”-qualified data. Nearly all of the concentrations that could be quantified were designated as “E,” meaning the concentrations were estimated.

Thiobencarb was detected in 10.6% of samples screened, broadly ranging from 0.001 to 4.38 µg/L (median, 0.01 ug/L).

Table 37. Detections and concentrations of thiobencarb reported in the NAWQA database.

Statistic	Thiobencarb
Number of Stations	74
Number of Observations	2256
Detects	241
Percent Detections	10.6%
Median (ug/L)	0.0102
Range (ug/L)	0.001-4.38
LRL (ug/L)	0.002-0.019
Year range	1992-2011

U.S. Geological Survey conducted a study during the 2002 and 2003 growing season to determine if changes in pesticide use on rice resulted in corresponding changes in pesticide concentrations in surface waters (Orlando and Kuivila 2004). This study is not included in the NAWQA database. Five surface water sites within the Sacramento Valley were analyzed for five rice pesticides, including thiobencarb. Samples were collected weekly from May through July. Maximum concentrations of all pesticides in 2003 were less than half of the 2002 levels. Higher peak pesticide concentrations and greater duration of exceedance of water quality protection goals observed in 2002 were attributed to differences in weather and pesticide use. Typically, little precipitation falls during the rice planting and growing season in the Sacramento Valley. However, in 2002 a significant storm in mid-May forced some early releases of field water including documented releases of thiobencarb that likely contributed to elevated concentrations in the Colusa Basin Drain and the Sacramento Slough. Additionally, spring rains in 2003 delayed planting and may have resulted in the observed decrease in thiobencarb use. The peak thiobencarb concentration observed during the study was 7.16 µg/L. Samples collected from the same locations in earlier years have revealed concentrations up to 170 µg/L in 1982. Regulations have since been implemented to reduce concentrations of rice pesticides in irrigation returns. Implementing holding times for pesticide-treated rice field water in the 1980s was credited in decreasing concentrations of thiobencarb and other pesticides in surface water in the Sacramento Valley (Orlando and Kuivila 2004).

We evaluated monitoring data from the California Department of Pesticide Regulation (CDPR), public database of pesticide monitoring data for surface waters in California[13]. Values in this database originate from monitoring studies conducted by CDPR, USGS, state, city and county water resource agencies along with some studies conducted by non-governmental or inter-governmental groups such as Deltakeeper. To avoid redundant use of these data, USGS data found in the CDPR database are excluded from the following data summary.

[13] (www.cdpr.ca.gov/docs/emon/surfwtr/surfdata.htm)

The database provided information on more than 2,800 sampling events for thiobencarb in California since 1994. Sampling has been conducted at 393 unique locations (Figure 31).

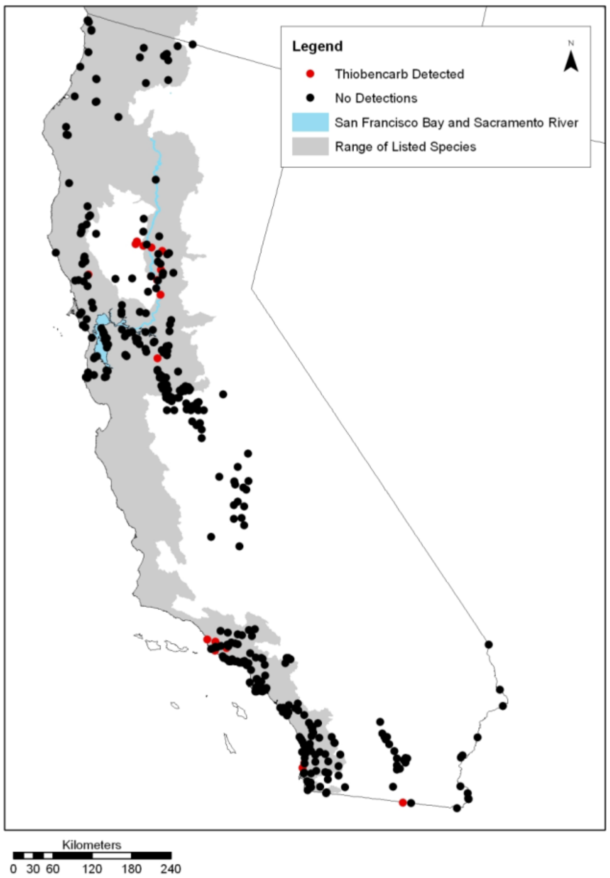


Figure 31. Distribution of CDPR monitoring sites that have sampled for the presence of thiobencarb relative to the range of threatened and endangered Pacific salmonids in California.

Since 1994, 28 unique locations where sampled within the range of the Central Valley spring-run Chinook and Sacramento River winter-run Chinook, and 66 locations were sampled within the range of the California Central Valley Steelhead (Table 38). Sample locations do not correspond well with the distribution of potential thiobencarb use sites. Rice is the only authorized use of thiobencarb in California and is grown almost exclusively in the Sacramento Valley and the central San Joaquin Valley. Detections of thiobencarb in southern California and in coastal areas of California are unexpected (Figure 31). These detections are presumed to represent misapplications or database entry errors since rice is not grown in those areas.

Table 38. Number of CDPR database sample sites within the distribution of listed Pacific salmonids in California, as determined through GIS analysis.

ESU / DPS	Kilometers of Stream Inhabited	Sample Sites in Freshwater Habitat (Sites with Detections)
Central Valley Spring – Run Chinook	2,212.94	28 (7)
Sacramento River Winter – Run Chinook	546.84	28 (7)
California Central Valley Steelhead	4,273.66	66 (10)

We conducted a simple GIS analysis to evaluate the spatial relationship between monitoring stations and rice fields. The thiobencarb monitoring sites were frequently located long distances from rice fields. A 1-km buffer was placed around each monitoring station and compared to the most recent crop data layer for rice (USDA-NASS Cropland Data Layer 2009). We found that 89% of the thiobencarb monitoring sites were more than 1 km (3281 ft) from the nearest rice field. This suggests that in general, the dataset would not be useful for evaluating peak concentrations that could occur in surface waters from near-field (<1000 ft) applications of thiobencarb.

Sampling effort was not equivalent among monitoring sites. More than half of the monitoring stations were sampled three times or less during the 17-year period. Five sample locations, roughly 1% of the monitoring stations, were sampled 51 to 203 times representing 22% of all the thiobencarb data collected. These stations and two others located within the Sacramento Valley rice-growing Region accounted for 95% of the thiobencarb detections (Figure 32). This region is within the range of the Central Valley Spring-Run Chinook, the Sacramento River Winter-run Chinook, and the California Central Valley Steelhead.

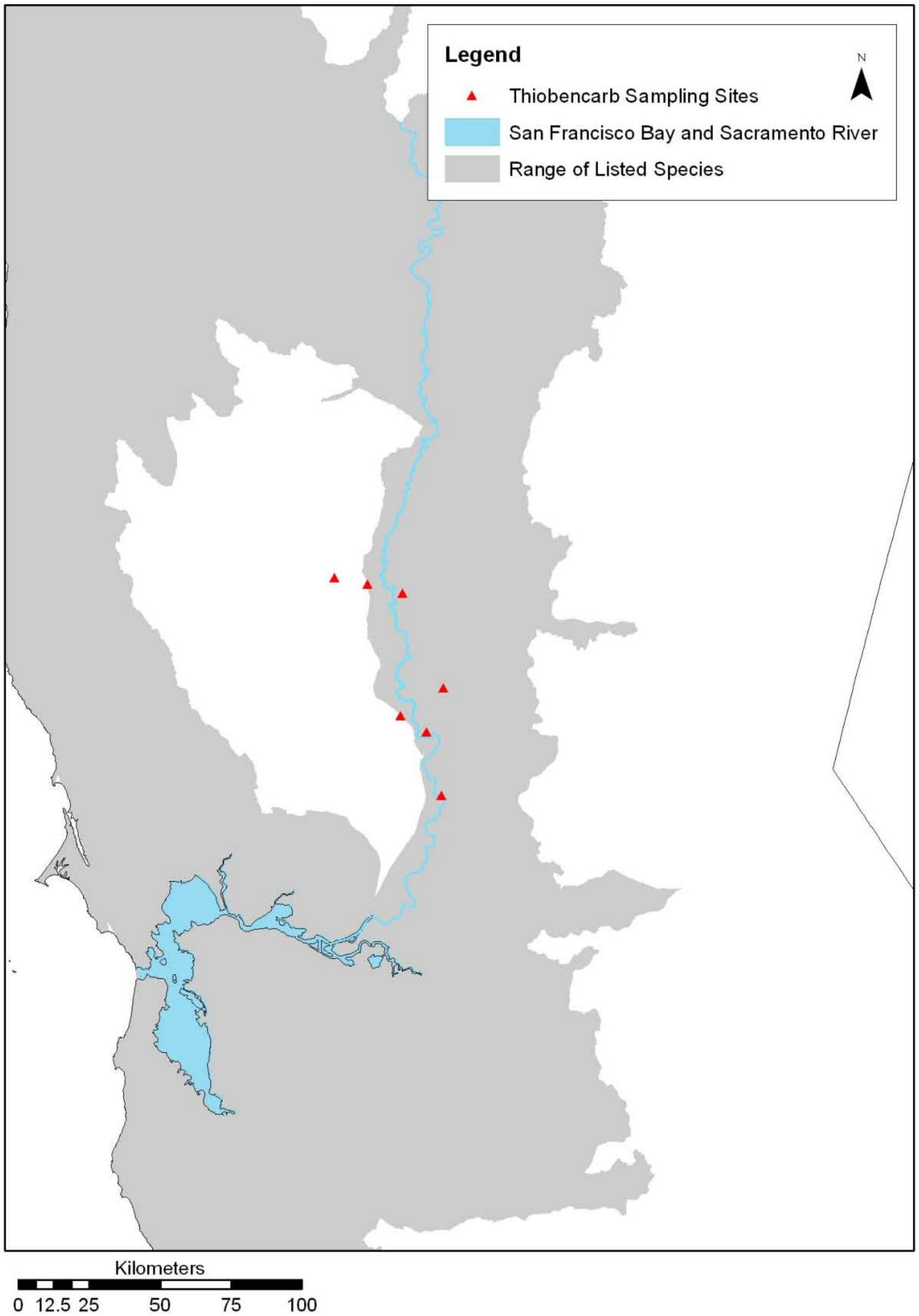


Figure 32. The location of seven sample sites accounting for 95% of the detections in thiobencarb in the CDPR monitoring database.

The contents of the database for thiobencarb are summarized in Table 39 while individual programs and studies are described below in the text. The CDPR requires a formal QA/QC protocol for data submitted or does a separate QA/QC review, thus only data subject to appropriate QA/QC procedures are included in the surface water database. Unlike the USGS NAWQA data set, the CDPR database may contain whole water samples as well as filtered samples. If whole water concentrations are reported for compounds that sorb significantly to the particulate phase, concentrations would appear higher than in a filtered sample, which represents only the dissolved phase. The database, last updated in June 2011, consists of approximately 378,000 data records. Each record reports a specific sampling site, date, and analyte. In this database, detections below the LRL are reported as 0 μg/L. Our summary statistics for the datasets were calculated on samples with values above the LRL. The number of observations is indicative of monitoring intensity rather than actual occurrence in surface waters.

Table 39. Detections and concentrations of thiobencarb from the CDPR database

Statistic	Thiobencarb
Number of Studies	25
Number of Stations	393
Number of Observations	2836
Detects	180
Percent Detections	6%
Median (ug/L)	1.275
Range (ug/L)	0.008-16.55
LRL (ug/L)	0.005-2
Year range	1994-2011

The CDPR database includes 2836 thiobencarb observations from 25 studies monitoring a total of 393 stations. Thiobencarb was detected in approximately six percent of samples screened for the pesticide (median 1.28 μg/L, range 0.008-16.55 μg/L, LRL=0.05-2 μg/L). Some studies reported in the database are annual monitoring results for multiyear monitoring programs, while other studies encompass several years. The Sacramento River Watershed Program was a single year monitoring effort that monitored for thiobencarb in 2002. Monitoring in this program screened for thiobencarb at five stations and detected thiobencarb in one of the five samples collected (2.1 ug/L, LRL=0.5 ug/L). The Central Valley Regional Water Quality Control Board included thiobencarb among the analytes screened for during its investigation into the sources and concentrations of diazinon in the Sacramento watershed during the 1994 spray season. Thiobencarb was detected in 35 of 64 samples taken from eight stations during this study (median 0.012 µg/L, range 0.008-0.042 µg/L, LRL=0.008 µg/L). The California State Water Resources Control Board Surface Water Ambient Monitoring Program (SWAMP) studies cover the years 2001 to 2008 and include 943 observations from 271 stations monitored for thiobencarb. Thiobencarb was detected nine times with a median concentration of 0.93 ug/L (range=0.109-11.8 µg/L, LRL=0.005-0.2 µg/L). The highest value at 11.8 µg/L was detected in a sample collected in May of 2006 from the New River at Boundary. The highest thiobencarb concentrations are reported in two monitoring studies targeting areas of high pesticide use which were sampled multiple times over the growing season or during peak irrigation. Such studies are more likely to capture peak or near peak pesticide concentrations. These studies are discussed in more detail in the following section on targeted monitoring.

The CDPR database includes data from two notable studies. The Central Valley Regional Water Quality Control Board monitoring project investigated water quality between 2004 and 2009 in agricultural drains of the Central Valley (hereafter "Irrigated Lands Study"), and CDPR monitored for pesticides used in rice cultivation between 1995 and 2002 (hereafter "Rice Pesticide Monitoring Program"). These two studies account for a majority of the thiobencarb data in the CDPR database; cumulatively they represent 64% of the observations that have screened for the presence of thiobencarb and 84% of thiobencarb detections. Although these studies do not target thiobencarb applications at the "field scale" or monitor exposure in salmonids, they are designed to evaluate surface water concentrations corresponding to regional agricultural practices.

Irrigated Lands Study. The irrigated lands study monitoring stations are located near agricultural drainages flowing into creeks or rivers in catchments dominated by return flow from mixed row crops and/or alfalfa, or areas where the primary land use was rice culture. The study design focused sampling efforts on periods of peak irrigation, especially the first major irrigation of the season. The purpose of the study was to evaluate the magnitude and extent of water quality problems in waters that receive agricultural drainage. The majority of detects of thiobencarb occurred in May and June which corresponds to thiobencarb's peak use (Table 40). Among 1419 samples collected from 114 stations, thiobencarb occurred in 72 samples from 28 stations at concentrations as high as 7.58 ug/L (LRL = 0.05-2 ug/L). The high concentration was detected within the Natomas Central Mutual Water District in the Sacramento Valley. At the time of sample collection, thiobencarb could legally be discharged into irrigation drains after a minimum six-day hold period because the district maintained a "closed system." The district was required to hold the water for the remainder of the 30-day hold period prior to discharge to creeks and rivers.

Table 40. Summary of Irrigated Lands Study data from the Central Valley Regional Water Quality Control Board (2004-2006).

Statistic	Thiobencarb
Number of Stations	114
Number of Observations	1419
Detects	72
Percent Detections	5%
Median (ug/L)	0.325
Range (ug/L)	0.055-7.58
LRL (ug/L)	0.05-2

Rice Pesticide Monitoring Program. CDPR implemented the Rice Pesticides Program (RPP) in 1983 to reduce thiobencarb and molinate discharges into surface waters. In 1990, the Central Valley Regional Water Quality Control Board and CDPR established performance goals for thiobencarb and several other pesticides used in rice production. The 1.5 µg/L performance goal for thiobencarb was established to help meet California's Maximum Concentration Limit of 1 µg/L in drinking water. This secondary MCL was established based on a threshold for impaired taste to drinking water. Data from the RPP are used to verify compliance with this and other performance goals and implement adaptive management (Bennett et al. 1998, Moran 2012).

Some of the sample locations have varied over the years. Recent monitoring evaluated by the RPP includes samples collected from five primary monitoring stations in the Sacramento Valley

(Figure 33). Additionally, samples collected at two water intake locations on the Sacramento River are provided by water treatment facilities.

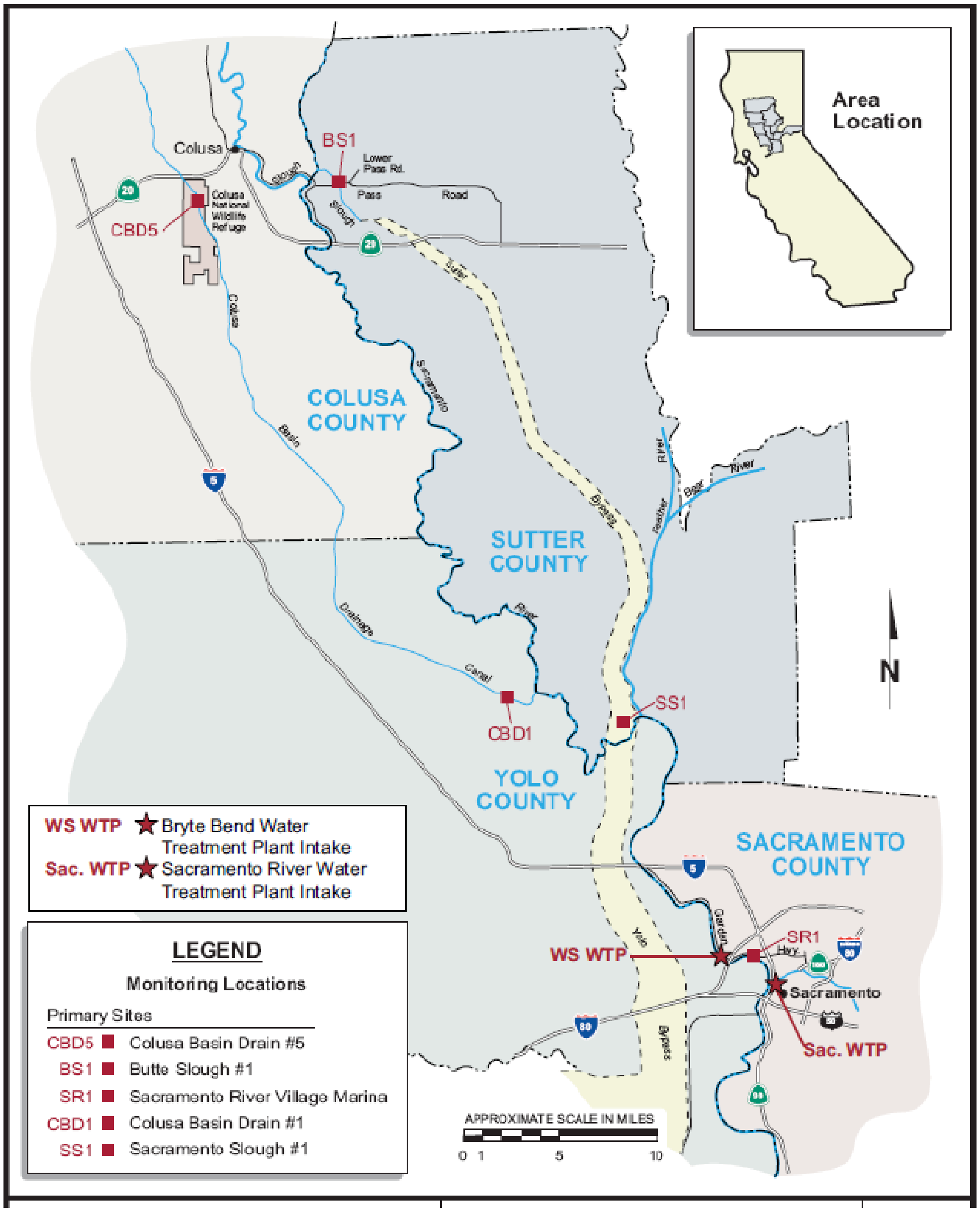


Figure 33. Sacramento Valley RPP monitoring locations (source: Moran 2012 modification of figure from CRC 2004 RPP Report)

CDPR prepared reports for the RPP monitoring program between 1995-2002. We searched the CDPR database for three stations that have been systematically monitored for thiobencarb since 1995. Water samples were collected yearly during the rice pesticide application period from three sites between 1995 and 2002 (LOQ 0.5-1 µg/L): Butte Slough at Lower Pass Road, Colusa Basin Drain #5, and from the Sacramento River at Village Marina/Crawdads Cantina. These areas receive irrigation returns from rice-growing areas (referenced in their metadata as studies 17, 30, 34, 40, 53, 67, 73 and 75). These stations were also sampled in later years by the California Rice Commission, and additional sample locations have been added to the Rice Pesticide Monitoring Program. Prior to 2002, detections occurred most frequently and at the highest concentrations at the Colusa Basin Drain site, with a 73 to 100% detection rate over the years sampled (Table 41). A seasonal increase in concentrations occurred each May followed by a decline by late June or early July. This seasonal increase corresponds with the period of thiobencarb application in California (Figure 24). Over the eight years of monitoring, thiobencarb was detected in 13-71% of samples from the Butte Slough site (peak concentration range 1.3-3.6 ug/L) and in 0-23% of samples from the Sacramento River site (peak concentration range 0.5-0.6 ug/L). Thiobencarb detections at the Butte Slough site generally occurred from mid-May to early June each year, while four of the six detections at the Sacramento River Site occurred in samples collected between May 21 and May 30 of 2002.

Table 41. Annual monitoring results for thiobencarb at stations associated with rice cultivation (1995-2002; LRL=0.5 ug/L)

		1995	1996	1997	1998	1999	2000	2001	2002
Sacramento River	# samples	16	17	17	17	8	7	12	26
	% detects	0%	0%	0%	0%	13%	0%	8%	15%
	Median	nd	nd	nd	nd	0.5	nd	0.5	0.52
	Min Max								0.29 0.6
Colusa Basin Drain no.5	# samples	22	23	24	23	15	14	23	24
	% detects	73%	70%	75%	61%	93%	100%	78%	63%
	Median	0.87	4.0	1.9	2.35	3.95	4.75	1.85	3.34
	Min Max	0.25 3.67	1 16.55	0.25 12.3	1.1 11	0.7 11.8	1.7 10.7	0.5 5.9	0.28 7.58
Butte Slough	# samples	15	19	16	17	8	7	12	23
	% detects	13%	37%	50%	35%	50%	71%	25%	30%
	Median	1.2	1.2	1.25	0.6	0.9	1.1	1.5	2
	Min Max	1.1 1.3	0.7 2.0	0.5 6	0.6 1.9	0.6 4.1	1 1.6	0.8 2.6	0.7 3.62

"nd" indicates median not determined because sample not detected above LRL
median and minimum values reflect statistics on sample detects

Monitoring by the California Rice Commission since 2003 indicates a pattern of lower detection frequencies and lower maximum concentrations compared to earlier monitoring (1995-2002). The peak concentration observed since 2003 was 3.6 µg/L in the Colusa Basin Drain no. 5 in 2004 (Table 42). Thiobencarb has generally not been detected at the primarly monitoring station on the Sacramento River, and detections at drinking water intakes on the Sacramento River have been ≤ 0.68 µg/L. These sample stations are on the mainstem of the river and not in the immediate vicinity of rice fields (Figure 33).

Table 42. Annual monitoring results for thiobencarb at stations associated with rice cultivation (2003-2011; Detection Limit 0.5 or 0.1 ug/L)

		2003	2004	2005	2006	2007	2008	2009	2010	2011
Sacramento River	# samples	23	14	17	15	16	16	16	16	15
	% detects	0%	0%	0%	0%	0%	6%	0%	0%	0%
	Median	nd	nd	nd	nd	nd	0.62	nd	nd	nd

		2003	2004	2005	2006	2007	2008	2009	2010	2011
	Min Max						na			
Butte Slough	# samples	17	15	17	14	17	16	16	15	16
	% detects	12%	0%	12%	28%	0%	13%	6%	7%	13%
	Median	0.56	nd	0.26	0.61	nd	1.56	0.50	0.80	0.55
	Min Max	0.51 0.6		0.22 0.30	0.56 0.70		1.2 1.99	na	na	0.5 0.6
Sacramento Slough	# samples	17	15	17	13	15	15	15	15	16
	% detects	0%	27%	12%	0%	0%	0%	0%	0%	0%
	Median	nd	0.75	0.47	nd	nd	nd	nd	nd	nd
	Min Max		0.5 0.9	0.33 0.6						
Colusa Basin Drain No. 1 (South end of Ag Drain)	# samples	23	15	15	15	16	15	15	15	14
	% detects	30%	20%	35%	20%	33%	53%	33%	40%	29%
	Median	0.9	1.1	0.59	0.86	0.53	0.81	1.75	65	0.9
	Min Max	0.5 2.3	0.7 1.6	0.20 0.67	0.60 0.90	0.50 0.76	0.56 1.80	1.24 1.84	0.50 1.8	0.6 1.2
Colusa Basin Drain No. 5 (North end of Ag Drain)	# samples	18	16	18	16	15	15	15	15	15
	% detects	28%	13%	17%	6%	7%	27%	40%	33%	13%
	Median	0.62	2.56	0.24	0.97	0.54	0.83	0.82	0.85	1.25
	Min Max	0.5 1.3	0.8 3.6	0.14 0.32	na	na	0.75 1.02	0.54 1.24	0.61 1.5	1.1 1.4

"nd" indicates median not determined because sample not detected above method detection limit
"na" indicates range not applicable because only one sample detected above method detection limit
median and minimum values reflect statistics on sample detects

A recent summary evaluated trends in thiobencarb detections in the Sacramento Valley (Moran 2012). The highest concentrations of thiobencarb were monitored in the 1980s, and between 1992 and 2002. The lowest levels occurred between 2003 and 2007. Increased concentrations of thiobencarb were noted starting in 2008. Additionally, while there were no documented incidents of exceeding the 1.5 µg/L performance goal between 2004 and 2007, the performance goal was exceeded on six occasions between 2008 and 2010. These exceedances were monitored at three sites (CBD1, CBD5, and BS1; Figure 33) at concentrations ranging from 1.5 – 2.0 µg/L. A number of factors were evaluated and considered unlikely sources of the increase including changes in thiobencarb use, management practices, rain events, river flows, wind, emergency releases, and violoations of water hold or seepage requirements. Moran (2012) concluded that transition from the 15G formulation of thiobencarb to the Bolero Ultramax formulation may have been the major factor in increased thiobencarb concentrations observed in the Sacramento Valley from 2008-2011.

All federal thiobencarb labels currently require water from thiobencarb treated fields be held a minimum of 14 days prior to release. Thiobencarb products labeled for California require holding periods of granular formulations of up to 30 days. Additionally California state regulatory bodies have further instituted, 19-d holding periods for liquid formulation i.e., Abolish®, to reduce surface water concentrations, meet performance goals, and permit requirements (Kelley 2000). Monitoring results suggest that these holding requirements have resulted in lower concentrations of thiobencarb in surface waters. The monitoring program has also documented concentrations of thiobencarb exceeding the 1.5 µg/L target concentration at some locations prior to the onset of water release. In some cases, concentrations in surface waters near treated fields have remained elevated above 1.5 µg/L for weeks. These occurrences have been attributed to a number of factors including thiobencarb drift, unplanned as well as planned emergency releases, and seepage (Kelley 2000).

Emergency releases of water from treated fields may be authorized by the state on a conditional basis (*e.g.*, to avoid crop damage associated with high salinity levels or due to heavy rainfall).

Seepage refers to the lateral movement of thiobencarb from the rice field to adjacent waters prior to release through structurally compromised weir boxes and levees. In California, County Agricultural Commissioners serve as local enforcement for pesticide use requirements and conduct a variety of monitoring activities including monitoring for seepage and compliance with pesticide hold requirements. Recent monitoring indicates the occurrence of seepage (Table 43), emergency releases of water from fields treated with thiobencarb (Table 44), and violations of label requirements have been relatively infrequent. However, where seepage and other early releases of water from thiobencarb treated sites do occur, they are likely to contribute to greater surface water concentrations of thiobencarb and may result in greater exposure to listed salmonids.

Table 43. Results of County Agricultural Commissioner's monitoring of seepage associated with thiobencarb applications and compliance with state regulations (2006-2010).

Year	Seepage Inspections			
	Thiobencarb Inspections	Seepage <5 gpm	Seepage >5 gpm	Enforcement actions
2006	929	29	5	1
2007	839	8	3	2
2008	250	25	2	3
2009	883	30	0	0
2010	964	46	4	3
Total	3865	138	14	9
Average	773	27.6	2.8	1.8

Table 44. Results of County Agricultural Commissioner's monitoring of compliance with state regulations associated with the use of thiobencarb (2006-2010).

Year	Emergency Releases		Water Hold		Application	
	Inquiries	Releases	Inspections	Civil Penalties	Inspections	Compliance Violations
2006	0	0	932	4	37	3
2007	4	1	839	2	30	0
2008	1	1	750	2	27	3
2009	2	1	1,012	0	22	0
2010	2	0	1,036	3	28	0
Total	9	3	4,569	11	144	6
Average	1.8	0.6	913.8	2.2	28.8	1.2

8.3.3. Targeted monitoring of thiobencarb during applications

Below we discuss studies that evaluated specific applications of thiobencarb to determine aquatic concentrations associated with edge of field drift, runoff, and discharge of thiobencarb.

8.3.3.1. *Lauck 1979 – MRID 25179*

A series of field studies characterized off-site movement of thiobencarb from applications to rice paddies near estuarine systems in Texas (Lauck 1979). The applications were made to 1900 acres of dry seeded rice in five rice growing areas in Brazoria County, Texas. Surface water, drift, and off-target vegetation were monitored to characterize off-site movement of thiobencarb following applications. Concentrations of thiobencarb were monitored from planned discharge (flushing) and from unanticipated releases associated with runoff from rain events. Thiobencarb was transported off-site by drift and runoff at all five study areas (Table 45). Concentrations in runoff were highest during unplanned releases immediately following applications. For example, a concentration of 8.9 mg/L was detected 2 hrs after application during a emergency release of rice water to a drainage ditch due to a heavy rain event. We consider this value as a worst case scenario. We note that current EPA labels do not authorize such a release. Although it may be possible to achieve comparable runoff from treated fields due to seepage and early releases, these events occur on an infrequent basis (Table 43 and Table 44). Therefore, exposure of salmonids and their designated critical habitat to concentrations of this magnitude are highly unlikely, occurring only rarely.

Many of the runoff releases occurred within a few days of application and frequently attained concentrations above 200 ug/L. The longer the holding time before release of rice water, the lower the concentrations of thiobencarb. As expected, after each release, subsequent releases showed lower thiobencarb concentrations. Thiobencarb was frequently detected in bayous (receiving waters) ranging from non-detectable to 385 ug/L. The maxima detected at each site resulted from releases that occurred on the same day of application or within a few days after application. In Area 2, two sites showed concentrations of 83 and 64 ug/L 14 days after application whereas in Area 3, 7.0 and 140 ug/L detections were measured 13 days after application. Thiobencarb concentrations detected in bayous, 26 to 30 days post application from all areas ranged from non-detectable to 6 ug/L.

Thiobencarb drifted off-site at each area and appeared dependent on wind speed (wind direction was not reported). Three areas were monitored for off-site drift (Table 45). As expected, the greatest drift occurred at distances nearest the center of the swath. Thiobencarb was detected at distances of up to 800 m, however most drift occurred within 50 m.

Non-target vegetation was collected from one of the study areas before and after applications. Cattail and turtle grass were the predominant vegetation within drainage ditches and emergent aquatic vegetation, respectively. Vegetation was collected at 12.5, 50, 100, 200, and 400 meters downwind from application. Grasses and broadleaf plants were collected including a rush, nutgrass, and hedge parsley. The vegetation was not noticeably affected by drift according to the study results. All species of levee vegetation were directly exposed to thiobencarb at 4.0 lbs/acre. Canary reed grass was the only plant that showed symptoms of injury (*i.e.*, size and maturity were reduced by 55% relative to untreated plants). According to the study, "other resident winter annuals, perennials, and woody plants within levees and bayous showed no symptoms of injury." The study did not report when the plants were assessed for damage, nor what type of effects were measured, how they were measured, or any raw data relating to plant responses.

Table 45. Texas field studies that evaluated offsite drift and runoff with thiobencarb in three tidally influenced bayous (Lauck 1979)

Bolero 8EC: 84.6% thiobencarb	Area 1 (592 acres)	Area 2 (451 acres)	Area 3 (435 acres)	Area 4 (894)	Area 5 (206 acres)
Application date	3/25/1979	3/20/1979	3/20/1979	3/20; 3/24; 4/08/1979	4/08/1979
Lbs/ Acre applied (nominal)	4 lbs/acre	4 lbs/acre	4 lbs/acre	4 lbs/acre	4 lbs/acre
Application type	aerial	aerial	aerial	aerial	Aerial
Number of applications	1	1	1	1	1
Drainage ditch distance between rice field and bayous	0.3 - 4 miles	200 - 2300 ft	750 -1650 ft	1650 - 2400 ft	Not reported
Weather conditions day of application	- Air temp = 16.1˚C - 0-2 mph	-Air temp = 24.7˚C - 5-10 mph	-Air temp = 29.2˚C - 4-8.5 mph	na	na
Spray drift card distance from application site (meters)	12, 25, 50, 100, 200, 400, 500, 700, 800, 1200	12, 25, 50, 100, 200, 400, 500, 700, 800, 1200	12, 25, 50, 100, 200, 400, 500, 700, 800, 1200	na	na
Drift on application day (percent of applied)	39 ft = 5.71 82 ft = 2.51 164 ft = 0.44 328 ft = 0.21 656 ft = 0.32 1312 ft = 0.05 1640 ft = nd 2297 ft = nd 2625 ft = nd 3937 ft = nd	39 ft = 7.06 82 ft = 2.30 164 ft = 1.48 328 ft = 0.29 656 ft = 0.05 1312 ft = 0.02 1640 ft = 0.01 2297 ft = 0.06 2625 ft = nd 3937 ft = nd	39 ft = 15.02 82 ft = 2.13 164 ft = 0.45 328 ft = 0.07 656 ft = 0.03 1312 ft = 0.02 1640 ft = nd 2297 ft = nd 2625 ft = 0.01 3937 ft= nd	na	na
Surface water monitoring of thiobencarb	Field outlets, drainage ditch: Maximum = **0.142 – 0.595 mg/L** Receiving water = **<2.0 – 40 ug/L**	Field outlets, drainage ditch: Maximum = **0.69 - 8.9 mg/L** application day Receiving water = **<2.0 – 83 ug/L**	Field outlets, drainage ditch: Maximum = **0.028 -0.405 mg/L** Receiving water = **<2.0 – 385 ug/L**	Field outlets, drainage ditch: Maximum = **0.010 -0.313 mg/L** Receiving water = **<2.0 – 55 ug/L**	Field outlets, drainage ditch: Maximum = **0.271 – 0.287 mg/L** Receiving water = **<2.0 – 64 ug/L**
na denotes not applicable, spray drift not monitored nd denotes not detected					

8.3.3.2. *Beiver 1994 – MRID 434040-03*

Two studies evaluated off-site transport of thiobencarb and its degradate, thiobencarbsulfoxide, following a single aerial application of Bolero 8EC to rice fields, one in Texas and one in Arkansas. Study results are presented in MRID 434040-03 and summarized in EPA RED, CRLF BE, and salmonid BE. Table 46 provides information on the two field studies including results of drift and surface water monitoring.

Drift was measured on application day with spray cards located along the fields perimeters. In the Arkansas study the greatest offsite drift occurred (3.9 -14.5% of the application amount) west of the rice field. A SE wind of 6 mph was measured during the application. Drift was less extensive east and south of the field. Drift cards alongside the bayou collected no detectable thiobencarb (<0.018 µg/L). In the Texas study, drift was less extensive, with a maximum of 7.4%, likely due to reduced winds (3-4 mph). Four drift cards were placed alongside Bernard Creek, two of which resulted in 0.2% of the applied thiobencarb detected (*i.e.*, 0.008 lbs thiobencarb per acre). The results supported the hypothesis that drift indeed occurs following aerial application of thiobencarb to rice and can be a significant pathway for offsite contamination. Wind speed and direction, application rate, distance to aquatic habitats, droplet size, and release height are all key determinants of drift.

Runoff was monitored in both studies for thiobencarb and thiobencarbsulfoxide. Results show that thiobencarb moved offsite during rain events and planned flushing events. Highest concentrations occurred during the first runoff events in both studies (Arkansas 380 µg/L and Texas 2300 ug/L). The three day average of thiobencarb in the Texas drainage ditch was 1120 µg/L (sampled multiple times per day), while the four day average in the Arkansas drainage ditch was 277 µg/L. Receiving waters (Bayou Bartholomew [AR] and West Bernard Creek [TX]), where aquatic life occurred, were also monitored in both experiments. No toxicity experiments were conducted with aquatic animals or plants in either experiment. Receiving water concentrations reached 42 µg/L in the Texas study and remained less than 1.0 µg/L in the Arkansas study, with the exception of one site which detected thiobencarb on day 2 and 3 post treatment at a concentration of 260 µg/L. Thiobencarbsulfoxide was consistently detected in both studies with maximum concentrations in the 60 µg/L range.

In aggregate, the results identify drift and runoff as contributing pathways to aquatic contamination from field applications. Additionally, the data show that early release of flooded rice fields, whether due to rain or flushing practices, can contribute highly toxic concentrations to off-site aquatic habitats. Elevated concentrations within drainage systems and proximate aquatic habitats remain for multiple days. Current federal labels, and more restrictive labels on thiobencarb products sold in California, require that water from thiobencarb treated fields be held 14 days or longer prior to release to aquatic habitats. Therefore, concentrations observed in receiving waters <14 days after application may not reflect likely exposure of listed salmonids and their designated critical habitat. Peak concentrations observed in receiving waters 14 days or more after applications were <30 ug/L.

Table 46. Registrant submitted field studies in Arkansas and Texas that evaluated off-site drift and runoff with thiobencarb and thiobencarbsulfoxide (MRID 434040-03)

Bolero 8EC: 84.6% thiobencarb	Bayou Bartholomew, AR April 30– June 27 1993 58 day experiment	East Bernard, TX May 12- June 26 1993 45 day experiment
Application date	May 1	May 14
Lbs/ Acre applied (nominal)	4 lbs/acre	4 lbs/acre
Application type	aerial	Aerial
Number of applications	1	1
Drainage ditch distance between rice field and receiving water	0.5 miles	300 ft
Weather conditions day of application	-Air temp = 21˚C -SE wind 9.7 km/hr (6 mph) -100% cloud cover with light rain	-Air temp = 26.7˚C -N-NE and N-NW wind 4.8-6.4 km/hr (3-4 mph) -Clear skies
Spray drift card distance from application site	100 ft, 12 cards deployed and 4 alongside receiving water	100 ft, 11 cards deployed and 4 alongside receiving water
Drift on application day (percent applied), drift cards placed (distance from field not reported)	Maximum= 14.5% Range = <0.018- 14.5% North side of field = 5.3-7.8% West side of field = 3.9-14.5% East side of field = none detected South side of field = <0.0018-0.032% Receiving water= <0.018 ug/L	Maximum= 7.4% Range = <0.017 – 7.4% North side of field=no drift cards deployed West side of field=6.8-7.4% East side of field= <0.6% (pilot did not spray on East side border South side of field= 0.89-5.2% Receiving water= <0.017-0.2%
Rain and flushing events causing runoff of thiobencarb Percent indicates % of runoff compared to total water volume	Day 1-4, 13% runoff Day 10-13, 33% runoff Day 25-28, 25% runoff Day 29-30, 34% runoff	Day 3-5, 30% runoff Day 9-11, 36% runoff Day 12-16, 17% runoff

Bolero 8EC: 84.6% thiobencarb	Bayou Bartholomew, AR April 30– June 27 1993 58 day experiment	East Bernard, TX May 12- June 26 1993 45 day experiment
Surface water monitoring Thiobencarb (ug/L)	Drainage ditch: Maximum = 380 ug/L 4 day mean = 277 ug/l 1 day PTI: 330-380 ug/L 13 days PTI: 10 ug/L 28 days PTI: <1.2 ug/L Receiving water = <0.05-260 ug/L -Max (260 ug/L) on 3 day PTI -26 ug/L on day 42 PTI	Drainage ditch: Maximum = 2300 ug/L 3 day mean = 1120 ug/L 3 days PTI: 1600-2300 ug/L 14 days PTI: 1.6-3.6 ug/L 30 days PTI: <0.8 ug/L Receiving water = <0.5-42 ug/L -Max (42 ug/L) on 12 days PTI -29 ug/L on day 14 PTI
Surface water monitoring Thiobencarbsulfoxide (ug/L)	Drainage ditch: Maximum = 61 ug/L 4 day mean = 45 ug/L Receiving water = <1 ug/L	Drainage ditch: Maximum = 67 ug/L 3 day mean = 49 ug/L Receiving water = <0.5-5.1ug/L

8.3.3.3. *Thiobencarb monitoring published in the open literature*

We located several studies from Japan on the use of thiobencarb in rice. In the first study, transport and partitioning of thiobencarb entering the inner Sugao marsh and mouth of the Iinuma River was monitored over the course of normal water discharge from surrounding rice paddies (Ibaraki Prefecture, Japan). Thiobencarb was detected in samples collected between late April to early May with suspended solids accounting for 3.5% of the estimated load of thiobencarb to the marsh, and fine suspended solids accounting for 59% of the total suspended solid fraction (Kawakami et al. 2006).

Dissipation and runoff of pesticides used in rice cultivation was monitored during the 2003 growing season at a rice paddy located in Higashi, Hiroshima City, Japan. Peak concentrations of thiobencarb (417.55-461.89 ug/L) were reached in paddy fields 3 days after application at a rate of 1.3 lbs a.i./A. Thiobencarb was detected downstream from the paddy up to 3 days after application at 0.02-0.08% of the applied concentration. Thiobencarb concentrations decreased to trace levels within one month after application (Parveen et al. 2005).

The concentrations, loadings and losses of pesticides used in rice fields were investigated between 1993 and 1997 in the Seta River, which is the only natural outlet of Lake Biwa, Japan.

The Lake Biwa catchment area is 20% paddy fields with six rivers flowing into the lake. The detection of thiobencarb in the influent rivers began in the middle of May and declined to detection limits after July. Thiobencarb was detected in 27% of the samples collected between late May and early August at concentrations ranging from 0.026-0.057 µg/L (Sudo et al. 2002).

A study evaluating continuous versus intermittent irrigation schemes as rice field water management practices indicated that the intermittent irrigation results in less losses of pesticide from rice paddies. The day after application at a rate of 1.3 lbs a.i./A, thiobencarb concentrations peaked in paddy drainage waters at 595 µg/L and rapidly declined over a period of two weeks (Watanabe et al. 2007).

8.3.4. Exposure to Other Action Stressors

Stressors of the action also include the metabolites and degradates of the a.i.s, other active and inert ingredients included in their product formulations, and tank mixtures and adjuvants authorized on their product labels. Below we summarize information presented in the BEs and provide additional information to characterize exposure to these stressors.

8.3.4.1. *Metabolites and degradates of thiobencarb*

EPA documents identified several degradates of thiobencarb (see previous section *Summary of Chemical Fate of Thiobencarb*). However, estimates quantifying potential exposure of listed salmonids and their habitat to these transformation products were limited and remain a considerable source of uncertainty. In general, failure to consider exposure to these breakdown products increases the likelihood that risk is underestimated.

8.3.4.2. *Other ingredients in formulated products*

NMFS reviewed all of the active labels of thiobencarb and found only one contained multiple a.i.s (EPA Reg. No. 71085-30). Active enduse products contain 10 – 84 % thiobencarb. Valent provided NMFS with Confidential Statements of Formula for Bolero Ultramax Herbicide (EPA Reg. No. 59639-79) and Bolero 15G (EPA Reg. No. 59639-112), the two thiobencarb products currently marketed in California. The specific chemical constituents in Bolero Ultramax Herbicide and other thiobencarb end-use products have not been communicated to us (Table 47).

Table 47. Examples of thiobencarb product ingredients.

EPA Product Registration Number	Active Ingredients %	Other Ingredients %
59639-79*	Thiobencarb 84	16
59639-80	Thiobencarb 10	90
59639-112*	Thiobencarb 15	85
63588-4	Thiobencarb 97.4	2.6
63588-6	Thiobencarb 84	16
63588-14	Thiobencarb 15	85
71085-30	Propanil 35, Thiobencarb 31	34
CA-930003	Thiobencarb 84	16

*Currently registered in California

Nonylphenol (NP) and nonylphenol polyethoxylates are "other ingredients" that may be part of a pesticide product formulation and are common adjuvant ingredients added during pesticide applications. NP and nonylphenol polyethoxylates are also ingredients in detergents, cosmetics, and other industrial products and are a common wastewater contaminant from industrial and municipal sources (Koplin et al. 2002). NP has been linked to endocrine disrupting effects in aquatic systems (Arsenault et al. 2004, Brown et al. 1999, Brown et al. 2003, Brown et al. 2005, Madsen et al. 2004, Schoenfuss et al. 2008a). A national survey of streams found that NP was among the most ubiquitous organic wastewater contaminants in the U.S., detected in more than 50% of the samples tested (Koplin et al. 2002).

Table 48. Detection and concentrations of nonionic detergent degradates in streams of the U.S. (Koplin et al 2002).

Chemical	Frequency Detected	Maximum (μg/L)	Median (μg/L)
4-nonylphenol	50.6	40	0.8
4-nonylphenol monoethoxylate	45.9	20	1
4-nonylphenol diethoxylate	36.5	9	1
4-octylphenol monoethoxylate	43.5	2	0.2
4-octylphenol diethoxylate	23.5	1	0.1

We are uncertain to what degree NP and NP-ethoxylates may or may not occur in pesticide products that contain thiobencarb and/or are added prior to application. Inert ingredients are often not specified on product labels. Additionally, NP and NP-ethoxylates represent a very small portion of the more than 4,000 inert ingredients that EPA permits for use in pesticide formulations (Koplin et al. 2002). Many of these inerts are known to be hazardous in their own right (*e.g.*, xylene is a neurotoxin and coal tar is a known carcinogen). Several permitted inerts are also registered a.i.s (*e.g.*, copper, zinc, chloropicrin, chlorothalonil). Inerts can be more than 50% of the mass of pesticide products, and millions of pounds of these products are applied to the landscape each year (Koplin et al. 2002). This equates to large contaminant loads of inerts that may adversely affect salmon or their habitat. Uncertainty regarding exposure to these ingredients will be qualitatively incorporated into our analysis.

8.3.4.3. *Tank Mixtures*

Several pesticide labels authorize the co-application of other pesticide products and other materials in tank mixes, thereby increasing the likelihood of exposure to multiple chemical stressors (see previous section *NMFS exposure estimates for pesticide mixtures*). In some cases specific application of other pesticide products or adjuvants are recommended. In all cases, tank mixtures are authorized unless specifically prohibited on the product label. These ingredients and the other inert ingredients in these products are considered part of the action because they are authorized by EPA's approval of the FIFRA label. Exposure to, and risk associated with, potential ingredients in tank mixtures were not addressed in EPA's BEs and remain a significant source of uncertainty.

8.3.4.4. *Environmental Mixtures*

As described in the *Approach to the Assessment*, we analyze the status of listed species in conjunction with the *Environmental Baseline* in evaluating the likelihood that action stressors will reduce the viability of populations of listed salmonids. This involves considering interactions between the stressors of the action and the *Environmental Baseline*. For example, we consider that listed salmonids may be exposed to the wide array of chemical stressors that occur in the various marine, estuarine, and freshwater habitats they occupy throughout their life cycle. Exposure to multiple pesticide ingredients most likely occurs in freshwater habitats and nearshore environments adjacent to areas where pesticides are used. As of 1997, about 900 a.i.s were registered in the U.S. for use in more than 20,000 different pesticide products (Aspelin and Grube 1999). Typically 10 to 20 new a.i.s are registered each year (Aspelin and Grube 1999). In a typical year in the U.S., pesticides are applied at a rate of approximately five billion pounds of a.i. per year (Kiely et al. 2004). Pesticide contamination in the nation's freshwater habitats is ubiquitous, and pesticides usually occur in the environment as mixtures (Gilliom et al. 2006a). Gilliom et al. (2006a) estimated that over "90% of the time, water from streams with agricultural, urban, or mixed-land-use watersheds had detections of two or more pesticides or degradates, and about 20% of the time they had detections of 10 or more." The likelihood of exposure to multiple pesticides throughout a listed salmonid's lifetime is great, considering the geographical range of their migration routes and habitats occupied during spawning and rearing.

Studies have suggested that assessment of pesticide mixture toxicity to aquatic life is needed given the widespread and common occurrence of pesticide mixtures, particularly in streams where the total combined toxicity of pesticide mixtures may be greater than that of any single pesticide compound (Gilliom 2007, Gilliom et al. 2006a). Exposure to multiple pesticide ingredients can result in additive and synergistic responses as described in the *Risk Characterization* section. It is reasonable to conclude that compounds sharing a common mode of action cause additive effects and in some cases synergistic effects. Exposure to these compounds and other baseline stressors (*e.g.*, thermal stress) was not a consideration in EPA's BEs, which only considered effects from single a.i.s. Therefore, risk to listed species may be underestimated in EPA's assessments.

8.4. Exposure Conclusions

Pacific salmon and steelhead use a wide range of freshwater, estuarine, and marine habitats and many migrate hundreds of miles to complete their life cycle. Thiobencarb is frequently applied to rice crops within the distribution of three listed salmonids that occur in the Central Valley of California: CV spring-run Chinook, Sacramento River winter-run Chinook, and CV steelhead. Thiobencarb and its degradates have been detected in habitats utilized by these species. Because the action proposes continued use on rice in California for the next 15 years, we expect thiobencarb will continue to be present within the freshwater distribution of these species, including times when salmon are present. Therefore we expect some individuals of each of these species, and their designated critical habitats, will be exposed to the thiobencarb and other stressors of the action.

We considered several sources of information to characterize the likely range of exposure to thiobencarb and the other action stressors (Table 49). Inherent in the modeling estimates is the assumption that the pesticide is applied in a location next to or draining into salmon-bearing waters. Monitoring data may reflect pesticide applications proximate to the waterbody (i.e., targeted monitoring), or resulting from more distant uses in the watershed (ambient monitoring). The surface water monitoring data used were not designed to determine exposure to listed salmonids. Monitoring studies were designed for other purposes and sample design did not reflect the spatial and temporal distribution of the listed species. Therefore, caution should be exercised in using these data for that purpose, especially when conducting probabilistic assessments. Defining exposure of the stressors of the action to the listed species is also complicated by uncertainty associated with the following factors:

- Product labels authorize the application of chemical mixtures that are not specified or clearly defined (*e.g.*, the ingredients of pesticide formulations are not fully disclosed, labels recommend tank mixture applications with other products, and tank mixtures with other pesticides are permitted unless specifically stated otherwise);
- Historical use of thiobencarb, including frequency of use, locations of use, and the amount of thiobencarb products applied may not reflect future use.

Table 49. Chemical exposure data ranges in monitoring data and modeling.

Exposure pathway	Value and Units
Runoff	
Peak concentrations measured within rice fields (Table 24) Thiobencarb Thiobencarb sulfoxide 4-chlorobenzylmethylsulfone	 13 - 576 µg/L 8.9 - 22 µg/L 5.2 – 8.3 µg/L
Dissipation half-life (Table 24) Thiobencarb Thiobencarb sulfoxide 4-chlorobenzylmethylsulfone	 5.8 – 8.7 d 2.8 - 3.4 d 6.0 – 10.4 d
Concentrations estimated at discharge (see *EPA Rice Model*)	

Exposure pathway	Value and Units
Day 0 Day 14 Day 30	2018 µg/L 350 µg/L 47 µg/L
Concentrations measured in discharge and receiving water (see *Targeted Monitoring*) Day 0 Day 14 Day 30	 Non-detect - 8,900 µg/L Non-detect -83 µg/L Non-detect – 6 µg/L
Frequency of early releases in Sacramento Valley (Table 44) Authorized Unauthorized Seepage < 5 gpm Seepage > 5 gpm	 0-3/year 0.2% of applications 3.6% of applications 0.4% of applications
Drift	
AgDrift predictions to floodplain habitats without buffer (Table 33) Ground application Aerial Application	 37 – 736 µg/L 96-1,910 µg/L
AgDrift Predictions to terrestrial habitats (Table 34) – *ground applications* with buffer 1 ft 10 ft 100 ft 1000 ft	 1.19 lbs/A; 29.68% applied 0.09 lbs/A; 2.24% applied 0.010 lbs/A; 0.24% applied 0.001 lbs/A; 0.03 % applied
AgDrift Predictions to terrestrial habitats (Table 34) – *aerial applications* with buffer 1 ft 10 ft 100 ft 1000 ft	 1.91 lbs/A; 47.76% applied 1.21 lbs/A; 30.34% applied 0.22 lbs/A; 5.61% applied 0.02 lbs/A; 0.55% applied
Measured thiobencarb drift from aerial application with buffer (Table 45) 39 ft 82 ft 164 ft 328 ft 656 ft 1312 ft 1640 ft 2297 ft 2625 ft 3937 ft	 5.71 - 15.02 % applied 2.13 – 2.51 % applied 0.44 – 1.48 % applied 0.07 – 0.29 % applied 0.03 – 0.32 % applied 0.02 – 0.05 % applied 0.00 – 0.01 % applied 0.00 – 0.06 % applied 0.00 - 0.01 % applied 0.00 % applied
Ambient monitoring in California surface water (see *USGS NAWQA Data* and *Monitoring Data from California*)	
Prior to establishing irrigation water holds Since establishing irrigation water holds	Not detected – 170 µg/L Not detected – 16.55 µg/L

Runoff of thiobencarb from rice fields, including planned discharge following label-specified holding periods of 14 and 30 days, will result in deposition of thiobencarb to salmonid habitats. Concentrations may attain levels as high as 350 µg/L at 14 days and 47 µg/L at 30 days in discharge waters (Table 49). Concentrations may be orders of magnitude higher in runoff if rice irrigation water is released within hours or days post application. This may occur due to emergency releases of irrigation water, and from unplanned and/or unauthorized discharges. Several field studies reported concentrations in the mg/L range following early releases, one attaining levels as high as 8.9 mg/L (Lauck 1979). We do not expect this to be a common occurrence based on reports over the last decade on the frequency of early releases.

Drift of the liquid formulation of thiobencarb into salmonid habitats is expected during the application period, primarily May and June, and to a lesser extent April and July. Concentrations will depend on the proximity to application area, application method, droplet size, release height, wind speed/directions, receiving water volume/flow, and interception by riparian vegetation. Based on drift studies, modeling exercises, and surface water monitoring studies, aquatic concentrations from drift may range from non-detectable to as high as 1910 µg/L (Table 49).

We assume that the exposure estimates provided by EPA in the BEs, and additional modeling and monitoring information provided above, represent realistic exposure levels for some individuals of the three listed species. Further, we assume the distribution within the range of exposures is a function of pesticide use and the duration of time listed salmonids spend in these habitats. All listed Pacific salmon and steelhead occupy habitats that could contain high concentrations of these pesticides at one or more life stages. However, the time spent in these habitats varies among the species that occur in the rice growing region. We are unable to accurately define exposure distributions for thiobencarb and the other stressors of the action given limitations of the available information. We assume the highest probability of exposure occurs in freshwater habitats in close proximity to Central Valley rice fields where thiobencarb is applied. We evaluated several spatial and temporal relationships to qualitatively assess the relative abundance of the three species in aquatic habitats near rice (Table 28, Table 30, Table 32, and Table 50).

Table 50. Relative abundance of listed salmonids near rice fields during the period when thiobencarb is applied and water is discharged from thiobencarb treated fields (April - July).

Life Stage	None Expected	Low	Medium	High
CV spring-run Chinook				
Adult Migration				X
Spawning	X			
Early rearing		X		
Juvenile migration		X		
Sacramento River winter-run Chinook				
Adult migration			X	
Spawning	X			
Early rearing		X		
Juvenile migration		X		
CV steelhead				
Adult migration		X		
Spawning	X			
Early rearing		X		
Juvenile migration			X	

The relative abundance estimates suggest that exposure will vary depending on the species and life stage. Given the available information, we conclude that some adults of the three listed species are likely to be exposed to thiobencarb during their spawning migration. However, spawning locations for these species occur in the watershed above the locations where rice is grown. Consequently, we consider it unlikely that any significant exposure to thiobencarb will occur during spawning, egg incubation, and fry emergence. However, some post-emergence fry, parr and pre-smolts may be exposed to the stressors of the action as they are displaced or migrate downstream to habitats that are near locations where thiobencarb is applied. These observations are carried forward to the Risk Characterization section to develop risk hypotheses and evaluate potential effects to individuals and populations. Additionally, we carry forward the following conclusions:

- The vast majority of exposure to thiobencarb drift occurs in May and June with less frequent exposures in April and July. Floodplain habitats alongside rice fields are at the greatest risk of receiving elevated concentrations from aerial and ground applications (i.e., 37 – 1910 µg/L);
- runoff containing thiobencarb typically occurs 14-30 days post application (after required holding periods), thus the greatest probability of exposure to the highest runoff concentrations likely occurs from mid-May through the end of July;
- riparian systems and multiple life stages of salmonids are likely exposed to thiobencarb from drift during applications and from runoff following release of rice irrigation water into drainage networks that ultimately return water to salmonid containing waters;
- migrating adults, rearing juveniles, and migrating juveniles overlap with peak thiobencarb applications in May and June; and
- monitoring data and modeling results show that thiobencarb contaminates salmonid habitats from drift and runoff with peak values occurring during thiobencarb applications.

Substantial data gaps in EPA's exposure estimates include estimates for "other ingredients" in pesticide formulations, other pesticide products authorized for co-application with thiobencarb, adjuvants, degradates, and metabolites. Although NMFS is unable to comprehensively quantify exposure to these chemical stressors, we are aware that exposure to these stressors is likely. We assume these chemical stressors may pose additional risk to listed Pacific salmonids. In order to ensure that EPA's action is not likely to jeopardize listed species or destroy or adversely modify critical habitat, NMFS analyzes potential exposure based on all stressors that could result from all uses authorized by EPA's action.

8.5. Response Analysis

In this section we evaluate toxicity information from the stressors of the action organized by assessment endpoints. The endpoints target potential effects to individual salmonids and their supporting habitats. The assessment endpoints represent biological and habitat attributes that, when adversely affected, lead to reduced fitness of individual salmonids or degrade PCEs such as prey abundance, water quality, and riparian vegetation (evaluated in *risk characterization for designated critical habitat* section). Uncertainties in the available toxicity information are discussed as they are encountered and summarized at the end of this section. Following the response analysis, we compare concentrations described in the exposure analysis with assessment endpoints to evaluate whether individual fitness or habitat endpoints might be compromised. Salmonid and designated critical habitat risk hypotheses are evaluated separately in the *Risk Characterization* sections.

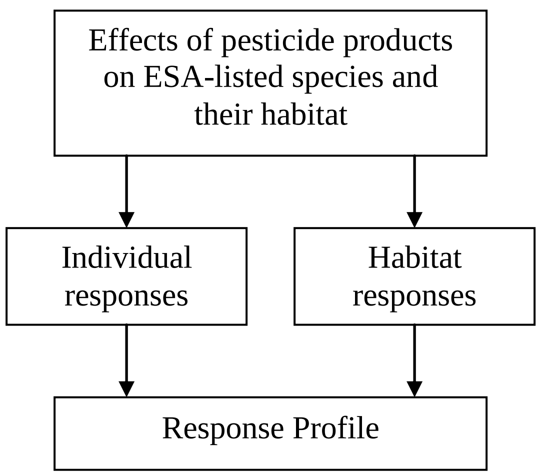


Figure 34. Response Analysis Conceptual Model

We begin the response analysis by describing the toxic mode and mechanism of action of thiobencarb, which sets the stage for what biological endpoints are assessed. Next, we summarize the toxicity data presented in the salmonid BE, RED, IRED, California Red Legged Frog BE, EFED science chapters, and open literature (Table 51). The information is organized by assessment endpoints (*e.g.*, survival, growth, migration, etc.). The information provided in

EPA documents primarily addressed aspects of survival, growth, and reproduction of some aquatic species following exposure to thiobencarb. Other information from selected field experiments on ecological endpoints was sometimes discussed within EPA documents. Since we found little information in EPA's documents regarding formulation, other ingredients, and mixture toxicity, we conducted our own literature review.

Under the ESA and implementing regulations NMFS evaluates all direct and indirect effects of a federal action. We therefore evaluate all aspects of an action that may reduce fitness of individuals or reduce PCEs of designated critical habitat. This includes toxicity information for thiobencarb, its degradates, other ingredients within thiobencarb formulations, and other pesticide active ingredients commonly combined in recommended tank mixtures. The evaluation includes information that EPA provided on survival, growth, or reproduction, and also encompasses a broader range of endpoints including behaviors, endocrine disruption, and other physiological alterations. The information we assessed is derived from published scientific journals, government agency reports, theses, books, applicant-submitted information, and independent reports. The most relevant study results are those that directly measure effects to identified assessment endpoints derived from studies with salmonids, preferably ESA-listed Pacific salmonids or hatchery surrogates. We also evaluate additional stressors that may influence the toxicity of the stressors of the action such as temperature.

8.5.1. Thiobencarb's Mode and Mechanism of Action

Thiobencarb belongs to a class of herbicides known as thiocarbamates which impair and kill plants by inhibiting the production of hormones called gibberellins. Thiobencarb also inhibits a plant's fatty acid and lipid biosynthesis (Weed Science Society of America 2007). Thiocarbamates affect the functioning of acetyl-CoA elongases, preventing the extension of the fatty acid chains (Gronwald 1991). Carbamothioates "may be rapidly metabolized or poorly translocated" and effects on plant lipid biosynthesis generally occur at the plant surface (Gronwald 1991). Thiobencarb is currently registered as an herbicide.

Thiobencarb is also an acetylcholinesterase inhibitor. USEPA included thiobencarb in its analysis of common mechanism of action with six other thiocarbamates: molinate, EPTC, trillate, butylate, pebulate, and cycloate (USEPA 2001). EPA concluded that thiobencarb shares a common mechanism of action for AChE inhibition with other thiocarbamates. Thus, we use other thiocarbamates as chemical surrogates when information is unavailable for thiobencarb. Empirical research indicates that AChE is inhibited by thiobencarb in fish and mammals (Fernandez-Vega et al. 1999, Pentyala and Chetty 1993). For example, thiobencarb reduced European eel plasma AChE activity by 50% following 96 hours of exposure at 220 µg/L (1/60th of the 96 hr LC_{50}) and remained depressed (<50% activity) following five days of recovery in clean water (Fernandez-Vega et al. 1999).

8.5.2. Temperature and toxicity

We located no information showing specific effects of temperature on thiobencarb's toxicity. Elevated temperatures typically increase the magnitude of toxic effects in fish particularly for pesticides that are transformed in the fish to more toxic metabolites such as organophosphates (Mayer and Ellersieck 1986). Carbamates including thiocarbamates have also been shown to impart greater toxicity to organisms when exposed to elevated temperatures (Altinok et al. 2006). Differences in toxicity due to temperature have been attributed to differences in respiration rate, chemical absorption, and metabolism. As discussed in the *Environmental Baseline*, temperature is a recognized stressor to salmonids in the Central Valley (Myrick and Cech 2005). Water temperatures in the lower Sacramento River regularly exceed 20 $^{\circ}$C by late spring, and statistical studies of coded wire-tagged juvenile Chinook show increased mortality as a function of temperature (Baker et al. 1995). Water temperatures higher than optimum levels can kill salmonids, increase physiological stress making them more susceptible to other stressors, increase predation, and affect salmonid's prey base. Thus temperature directly affects survival, growth rates, distribution, and developmental rates. We therefore discuss Chinook and steelhead fitness implications in the context of elevated temperatures enhancing thiobencarb's toxicity.

8.5.3. pH and toxicity

We located no studies that tested pH's effect on thiobencarb's aquatic toxicity. However, pH influenced acute lethality of N-methyl carbamates aminocarb, carbofuran, and carbaryl (Mayer and Ellersieck 1986). We are uncertain whether thiobencarb, a thiocarbamate, will behave similarly as the three N-methylcarbamates. Thus this is a recognized data gap and we cannot definitively determine whether biological endpoints will be more or less affected by pH.

Studies with mixtures of AChE inhibiting herbicides. We located information on mixture toxicity indicating additive toxicity when fish are jointly exposed to thiobencarb and another thiocarbamate, molinate (Finlayson and Faggella 1986). Acute lethality tests (96-hr LC_{50}s) of a thiobencarb and molinate mixture showed additive toxicity to juvenile steelhead, Chinook salmon, and channel catfish. Thiobencarb and molinate were twice as lethal when present together at 1:1 LC_{50}-ratios than they were individually (Finlayson and Faggella 1986). Thiobencarb was 18-22 times more toxic than molinate based on individual LC_{50}s for the three fish species. Thiobencarb concentrations used in the ratios were 360 (steelhead), 430 (Chinook), and 990 µg/L (channel catfish). These results corroborate additive mixture toxicity when pesticides share a common mechanism/mode of toxic action. Although use of molinate has been prohibited since 2009, other thiocarbamate pesticides are currently used in California and may co-occur in the environment with thiobencarb in the future (*e.g.,* EPTC and cycloate; (CDPR 2010b) .

8.5.4. Herbicide effects to salmonids and their habitats

In previous Opinions, we have addressed organo-phosphorus (OP) and carbamate insecticides (NMFS 2008a, NMFS 2009e, NMFS 2010a). Although used to control insects, these pesticides have a mode of action (cholinesterase inhibition) expected to directly affect salmon and other non-target organisms, such as aquatic and terrestrial invertebrates that provide a forage base for salmon. While thiobencarb is also a cholinesterase inhibitor that may have direct effects on fish and invertebrates, we also investigated the indirect effects caused by its herbicidal mode of action. We surveyed available literature regarding herbicide effects in the environment, considering a broad range of herbicides, including ones not addressed in this Opinion. A summary of this survey and conceptual models based on information gleaned from this survey are presented below.

8.5.5. Importance of plants and other photosynthetic organisms in fueling secondary production within salmonid habitats

Secondary production within aquatic systems, including production of juvenile salmonids, is ultimately fueled by plants and other photosynthetic organisms (*e.g.,* green algae, diatoms, cyanobacteria). In salmonid freshwater and estuarine habitats, this energy comes from two sources: (1) primary production within aquatic habitats (autochthonous inputs), as well as (2) inputs of organic matter from adjacent terrestrial ecosystems (allochthonous inputs) (Allan and Castillo 2008). Plants and other photosynthetic organisms are primary producers, and can be consumed (by "consumers") as living tissue that is grazed from the benthos (periphyton), as living tissue collected from the water column (phytoplankton), or as dead tissue that is consumed after being colonized by microbial communities (detritus). Invertebrates and fish are specific with regard to their ability to feed on these various food resources, and these distinctions help define functional feeding groups that include grazers, shredders, and predators, among others. Therefore, although there is great diversity in the pathways energy takes in an aquatic system, much of the energy that fuels production in aquatic habitats derives ultimately from plants and other photosynthetic organisms.

Fish can consume a very high proportion of the invertebrate secondary production in aquatic habitats (Huryn 1996, Huryn 1998). Juvenile salmonids are predators that can consume a wide range of invertebrates, including those from all functional feeding groups. Changes in the production of any of these groups could change prey availability for these fish. For example, a reduction in periphyton production on rocks in a stream could reduce invertebrate grazer production. Likewise, a change in the quantity or quality of terrestrial leaf litter falling from a riparian buffer could alter the production of invertebrate shredders downstream. In addition to being the ultimate source of food for much of the invertebrate community, plants also provide habitat for invertebrates and fish, including but not limited to substrate for them to shelter on and under (*e.g.,* macrophytes, root wads). Plants and other photosynthetic organisms within and adjacent to salmonid freshwater and estuarine habitats are therefore essential components of productive salmonid habitats. Actions that affect the diversity, biomass and/or the production of primary producers in and around salmonid habitats may limit or alter secondary production within those systems as well (Figure 35, Figure 36).

As food resources, living plants and other living photosynthetic organisms are especially nutritious for grazing invertebrates and herbivorous fish (Torres-Ruiz et al. 2007), and they often contribute more to overall secondary production within a system than would be expected simply by their standing stock at any one point in time (Allan and Castillo 2008). Because of this high nutritional value, the autochthonous production of plants and other photosynthetic organisms can be limited by grazers, though abiotic factors such as light, nutrients, and water velocity are also often limiting (Blanchet et al. 2008, Rosemond et al. 2000, Sanderson et al. 2009). The relative importance of these biotic and abiotic factors in limiting primary production varies by system and can change seasonally (Huryn 1998, Sanderson et al. 2009).

When primary production is limited or low, consumer production can be limited. This has been demonstrated primarily by amending a limiting resource to the point at which it is no longer limiting. For example, when nutrients are added to nutrient-limited systems, primary production and consequently secondary production can increase (Harvey et al. 1998, Mundie et al. 1991). Fewer studies have examined explicitly how *reductions* in primary producers (or primary production) affect fish and invertebrates, as would potentially occur when sensitive photosynthetic organisms at the base of an aquatic food web are exposed to herbicides. In some cases when algal biomass is reduced by disturbances, invertebrate grazer growth and abundance decline. Higher trophic levels can be affected by these bottom-up effects, as Perry and colleagues (2003) observed in juvenile Chinook salmon. In small tributaries of the Yukon River, a fire and flood reduced the proportion of high quality autochthonously derived energy that salmon consumed, suggesting there may be direct and indirect effects of disturbances on energy transfer among trophic levels including salmon (Perry et al. 2003).

The loss or reduction of inputs of organic matter (including leaf litter, woody debris, and terrestrial insects) from adjacent terrestrial ecosystems can also significantly reduce invertebrate secondary production and potentially fish production (Wallace et al. 1999, Allan et al. 2003). This was demonstrated by Wallace et al. (1999) when they excluded terrestrial leaf litter from a forest stream in the southeast for four years and found that invertebrate production in the affected habitats declined by 78%. Although there were no fish in these systems, they did observe reductions in the top invertebrate predators, illustrating that bottom-up effects of this exclusion of plant material permeated throughout the food web (Wallace et al. 1999). Similarly, Fischer et al. (2010) suggested that differences in food availability associated with the presence or absence of riparian buffers likely affected differences in observed fish growth. In systems where allochthonous inputs sustain secondary production (including shaded, forested streams that provide rearing habitat for some salmonids), a reduction in allochthonous inputs could reduce secondary production, and consequently affect fish production. In addition to organic inputs, riparian vegetation provides shade for aquatic habitats, increases bank stability, helps buffer aquatic habitats from contaminants present upland, and helps maintain natural flow dynamics of water, nutrients and sediment (Richardson et al. 2010).

Numerous studies illustrate the trophic linkages among plants and other photosynthetic organisms and the secondary production of fish and their prey. While it is logical that reductions in autochthonous and/or allochthonous food resources could limit consumers and predators, including juvenile salmonids, there are often a number of factors that affect the magnitude and even the direction of change within complex aquatic food webs. These relationships may be directly or indirectly affected by herbicides. The following sections briefly review some of these impacts and discuss the challenges faced in predicting how herbicides may affect salmonids and their critical habitats.

8.5.6. Effects of herbicides on non-target aquatic communities

Potential effects of herbicides on aquatic and riparian communities are illustrated in Figure 35 and in Figure 36. The range of effects includes direct effects (primarily negative) on photosynthetic organisms and water quality parameters, as well as indirect effects (positive and negative) on multiple trophic levels and water quality. Generally, if an herbicide exposure is great enough to reduce primary production within or adjacent to aquatic habitats, there may be effects on multiple trophic levels, including salmonids. A number of factors contribute to the magnitude and direction of effects, although it’s difficult to predict and identify patterns within and across aquatic systems.

Numerous studies using standard toxicity tests have demonstrated that herbicides reduce the growth and biomass of photosynthetic organisms. Plants and photosynthetic organisms are typically more sensitive to herbicides than invertebrates and fish because of the herbicides' various mechanisms of toxic action. For example, Brock and others (Brock et al. 2004) determined HC_5s (hazardous concentrations for 5% of the species) for two herbicides (metribuzin and metamitron) on a variety of taxa, and found, not surprisingly, that the algae and macrophytes were >100 to >1000x more sensitive than invertebrates and fish. Similarly, Van den Brink et al. (Van den Brink et al. 2006) found that herbicides varied in their toxicity, and that relative sensitivities (based on short-term toxicity growth tests) of the taxonomic groups included in the study were algae ≥ macrophytes > invertebrates > vertebrates. For some herbicides, algae and macrophytes were similar in their sensitivities, *e.g.*, for atrazine and diquat, (Van den Brink et al. 2006), but for others, such as 2,4-D (an auxin simulator), macrophytes were significantly more sensitive than all of the algae taxa included in the analyses (Van den Brink et al. 2006). In their extensive review of herbicides, Brock et al. (2000) also concluded that auxin simulators like 2,4-D were generally more toxic to macrophytes than other photosynthesis inhibitors. Neither paper reviewed studies on thiobencarb, however Brock et al. (2000) reviewed studies on triallate (a thiocarbamate) and remarked that the acute 48 h survival EC_{50} for *Daphnia magna* (57 µg/L ; (Johnson 1986)) is similar to that of a standard test algae, *C. selenastrum* (47 µg/L ; (Fairchild JF et al. 1997)).

The direct effects of herbicides on diverse communities of aquatic primary producers can be highly variable. For instance, Gruessner and Watzin (Gruessner and Watzin 1996) exposed stream communities in microcosms to a low concentration of atrazine (5 µg/L) for 14 days, but found no effect on algal biomass. In other studies, the species composition of primary producers changes after exposure while abundance may increase or decrease. Wendt-Rasch et al. (Wendt-Rasch et al. 2003) found that even though macrophyte root growth in mesocosms declined following exposure to metsulfuron methyl, the biomass of periphytic algae on those macrophytes actually increased. In addition, the algal species composition was significantly different in the mesocosm exposed to the highest dose (Wendt-Rasch et al. 2003). Hartgers et al. (Hartgers et al. 1998) observed an initial decline in the abundance of some phytoplankton taxa following exposure to a mixture of herbicides (atrazine, diuron and metolachlor), but by 14 days post-application several phytoplankton taxa had actually increased in abundance.

In addition to direct effects on primary producers, there may be direct effects of herbicides on microbial communities. The processing of organic matter by microbial communities – which includes, in part, making leaf litter palatable to some invertebrates – is a critical energy pathway within aquatic food webs. Despite their importance, there are relatively few studies examining the effects of pesticides on microbial communities. Of the few studies regarding herbicides, it appears there may be some direct and indirect effects at relatively low concentrations. Microbial communities were altered following exposure to various concentrations of atrazine, with some taxa becoming more abundant and productive while others declined (DeLorenzo et al. 1999). In another study, the herbicide diuron limited algal growth in mesocosms, and the abundance, diversity, and activity of the associated microbial community was also limited (Pesce et al. 2006). The authors suggested that diuron decreased the capacity of the microbial community to recover when favorable conditions were provided (as was the case in the control mesocosms), and this reduced the efficiency of the microbial food web (Pesce et al. 2006). Although it is difficult to extrapolate short-term mesocosm studies to potential longer-term effects in the natural environment, these studies suggest that exposure to herbicides can directly affect the structure as well as function of the diverse communities that are the base of aquatic food webs.

The effects of herbicides, either by reducing primary producers or by changing the processes and paths through which energy flows, can have significant effects on higher trophic levels. For example, herbicides are commonly found to reduce the abundance (or biomass or growth rates) of consumers. Interestingly, these indirect effects of herbicides are often reported at concentrations well below those found to have direct effects on consumers. The population growth rate of an aquatic oligochaete *Lumbriculus variegatus* was reduced by 50% after being exposed to only 6 µg/L of the herbicide terbutryn (Brust et al. 2001). This effect was attributed to the reduction of the food source of the oligochaete by the herbicide at a concentration three orders of magnitude lower than the concentration that caused acute toxicity to the oligochaete itself. Similarly, Dewey (Dewey 1986) found that multiple trophic levels within experimental ponds were impacted by atrazine, though effects on higher trophic levels were likely due to indirect effects (reduction in food resources). These effects throughout the food web were found at concentrations one order of magnitude lower than acute toxicity values for a common midge (Dewey 1986). Brock et al. (Brock et al. 2004) observed long-term (lasting >8 weeks) changes in the macroinvertebrate communities within mesocosms treated with metribuzin at concentrations 20x lower than the HC_5s for aquatic invertebrates. In a similar study, predatory ciliates were relatively more affected by the reduction of their prey (phototrophic flagellates) due to exposure to the herbicide prometryn than by the direct toxicity (Liebig et al. 2008). Finally, a number of studies have documented declines in zooplankton densities due to reductions in their phytoplankton food sources following exposure to herbicides (DeNoyelles et al. 1982, Juttner et al. 1995, Kasai and Hanazato 1995).

These examples illustrate that reduced primary production due to herbicide exposure results in bottom-up effects. Alternatively, if an herbicide is directly toxic to consumers, primary production may actually increase as grazing pressure declines (Rohr and Crumrine 2005). In addition, sublethal effects of herbicides on invertebrates have also been found at environmentally relevant concentrations, and this may also have effects throughout the food web. For example, Cook and Moore (Cook 2008) found the herbicide metolachlor (at an environmentally relevant concentration of 80 μg/L) altered agonistic behavior in crayfish.

Effects on water quality are also often reported. These changes are due in part to changes in community metabolism (Brock et al. 2000). For example, if photosynthetic efficiency declines, it is expected that oxygen concentrations and pH decrease (Hartgers et al. 1998, Brock et al. 2000). These effects have been shown to be dose-dependent by Pratt et al. (Pratt et al. 1997), as oxygen levels decreased most significantly in microcosms exposed to the highest doses of the herbicide diquat. Changes in water quality, especially significant declines in dissolved oxygen, may affect sensitive taxa, but it is unclear how often this may occur in salmonid habitats.

Brock et al. (Brock et al. 2000) concluded in their review of herbicides that indirect effects of photosynthetic inhibitors on consumers and predators occur at concentrations around the EC_{50} for standard algae taxa; these impacts on consumers and predators are likely due to reduced availability of food resources and the effects may be delayed relative to the exposure event.

Other effects on the ecosystem (*e.g.,* blooms of insensitive algae) can occur at lower concentrations (*e.g.*, 0.1 of the EC_{50} of standard algae), and these effects may also be delayed. When macrophytes are impacted, organisms using those macrophytes as habitat are immediately impacted. Some studies published after the Brock et al. (2000) review noted indirect effects at surprisingly low concentrations, and, in general, papers published since their review corroborate their findings. As mentioned previously, Brock et al. (2000) reviewed studies on triallate (a thiocarbamate) and remarked that the acute 48 h survival EC_{50} for *Daphnia magna* (57 µg/L; (Johnson 1986)) is similar to that of a standard test algae, *C. selenastrum* (47 µg/L; (Fairchild JF et al. 1997)). It appears that triallate is an order of magnitude more toxic than thiobencarb based on *D. magna* survival, *i.e*., 10 µg/L vs. 101 µg/L, respectively. We present thiobencarb's toxicity to algae and invertebrates in Table 51.

8.5.7. Challenges in scaling up effects and making predictions across salmonid habitats

The current literature describes a wide range of effects of herbicides. While it is difficult to generalize across these studies, it is clear that many studies illustrate that herbicides can have direct and indirect effects on multiple trophic levels within aquatic food webs, and often these effects occur at concentrations well below concentrations expected based on single-species acute toxicity tests. However, it is difficult to predict the magnitude, duration, and direction that these effects may have on juvenile salmonids and their habitat because multiple factors influence these effects. These factors include, but are not limited to, the composition and relative abundances of taxa at the time of exposure (Relyea 2009), the functional redundancy among taxa within the system, and the resilience of the various communities within the system (Brock et al. 2000). In addition, the abiotic conditions, the presence of other stressors, and the properties of the herbicides themselves (*e.g.,* mode of action, persistence) can affect the magnitude, duration and direction of effects.

Juvenile salmonids are generally opportunistic drift-feeders, and are therefore sensitive to factors that influence the general quantity and quality of invertebrate prey items. If, for instance, there were reductions in the production of invertebrate grazers or the inputs of invertebrate prey from riparian vegetation, salmonids may be forced to alter their foraging behavior (*e.g.,* take more risks, select less energy-rich prey) (as shown in Figure 35 and in Figure 36). Alternatively, if there were shifts, rather than reductions, in the abundances and composition of the prey community within riparian and aquatic habitats, indirect impacts on salmonids may be minimal if foraging behaviors were not altered. Whether or not production of prey decreases or shifts (or increases) after exposure to herbicides will depend in part on the composition of the community (structure and function) and the relative sensitivities of those taxa. Multiple experiments conducted in mesocosms have demonstrated that the particular composition of the community at the time of exposure influences the magnitude of the impact as well as the trajectory of the recovery (Jenkins and Buikema 1998, Pesce et al. 2006, Relyea 2009, Rohr and Crumrine 2005), and this would likely be the case as well in salmonid habitats.

Abiotic conditions may also affect how herbicides directly and indirectly affect salmonids and their habitats. For instance, herbicides can affect water quality parameters that may indirectly affect aquatic communities. Austin *et al.* (1991) suggest that increased algal production in oligotrophic systems after exposure to glyphosate may be due to the addition of phosphorous (in the glyphosate), and they suggest this could lead to eutrophication of salmonid habitats. Likewise, total phosphorous increased eightfold in earthen mesocosms treated with glyphosate (Perez et al. 2007). In forested watersheds in the southeastern United States, nitrogen concentrations were elevated in streams for two years after herbicides were applied (Neary et al. 1993). This effect was likely due to the increased leaching from the terrestrial environment and/or reduction in uptake within the stream. Regardless of how nutrients become elevated (from the herbicide itself or from changes in biogeochemical cycles within the watershed) elevated nitrogen and phosphorous concentrations can stimulate periphyton growth in nutrient-limited systems and consequently affect higher trophic levels. Indirect effects from herbicides may also include an increase in stressful water temperatures due to reduced shading and long-term reductions in woody debris used for cover by salmonids from loss of riparian vegetation. If herbicides were used to reduce plant growth over a large area within a watershed, instream flow dynamics may be impacted enough to affect salmonids and their habitats (*e.g.*, (Likens et al. 1970). Finally, changes such as increased turbidity (due to reduced bank stability) or decreased dissolved oxygen could have impacts on primary producers as well as consumers within salmonid habitats (Figure 35 and Figure 36).

In addition to the uncertainties associated with variable and diverse communities and the range of sensitivities they have to various abiotic conditions, there are uncertainties about how herbicides may affect aquatic systems affected by other stressors. When experiments are used to examine multiple stressors, the results are often variable and again (like simpler experiments) often depend on the abiotic and biotic conditions at the time. In a series of experiments, Rohr et al. (Rohr et al. 2004) found few interactions among food availability, drying conditions and atrazine (at 4 concentrations) on a streamside salamander, but they did find that the lethality of atrazine varied by year and may be condition dependent. Figure 35 and Figure 36 illustrate the direct and indirect effects stemming from herbicide exposure, but they do not attempt to capture the complex web of interactions that may arise when multiple stressors affect a system.

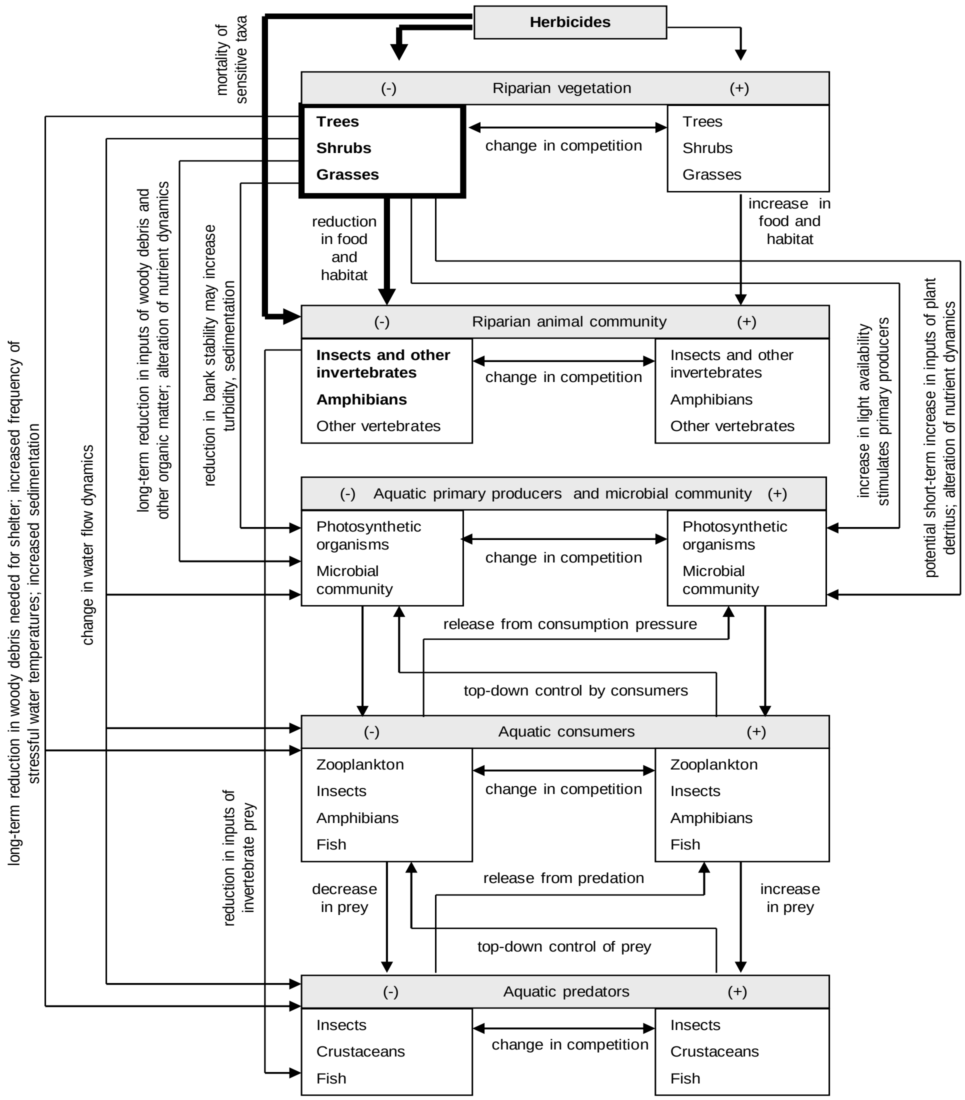


Figure 35. Part I of a conceptual model of potential effects of herbicides to aquatic communities. This figure focuses on potential effects of herbicides applied to riparian areas adjacent to salmonid habitats. Bolded arrows and text note those effects that are most likely to occur based on the frequency that they are reported in the literature.

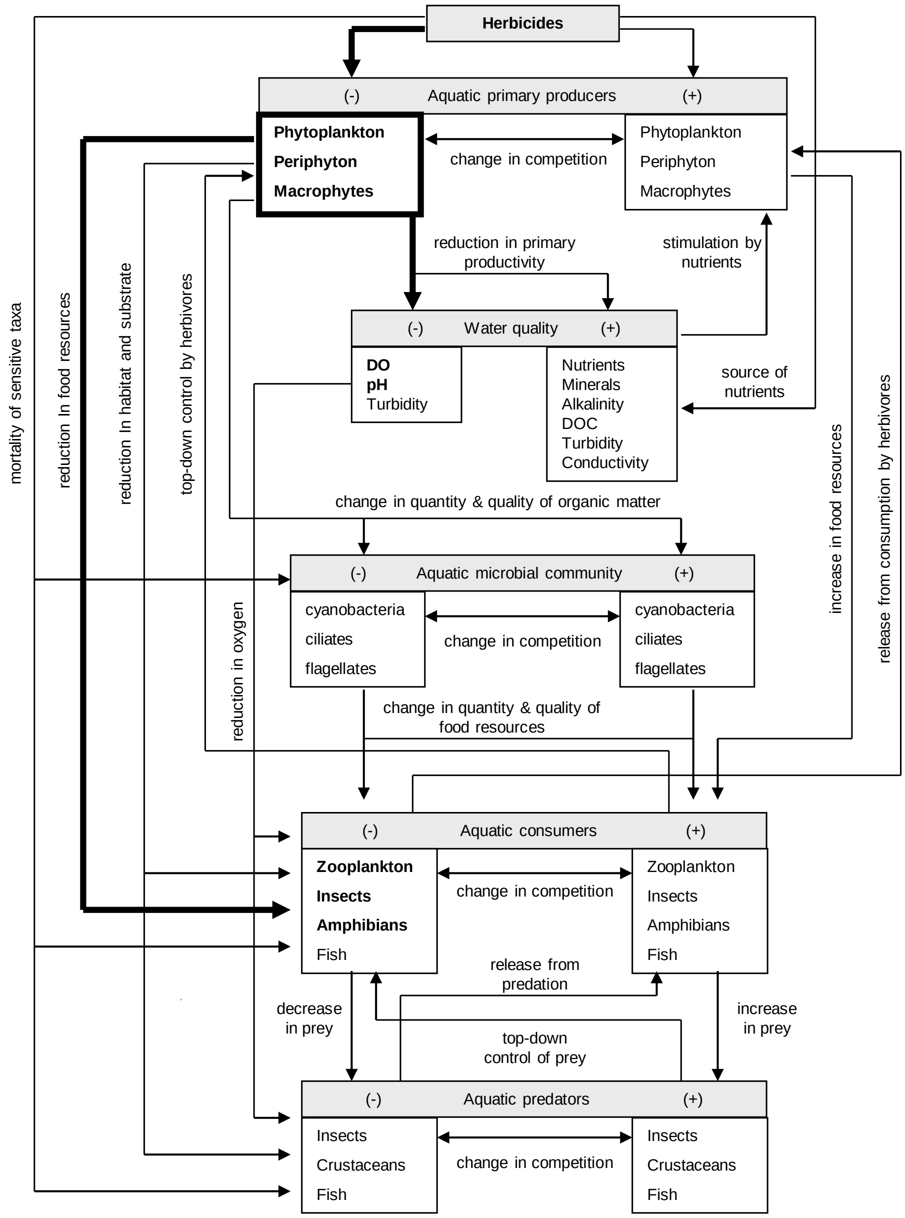


Figure 36. Part II of a conceptual model of potential effects of herbicides to aquatic communities. This figure focuses on potential effects of herbicides that are applied to or otherwise reach salmonid habitats. Bolded arrows and text note those effects that are most likely to occur based on the frequency that they are reported in the literature.

Mixtures of pesticides present a particular challenge. Most of the experiments described above were conducted in mesocosms with a single exposure of a single herbicide. In field surveys in the Unites States as well as throughout Europe, herbicides are often among the most concentrated pesticides detected, but they are almost always found in mixtures with insecticides and fungicides (Gilliom 2007, Schafer et al. 2007). Although it is becoming more apparent that herbicides are often found in mixtures, the toxicity of the herbicides within those mixtures may depend on the composition of the mixture itself. When Van den Brink et al. (2009) examined the effects of a simple herbicide-insecticide mixture on mesocosm communities, they found the herbicide (atrazine) had fewer effects than expected, and they suggested that this effect may have been due to the reduced grazing pressure that resulted directly from the reduction in invertebrates caused by the insecticide (lindane). In a series of experiments comparing effects of single herbicides, single insecticides and mixtures of these, Relyea (2009) found that a mixture of five herbicides had relatively few effects on mesocosm communities compared to several individual insecticides, a mixture of 5 insecticides and a mixture of all 10 pesticides. One effect he did find was that chlorophyll concentrations in phytoplankton were similarly reduced after 16 days in both the acetochlor-alone treatment as well as the 5-herbicide mixture treatment. This suggests acetochlor alone, and not the other four herbicides, likely contributed to the overall toxicity of the mixture for this response variable. It is unclear, however, how other communities exposed to the numerous possible combinations of mixtures would respond. Finally, in addition to the composition of the mixture, the dose of the mixture may also be important in determining the direction of effect. In a study on eelgrass, low concentrations of a mixture of three herbicides (glyphosate, benzatone, and MCPA) were synergistic but high concentrations had an antagonistic effect (Nielsen and Dahllof 2007).

A final consideration and uncertainty in how herbicides may impact salmonids and their habitats is the question of resiliency of these aquatic ecosystems. The recovery of primary and secondary production – to rates observed prior to exposure – depends on the communities themselves and the exposure. For instance, if herbicides persist in the landscape, exposures may occur repeatedly (or continuously) depending on application rate, precipitation, and conditions in the watershed. Michael et al. (2006) found exposures of sulfometuron occurred repeatedly, due to wash off from the upstream forest after a single application (see also (Michael et al. 1999, Michael 2003). The persistence of an herbicide can affect the recovery of a community, as seen when the herbicide 3,4-dichloroaniline was added to mesocosms (Maund et al. 2009). This herbicide was initially added at a concentration equal to the median LC_{50} value of taxa in the mesocosms, but it persisted several months (median dissipation time was estimated at 30 d). The lack of recovery of populations within the mesocosms by 10 months and the delay of recovery even when colonists were added following exposure was attributed to the persistent toxicity (Maund et al. 2009). Generally, photosynthesis has been found to resume rapidly once exposure stops, while indirect effects on longer-lived taxa can persist much longer (Brock et al. 2000, Brock et al. 2004). This difference can lead to dynamics in trophic interactions (*e.g.*, alterations between top-down and bottom-up control). These fluctuations have been found to stabilize in mesocosms within weeks to months, but for juvenile salmonids that require reliable food resources daily, this time period of recovery may be too long.

These uncertainties make it difficult to predict how herbicides will affect salmonids and their critical habitats, but they do not change NMFS' assessment that there may be an adverse impact.

8.5.8. Toxicity of Thiobencarb (Assessment Endpoints)

8.5.8.1. *Direct Effects to Salmonids*

We evaluate effects to salmonids based on toxicity information presented in the salmonid BE (EPA 2002a), the more recent California red-legged frog BE (EPA 2009b), REDs (EPA 1997a), ECOTOX database, and open literature.

8.5.8.2. *Survival*

Individual survival is typically measured by incidences of death following 96-hour (h) exposures (acute test) and incidences of death following 21-day (d), 30-d, 32-d, and "full life cycle" exposures (chronic tests) to a subset of freshwater and marine fish species reared and exposed in laboratories under controlled conditions (temperature, pH, light, salinity, etc.) (EPA 2004). Lethality of the pesticide is usually reported as the median lethal concentration (LC_{50}), the statistically-derived concentration sufficient to kill 50% of the test population. It is derived from the number of surviving individuals at each concentration tested following a 96 h exposure and is usually estimated by probit or logit analysis and more recently by non-linear curve fitting techniques. Ideally, to maximize the utility of a given LC_{50} study, a slope, variability around the LC_{50}, and a description of the experimental design, such as experimental concentrations tested, number of treatments and replicates used, solvent controls, etc., are needed. The slope of the observed dose-response relationship is particularly useful in interpolating incidences of death at concentrations below or above an estimated LC_{50}. The variability of an LC_{50} is usually depicted by a confidence interval (95% CI) or error (standard deviation or standard error) and is illustrative of the degree of confidence associated with a given LC_{50} estimate (*i.e*., the smaller the range of uncertainty, the higher the confidence in the estimate). Without an estimate of the variability, it is difficult to infer the precision of the estimate. Furthermore, survival experiments are of most utility when conducted with the most sensitive life stage of the listed species or a representative surrogate.

In the case of ESA-listed Pacific salmonids, there are several surrogates including hatchery reared coho salmon, Chinook salmon, steelhead, and chum salmon, as well as rainbow trout[14]. Unfortunately, slopes, estimates of variability for an LC_{50}, and experimental concentrations frequently are not reported. In our review of the BEs, we did not locate any reported slopes of dose-response curves. Consequently, to insure that EPA's action is not likely to jeopardize listed species, we must select LC_{50}s from the lower range of available studies. We evaluate the likelihood of concentrations that are expected to kill fish and apply qualitative and quantitative methods to infer population-level responses of ESA-listed salmonids within the *Risk Characterization* section. Thiobencarb (in technical products and formulations) has been tested extensively in acute lethality toxicity tests with numerous fish species (Table 51). EPA reported some of these data in Appendix I of the California red-legged frog BE (EPA 2009b).

Data were typically from registrant submitted guideline studies and open literature studies. EPA constructed a species-sensitivity distribution for acute LC_{50}s from studies they deemed acceptable in Appendix I, pg. 8 (EPA 2009b). 96 h LC_{50}s ranged from 260 µg/L for white sturgeon (*Acipenser transmontanus*) to 13,200 µg/L for the European eel (*Anguilla Anguilla*). Most data were from 96 h tests, although some were from 24 h, 48 h, and 72 h tests. Rainbow trout (LC_{50} 790 – 1,200 µg/L, n=5), steelhead (LC_{50} = 790, n=1), and Chinook salmon (LC_{50}

[14] Rainbow trout and steelhead are the same genus species (*Oncorhynchus mykiss*), with the key differentiation that steelhead migrate to the ocean while rainbow trout remain in freshwaters. Rainbow trout are therefore good toxicological surrogates for freshwater life stages of steelhead, but are less useful as surrogates for the life stages that use estuarine and ocean environments.

760, n=1) were among the more sensitive species tested based on lethality. Many LC_{50} assays on freshwater and estuarine fish species in California were conducted with thiobencarb-containing formulations to evaluate potential aquatic toxicity from rice growing operations in California's Central Valley (Harrington 1990). Additionally, three species of atherinid fishes (coastal and estuarine species) were exposed to 96 h thiobencarb in static and flow through systems at day-of-hatch, 7 d, and 14 d (Borthwick et al. 1985). Treatment levels were verified analytically. Acute sensitivity was greatest for 7-d old fish where LC_{50}s ranged from 204-464 µg/L flow through and from 396-483 µg/L static (Borthwick et al. 1985). Based on EPA toxicity categories for acute LC_{50}s, EPA classified thiobencarb as highly toxic to salmonids.

Table 51 Thiobencarb toxicity values (µg/L) for aquatic organisms and plants reported in EPA salmonid BE, CRLF BE, RED, IRED, EFED science chapter, and ECOTOX. Abbreviations as follows: a.i. = active ingredient; NR = Not Reported; T= Technical grade; F = Formulated product; sw = estuarine/marine species; [ ] = 95% Confidence interval.

Assessment Endpoint		Concentration (µg/L aquatic tests; lbs a.i./acre terrestrial tests)	
		Thiobencarb	
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)
Survival Source(s): EPA RLF 2009 Appendix I; ECOTOX # 12136; ECOTOX # 15172 EPA RLF 2009 Appendix I; MRID #s 139051, 00080851, 00080851, 00050664 Registrant submitted	salmonid LC_{50} (96 h)	Rainbow trout (*O. mykiss*) (95.5%; T) = 1150, formulation IMC-3590	Rainbow trout (*O. mykiss*) (85.2%; F) = 1200, [700-1600], Bolero 8EC formulation Rainbow trout (*O. mykiss*) (84%; F) = 1050, Bolero 8EC formulation Rainbow trout (*O. mykiss*) (10%; F) = 1500, [1200-1900] Bolero G (granular) Rainbow trout, v. Donaldson trout (*O. mykiss*) (% a.i. not reported)= 1200 Chinook Salmon (*O. tshawytscha*) (% 85.2; F) = 760; Bolero EC Steelhead (*O. mykiss*) (% 85.2; F = 790; Bolero EC Chinook salmon (*O. tshawytscha*) (85.2%; F): survival of fry NOEC = 140, LOEC = 250; Bolero 8EC; E0015472 (Faggella and Finlayson 1988)
Survival	Non-salmonid freshwater, estuarine, and	Bluegill sunfish (*L. macrochirus*) (95.5%; F) = 2480, IMC-3590	Bluegill sunfish (*Lepomis macrochirus*) (94%; T) = 2600

Assessment Endpoint		Concentration (µg/L aquatic tests; lbs a.i./acre terrestrial tests)	
		Thiobencarb	
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)
	marine fish LC_{50} (96 h)		Bluegill sunfish (*L. macrochirus*) (10%; F) = 560 [330-1200], Bolero 10G (granular) Bluegill sunfish (*L. macrochirus*) (84%; F) = 1660, Bolero 8EC Bluegill sunfish (*L. macrochirus*) (85.2%; F) = 1700 [1200-2300], Bolero 8EC Striped bass (*Morone saxatilis*) (sw) (85.2%; F): Mean = 731; median = 760; range = 430-1000; n=17; Bolero 8EC, ages 7 – 45 post emergence; Bolero 8EC; Fujimura et al. 1991 White sturgeon (*Acipenser transmontanus*) (T) = 260 [230-300], Bailey 1984 MRID #40651315
Reproduction or larval survival	NOEC/LOEC	Fathead minnow (*Pimephales promelas*) (96.5%; T) = 53/110; 2 replicates/treatment only; lifecycle test (260 day exposure); Endpoints affected: survival, growth of F0 generation, # eggs per spawn, hatching success, growth, survival F1 generation MRID 45695101 Sheepshead minnow (C. variegates) (sw) (95.2 %; T) = <150/150 28 d post hatch survival 370/600 hatching success MRID 00079112	
Fish growth and development	NOEC/LOEC	Fathead minnow (*Pimephales promelas*) (96.5%; T) = 53/110; 2 replicates/ treatment; lifecycle test (260 day exposure); Endpoints affected: survival, growth of F0 generation, # eggs per spawn, hatching success, growth, survival F1 generation MRID 45695101 Zebrafish (*Danio rerio)*(96.5%; T) = 800 LOEC, 96 h exposure, 100% abnormality and reduced length Appendix 5, (NMFS 2011c)	Chinook salmon (*O. tshawytscha*) (85.2%; F) = 28/ 49; survival of fry 140/ 250 Bolero 8EC; 88 day exposure 28 day pre hatch through 60 day post hatch. E0015472 Fujimura et al 1991 Striped bass (*M. saxatilis*) (sw) (85.5%; F): NOEC=21, LOEC=36; NOEC=<23, LOEC=23; NOEC=58, LOEC= 91 (n=3); Bolero 8EC, reduced dry weight, early life stage study. Fujimura et al 1991

Assessment Endpoint		Concentration (μg/L aquatic tests; lbs a.i./acre terrestrial tests)	
		Thiobencarb	
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)
Fish respiration	gill damage (micropathological assessment)		Mosquitofish (*Gambusia affinis)*, (Bolero 8EC[15]; formulation): @ 19, 90, 576, 1200, 1800 μg/L Bolero8EC inflamed gills, dose dependent response
Habitat-salmonid prey	invertebrate survival (48 h EC/LC_{50})	Amphipod (*Gammarus pseudolimnaeus*) (95.5%; F) = 720, IMC-3590 Midge (*Chironomus tentans*) (97.2%;T) = 364 [322-413], probit slope = 3.42 [2.88-4.16]; Survival NOEC = 135 Opossum shrimp (*Neomysis mercedis*) (sw) (NR; T): 96 h LC_{50} = 304; 7 d LC_{50} = 214; 14 d LC_{50} = 91 (Bailey 1993)	Water flea (*Daphnia magna*) (94.4%) = 101.2 [73.8 – 138.7] Water flea (*D. magna*) (82.25%; F) = 210.7 [175.7 – 252.7] Bolero 8EC Water flea (*D. magna*) (85.2%; F) = 1200 [400-3100] Bolero 8EC Amphipod (*G. pseudolimnaeus*) (85.2%; F) = 1000 [600-1700], Bolero 8EC Amphipod (*G. pseudolimnaeus*) (84%; F) = 1000, Bolero 8EC Midge (*C. tepperi*) (% not reported,; F) = 188 (LOEC emergence); 375 (LOEC developmental time, wing length) (Burdett et al. 2001)
	Invertebrate reproduction NOEC/LOEC (21 d life-cycle test)	Water flea (*D. magna*) (96.9%; T) = 48.0/ 90.0; # of offspring produced; MRID 00079098 Water flea (*D. magna*) (96.2%; T) = na/ 38; # of offspring produced; MRID 41636101[16] Water flea (*D. magna*) (95.2%; T) = 1.0/ 3.0; # of offspring produced; MRID 241483 Midge (*Chironomus riparius*) (97.2%; T) = 180/ 420; 28 day sediment toxicity assay Opossum shrimp (*N. mercedis*) (sw) (NR; T): 3.2/6.2 (Bailey 1993) Mysid shrimp (*Mysidopsis bahia*) (sw)	

[15] Thiobencarb per cent not reported in paper. Other Bolero 8EC studies report 85.2% thiobencarb.

[16] MRID 41636101, McNamara, P. (1990) Bolero Technical: The Chronic Toxicity to *Daphnia magna* under Flow-Through Conditions: Lab Project Number: 90- 8-3444. Unpublished study prepared by Springborn Labs, Inc. 80 p. The experiment showed effects at the lowest concentration tested, 38 ug/L, therefore a NOEC cannot be determined.

Assessment Endpoint		Concentration (μg/L aquatic tests; lbs a.i./acre terrestrial tests)	
		Thiobencarb	
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)
		(95.1%): NOEC not determined, EC_{05} = 9.8; survival of offspring[17]	
Habitat: Riparian Vegetation MRID 41690902 Hoberg, J.R. 1990 Sup	Vegetative vigor (lbs a.i./acre):	Monocots (96.6%; T) EC_{25}: corn = 2.2, oat = 0.17, onion = 1.2, ryegrass = 0.073; NOEC: corn = 2.2, oat = 0.12, onion = 0.80, ryegrass = 0.02 Dicots (96.6%; T): EC_{25}: carrot = > 2.2, cabbage = 1.2, cucumber = na, lettuce = 1.3, soybean = 1.2, tomato = 1.8; NOEC: carrot = 2.2, cabbage = 1.4, cucumber = <0.12, lettuce = 0.80, soybean = 0.80, tomato = 2.2	
Habitat: Riparian Vegetation Sources: MRID 41690902 Hoberg, J.R. 1990; MRID 44846201 Chetram, R.S. 1999 Acceptable DER 11/16/2002	Seedling emergence (lbs a.i./acre): shoot length or mortality EC25	Monocots (96.6%; T): -Shoot length EC25: corn = >1.7, oat = 0.086, onion = 2.0; NOEC: corn =1.7, oat = 0.055, onion 0.94; -Mortality (rye grass) EC_{25} = 0.019; NOEC = 0.0051 (17% mortality at 0.011) Dicots (96/6%; T): -shoot length EC25: carrot = >3.1, cabbage = 0.082, cucumber = >1.7, soybean = >1.7, tomato = 1.1; NOEC: carrot = 2.1, cabbage = 0.071, cucumber = 0.16 soybean = 0.94, tomato = 0.94. -mortality (lettuce); EC_{25} = 0.27	
Habitat: In-stream Primary Productivity Source MRID 41690901 Giddings, 1990 DER 09/18/1995	Aquatic plant growth: cell density EC_{50} & NOEC	Green algae (*Selenastrum capricornutum*) (96.6%; T) 120 hr EC_{50} = 17 [12-26], NOEC = 13 Freshwater diatom (*Navicula pelliculosa*) (96.6%; T) 120 hr EC_{50} = 380 [240-610]; NOEC = 65 Duckweed (*Lemna gibba*) (96.6%; T) 14 day frond production EC_{50} = 770 [380-1600]; NOEC = 140	

[17] EPA RED for thiobencarb concluded that a NOEC could not be determined because control had no replication. A nonlinear regression analysis was used by EPA to calculate an EC_{05} which was used in lieu of the NOEC.

Assessment Endpoint		Concentration (µg/L aquatic tests; lbs a.i./acre terrestrial tests)	
		Thiobencarb	
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)
		Green algae (*Scenedesmus acutus*) (100%; analytical standard) 96 hr EC_{50} = 17 [16-19] (Sabater and Carrasco 1996) Green algae (*Chlorella saccharophila*) (100%; analytical standard) 96 hr EC_{50} = 4000 [3800-4100] (Sabater and Carrasco 1996) Algae (*Pseudanabaena galeata*) (100%; analytical standard) 96 hr EC_{50} = 370 [350-400] (Sabater and Carrasco 1996)	

A species sensitivity distribution based on reported 96 hr LC_{50}s shows the position of salmonids in relation to other species tested

). The distribution contains those studies that EPA reviewed and ranked as acceptable, therefore not all fish LC_{50}s are included. Salmonids are at the lower end of the distribution indicating they are more sensitive than many of the other species tested based on 96 hr lethality assays. Based on the confidence bands (shown in gray) around the means, substantial variation exists between and among species.

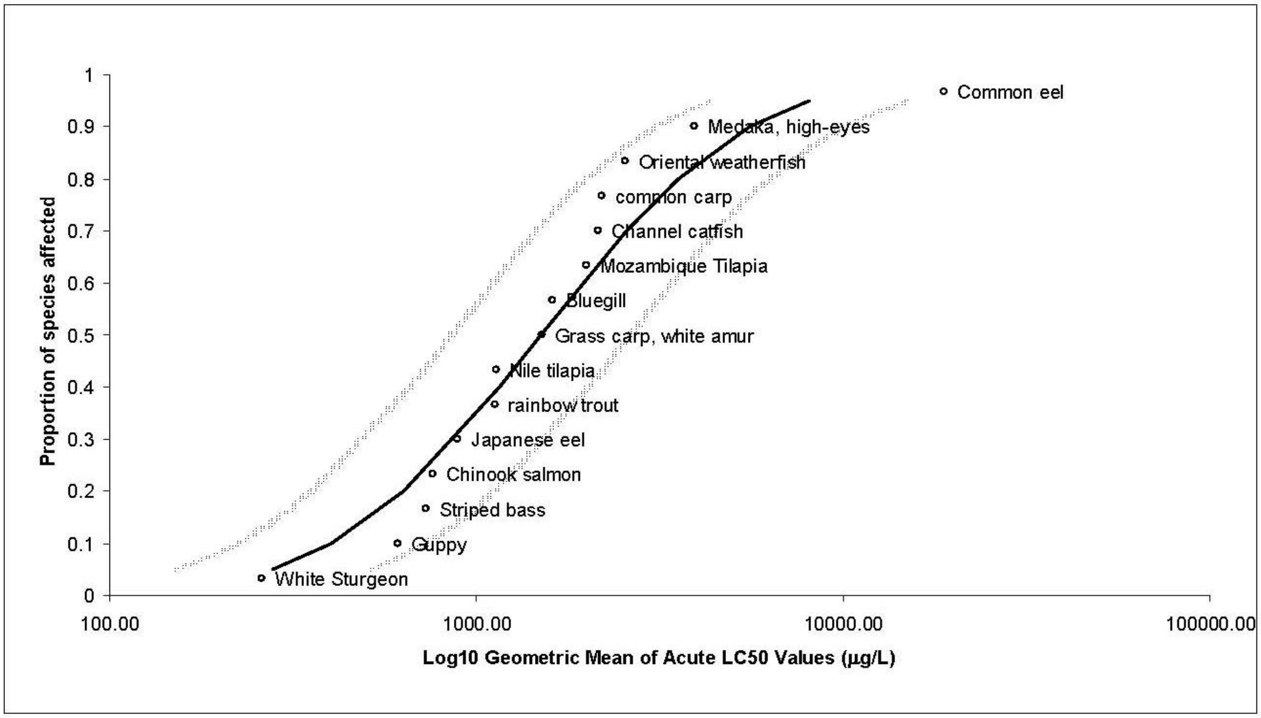


Figure 37. Species sensitivity distribution of freshwater fish 96 h LC50s derived from EPA denoted "acceptable" studies[18]

8.5.8.3. *Growth and Reproduction*

Growth and reproduction endpoints are typically evaluated in FIFRA guideline tests conducted by the chemical company registrants. In these tests, fish are exposed to the a.i. for variable durations depending on species tested. Fish are fed twice daily, *ad libitum* (*i.e.*, an over abundance of food is available at time of feeding). The lowest concentration eliciting a statistically significant difference from controls (no treatment) to growth or reproductive endpoints is recorded (*i.e.*, the Lowest Observable Effect Concentration (LOEC)), as well as the lowest exposure concentration tested that is not different than the control (*i.e.*, the No Observable Effect Concentration (NOEC)). Many researchers have commented on the poor application of environmental statistics and laboratory testing regarding NOECS and LOECs (Laskowski 1995, Chapman 1996, Kooijman 1996), (Suter 1996) and (Landis and Chapman 2011). Prominent limitations include: (1) NOECs and LOECs are statistically derived, a function of the

[18] EPA developed this SSD using the Species Sensitivity Generator v.1.0, Causal Analysis/Diagnosis Decision Information System (CADDIS), EPA 2009a. Plot shows curve fit with bounds in gray, however no information was provided on how curve fit was done or what bounds represent.

concentrations selected by the experimenters, and are inconsistent between studies; and (2) NOECs and LOECs ignore the fundamental model of toxicology; (3) ignore critical data at other treatments; (4) use a lack of evidence as a no-effect; and (5) are limited to the concentrations tested. NOECs typically correspond to an EC_{10} to EC_{30} on an exposure-response curve (Moore and Caux 1997). A 30% affect rate within a population can be striking, particularly if the effect is on a critical biological endpoint such as reproduction, growth, migration, or olfaction. Previous salmonid population modeling suggests that when 14% mortality occurs to juveniles population growth rate is substantially affected (NMFS 2009e) . We therefore exercise caution in interpreting a NOEC as a true “no response” to an organism.

Growth of individual organisms is an assessment endpoint derived from chronic fish and invertebrate toxicity tests summarized in the BE. Reproduction, at the scale of an individual, can be measured by the number of offspring per female (fecundity), and at the population scale by measuring the number of offspring per female in a population over multiple generations. The BEs summarized reproductive endpoints at the individual scale from chronic, freshwater fish experiments. Other assessment measures of reproduction include egg size, spawning success, sperm and egg viability, gonadal development, and hormone levels--most of which are rarely measured in standardized toxicity tests conducted pursuant to pesticide registration.

8.5.8.4. *Respiratory Effects*

Gills are a well known target of toxic insult in fish and respond quickly to degraded environmental conditions (Poleksic and Mitrovic-Tutundzic 1994). Following 7-d static exposures to Bolero 8EC at concentrations ranging from 19 – 1900 µg/L (~16.2-1620 µg/L thiobencarb; based on 85.2% by volume), gills of mosquitofish were inflamed (Persichino et al. 1998). Upon examination by light microscopy, the inflammatory response was characterized by hypertrophy of respiratory epithelium, hyperplasia of mucous cells, and empty mucous cells (Persichino et al. 1998). At concentrations of 1200 µg/L and above, mosquitofish died. At 19 µg/L Bolero 8EC, two of three fish perished due to combination of parasites and Bolero—no parasites were observed in control fish. Limitations in applying these experimental results include lack of verification of treatment concentrations by analytical chemistry methods, small treatment size, n=3, and no replication. Although the study's limitations reduce its significance, the data do suggest adverse responses in fish gills which may be an additional mechanism of action.

8.5.8.5. *Acetylcholinesterase Inhibiting Effects*

We located no salmonid studies that tested thiobencarb exposure on AChE activity, thus we broadened the search to other fish species. A suite of experiments evaluated the effects of thiobencarb on European eels, also referred to as yellow eels (Fernandez-Vega et al. 1999, Fernandez-Vega et al. 2002, Bretaud et al. 2001, Sancho et al. 2000, Babu et al. 1989).

Eels were fairly insensitive to thiobencarb's capacity to kill them over 96 h, LC_{50}s ranged from 13.3 – 25 mg/L (Fernandez-Vega et al. 1999). Eel nervous systems were much more sensitive and experienced adverse effects at 1/60th of the LC_{50}, a concentration of 220 µg/L. Juvenile eels exposed for 96 h to thiobencarb in flow through conditions (continuous exposure) had more than a 50% reduction in their blood AChE activity. Statistically significant inhibition occurred at the first time point measured, 2 h, and continued through the 96 h. Normal activity returned at 72 h post-exposure indicating, similar to other carbamates, that AChE activity is reversible following exposure to carbamates. Eels also showed signs of lethargy and agitation during the experiment, common symptoms of nervous system toxicants. Brain, muscle, and gill AChE activities were inhibited by 30 – 50% following 96 h of continuous exposure (Fernandez-Vega et al. 2002). Recovery occurred within eight days for brain and muscle activities, while gill AChE activity had minimal recovery of < 29 %. Eels also exhibited behavioral impacts including agitation, loss of equilibrium, increased respiratory rhythm, uncoordinated swimming movements, and many remained at the bottom of exposure tanks. These aberrant behaviors were not observed in control exposures.

AChE activity in eel eyes was inhibited by thiobencarb at 220 µg/L (Sancho et al. 2000). Following 220 µg/L exposures (1/60th of the 96 hr LC_{50}) at 2, 12, 24, 48, 72, and 96 h, total eye AChE and specific eye AChE were measured. Eye AChE was inhibited in a time-dependent manner with onset of inhibition occurring at 2 h and remaining throughout the exposure period. Inhibition occurred at 2 h and did not return to its normal levels over the six day recovery period in clean water. Maximal inhibition occurred at 12 h with more than 80% of activity inhibited. Eels were lethargic, hypoactive, and exhibited reduced opercular movements. Following a review of these findings, we found several limitations including the selection of the eye as a target for AChE inhibition, the methodology used to detect cholinesterase activity, the levels of activity within the eye, reported standard deviations of 0.00, and the apparent lack of a positive control. In the vertebrate retina, acetylcholine is used as the neurotransmitter for only a subset of one class of neurons, the amacrine cells. The majority of the neurons (e.g., rods, cones, horizontal, bipolar) use other neurotransmitters such as glutamate, gamma aminobutyric acid (GABA), and glycine. Acetylcholine is likely the neurotransmitter used to control the muscles that control the iris. Thus, only a small portion of the eye is likely to require AChE, making the consequence of AChE inhibition to overall eye function difficult to assess and likely subtle. The method used to quantify AChE detects multiple types of cholinesterase activity. For example, the method does not distinguish between AChE and butyrylcholinesterase. The authors do not discuss the possibility that much of the inhibition they detect by thiobencarb may be occurring to an enzyme other than AChE. Additionally, many of the mean values for specific activity in the eye have a standard deviation of 0.00. It is difficult to believe that there was zero variability around the means. The results of this study do show an inhibitory effect (reduction in AChE activity) of thiobencarb on eel eyes, however further research would be needed to confirm these results. Due to these limitations, we have low confidence in these results.

As no studies were found on thiobencarb's effect on salmonids, we broadened our search to other thiocarbamates including EPTC, molinate, pebulate, trillate, butylate, and cycloate. Unfortunately, no additional empirical data were located on AChE inhibition in salmonids or other fish.

These studies showed that (1) biochemical endpoints (AChE inhibition in blood, brain, muscle, gill and eyes) were inhibited in eels at concentrations well below the 96 hour LC_{50}, at $1/60^{th}$ of eel LC_{50}; (2) adverse behavioral changes (lethargy) occurred at that concentration, (3) effects occurred within a couple of hours after exposure; and (4) recovery of AChE activity is tissue dependent (minimal recovery seen in whole eyes over a six day period and recovery in plasma activity occurring in three days). Although thiobencarb exposure to 220 µg/L would not kill eels outright, it could cause disorientation, loss of visual competence, and loss of locomotion within a few hours of exposure, which may lead to predation, reduced foraging, and impaired migration. We recognize that listed salmonids' AChE activity following exposure to thiobencarb is an uncertainty given that no experiments were identified that tested this response. We expect that thiobencarb will inhibit AChE, although the concentration that would induce this effect remains unknown. European eels, the only fish species we have AChE activity toxicity data for, are among the most tolerant of the various fish species for which LC_{50}s are available. If the 1/60 factor is presumed to apply to salmonid species, similar effects could be expected at roughly 12–19 µg/L, based on the median LC_{50} values for Chinook salmon and rainbow trout (Table 51).

8.5.8.6. *Swimming*

Swimming is a critical function for anadromous salmonids necessary to complete their lifecycle. Impairment of swimming may affect feeding, migrating, predator avoidance, and spawning. It is the most frequently assessed behavioral response of toxicity investigations with fish (Little and Finger 1990). Swimming activity and swimming capacity of salmonids have been measured following exposures to a variety of AChE-inhibiting insecticides including the OPs chlorpyrifos, diazinon, and malathion (reviewed in NMFS 2008) and the carbamates carbaryl, methomyl, and carbofuran (reviewed in NMFS 2009). Swimming capacity is a measure of orientation to flow as well as the physical capacity to swim against it (Howard 1975, Dodson and Mayfield 1979). Swimming activity includes measurements of frequency and duration of movements, speed and distance traveled, frequency and angle of turns, position in the water column, and form and pattern of swimming. A review paper published in 1990 summarized many of the experimental swimming behavioral studies and concluded that effects to swimming activity generally occur at lower concentrations than effects to swimming capacity (Little and Finger 1990). Therefore, measurements of swimming activity are usually more sensitive than measurements of swimming capacity. A likely reason is that fishes with impaired swimming to the degree that they cannot orient to flow or maintain position in the water column are moribund (*i.e.*, death is imminent). The authors of the review also concluded that swimming-mediated behaviors are frequently adversely affected at 0.3 – 5.0 % of reported fish LC_{50}s, and that 75% of reported adverse effects to swimming occurred at concentrations lower than reported LC_{50}s. Both swimming activity and swimming capacity are adversely affected by AChE inhibiting insecticides. We located no studies that measured impacts to salmonid swimming behaviors from exposure to thiobencarb. Several studies on eels (discussed previously) showed AChE inhibition at 1/60th of the LC_{50} (200 µg/L) and observed that affected fish were lethargic, hypoactive, and exhibited reduced opercular movements, all classic symptoms of AChE poisoning.

8.5.9. Indirect Effects to Salmonids (Prey effects and Habitat Modifications)

Indirect effects on salmon from exposure to thiobencarb include reductions in prey base (aquatic invertebrates), disruptions in primary productivity in salmon habitats (phytoplankton and macrophytes), and effects on riparian vegetation (reduced cover and increased sedimentation).

8.5.9.1. *Effects to Salmonid prey*

Prey Survival. Data from several freshwater and marine/estuarine aquatic invertebrates are available to determine their sensitivity to thiobencarb and presented in Table 51. Ranges in survival EC50s were observed for both freshwater species (~100-1200 µg/L, Table 51) and marine/estuarine species (150 – 4400 µg/L19). Several of the freshwater species tested are known salmonid prey including water fleas and amphipods. We located few studies that tested native macro-invertebrates to CA's Central Valley. This is a notable data gap for effects to salmonid prey following exposure to thiobencarb.

Reproduction and Growth. We located four laboratory studies on the effects of thiobencarb to two freshwater species, a water flea and a chironomid, both of which are fed upon by young salmonids. The FIFRA guideline protocol for a 21-d chronic test on reproduction and growth was carried out in three separate studies with *D. magna*. The number of offspring was reduced by thiobencarb at 3, 38 and 90 µg/L (LOECs) with corresponding NOECs of 1, not determined, and 48 µg/L in the three studies, respectively. In the study with a LOEC of 38 ug/L (MRID 41636101), the NOEC could not be determined because the lowest concentration tested, 38 ug/L,

[19] Individual values obtained from EPA Registration Eligibility Decision, page 43, Table 17.EPA 1996. 'Ecological Effects Test Guidelines OPPTS 850.4000, Background -Nontarget Plant Testing.' in *Ecological Effects Test Guidelines OPPTS 850.4000, Background -Nontarget Plant Testing*, 15.

elicited a statistically significant effect. Growth endpoints were not reported in EPA documents. The limitations with NOECs were discussed previously and should be used with caution as typically they are not a true “no effect” concentration. Thus we expect some level of effect at 1 and 48 µg/L for water flea reproduction. In the chironomid study, sediments were spiked with thiobencarb and chironomids were exposed to contaminated sediments for 28 days. The treatment levels for aqueous thiobencarb representing the NOEC and LOEC were 180 and 420 µg/L, respectively (Table 51). Other studies summarizing study designs and results with aquatic species can be found in Table 52.

Table 52 Study designs and results with aquatic species

Taxa/species	Assessment measures	Concentrations tested (µg/L)	Exposure duration	Effects	Data source
Freshwater pond invertebrate communities	Species richness and abundance of groups: Corixidae, Ostracoda, larval Hydrophilidae, larval Dytiscidae, adult Dytiscidae, chironomids, ceratopogonids, calanoids, cyclopoids	Sprayed once at 3.2 lbs/acre (3600 g/HA) = 2700 µg/L; 3 replicates	5 weeks; sampled at 1 and 5 wks	@ 1 wk reduced abundance of chironomids, calanoids, and cyclopoids; @ 5 wks highest and lowest species richness and abundance in thiobencarb treatments	(Burdett et al. 2001)
Americamysis bahia (mysid)	Life cycle test: life stage-specific survival and reproduction; Modeled population impacts	0, 22, 35, 76, 181, 345, 734	28 days	22 µg/L = no effects; 35 µg/l = delayed reproduction; 76 µg/L=reduced overall reproduction; 181 µg/L = reduced overall reproduction; 345 µg/L = reduced overall reproduction and survival; 734 µg/L= reduced overall reproduction and survival; Population modeling results: @ 76 µg/L and above population declines anticipated	(Raimondo and McKenney 2006)

Taxa/species	Assessment measures	Concentrations tested (µg/L)	Exposure duration	Effects	Data source
Ceriodaphnia dubia (daphnid)	48 h survival tests with field collected water toxicity identification evaluation (TIE);	No thiobencarb detected in field-collected water, TIE experiments; Methylparathion, carbofuran, molinate detected	48 h	No reported effects from thiobencarb as none detected in samples. Carbofuran and methyl parathion explained toxicity	(Norberg-King et al. 1991)
	24 h & 48 h survival EC50 with spiked laboratory water	Treatment concentration not reported	24 & 48 h exposures	24 h survival EC_{50} = 580 µg/L (95%CI 430-790) 48 h survival EC_{50} = 510 µg/L (95% CI 400-650)	

8.5.9.2. *Aquatic Plants (Phytoplankton and Vascular Plants)*

As an herbicide, thiobencarb may impair growth or kill aquatic and riparian plants within salmon habitats. Table 51 summarizes toxicity data for aquatic primary producers including various types of algal species and vascular plants. Very few data were identified and discussed by EPA regarding the effects of thiobencarb on salmonid habitats. The available information is sparse, non-specific, and difficult to translate to habitat assessment endpoints. For example, phytoplankton provides energy to aquatic systems, while macrophytes provide structural components such as attachment sites for aquatic invertebrates and refugia for salmonids. Data from laboratory bioassays on one species of algae exposed to thiobencarb fail to encompass the complexity of aquatic systems where hundreds of algae make up but one component of a food web. We located five studies on five species of algae that report reductions in growth as measured by cell density. The EC_{50}s (derived from 48-120 h exposures) ranged from 17- 4000 µg/L, and include three species green algae, a cyanobacteria, and a freshwater diatom.

Based on these data, species of green algae were the most and least sensitive (*i.e.*, 17 µg/L and 4000 µg/L). Algae serve as important food for zooplankton and early life stages of aquatic invertebrates. If these test species are representative of aquatic systems, then reductions in primary productivity or modifications in community structure via removal of sensitive species may result in “bottom-up” trophic cascades which may adversely affect salmonids. It is difficult to determine the level of effect as numerous site and species characteristics drive community responses.

Loss of structure provided by macrophytes may result in decreased populations of aquatic invertebrates or increased predation on juvenile salmonids. One experimental result is available with the aquatic vascular plant, duck weed (*Lemna gibba*), that addresses loss of macrophytes within aquatic systems. Thiobencarb is predicted to reduce frond production by 50% at 14 days following an exposure of 770 µg/L with a NOEC of 170 µg/L. We located no other studies with aquatic plants. This is a significant data gap given the importance of aquatic plants to healthy salmonid habitats.

8.5.9.3. *Effects on Riparian Vegetation*

Riparian vegetation is important for providing shade to the stream, stabilizing the stream banks, reducing sedimentation, and providing organic material inputs, both in terms of plant material and terrestrial insects. Riparian vegetation is a major focus of restoration efforts within California, and when present can reduce pesticide loading into aquatic resources. Riparian vegetation is an important assessment endpoint for herbicidal impacts on salmon habitats. Generally there are sparse data regarding the effects of herbicides on wild plants within riparian systems, other than weed species. EPA requires submission of crop effects data as part of the registration process (EPA 1996). This information currently provides the only basis for evaluating effects on herbaceous plants unless data are available from other sources. Registrant-submitted guideline studies report vegetative vigor and seedling emergence EC_{25}s for thiobencarb end-use products (EPA 1996). The overall assumption is that the sensitivity of plant species tested (typically plants used in agriculture) in the guideline studies will be representative of riparian species. There is no way to know this is the case, therefore a high degree of uncertainty regarding the toxicity of thiobencarb to riparian vegetation exists. We note that a Science Advisory Panel in 2001 (now more than 11 years ago) was convened by EPA to review non-target plant toxicity tests under the North American Free Trade Agreement. A host of recommendations were made as well as several research recommendations to fill identified data gaps. We were not able to confirm if any of the recommendations or research initiatives were undertaken although if implemented several would have informed the current consultations with effects on thiobencarb to aquatic and riparian plants (EPA 2002b).

We calculated medians for the range of crop data presented in the EPA assessments, and these are presented in Table 51. Overall, tested monocots (grasses) appear more sensitive than dicots (broadleaf plants) to thiobencarb. For monocots, the seedling emergence endpoint is more sensitive than the vegetative vigor based on NOECs, but based on EC_{25}s, the sensitivity of the two endpoints is similar. Thiobencarb affected terrestrial crops at concentrations as low as 0.019 lbs/acre indicating high sensitivity of some plants. Vegetative vigor, shoot length, mortality, and seedling emergence were all affected at application rates well below authorized label rates. In the tier II seedling emergence test, mortality of test plants occurred in the tests with ryegrass and lettuce. Mortality was the most sensitive toxic endpoint for these species (plants tended to die shortly after emerging). The most sensitive species was ryegrass, a monocot, for which the LC_{25} was 0.019 lb thiobencarb/acre. The most sensitive dicot was cabbage (EC_{25} based on shoot length was estimated to be 0.082 lb thiobencarb/acre). Based on the results of the terrestrial plant toxicity tests, it appears seedling emergence is the most sensitive endpoint for both dicots and monocots. No studies were located on the effects of thiobencarb on riparian vegetation, a significant uncertainty. Additionally, no information was located on effects to established riparian vegetation.

8.5.10. Degradate Toxicity

Two "major"[20] degradates were identified in EPA's California red legged Frog BE, 4-chlorobenzoic acid and 4-chlorobenzaldehyde, for which EPA located no toxicity data based on an ECOTOX database search. Therefore, EPA estimated potential effects using a structure activity relationship model, ECOSAR version 1.0. The model predicted aquatic toxicity concentrations for eight degradates: 4-chlorobenzoic acid, 4-chlorobenzaldehyde, thiobencarb

[20] degradates are considered "major" by EPA if the degradate makes up more than 10% of the parent active ingredient applied following standardized, registrant-submitted laboratory degradation studies.

sulfoxide, 2-hydroxythiobencarb, 4-chlorobenzylmethylsulfone, 4-chlorobenzylmethylsulfoxide, bencarb, and desmethyl thiobencarb.

The ECOSAR model estimated that the two major degradates, 4-chlorobenzoic acid and 4-chlorobenzaldehyde, were less toxic than thiobencarb. As none of the other degradates assessed by ECOSAR were considered "major" degradates, EPA assumed that degradate toxicity would be minimal. Overall EPA concluded that none of the degradates would alter its effect determinations for the California Red-legged Frog (EPA 2009b).

Thiobencarb's lowest empirical 96-hr LC_{50} for freshwater fish (white sturgeon) is 260 µg/L, while the ECOSAR model predicted LC_{50}s of 2.9 mg/L (an order of magnitude less toxic) suggesting that the model under predicts toxicity. The model was also used to predict freshwater fish 96-hr LC_{50}s for the two major degradates, 421.7 mg/L for 4-chlorobenzoic acid, and 5.5 mg/L for 4-chlorobenzaldehyde. The lowest predicted degradate 96-hr LC_{50} was 1.6 mg/L for bencarb. It is difficult to determine what significance should be placed on EPA's toxicity modeling results given the lack of discussion of the non-major degradates and the absence of a discussion with in the BE. Bencarb, a carbamate ester, was consistently the most toxic degradate compared to the other degradates across aquatic invertebrate and fish endpoints that were modeled using ECOSAR (Appendix B, (EPA 2009b)). Bencarb achieved levels just below 10% in degradation studies, *i.e.*, 8.3% at 21 days and 8.1% at thirty days in a photolysis study. Chemical fate and persistence data were not provided. These data indicate that bencarb may be of aquatic concern to salmonids and their habitat, however definitive conclusions on its toxicity is complicated by the lack of empirical information.

8.5.11. Tank Mixtures

Thiobencarb is often co-applied with other pesticides, including products that contain propanil, copper sulfate, and many others (Appendix 4). Several thiobencarb labels specifically recommend co-application with propanil. Propanil is an herbicide with a wide range of acute LC_{50}s for freshwater fish (2.1– 12.7 mg/L) and invertebrates (1.2 – 16 mg/L) (EPA 2009a). Rainbow trout are the most sensitive freshwater fish tested with an LC_{50} of 2.1 mg/L. Propanil is highly toxic to aquatic (EC_{50} 0.016 -0.11mg/L) and terrestrial primary producers (EC_{25} 0.09-12 lbs propanil/ acre). Propanil's degradate, 3,4-DCA, is 15-fold more toxic than propanil based on *D. magna* EC_{50} survival test (48-h EC_{50} = 0.528 µg/L parent; 0.035 mg/L degradate). We located no fish toxicity studies with 3,4-DCA and no studies on the potential combinatorial toxicity of thiobencarb and propanil.

While labels do not specifically recommend co-application of thiobencarb and copper sulfate products, such mixtures are not prohibited and they are co-applied as tank mixes. In California, pesticide use report data indicate approximately 7% of all sites that receive thiobencarb applications are also treated with copper sulfate on the same day (Appendix 4). Dissolved copper is highly toxic to fish and their invertebrate prey, particularly the olfactory sensory system of salmonids (Baldwin et al. 2003a, Baldwin et al. 2011, De Boeck et al. 1997, Hansen et al. 1999, Hansen et al. 2002, Linbo et al. 2006, Linbo et al. 2009, McIntyre et al. 2008, Mebane and Arthaud 2010, Sandahl et al. 2004, Tierney et al. 2006, van der Geest et al. 2000, van der Geest et al. 2002). Olfaction conveys critical environmental information that fishes use to mate, locate food, discriminate kin, avoid predators, and home. Any or all of these essential olfactory-mediated behaviors may be affected by exposure to copper. Numerous studies spanning several species have shown ecologically relevant exposures to copper can interfere with fish olfaction, disrupting life history processes that determine individual survival and reproductive success. For example, copper impairs and destroys salmonid olfactory sensory neurons in a matter of minutes at low µg/L levels and effects persist for hours to weeks depending on exposure concentration and duration (Baldwin et al. 2003b). Measured behavioral effects in salmonids from impaired olfaction include compromised alarm response, loss of ability to avoid copper, interrupted spawning migrations, loss of homing ability, and delayed and reduced downstream migration of juveniles (Baldwin et al. 2003b, Baldwin et al. 2011, Hansen et al. 1999, McIntyre et al. 2008, Mebane and Arthaud 2010, Sandahl et al. 2004). Disruption of these essential behaviors reduces the likelihood of an individual salmonid completing its life cycle. If sufficient numbers of individuals are compromised, a population's abundance and productivity can be reduced.

The effect of environmental mixtures of thiobencarb with propanil and copper sulfate is uncertain as toxicity information assessing these mixtures was not located. Additionally the type of mixture toxicity is difficult to predict as these compounds do not share a mechanism of action.

Given that propanil is an authorized tank mix and copper sulfate is frequently applied the same day to the same area, we conclude that impacts of such mixtures to aquatic plant communities will be greater than the impact of thiobencarb alone. Additionally, laboratory studies suggest propanil and copper are more toxic to a broader range of aquatic flora and fauna than thiobencarb suggesting that in combination, community effects may be increased compared to thiobencarb alone.

8.5.12. Adjuvant Toxicity

Although no data were provided in the EPA available documents related to adjuvant toxicity, an abundance of toxicity information is available on the effects of the alkylphenol polyethoxylates, a family of non-ionic surfactants used extensively in combination with pesticides as dispersing agents, detergents, emulsifiers, adjuvants, and solubilizers (Xie et al. 2005). Two types of alkylphenol polyethoxylates, nonylphenol (NP) ethoxylates and octylphenol ethoxylates, degrade in aquatic environments to the more persistent, toxic, and bioaccumulative degradates NP and octylphenol, respectively. We did not receive information on the presence or absence of alkylphenol polyethoxylates in thiobencarb-containing formulations. Adjuvants are frequently mixed with formulations prior to applications, so although they may not be present in the formulations they could still be co-applied. Below we discuss NP's toxicity as an example of potential adjuvant toxicity, as we received no information on adjuvant use or toxicity within the BEs provided by EPA.

We queried EPA's ECOTOX online database and retrieved 707 records of NP's acute toxicity to freshwater and saltwater species. The lowest reported LC_{50} for salmonids in ECOTOX was 130 µg/L for Atlantic salmon. Aquatic invertebrates, particularly crustaceans, were killed at low concentrations of NP, with the lowest ECOTOX reported LC_{50} = 1 µg/L for *Hyallela azteca*. These data indicate that an array of aquatic species is killed by NP at low µg/L concentrations. We also queried EPA's ECOTOX database for sublethal toxicity and retrieved 689 records of freshwater and saltwater species tested in chronic experiments. The lowest fish LOEC reported was 0.15 µg/L for fathead minnow reproduction. Numerous fish studies reported LOECs at or below 10 µg/L.

Salmonid prey species appear highly sensitive to sublethal effects of NP at low concentrations. The amphipod, *Corophium volutator*, grew less and had disrupted sexual differentiation at 10 μg/L (Brown et al. 1999). Multiple studies with fish indicated that NP disrupts fish endocrine systems by mimicking the female hormone 17β-estradiol (Arsenault et al. 2004, Brown et al. 2003, Lerner et al. 2007a, Lerner et al. 2007b, Luo et al. 2005, Madsen et al. 2004, McCormick et al. 2005, Hutchinson et al. 2006, Jardine et al. 2005, Segner 2005). NP induced the production of vitellogenin in fish at concentrations ranging from 5-100 μg/L (Arukwe and Roe 2008, Hemmer et al. 2002, Ishibashi et al. 2006, Schoenfuss et al. 2008b). Vitellogenin is an egg yolk protein produced by mature females in response to 17β-estradiol, however immature male fish have the capacity to produce vitellogenin if exposed to estrogenic compounds. As such, vitellogenin is a robust biomarker of exposure. A retrospective analysis of an Atlantic salmon population crash suggested the crash was due to NP applied as an adjuvant in a series of pesticide applications in Canada (Brown and Fairchild 2003, Fairchild et al. 1999). Additionally, processes involved in sea water adaptation of salmonid smolts are impaired by NP (Lerner et al. 2007a, Lerner et al. 2007b, Luo et al. 2005, Madsen et al. 2004, McCormick et al. 2005, Jardine et al. 2005).

These results demonstrate NP is of concern to aquatic life, particularly salmonid endocrine systems involved in reproduction and smoltification. This summary is for one of the more than 4,000 inerts/other ingredients and adjuvants currently registered for use in pesticide formulations and there are likely others with equally deleterious effects. Unfortunately we received minimal information on the constituents found in thiobencarb formulations. Consequently, the effects that these other ingredients may have on listed salmonids and designated critical habitat remain an uncertainty and are a recognized data gap in EPA's action under this consultation.

8.5.13. Uncertainties and Data Gaps Identified from Review of Available Toxicity Information for Thiobencarb[21]

1. No translation of aquatic primary producer toxicity data from single species to salmonid aquatic habitats;
2. No AChE activity measurements with salmonids;
3. No empirical data on effects of thiobencarb degradates on fish or habitat assessment endpoints;
4. No empirical data on impacts to salmonids' olfaction, endocrine system, immune-suppression capacity, migration, spawning, or smoltification;
5. No empirical data on effects to riparian plant species; and,
6. No information on other ingredient toxicity within thiobencarb-containing formulations.

8.5.14. Evaluation of Data Available for Response Analysis

We summarize the available toxicity information by assessment endpoint in Table 53. Data and information reviewed for each assessment endpoint was assigned a general qualitative ranking of either "low," "moderate," or "high." To achieve a high confidence ranking, the information stemmed from direct measurements of an assessment endpoint, conducted with a listed species or appropriate surrogate, and was from a well-conducted experiment with stressors of the action or relevant chemical surrogates. A moderate ranking was assigned if one of these three general criteria was absent, and low ranking was assigned if two criteria were absent. Evidence of adverse effects to assessment endpoints for salmonids and their habitat from thiobencarb was available for acute lethality to salmonids and aquatic invertebrates, and highly variable for the other assessment endpoints. However, much less information was available for other ingredients, due in part to the lack of formulation information provided in EPA's BEs as well as the statutory mandate under FIFRA for toxicity data on thiobencarb to support registration. We did locate a substantial amount of data on one group of adjuvants/surfactants, the NP ethoxylates. However, we received minimal information detailing tank mixes and other ingredients within formulations.

[21] A finding of no information for biological assessment endpoints e.g., AChE inhibition in salmonids, does not mean that salmonid AChE is not affected by thiobencarb.

Table 53. Summary of Toxicity Data for Thiobencarb

Assessment Endpoint	Evidence of adverse responses (yes/no)	Concentration range of observed effect or concentrations tested showing absence of effect (µg/L)	Degree of confidence in effects (low, moderate, high)
Salmonid survival (LC_{50})	Yes	760-1200, n=7, lowest NOEC=140 (88 d exposure)	High
Growth (LOEC)	Yes	23-800 n=6	Moderate
Reproduction (LOEC)	Yes	110, n=1	High
Respiration	Yes	19, n=1	Low
Swimming	-	-	-
Olfactory-mediated behaviors	-	-	-
Endocrine disruption	-	-	-
AChE inhibition	Yes	220 is 1/60th of eel LC_{50} (30-50% AChE inhibition), n=1; estimate for salmonids at 12-19	Low
Prey survival	Yes	101-1200, n=8	High
Prey reproduction and growth (LOEC)	Yes	3-420, n=3	High
Aquatic primary production (EC_{50})	Yes	17-4000, n=6	High
Riparian vegetation (terrestrial EC_{25})	Yes	0.019-3.1 lbs thiobencarb/acre, n=18	High

Assessment Endpoint	Evidence of adverse responses (yes/no)	Concentration range of observed effect or concentrations tested showing absence of effect (µg/L)	Degree of confidence in effects (low, moderate, high)
Additive toxicity of thiocarbamates (molinate+thiobencarb)	Yes	Multiple, n=2	High
Degradate Toxicity	Yes	Toxicity responses modeled using EPA ECOSAR model	Low
Adjuvant toxicity: Nonylphenol Survival Reproduction Smoltification Endocrine disruption Prey survival	 Yes Yes Yes Yes Yes	 130 - >1000 0.15 -10 5 -100 5 - 100 1 - >1000	 High High Moderate High High

n indicates number of studies
- indicates no information found on assessment endpoint

9. Risk Characterization

In this section we integrate our exposure and response analyses to evaluate the likelihood of adverse effects to individuals and populations (Figure 38). We combined the exposure analysis with the response analysis to: (1) determine the likelihood of salmonid and habitat effects occurring from the stressors of the action; (2) evaluate the evidence presented in the exposure and response analyses to support or refute risk hypotheses; and (3) translate fitness level consequences of individual salmonids to population-level effects. The risk characterization section concludes with a general summary of species responses from population-level effects. We then evaluate the effects to specific ESUs (*i.e.*, species), in the *Integration and Synthesis* section.

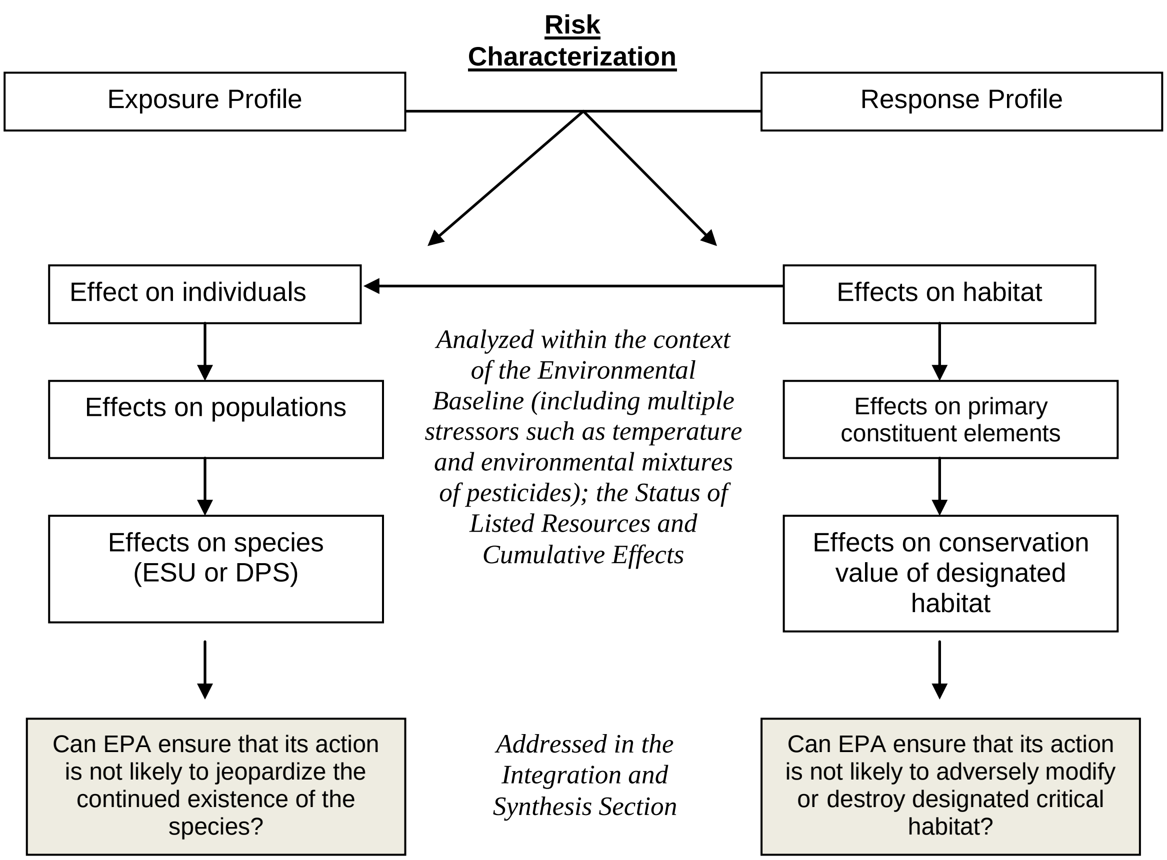


Figure 38. Schematic of Risk Characterization Phase.

Field incidents of dead fish and crop damage reported in EPA incident database made available to NMFS

We also evaluate incident data supplied by EPA for thiobencarb as a separate line of evidence. No incidents of fish kills were submitted to us from EPA.

Seven incidents of crop damage were identified. In most cases, no information was available to characterize drift. EPA ranked one incident as highly probable, three as probable, and three as possible. We are uncertain how to use these data other than adjacent crops have been affected by drifting thiobencarb.

Field studies with thiobencarb from rice irrigation water discharge

A series of studies in Halls Bayou were conducted in 1979 to test the effects of Bolero on aquatic organisms following releases of rice irrigation water (Harper et al. 1979). Fish, phytoplankton, and macroinvertebrates were evaluated. Much of the study reprint is illegible. What can be discerned is that caged fish studies with a variety of species (black drum, striped mullet, grass shrimp, and bullhead minnows) showed mortalities following release of thiobencarb contaminated water a few days after application due to heavy rains. Mortalities ranged from 0 to 100% depending on the species and geographic area. No reference sites were identified in the study, thus the use of much of the data is questionable. We note that concentrations of thiobencarb may have achieved sufficient levels to kill sensitive fish and grass shrimp (2 – 83 µg/L) at cage sites, however the authors note that other pesticides were also present. No laboratory acute toxicity data were available for the species tested. These unpublished data are of limited utility based on flawed experimental design.

Mixture analysis based on AChE inhibition

In previous biological opinions with AChE inhibiting insecticides, we analyzed the joint effects of multiple AChE inhibiting insecticides using simple, quantitative dose-response relationships based on empirical data (NMFS 2008a, NMFS 2009d, NMFS 2010a). We located no experimental data on thiobencarb's capacity to reduce salmonid AChE activity and therefore lack the information necessary to conduct a mixture analysis. This does not mean that mixture toxicity does not occur, we expect it does given thiobencarb's common mode of action. When thiobencarb co-occurs with other AChE inhibitors, we expect additivity, and potentially synergism with certain combinations, although we cannot quantify the effects.

9.1. Exposure and Response Integration

In Figure 39 we show the overlap between exposure estimates for thiobencarb and concentrations that affect assessment endpoints. This portion of the analysis primarily focuses on thiobencarb, although we do present assessment endpoints for a common adjuvant, nonylphenol, mixture toxicity, and degradate information. The figure shows the exposure concentration ranges (minimum to maximum values) gleaned from the three sources of exposure data we analyzed: EPA's estimates presented in the BEs that represent crop uses; NMFS' modeling estimates for flood plain habitats; and surface water monitoring data from ambient monitoring programs and from targeted monitoring. In addition to the salmonid BEs submitted to NMFS, we also considered the exposure estimates developed by EPA in the BEs for the California red-legged frog. The effect concentrations are values taken from the toxicity data reviewed in the *Response Analysis* section. For the survival assessment endpoint, effect concentrations are LC_{50}s, thus death of individuals occurring at concentrations below LC_{50}s are not represented. Consequently, when LC_{50} effect concentrations are not exceeded by the exposure estimates, it does not mean there are no incidences of mortality. Thus for those instances where LC_{50}s do not overlap or are not exceeded by exposure estimates, we discuss the difference in magnitude of the two metrics and apply best professional judgment on whether death of individuals is expected.

This coarse analysis does not present temporal aspects of exposure nor does it show the distribution of toxicity values. It is also predicated primarily on standard toxicity endpoints as we located few ecologically relevant sublethal or field-collected data, a noted uncertainty with this analysis. However, the analysis does allow us to systematically address which assessment endpoints are likely to be affected by exposure to thiobencarb. Where significant uncertainty arises, we highlight the information and discuss its influence on our inferences and conclusions.

Most of the concentration ranges overlap with assessment endpoints indicating that adverse effects are expected in salmonids and to their habitat if exposed for sufficient durations (Figure 39). The maximum reported values from CDPR data (16.55   g/L) and NAWQA California data (4.38 µg/L) did not exceed effects concentrations for prey survival, fish reproduction, fish growth, or salmonid survival. Concentrations from targeted studies, EPA estimates, flood plain estimates, and early release of rice irrigation water exceeded effect concentrations for all assessment endpoints. Careful analysis is required to determine if exposure to thiobencarb is expected to exceed effect concentrations. For example, the targeted monitoring studies (n=10), reported a value of 8.9 mg/L which represents the highest concentration reported for thiobencarb in water. The value is the result of an early release of rice irrigation water 2 hrs after application due to a flood event. Thiobencarb labels generally require a holding period of at least 14 days prior to discharge for liquid formulations and up to 30 days for granular formulations, thus we did not rely heavily on 8.9 mg/L value to characterize exposure. We note that this value would only pertain to early releases of treated water and expect this to happen infrequently, which monitoring data indicate are reasonably certain to occur despite product labeling (Table 43). Similarly, the exposure numbers presented for floodplain habitats were generated using a field-scale transport model to assess potential exposure among individuals and habitats. These estimates are extremely relevant given the species habitat requirements and the lack of EPA label requirements for set-backs (*i.e.*, buffers) to salmonid habitats. However, we recognize that these estimates do not reflect the likely exposure of all individuals and all habitats, including exposure to individuals occupying deeper habitats and habitats that are not adjacent to thiobencarb applications.

The most sensitive assessment endpoint for direct effects on salmonids was the AChE inhibition (EC_{50}) which was estimated at 12-19 µg/L. However, we had low confidence in this study as the information stemmed from eels and was conducted with only one thiobencarb treatment at 220 µg/L. It is uncertain whether salmonids would show the same proportional relationship between LC_{50} and AChE EC_{50} exhibited by eels. The next most sensitive assessment endpoint was aquatic primary production where 17 µg/L thiobencarb inhibited algal growth, while other algae were much more tolerant of concentrations up to 4000 µg/L. Survival based on LC_{50}s of salmonids exposed to thiobencarb in various formulations was encompassed by the upper end of concentration ranges observed in targeted studies, EPA estimates, and flood plain habitat estimates. We do not disregard any of these effects at this point in the analysis as the evidence supports evaluation of whether an individual's fitness or habitat endpoints are compromised.

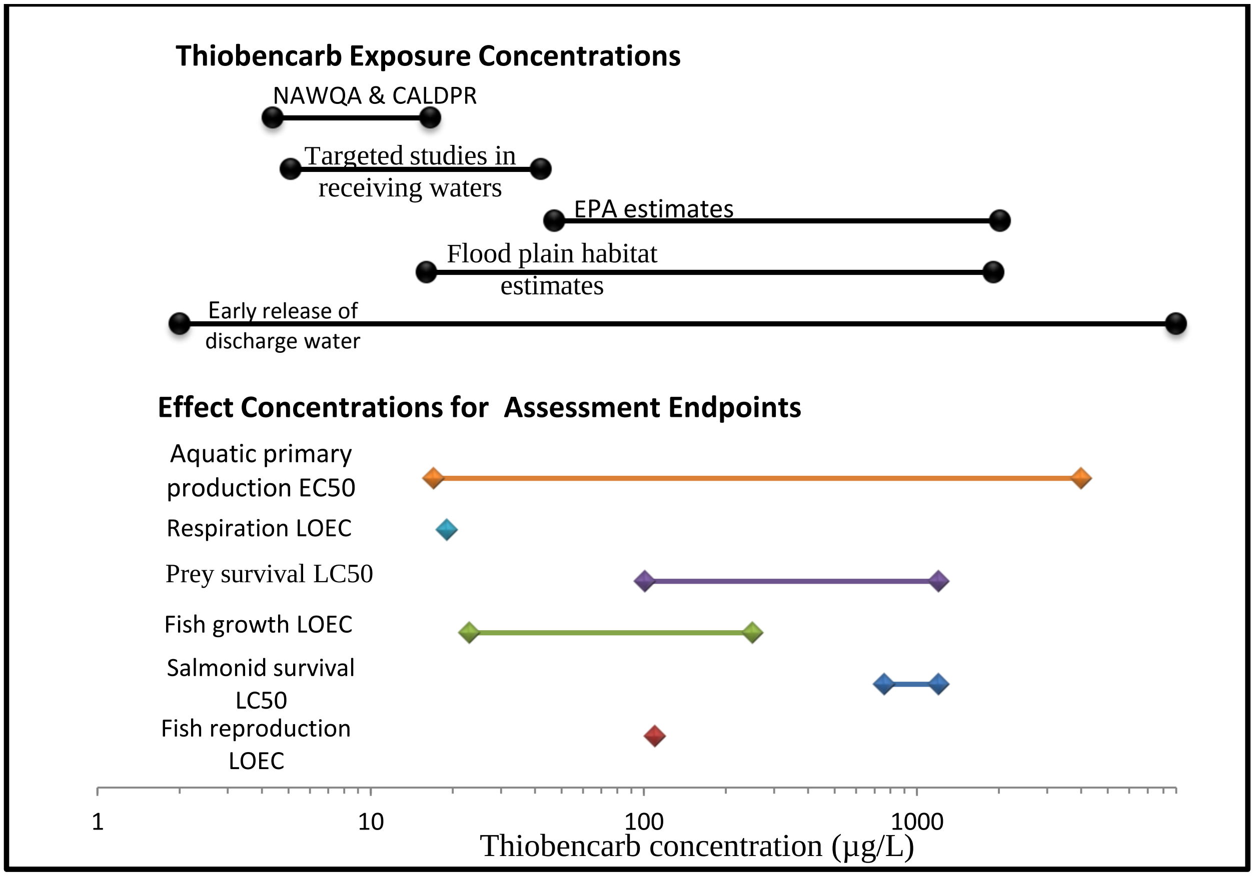


Figure 39. Thiobencarb exposure concentrations and effect concentrations for assessment endpoints

Key Findings of Exposure Analysis:

- The vast majority of exposure to thiobencarb drift is expected to occur during the peak application period of May and June with less frequent exposures expected in April and July. Applications during other months are allowed but are extremely rare (state-wide average < one application per year). Floodplain habitats alongside rice fields are at the greatest risk of receiving elevated concentrations from aerial and ground applications, i.e., from 37 – 1910 µg/L.

- Thiobencarb in runoff typically occurs 14-30 days post application (after required holding periods), thus the greatest probability of exposure to the highest runoff concentrations likely occurs from mid-May through the end of July.
- Riparian systems and multiple life stages of salmonids are likely exposed to thiobencarb from drift during applications and from runoff following release of rice irrigation water into drainage networks that ultimately return water to salmonid containing waters.
- Migrating adults, rearing juveniles, and migrating juveniles overlap with peak thiobencarb applications in May and June.
- Spawning adults, incubating eggs, and emerging fry exposure to thiobencarb is species dependent.

Key Findings of Response Analysis:

- Aquatic, laboratory toxicity tests indicate that thiobencarb is acutely toxic to fish (multiple endpoints; 19-1200 µg/L), invertebrates (survival EC_{50}s; 100-1200 µg/L), and plants (EC_{25}s; 17 – 4000 µg/L).
- AChE inhibition in salmonids is likely, but unquantifiable as no studies with salmonids were available.
- Riparian plants, particularly emergent vegetation, are highly sensitive to thiobencarb drift given agricultural plants sensitivity to thiobencarb. At the labeled rate of 4 lbs a.i./A, 0.55% of the applied material is predicted to drift ~ 1000 ft from an aerial application. That amount of drift would be expected to reduce emergent vegetation seedling growth by 25%.

9.2. Evaluation of Risk Hypotheses

In this phase of our analysis we examine the weight of evidence to determine whether it supports or refutes a given risk hypothesis (Table 54). This is not a statistical analysis, but rather a qualitative assessment of the available information presented in the response and exposure sections. We also highlight general uncertainties and data gaps associated with the data. In some instances there may be no information specifically related to a given hypothesis. In some cases, if information on a similar endpoint or chemical is available, and it is reasonable to do so, we extrapolate from the available data to fill gaps, recognizing that this may introduce additional uncertainty into the analysis. If the evidence supports the risk hypothesis, we determine whether it warrants an assessment at the population level or affects PCEs to a degree that warrants analysis of the potential to reduce the conservation value of designated critical habitat.

The information available to characterize thiobencarb exposure included surface water monitoring data and estimates from pesticide transport models (runoff and drift). We combine this information with the distribution and life-history characteristics of ESA-listed Pacific salmonids by ESU. As discussed in the *Exposure Analysis* section above, each source of information has inherent limitations and uncertainties. For example, the pesticide monitoring data were generally not designed to quantify peak exposure concentrations or distributions of exposure in salmonid habitats within the Central Valley. Consequently, pesticide exposure models were used to supplement monitoring data, and together this information was used to describe the potential range of pesticide concentrations. We conducted AgDrift model runs to provide estimates for concentrations resulting from drift to a shallow and narrow body of water, such as those found in floodplain habitats of the Central Valley. Small streams and many floodplain habitats such as the Yolo bypass are more susceptible to higher pesticide concentrations than larger, high flow systems as their physical characteristics provide less dilution.

We recognize that pesticide concentrations will vary greatly among salmonid habitats in the Central Valley, and exposure duration and concentration will be reduced by higher water volumes and velocities.

Standardized toxicity tests for pesticide registration are typically poor predictors of real world aquatic ecosystems as test organisms are exposed to constant pesticide concentrations for arbitrary durations (*e.g.,* acute, 96-h and chronic, 21- or 28-d) that may poorly reflect field exposures, which tend to be repeated exposures Additionally, the tests do not account for exposure to stressors, pesticides and others, already in the water. The response of fish and their prey to different durations of exposure, and exposure representing different environmentally relevant dissipation patterns of thiobencarb is a prominent data gap. Exposure durations sufficient to elicit toxicological responses can occur at durations much shorter than standard toxicity tests. We therefore did not average exposure concentrations over time, *i.e.*, time-weighted averages, because adverse responses to short term thiobencarb exposures such as pulses would likely be masked.

Large spatial and temporal variability exists in the use of aquatic habitats by listed Pacific salmonids in the Central Valley. These differences occur at multiple scales of biological organization (*i.e.,* individual, population, and species). Both an individual's life stage and its life history are important considerations in its use of aquatic habitats. This natural variation is overlaid with the inherent variation of environmental factors including climate (*e.g.,* precipitation patterns), habitat stressors, and land use. Given this biological and environmental variability, it is difficult to predict the precise exposure to thiobencarb for any one individual, let alone for a population or species.

Consequently, we used the life history information of the three species to evaluate potential exposure in their myriad aquatic habitats (Table 28, Table 30, and Table 32). For example, we found that adult Chinook migrating upstream to reach spawning grounds and juvenile steelhead moving downstream were most at risk of thiobencarb exposure.

We evaluated the potential for individual fitness consequences (*i.e.,* assessment endpoints) by comparing the range in expected exposure concentrations with adverse effect levels in the context of aquatic habitat utilization. Given the geographic specificity of thiobencarb use in California, we focused on habitats used for rearing and migrating which range from first order streams to large rivers—the Sacramento River, San Joaquin River, and the Delta. These habitats are critical to successful adult migration, the development and growth of young fish, and for providing safe passage to and from the ocean.

Salmonids also rely on the San Francisco Bay and the Pacific Ocean for migration and rearing prior to moving into open ocean areas. In general, thiobencarb exposure will be less intense in these areas compared to freshwater systems in the Central Valley. Exceptions may include the Sacramento-San Joaquin Delta where juveniles are rearing for extended periods (weeks-months) proximate to rice fields that may prematurely release thiobencarb-contaminated water due to flooding.

This framework allows us to evaluate risk hypotheses based on the spatial and temporal nature of exposure to thiobencarb across CA's Central Valley. Below we make a determination of whether fitness of individuals is compromised warranting a subsequent analysis at the population level.

DRIFT:

Drift of thiobencarb into salmonid habitats occurs during its application period, primarily May and June, and to a lesser extent in April and July. Concentrations depend on proximity to application area, application method, droplet size, release height, wind speed/direction, receiving water volume/flow, and interception by riparian vegetation. Based on drift studies, modeling exercises, and surface water monitoring studies, concentrations may range from non-detectable to 1910 µg/L (aerial application at edge of field).

RUNOFF:

Runoff from rice fields, including planned discharge following holding periods from 14, 19, and 30 days, will result in thiobencarb entering salmonid habitats. Concentrations may attain levels as high as 350 µg/l at 14 d and 47 µg/L at 30 d (both EPA modeling estimates) in discharge waters. Final concentrations within salmonid habitats receiving the discharge will likely be less than discharge concentrations and will be dependent on factors including receiving water flow and depth. Concentrations may be orders of magnitude higher in runoff if rice irrigation water is released within hours or days post application due to emergency releases. Several field studies reported concentrations in the mg/L range following early, emergency related releases, one attaining levels as high as 8.9 mg/L. We do not expect this to be a common occurrence based on reports over the last decade on frequency and measured concentrations from early releases (Table 43).

Table 54. Risk Hypotheses

Effects to salmonids
1. Exposure to thiobencarb via drift or runoff is sufficient to:
a. Kill salmonids from direct exposure
b. Reduce salmonid survival through impacts to growth
c. Reduce reproduction
d. Impair swimming
e. Impair respiration
f. Reduce salmonid growth through impacts on the availability and quantity of prey
Effects to salmonid habitats
2. Exposure to thiobencarb via drift or runoff is sufficient to:
a. Reduce numbers of aquatic primary producers, thereby affecting salmonid prey communities, salmonids, and salmonid instream cover.
b. Reduce riparian vegetation to such an extent that stream temperatures are elevated, erosion increases, and reductions in natural coverage results through reduced inputs of large woody debris and vegetation.
c. Reduce water quality and prey resources in estuarine environments.
Effects from other stressors of the action and contributing environmental factors
3. Exposure to degradates of thiobencarb will cause adverse effects to salmonids and their habitats.
4. Exposure to adjuvants, tank mixtures and other chemicals within pesticide products containing thiobencarb will cause adverse effects to salmonids and their habitats.
5. Exposure to other pesticides present in the action area can act in combination with thiobencarb to increase effects to salmonids and their habitats.
6. Exposure to elevated temperatures will enhance the toxicity of the stressors of the action.

9.2.1. Risk Hypotheses

1. Exposure to thiobencarb via drift or runoff is sufficient to:

A. Kill salmonids from direct exposure

A substantial body of laboratory toxicity data indicates that 50% of a test population of salmonids die following short term (<96 hrs) exposures to thiobencarb at levels at or above 760 µg/L (lowest reported salmonid LC_{50}). Thiobencarb kills salmonids at concentrations well below LC_{50}s in 96-hr laboratory studies; however few test results reported the number of dead fish at treatment concentrations below an LC_{50}. Exposure duration, health condition of individuals, and concentrations are key determinants of lethality in real world aquatic systems, as are the presence of other stressors of the action and stressors present in the environmental baseline. We located a single embryo-to-fry study that reported a survival NOEC of 140 µg/L for salmonid fry (28 days old) following 88 days of continuous flow through exposure from 28 days prehatch to 60 days post hatch (Fujimura et al. 1991). At 250 µg/L (LOEC), survival was reduced by 37% and at 140 µg/L by 13% (no statistical difference from controls) (Fujimura et al. 1991). We don't expect thiobencarb concentrations to persist at 140 µg/L for 88 days in Central Valley surface waters based on the monitoring data and pesticide exposure models.

We expect that adults and juveniles will die if exposed to thiobencarb-containing drift from applications proximate to salmonid bearing habitats. We expect that 14-d releases will kill sensitive individuals if they are located directly downstream from a discharge or are present within the drainage network. We also expect that early releases of thiobencarb-containing rice water into aquatic habitats are sufficient to kill juvenile and adult salmonids (Table 45). For these reasons, we discuss loss of juveniles and adults at the population scale.

B. Reduce salmonid survival through impacts to growth

Thiobencarb reduced Chinook salmon growth by 22% at 49 µg/L compared to unexposed fish following 88 days of continuous exposure in a laboratory experiment (Fujimura et al. 1991). It's unlikely that concentrations of 50 µg/L or more will be attained and sustained based on available monitoring and pesticide modeling data. Chronic toxicity studies indicate that juvenile fathead minnow and striped bass growth are affected at concentrations ranging from 23 to 250 µg/L (LOECs). Exposure duration in these tests varied from 260 days in the fathead minnow tests to between 45 and 52 days for experiments with striped bass. We do not expect any juvenile salmon to be exposed to these concentrations for chronic durations that have been shown to cause impacts to growth (*e.g.*, 45 days or more). Some juvenile salmon are expected to be exposed to these concentrations for shorter durations (*e.g*. 96 hrs or less), however we located no experiments that measured growth over these durations and this assessment endpoint remains highly uncertain. Zebrafish length measured following 96 h exposures to thiobencarb at 0.8, 8.0, 80, and 800 µg/L showed statistically significant reductions at 800 µg/L (NMFS 2011d). Additionally, 100% of zebrafish exposed to 800 ug/L were observed as abnormal. Example abnormalities included edema, unhatched eggs, lethargic, deformed fins, and deformed tails (NMFS 2011d).

Growth effects have been seen with other AChE inhibitors, such as organophosate and carbamate insecticides, however, thiobencarb has low potency compared to these compounds. For these reasons, we conclude that growth will not be sufficiently compromised to affect an individual's fitness.

C. Reduce reproduction

We located one study that reported effects to fish reproduction in freshwater fish (MRID 45695101; (EPA 2009b). The LOEC was 110 µg/L (LOEC) following a 260-day exposure in flow through conditions. Concentrations from drift and early runoff may attain 110 µg/L, however durations of exposure will depend on the receiving habitat and factors such as flow, depth, etc. We also note that the majority of known spawning areas occur outside of rice growing areas. Although, significant reproduction responses such as reductions in the number of eggs spawned and hatching success were observed in this study, there is no information available to assess reproductive responses in salmonids exposed to thiobencarb for relevant exposure durations. Additionally, no adverse responses to reproduction were noted following exposure to 53 µg/L for 260 days (LOEC). For these reasons, we do not expect reproduction to be affected and therefore do not anticipate an individual's fitness to be compromised based on effects to reproductive endpoints.

D. Impair swimming

Swimming is critical to salmonids at all life stages following hatching. We found no studies that tested swimming capacity or performance following exposure to thiobencarb. We found no direct information on the AChE response in salmonids following environmentally relevant exposures, which represents a significant data gap. Since thiobencarb is a likely AChE inhibitor, we do expect swimming to be compromised well below salmonid LC_{50}. We note that if eels are an appropriate surrogate for AChE inhibition, then salmonids exposed to 1/60th of the LC_{50} would experience 40-50% inhibition of AChE. However, we located no evidence to refute or support this assertion. If eels are appropriate surrogates, then salmonids would experience neurotoxic effects such as reduced swimming ability at concentrations of 12-19 µg/L thiobencarb (i.e., 1/60th of the LC_{50}) after a few hours of exposure. We conclude that salmonids likely experience AChE inhibition at concentrations below salmonid LC50s (i.e., 720 µg/L), however we have no empirical information to further refine effect thresholds.

Given the targeted monitoring data, EPA's model results, and NMFS flood plain habitat modeling results, swimming is likely affected at concentrations well below LC_{50}s, less than 720 µg/L, particularly for sensitive juveniles. We therefore conclude that adult and juvenile fitness are compromised from impaired swimming and discuss population level consequences.

E. Impair respiration

Functional breathing is critical to salmonid survival. We located one study that tested this hypothesis. The study showed that at 19 µg/L BoleroEC gills were abnormally inflamed compared to control fish. Study limitations included lack of verification of treatment concentrations, small treatment size (n = 3 fish), and no replication within treatments. Although the study's limitations reduce its utility, the data do support adverse responses in fish gills and suggest that thiobencarb is a gill toxicant. If salmonids are as sensitive to BoleroEC as mosquito fish, respiration is likely impaired due to gill damage. Seven day static exposures maintaining an average concentration of ~20 µg/L is a potential scenario in lower flow, shallow, flood plain habitats as well as in salmonid habitats receiving continuous discharges of thiobencarb from rice growing operations. We therefore conclude that damaged gills reduce an individual's fitness and discuss potential population level consequences.

F. Reduce salmonid growth through impacts on the availability and quantity of prey

Thiobencarb inhibits acetylcholinesterase activity in exposed invertebrates. Therefore, we evaluate toxic responses experienced by salmonid prey. The majority of data derive from laboratory bioassays with standard aquatic invertebrates (Table 51). Toxicity values for survival, growth, and reproduction were available and indicate that thiobencarb adversely affects aquatic invertebrates. Concentrations reducing survival ranged from 101-1200 µg/L following 48 - 96 hr exposures (EC_{50}s). Reproduction and growth endpoints were affected at lower concentrations, 3-420 µg/L following 21 d exposures.

Salmonid habitats proximate to rice fields and particularly shallow, low-flow flood plain habitats are most at risk of receiving drift. In these areas, thiobencarb levels in drift may attain levels sufficient to kill aquatic invertebrates. Additionally, early releases of rice irrigation water containing thiobencarb may also kill salmonid prey. Therefore, we discuss the impact on populations of juveniles that rely on invertebrate communities for food.

2. Exposure to thiobencarb is sufficient to:

A. Reduce numbers of aquatic primary producers thereby affecting salmonid prey communities, salmonids, and salmonid instream cover;

Aquatic primary producers showed reduced growth (EC_{50}s) at concentrations as low as 17 µg/L and as high as 4000 µg/L (Table 51). Both values were reported for species of green algae. No studies were available that tested aquatic community responses to thiobencarb in running water systems or evaluated salmonid habitats within CA's Central Valley. The bioassay results indicate that following 48-hr continuous exposure to thiobencarb, reductions in aquatic primary producers can occur. Translating this information to real world systems is complicated by many factors including condition and composition of existing instream plant community, community responses to reductions in primary production, and prediction of bottom up effects in systems that are severely and continuously compromised by multiple stressors. We conclude that effects to primary producers may occur, particularly in salmonid habitats receiving drift and those exposed to early release of rice irrigation water. Therefore, we do not anticipate individual fitness of salmonids to be compromised by anticipated infrequent reductions in instream primary productivity.

B. Reduce riparian vegetation to such an extent that stream temperatures are elevated, erosions increases, and reductions in natural coverage results through reduced inputs of woody debris and vegetation.

Amount and frequency of thiobencarb drift are the predominant factors in evaluation of riparian vegetation responses. Thiobencarb adversely affects emerging plants, both monocots and dicots, at concentrations as low as 0.019 lb a.i./A, which represents < 1% of the labeled application rate and is comparable to the drift expected 1000 feet from an aerial application of thiobencarb. If riparian vegetation is as sensitive as the crops tested, they are extremely susceptible to thiobencarb's herbicidal properties. Thiobencarb is applied both aerially and by ground throughout the application period and intensely in May and June. Approximately 25% of known thiobencarb application occur with liquid formulations. Riparian zones that are proximate to applications are at the greatest risk of exposure, but even zones that are 1000 ft away from application areas may receive toxic levels. Reductions in riparian systems likely compromise fish fitness by increasing temperatures, increasing erosion/sedimentation, and reducing natural cover.

We therefore conclude that loss of riparian vegetation affects salmonid habitats and salmonids and discuss potential effects to salmonid populations and to designated critical habitats.

C. Reduce water quality and prey resources in estuarine environments.

We located monitoring data within the San-Francisco Bay Estuary and Sacramento-San Joaquin Delta as well as toxicity information from a variety of estuarine and marine organisms. Based on our review of these data we expect that thiobencarb will reach CA's estuaries; however measured levels attained typically are in the low ng/L range (maximum concentration of 66 ng/L) (Kuivila and Jennings 2007), orders of magnitude below adverse effect levels. Given these data we do not anticipate water quality in estuaries to be reduced by thiobencarb so we do not evaluate impacts to populations.

3. Exposure to degradates of thiobencarb will cause adverse effects to salmonids and their habitats.

Many degradates are formed as thiobencarb breaks down in aquatic systems. We located no aquatic toxicity information for these degradates, even for those that represent more than 10% of thiobencarb applied. EPA applied structure activity models that identified several degradates as highly toxic to fish, invertebrates, and plants. Additionally, EPA predicted that some degradates, such as bencarb may impair salmonids and their habitats. We therefore conclude that thiobencarb degradates will cause adverse effects to salmonids and their habitats beyond those predicted with thiobencarb alone.

4. Exposure to adjuvants, tank mixtures and other chemicals within pesticide products containing thiobencarb will cause adverse effects to salmonids and their habitats.

In addition to exposure to thiobencarb, which is currently the only stressor of the action incorporated in the EPA's risk assessment, salmonids and their habitat are likely exposed to other stressors of the action, including additional active ingredients in formulated products and tank mixes. Salmonid habitats may also be exposed to a number of the approximately 4,000 inert ingredients approved for use in end-use pesticide products by EPA, including adjuvants, surfactants, and other products that are applied as tank mixtures. Once the mixture (formulated pesticide or tank mix) is introduced into the environment, physico-chemical properties influence transport and resulting concentrations in sediment, water, and organisms. We expect some of these chemicals to reach salmonid habitats from spray drift deposition and from runoff events following application. Salmon and their habitats exposed to these multiple stressors are expected to show a greater adverse response than laboratory animals exposed only to thiobencarb, thus available toxicity data generally underestimate the response from a field-applied pesticide mixture.

4.a. Additional Active Ingredients within Tank Mixes

Some of the active ingredients used in tank mixtures may be more toxic than the active ingredient, for example, copper sulfate. EPA has not consulted with NMFS on most of these active ingredients, and NMFS in this consultation is not evaluating the effects of each of the separate active ingredients with which thiobencarb could be mixed. This is a significant source of uncertainty in the analysis, and NMFS described some of its concerns in this Opinion. Several labels recommend propanil as a tank mixture. Also, copper sulfate is frequently mixed with thiobencarb based on CDPR data. Propanil is an herbicide with a wide range of acute LC_{50}s for freshwater fish (2.1– 12.7 mg/L) and invertebrates (1.2 – 16 mg/L) (EPA 2009a). Rainbow trout appear the most sensitive freshwater fish tested with an LC_{50} of 2.1 mg/L. Propanil is highly toxic to aquatic (EC_{50} 16-110 µg/L) and terrestrial primary producers (EC_{25} 0.09-12 lbs propanil/ acre). Propanil's degradate, 3,4-DCA, is 15 fold more toxic than propanil based on *D. magna* EC_{50} survival test (48-h EC_{50} = 528 µg/L parent; 34.9 µg/L degradate). Drift of propanil into riparian areas and salmonid habitats will likely impair primary producers.

Copper sulfate (and other copper organic complexes) readily disassociates in water once transported to aquatic habitats from either drift or runoff. Dissolved copper is acutely toxic at low µg/L concentrations to a wide range of salmonid assessment endpoints, salmonid prey, and primary producers (Fernandes and Henriques 1991, Baldwin et al. 2003b, Baldwin et al. 2011, Hansen et al. 1999, Mebane and Arthaud 2010). NMFS believes that for many of these endpoints copper could be significantly more toxic than thiobencarb. Given the levels of copper used for pest control in CA's Central Valley, copper is potentially a major toxicant affecting listed salmonids. However, as noted above, EPA is not consulting on each active ingredient with which thiobencarb can be mixed, and NMFS is not in a position to evaluate this hypothesis with respect to the populations at issue in this opinion. Further, measures to keep thiobencarb out of the water should also reduce concentrations of other active ingredients.

4. b. Inert/other ingredients

Labels for thiobencarb list the percentage of other ingredients, and can be as much as 90% of the total volume (Table 47). In past Opinions, other ingredients were referenced on labels and included highly toxic compounds such as aromatic solvents, xylene solvents, and petroleum distillates. PAHs (in some aromatic solvents) are known aquatic contaminants, and some have been linked to carcinogenic and immunogenic effects. Other aromatic solvents may have a narcotic effect. For example, EPA noted that mixtures of “xylenes and xylene isomers are moderately to highly toxic to aquatic species” (EPA 2005). Toxic effects vary dependent on the specific chemical. The likelihood of these compounds co-occurring in the water column is difficult to determine with any specificity, but can reasonably be presumed to occur in spray drift deposition and runoff.

In addition to other/inert ingredients listed on the labels for the a.i.s considered in this Opinion, thousands of other compounds are approved by EPA as additives to pesticide products without any specific requirement for the compound identity or amount to be listed on the labels. One example of these ingredients are nonylphenol polyethoxylates, which have been linked to endocrine disruption. These chemicals were addressed at length in previous Opinions on EPA pesticide registrations (NMFS 2008c, NMFS2009b). There are however, myriad others, some of which may increase the toxicity of thiobencarb. The majority of a pesticide formulation is often composed of inert ingredients. For example, one thiobencarb label considered in this Opinion has more than >90% other ingredients. Consequently, salmonid exposure to these ingredients may be greater than exposure to thiobencarb. EPA currently has no specific method of accounting for this potential additional toxicity and risk, but it cannot be ignored. NMFS has opted to address the uncertainty associated with these ingredients qualitatively.

Collectively, the available lines of evidence support the overall hypothesis that other stressors of the action cause adverse effects to salmonids and their habitat. From our review of the available information it is not possible to accurately quantify the contribution of other stressors of the action. These stressors include propanil and copper, as well as inert/other ingredients in pesticide formulations. These stressors of the action are an important consideration when assessing potential effects on listed salmonids and their habitat. Thus, we carry forward effects from these other stressors of the action when we discuss effects to salmonid populations.

5. Exposure to other pesticides present in the action area can act in combination with thiobencarb to increase effects to salmonids and their habitats.

The available toxicity and exposure data support this hypothesis. Other AChE inhibitors (*e.g.*, carbamates and organophosphates) found in the action area likely result in additive or synergistic effects to exposed salmonids and aquatic invertebrates. The magnitude of effects will depend on the duration and concentrations of exposure. Effects of exposure to other pesticides are carried forward to the population-level analysis.

6. Exposure to elevated temperatures can enhance the toxicity of the stressors of the action.

We reviewed the available information to determine whether empirical data indicated enhanced toxicity at elevated temperatures for carbamates in general, and, in particular, for thiobencarb. Multiple experimental results indicated that increases in temperature resulted in lower LC_{50}s for fish, including salmonids (Mayer and Ellersieck, 1988). Acute lethality bioassays with carbamates (aminocarb and carbaryl), showed a distinct relationship between toxicity (measured by 96 h LC_{50}s) and temperature (Mayer and Ellersieck, 1988). These experiments were conducted with brook trout and yellow perch. We recognize that aminocarb and carbaryl are both N-methylcarbamates, a separate carbamate class than thiocarbamates (thiobencarb's class), thus uncertainty exists whether thiocarbamates will respond to temperature in a similar fashion. These data support the hypothesis and we therefore carry forward temperature effects when we discuss effects to salmonid populations.

Table 55 summarizes the preceding discussion covering our risk hypotheses as to whether or not the stressors of the action may compromise individual fitness to the listed salmonids within California's Central Valley.

Table 55. Summary of determinations to risk hypotheses.

Effects to salmonids	Individual fitness compromised
1. Exposure to thiobencarb via drift or runoff is sufficient to:	
a. Kill salmonids from direct exposure	yes
b. Reduce salmonid survival through impacts to growth	no
c. Reduce reproduction	no
d. Impair swimming	yes
e. Impair respiration	yes
f. Reduce salmonid growth through impacts on the availability and quantity of prey	yes
Effects to salmonid habitats	
2. Exposure to thiobencarb via drift or runoff is sufficient to:	
a. Reduce aquatic primary producers thereby affecting salmonid prey communities, salmonids, and salmonid instream cover	no
b. Reduce riparian vegetation to such an extent that stream temperatures are elevated, erosion increases, and reductions in natural coverage results through reduced inputs of large wood and vegetation.	yes
c. Reduce water quality and prey resources in estuarine environments.	no
Effects from other stressors of the action and contributing environmental factors	
3. Exposure to degradates of thiobencarb will cause adverse effects to salmonids and their habitats.	yes
4. Exposure to adjuvants, tank mixtures and other chemicals within pesticide products containing thiobencarb will cause adverse effects to salmonids and their habitats.	yes
5. Exposure to other pesticides present in the action area can act in combination with thiobencarb increase effects to salmonids and their habitats.	yes
6. Exposure to elevated temperatures will enhance the toxicity of the stressors of the action	yes

9.2.2. Population Level Responses

In this section, we discuss the likelihood of individual salmonid fitness consequences rising to the level at which population-level effects occur[22]. To forecast potential effects at the population level, first we determine the number and life stage of salmonids likely exposed to thiobencarb and the other stressors of the action. Since the overwhelming majority of thiobencarb applications are expected to occur between April and July (Figure 24), we overlay relative abundances of listed salmonid populations during these months in rice-growing areas of the Central Valley (Table 56). Next we evaluate the potential for population level consequences based on the relative abundance information. The more individuals within a population that overlap with high thiobencarb use, the greater the likelihood for population level consequences. Rearing salmonids showed low abundances in aquatic areas prone to thiobencarb drift and runoff. Migration timing and areas overlapped with thiobencarb use. Juvenile migration abundances were ranked as low for Chinook populations and medium for Central Valley steelhead populations. Migrating juvenile steelhead, therefore have a greater likelihood of exposure to thiobencarb than migrating Chinook. We do not expect spawning to co-occur within rice growing areas for Chinook populations, however upstream migrations of pre-spawn adults through rice growing areas showed high abundances for spring-run Chinook, moderate abundances for winter-run Chinook, and low abundance for steelhead between April and July.

[22] In previous Opinions we used population modeling to evaluate consequences to the population growth rate (lambda) from reduced survival of subyearling salmonids and reduced growth due to AChE effects. We do not employ these models in this Opinion because overlap of thiobencarb use areas with rearing subyearlings is not expected to a great extent. Therefore we expect much lower number of juveniles to be exposed. This was not the case for other pesticides in previous Opinions.

Table 56. Relative abundance of listed salmonids in rice areas during thiobencarb use periods, April-July.

Listed Species Populations	Life-Stage Behaviors	Expected Abundance			
		None	Low	Med	High
Central Valley Steelhead:	Adult migration		X		
1 population	Spawning		X		
	Early rearing		X		
	Juvenile migration			X	
Sacramento River Winter-run Chinook	Adult migration			X	
1 population	Spawning	X			
	Early rearing		X		
	Juvenile migration		X		
Central Valley Spring-run Chinook	Adult migration				X
3 populations	Spawning	X			
	Early rearing		X		
	Juvenile migration		X		

Based on the expected abundances of juveniles of each species co-occurring with thiobencarb application periods, we determined that population level consequences are not anticipated in the Central Valley from rice applications of thiobencarb (Table 57).

The two primary drivers for high concentrations of thiobencarb, drift into shallow flood plain habitats and early release of rice irrigation water are to likely occur infrequently, and exposure to such concentrations is expected to be infrequent given information on their occurrence and/or on spatial and temporal considerations. Most life stages of the three listed salmonids are not expected to occur in close proximity to thiobencarb use sites during the period of application and discharge, or are only expected to occur in relatively low abundance. The exceptions are migratory adult Chinook.

A high proportion of spring-run Chinook adults migrate up the Sacramento River to spawn during thiobencarb application periods, and a moderate number of winter-run Chinook pass through these areas during peak application months. Adult migration through the rice growing areas occurs over a 7-8 month period compared to the relatively short application period of thiobencarb, which largely occurs during a span of 6-8 weeks. Migrating adults typically follow the thalweg where water is faster and deeper than other areas of the river possibly reducing exposure to edge of field concentrations and decreasing exposure duration, *i.e.*, adults may co-occur with thiobencarb albeit where dilution is greatest (Table 21). Overall, we expect adults to be exposed to an array of thiobencarb concentrations, many of which would be in the ng/L range, and potentially some in the ug/L range. Duration of exposure will depend on individual adult behaviors, *e.g.*, feeding, holding, and migrating, which modulate the toxicity experienced by an individual. Given the toxicity information, we expect some adults to show fitness level effects, however population level consequences are not anticipated.

Table 57. Salmonid risk hypotheses and population level effects.

Effects to salmonids	Individual fitness compromised (yes/no)	Population level effects anticipated (yes/no)
1. Exposure to thiobencarb via drift or runoff is sufficient to:		
a. Kill salmonids outright	yes	no
b. Reduce salmonid survival through impacts to growth	no	--
c. Reduce reproduction	no	--
d. Impair swimming	yes	no
e. Impair respiration	yes	no
f. Reduce salmonid growth through impacts on the availability and quantity of prey	yes	no
Effects to salmonid habitats		
2. Exposure to thiobencarb via drift or runoff is sufficient to:		
a. Reduce aquatic primary producers thereby affecting salmonid prey communities, salmonids, and salmonid instream cover	no	--
b. Reduce riparian vegetation to such an extent that stream temperatures are elevated, erosion increases, and reductions in natural coverage results through reduced inputs of large wood and vegetation.	yes	no
c. Reduce water quality and prey resources in estuarine environments.	no	--
Effects from other stressors of the action and contributing environmental factors		
3. Exposure to degradates of thiobencarb will cause adverse effects to salmonids and their habitats.	yes	no
4. Exposure to adjuvants, tank mixtures and other chemicals within pesticide products containing thiobencarb will cause adverse effects to salmonids and their habitats.	yes	no
5. Exposure to other pesticides present in the action area can act in combination with thiobencarb increase effects to salmonids and their habitats.	yes	no
6. Exposure to elevated temperatures will enhance the toxicity of the stressors of the action	yes	no

10. 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 by this Opinion (50 CFR 402.02). Future federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to section 7 of the ESA.

During this consultation, NMFS searched for information on future state, tribal, local, or private actions that were reasonably certain to occur in the action area. NMFS conducted electronic searches of business journals, trade journals, and newspapers using Google and other electronic search engines. Those searches produced reports on projected population growth, commercial and industrial growth, and global warming. Trends described below highlight the effects of population growth on existing populations and habitats for the Central Valley ESUs/DPSs. Changes in the near-term (five-years; 2017) are more likely to occur than longer-term projects (10-years; 2022). Projections are based upon recognized organizations producing best available information and reasonable rough-trend estimates of change stemming from these data. NMFS analysis provides a snapshot of the effects from these future trends on listed ESUs.

California's Central Valley, like other regions on the west coast is projected to have some of the most rapid growth of any area in the U.S. within the next few decades. California, Idaho, Oregon, and Washington are forecasted to have double digit increases in population for each decade from 2000 to 2030 (USCB 2005). According to U.S. Census Bureau figures, California is in step with this prediction with a growth of 10 % between 2000 and 2010. The major rice growing counties are listed in Table 58. These counties surpassed the state average for population growth. The overall growth rate was 17.4 percent. Placer County grew the most rapidly among the major rice growing counties with a 40.3 percent growth rate (Census Bureau 2010).

Table 58. Population growth rates among major rice growing counties.

County	2000 Census	2010 Census	Percent Growth
Tehama	56,039	61,136	9.1
Glenn	26,453	28,229	6.7
Butte	203,171	220,577	8.6
Colusa	18,804	21,321	13.4
Sutter	78,930	92,614	17.3
Yuba	60,219	72,925	21.1
Yolo	168,660	199,407	18.2
Placer	248,399	348,552	40.3
Sacramento	1,223,499	1,400,949	14.5
All Combined	2,084,174	2,445,710	17.4

The Delta, East Bay, and Sacramento regions are expected to increase in population by nearly three million people by the year 2020 (California Commercial 2002). Increases in urbanization and housing developments can impact habitat by altering watershed characteristics, and changing both water use and stormwater runoff patterns. The anticipated growth will occur along both the I-5 and US-99 transit corridors in the east and Highway 205/120 in the south and west. Increased growth will place additional burdens on resource allocations, including natural gas, electricity, and water, as well as on infrastructure such as wastewater sanitation plants, roads and highways, and public utilities. Some of these actions, particularly those which are situated away from waterbodies, will not require Federal permits, and thus will not undergo review through the section 7 consultation process with NMFS.

Urban runoff from impervious surfaces and roadways may contain oil, heavy metals, PAHs, and other chemical pollutants and flow into state surface waters. Inputs of these point and non-point pollution sources into numerous rivers and their tributaries will affect water quality in available spawning and rearing habitat for salmon. Based on the increase in human population growth, we expect an associated increase in the number of NPDES permits issued and the potential listing of more 303(d) waters with high pollutant concentrations in state surface waters. Continued growth into forested and other natural areas will continue the cycle of altering landscapes to the detriment of salmon habitat. Altered landscapes adversely affect the delivery of sediment and gravel and significantly alter stream hydrology and water quality.

Increased urbanization also is expected to result in increased recreational activities in the region. Among the activities expected to increase in volume and frequency is recreational boating. Boating activities typically result in increased wave action and propeller wash in waterways. This potentially will degrade riparian and wetland habitat by eroding channel banks and mid-channel islands, thereby causing an increase in siltation and turbidity. Wakes and propeller wash also churn up benthic sediments thereby potentially resuspending contaminated sediments and degrading areas of submerged vegetation. This, in turn, would reduce habitat quality for the invertebrate forage base required for the survival of juvenile salmonids moving through the system. Increased recreational boat operation in the Delta is anticipated to result in more contamination from the operation of gasoline and diesel powered engines on watercraft entering the water bodies of the Delta. California is widely known for its scenic and natural beauty. Natural areas are used recreationally by residents and tourists. Increases in use could place additional strain on the natural state of park and nature areas that are also occupied by protected species.

In the western states, which includes California, a 2.2 percent rise in agricultural output is anticipated (Figueroa and Woods 2007). Increases in agricultural practices may negatively affect riparian and wetland habitats through upland modifications that lead to increased siltation or reductions in water flow in stream channels flowing into the action area, including the Sacramento River and Delta. Grazing activities from dairy and cattle operations can degrade or reduce suitable critical habitat for listed salmonids by increasing erosion and sedimentation, as well as introducing nitrogen, ammonia, and other nutrients into the watershed, which then flow into receiving waters. Stormwater and irrigation discharges related to agricultural activities can contain a heavy nutrient load, numerous pesticides and herbicides that may negatively affect water quality, prey, salmonid reproductive success and survival rates (Dubrovsky et al. 1998). Increased output and water diversions for agriculture may also place greater demands upon limited water resources. Water diversions will reduce flow rates and alter habitat throughout freshwater systems. As water is drawn off, contaminants will become more concentrated in these systems, exacerbating contamination issues in habitats for protected species.

Mining has historically been a major component of California's economy. With national output for metals projected to increase by 4.3% annually, output of western mines should increase markedly (Figueroa and Woods 2007). Increases in mining activity will add to existing significant levels of mining contaminants entering river basins. Given this trend, we expect existing water degradation in California streams that feed into or provide spawning habitat for threatened and endangered salmonid populations will be exacerbated.

The above non-federal actions are likely to pose continuous unquantifiable negative effects on listed salmonids addressed in this Opinion. Each activity has negative effects on water quality. They include increases in sedimentation, increased point and non-point pollution discharges, decreased infiltration of rainwater (leading to decreases in shallow groundwater recharge, decreases in hyporrheic flow, and decreases in summer low flows).

Non-federal actions likely to occur in or near surface waters in the action area may also have beneficial effects on the species. They include implementation of riparian improvement measures, fish habitat restoration projects, and best management practices (*e.g.*, associated with timber harvest, grazing, agricultural activities, urban development, road building, recreational activities, and other non-point source pollution controls).

11. Integration and Synthesis for Listed Species

In this section we describe the potential for thiobencarb and its associated stressors to reduce the reproduction, numbers, or distribution of listed anadromous salmonids within California's Central Valley, taking into account the status of the three species, the environmental baseline, and cumulative effects.

Our past Biological Opinions on pesticides and Pacific salmonids assessed pesticides authorized for use across the Pacific NW and California on multiple land types, including multiple crops, which could be applied throughout the year. Due to their broad spatial and temporal use we did not conduct single crop/use analyses.

Thiobencarb's authorized use is narrow. In the Western United States it is used solely on rice grown exclusively in the Central Valley of California where soil type, temperature, other climatic factors, and water availability make rice growing possible. Extensive use information for thiobencarb has been collected by CDPR which provides primary information on when, where, and how much has been applied to rice. For these reasons, we focused the analysis both temporally and spatially to California's Central Valley which serves as habitat for three species of listed anadromous salmonids, Central Valley Spring-run Chinook, Sacramento River Winter-run Chinook, and California Central Valley Steelhead.

Threats to all three species include loss of historical spawning habitat, degradation of remaining habitat, and threats to genetic viability. Each species requires cool, clean freshwater for successful development. In the Central Valley, summer water temperatures are suitable above 150-500 m elevations; however access to these habitats is typically blocked by dams. In the case of steelhead, dams also have reduced expression of the anadromous life history form, greatly reduced the abundance of anadromous individuals, and prevented exchange of migrants among resident populations (i.e., populations located above dams).

In addition to loss of habitat, each species must contend with widespread habitat degradation and modification of both rearing and migration habitats caused by water development and land-use practices. Natal streams do not have large impassable dams, like many CV streams, but they do have small hydropower dams and water diversions that have greatly reduced or eliminated in-stream flows during critical migration periods. Threats in freshwater migratory corridors include unscreened or inadequately screened water diversions, predation by native and non-native species, excessively high temperatures, and pollution from agricultural, urban, and mining land-use.

Past hatchery practices have influenced the genetic viability of steelhead and spring-run Chinook. The Feather River spring-run populations depend on the Feather River Hatchery where production is likely a hybridization between spring-run and fall-run Chinook salmon. A declining proportion of wild CV steelhead spawners has resulted in reduced genetic viability, likely caused by increased interactions with hatchery fish. The winter-run Chinook population is the last surviving population of the species and is dependent on regulated flows of the Sacramento River. The recovery target for this species is 110,000 annual adult returns. The small number of winter-run Chinook (approximately 8000 annual natural adult returns based on the1992-2007 average) makes the species vulnerable to threats and reduces its capacity to recover following population perturbations. Sustained population declines over past decades raise concerns about the population’s genetic integrity and its capacity to recover from catastrophic events.

River and stream restoration programs, reductions of toxic materials into Central Valley waters, and improved water management are some examples of beneficial activities that have occurred in the Central Valley. Additionally, California’s regulatory agencies identified thiobencarb as an aquatic risk to both humans and aquatic life more than 20 years ago and have implemented a myriad of actions to reduce its potential to contaminate surface waters of the Central Valley. Measures include mandatory holding periods, application restrictions, a surface water monitoring program, an education program for applicators, and an enforcement program. In aggregate, these activities have been successful at reducing and tracking thiobencarb entering surface waters.

These activities have reduced exposure to toxic concentrations of thiobencarb to the three species.

We anticipate a variety of threats over the next 15 years (described in the *Cumulative Effects Section*). The human population of CA's Central Valley grew by more than 17% between 2000 and 2010 and is expected to increase annually. The more people in the Central Valley, the larger the urban and residential footprint. Listed salmonids will likely encounter altered stream and river flows, elevated water temperatures, and more pollutants. High intensity agricultural practices are expected to continue and, left unchecked, can, and do, impact riparian and wetland habitats, cause erosion and sedimentation, and contaminate surface waters with excess nutrients and pesticides. Increases in population and a continuation of high intensity agriculture will likely affect water quantity and quality, salmonid prey, and ultimately may affect the species' recovery trajectories.

We present key findings of the effects analysis which we apply to the final assessment at the species level (Table 59). For each of the three listed species, we rank the potential of thiobencarb to reduce the reproduction, numbers, or distribution as Low, Medium, or High. For threatened species, a "high" designation equates to a jeopardy determination. For endangered species, which are more vulnerable to extinction, a medium or high designation equates to a jeopardy determination. In the *Conclusion Section* we present jeopardy or no jeopardy determinations.

Table 59. Key Findings of Effects of the Proposed Action Section

Section of Biological Opinion	Summary Finding
Exposure	Individual salmonids from each of the three species are expected to be exposed to thiobencarb and its associated stressors.
Response	Anticipated levels of thiobencarb and its associated stressors are sufficient to impair several endpoints linked to individual fitness of salmonids.
Risk Characterization	Individual fitness consequences are likely, but not expected, to affect populations of the three species.
Environmental Baseline	Pesticides, elevated temperature, water flow, other pollutants, and dams are some of the stressors affecting salmonid habitats within the Central Valley. Water quality is severely degraded throughout the Central Valley. California's current risk reduction measures for thiobencarb reduced exposure to listed salmonids and their habitats.
Cumulative Effects	Increases in human population and a continuation of high intensity agriculture are anticipated within the Central Valley. Some beneficial activities are also anticipated including river restoration projects and best management practices to improve water quality.

Juveniles and adults that encounter thiobencarb, its degradates, other ingredients within formulations, tank mixture chemicals within the Central Valley may or may not experience adverse effects. Whether a salmonid is affected is a function of how much it is exposed to and for how long. The longer the exposure period the greater the likelihood of experiencing reduced fitness. Similarly, the higher the concentrations of the stressors of the action the greater the likelihood of experiencing reduced fitness.

We found that if salmonids rear during thiobencarb's high use periods proximate to rice fields, they are at risk from drift. In rice growing areas, concentrations in flood plain habitats and small streams may attain levels that impair swimming, impair breathing, kill salmonid prey, and in some cases kill salmonids. We expect these occurrences to be infrequent, although not absent. We also expect that these effects to be magnified when other pesticides are present or when water temperatures are elevated. Waters that show elevated temperatures and/or contain other pesticides that share a mode of action with thiobencarb i.e., other AChE inhibitors, effects of the action will be more pronounced and dependent on potency of the chemicals as well as the magnitude of elevated temperature.

The other high risk scenario involves early releases of thiobencarb-containing irrigation water. Rice water may be released into drainages prematurely during high rain events or from leaky dikes. High concentrations that would kill salmonids and their prey may occur if rice water is released prior to the end of holding periods, particularly rice water released within a few days of application (Table 46). The magnitude of effects depends, in part, on how soon the water is released after application of thiobencarb, how far salmonid-bearing waters are located downstream from the release, how many salmonids are exposed, and how long they are exposed. We anticipate these occurrences as rare given the information reviewed and the current federal and state regulations already in place.

Central Valley Spring-run Chinook (Threatened species)

The ESU includes three populations in the upper Sacramento River and its tributaries as well as a population in the Feather River. The spatial distribution has been greatly reduced through extirpation of populations and dams blocking fish passage. Genetic diversity was similarly reduced with the extirpation of all San Joaquin runs. Abundance levels are severely depressed from historic estimates. Time series data show that the three tributary populations currently have growth rates just above replacement while the Feather River population is dependent on hatchery augmentation.

We rank the potential for thiobencarb exposure to cause species level consequences to CV Spring-run Chinook as low for the following reasons:

- Exposure of migrating and spawning adults to thiobencarb is limited. Adults returning to their spawning habitats do migrate through rice areas of the Central Valley and are likely to be exposed to thiobencarb based on their high relative abundance during peak thiobencarb applications (Table 28). However, while the levels and durations of thiobencarb exposure experienced by migrating adults may adversely affect individuals, they are not expected to affect populations because exposures are expected to be confined to infrequent incidents of early release of water from treated fields and drift of liquid formulations to floodplain habitats (Table 36).

- Exposure of juveniles to thiobencarb is limited. While some individuals may experience adverse effects of exposure, the levels and durations of such exposure are not expected to affect juvenile populations because rearing and migrating juveniles have a low probability of exposure to thiobencarb applied near their habitat (reference table 7, table 8).
- Anticipated effects from reduced riparian zone function will likely be limited based on the narrow application period of thiobencarb May-June and will primarily be affected where riparian areas are exposed to drift.

Sacramento River Winter-run Chinook (Endangered species)

The Sacramento River Winter-run Chinook salmon ESU is now comprised of a single population. As there is only one population, any population level effects will directly impact the species. This population spawns and rears in the main stem of the Sacramento River below Keswick Dam. Abundance and productivity have fluctuated greatly over the past two decades. The genetic diversity of this population has been reduced substantially through small population sizes and the influence of hatchery fish. The large fluctuations in productivity and abundance indicate that the species is highly vulnerable to extinction.

We rank the potential for thiobencarb exposure to cause species level consequences to Sacramento River Winter-run Chinook as low for the following reasons:

- Exposure of migrating and spawning adults to thiobencarb is limited. While some adults migrating through rice growing areas will be exposed to thiobencarb, all known spawning is located at higher elevations than rice fields (current Figure 6). We expect that some individuals will be exposed to thiobencarb drift and runoff during migration (current Tables 9, 10), possibly experiencing adverse effects, but, overall, we do not expect population level consequences for adult winter-run Chinook (Table 36).
- Exposure of juveniles to thiobencarb is limited. Spawning grounds are located up-stream of rice growing areas (Current Figure 6), and most juvenile migrations through Central Valley rice-growing areas do not occur during the time period when thiobencarb is

applied (Snider and Titus 2000, Bajjaliya and Vincik, 2008, USFWS 2010, 2012). Thus, the probability of exposure to thiobencarb via drift and runoff into aquatic habitats in close proximity to rice fields is low.

- Anticipated effects from reduced riparian zone function will likely be limited based on the narrow application period of thiobencarb May-June and will primarily be affected where riparian areas are exposed to drift.

California Central Valley Steelhead (Threatened species)

The DPS consists of 81 historical and independent populations. Its spatial structure has been greatly reduced by loss of habitat diversity and tributary access from dams. Available information shows a significant long-term downward trend in abundance (NMFS, 2009a). Population losses and reduction in abundance have reduced the genetic diversity that existed within the DPS.

We rank the potential for thiobencarb exposure to cause species level consequences to Central Valley steelhead as low for the following reasons:

- Exposure of migrating and spawning adults to thiobencarb is limited. Co-occurrence of thiobencarb application with spawning adults is unlikely based on the location of known spawning areas relative to where rice is grown (current Figure 7). There is also little temporal overlap between thiobencarb application and adult migration through rice growing areas (currently Table 11).
- Exposure of rearing and migrating juveniles to thiobencarb is limited. Co-occurrence of thiobencarb with egg incubation, fry emergence, and early rearing is unlikely based on the location of known spawning areas relative to where rice is grown (current Figure 7). There may be some exposure of outmigrating juveniles to thiobencarb, especially in the San Joaquin and Mokelumne Rivers, however very little information is available on steelhead use of these rivers (current Tables 11, 12). Overall, while some individual juveniles may experience adverse effects during migratory periods, we do not expect population level consequences (Table 36).

- Anticipated effects from reduced riparian zone function will likely be limited based on the narrow application period of thiobencarb May-June and will primarily be affected where riparian areas are exposed to drift.

Table 60. Summary Findings for Species

Species	Co-occurrence of action with rice growing areas in California	Potential for reduction in reproduction, numbers, distribution	
		Populations	Species
Central Valley Spring-run Chinook (T)	Yes	Low	Low
Sacramento River Winter-Run (E)	Yes	Low	Low
California Central Valley Steelhead (T)	Yes	Low	Low

T = threatened; E = endangered

12. Effects of the Proposed Action on Designated Critical Habitat

NMFS' critical habitat analysis determines whether the proposed action will likely destroy or adversely modify critical habitat for ESA-listed species by examining potential reductions in the conservation value of the essential features of designated critical habitat. Our analysis does not rely on the regulatory definition of "adverse modification or destruction" of critical habitat. Instead, we rely on the statutory provisions of the ESA, including those in section 3 that define "critical habitat" and "conservation", those in section 4 that describe the designation process, and those in section 7 setting forth the substantive protections and procedural aspects of consultation. In this section NMFS evaluates the potential consequences to designated critical habitat from exposure to the stressors of the proposed action. The pathway of analysis is presented in Figure 40. It is similar in structure to the jeopardy analysis, but focuses on whether the proposed action is likely to destroy or adversely modify designated critical habitat for listed Pacific salmonids. We first determine the potential for critical habitat to be exposed to the stressors of the proposed action. If we conclude that critical habitat is likely to be exposed, we assess the consequences of that exposure on the quality, quantity, or availability of one or more of those primary constituent elements (PCEs) that comprise critical habitat. Water quality, forage (prey availability), and natural cover (riparian plants) are key attributes of salmonid PCEs that are susceptible to the stressors of the action. Water quality encompasses a range of typically measured parameters, including dissolved oxygen, temperature, turbidity, and presence of chemical contaminants in sufficient concentrations to adversely affect aquatic organisms.

Because the proposed action would degrade water quality by introducing thiobencarb and other chemicals, we evaluate concentrations of thiobencarb likely to adversely affect fish, prey, and terrestrial and aquatic plants as measures of degraded water quality. This analysis is conducted by comparing toxicity information reviewed earlier in the Response section with expected concentrations in salmonid habitats. Similarly, we evaluate adverse effects to salmonid prey to determine the effects of the action on prey availability and forage, a key attribute for many salmonid PCEs.

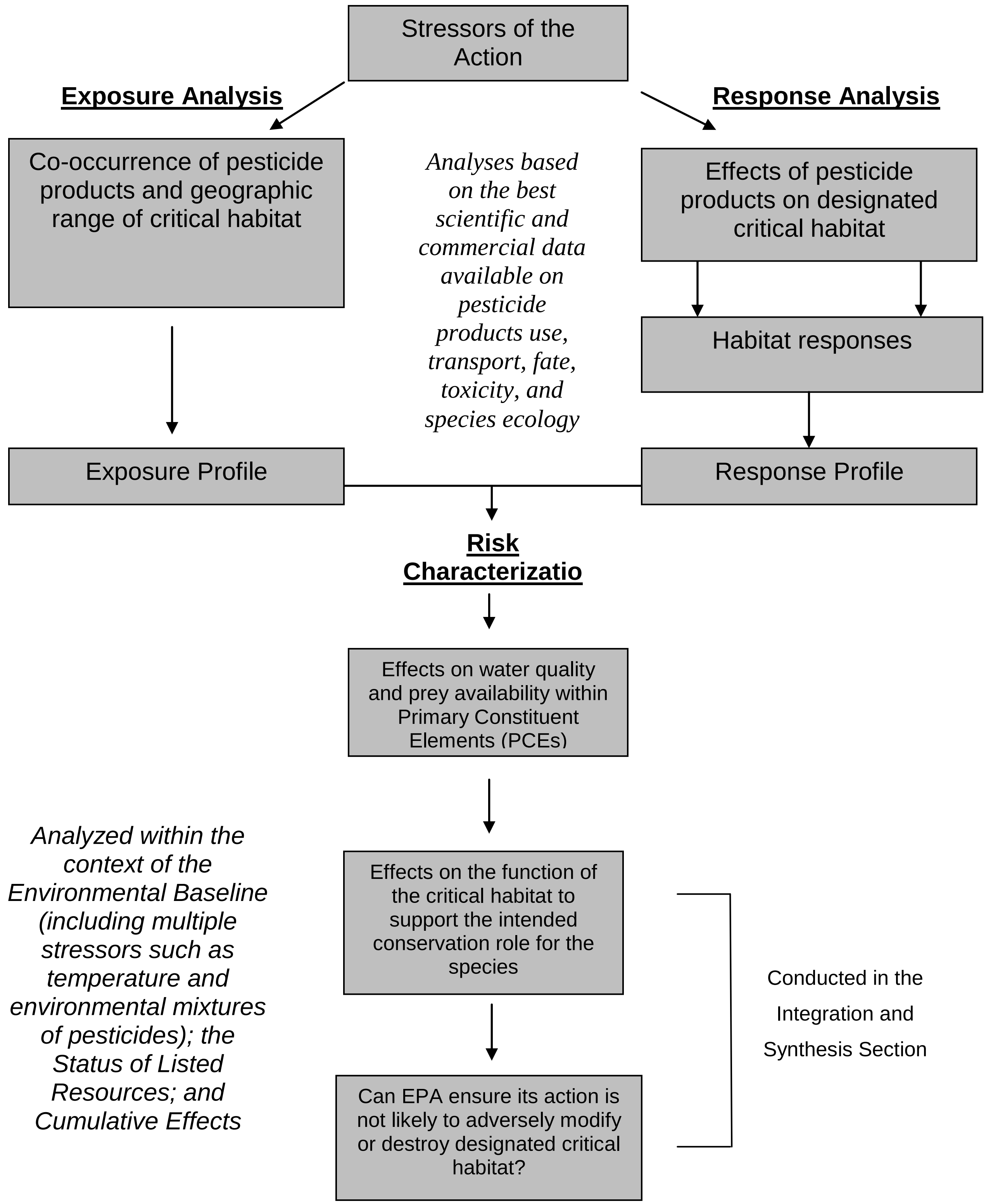


Figure 40. Assessment Framework for Designated Critical Habitat Assessment Framework for Designated Critical Habitat.

We formulated several risk hypotheses to assess potential changes in PCEs of designated critical habitat based on: 1) the likely concentrations that would be observed where critical habitat is exposed to chemicals derived from pesticide applications; and 2) the response of PCEs to the anticipated concentrations.

NMFS used the assigned conservation values (high, medium, and low) of watersheds within each ESU/DPS for the PCEs of critical habitat identified for each life stage common to listed salmonids (described in the *Status of Listed Resources* section). Because watersheds with high conservation value are essential to the conservation of the species, reductions in the quantity, quality, or distribution of the PCEs supporting that watershed would be expected to adversely affect the function of critical habitat to support its intended conservation role. We assess these watersheds within the *Integration and Synthesis for Designated Critical Habitat* section.
NMFS has designated critical habitat for each of the three species. The action area for this Opinion encompasses all designated critical habitat for the three listed Pacific salmonids in the Central Valley. The PCEs for each listed species are described in the *Status of Listed Resources* section of this Opinion. As the species of salmonids addressed in this Opinion have similar life history characteristics, they share many of the same PCEs. These PCEs include sites that support one or more life stages (sites for spawning, rearing, migration, and foraging) and contain physical or biological features essential to the conservation of the ESU/DPS, including:

1. freshwater spawning sites;
2. freshwater rearing sites;
3. freshwater migration corridors;
4. estuarine areas;
5. nearshore marine areas[23]; and
6. offshore marine areas.

[23] Nearshore marine areas are free of obstruction and excessive predation with: (i) water quality and quantity conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation; and (ii) natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, and side channels (70 FR 52488; 73 FR 7816).

Water quality, prey availability (forage), and natural cover in freshwater and estuarine areas are susceptible to the effects of the proposed action where they overlap with the stressors of the action. Effects to water quality and prey availability will be evaluated to determine the likelihood of reducing the quality of PCEs such as spawning and rearing sites, or migration corridors. Given the use and environmental fate profile of the pesticide formulations containing thiobencarb, we do not expect offshore marine areas and nearshore marine areas to be affected. Therefore, a risk hypothesis was not developed for offshore marine areas or for nearshore marine areas and further evaluation of these PCEs are unwarranted.

Good water quality is a necessary attribute of all PCEs to support the conservation role of designated critical habitat. Water quality is clearly degraded when pesticides and other stressors of the action reach levels in salmonid habitat that are sufficient to affect aquatic organisms, including those that reduce individual fitness of exposed salmonids. Impacts to salmonid fitness were evaluated earlier in the document and these impacts are used as indicators of degraded water quality. We evaluate exposure and effect concentrations presented earlier in the *Effects of the Proposed Action* section to determine whether PCEs are affected. We re-evaluate the information to determine potential effects to PCEs.

We also evaluate effects on salmonid prey because forage is an essential attribute of all PCEs except in spawning sites. Freshwater juvenile rearing and migratory habitats as well as some estuarine and nearshore marine areas must provide sufficient forage that support salmonid growth and development in the Central Valley. Reductions in the abundance of prey items can decrease the quality of rearing, migration, and estuarine PCEs, as they will support fewer individuals, especially during a salmonid's first year of survival. Reductions in prey can reduce a PCE's potential to support salmonids (juvenile development, growth, maturation, survival), thereby reducing the carrying capacity of critical habitat.

We evaluated toxicity assessment endpoints including prey survival (EC_{50}/LC_{50}), prey growth, prey drift, prey reproduction, prey abundance, health condition of invertebrate aquatic communities (using indices of biological integrity), aquatic primary producers, terrestrial plants in riparian zones, and recovery of aquatic communities following thiobencarb exposure to determine whether expected concentrations are sufficient to affect PCEs for salmonid critical habitats.

Exposure of designated critical habitats to the stressors of the action:
Designated critical habitat for the three species is located within the action area. Many freshwater areas overlap with the allowable uses of thiobencarb authorizations (Appendix 6). The stressors of the action contaminate these habitats primarily via drift and runoff (including from irrigation returns), and to a lesser extent from atmospheric deposition. Once in salmonid habitats of the Central Valley, thiobencarb persists for a period of time; the extent of which is dependent on the chemical, biological, and physical environment of the contaminated aquatic habitats. Expected concentrations of other/inert ingredients and adjuvants added to formulations prior to application remain unknown - a data gap.

Table 49 (summary exposure section depicting concentrations) shows expected concentrations of thiobencarb and its degradates that were derived from EPA BEs, EPA incident data, surface water monitoring data, and NMFS exposure modeling estimates. These data will be discussed in the context of spawning, rearing, migrating, and estuarine PCEs. The vast majority of exposure information applies more readily to freshwater habitats compared to estuarine where much less information is available.

Responses of salmonid habitats to the stressors of the action
If PCEs are exposed to the stressors, we evaluate the level at which thiobencarb, any toxic degradates, and inert/other ingredients adversely affect water quality, prey availability, and natural cover (terrestrial and aquatic primary production). For many of the other ingredients, recommended tank mixtures, and degradates, not only was there no available exposure information, but also little to no toxicity information. In the *Response* and *Risk Characterization* sections of the *Effects of the Proposed Action*, we showed that applications of thiobencarb can result in concentrations that may reduce salmonid survival, growth, swimming, respiration, and reduce riparian vegetation, all which independently translate to a degradation of water quality Table 53. These types of individual fitness consequences demonstrate a degradation of water quality in affected habitats.

We summarized the available toxicity information in the *Response Analysis* (Table 53). It is important to note that the toxicity of thiobencarb is variable depending on the biological endpoint (*e.g.*, acute lethality to fish and invertebrates), the levels expected in salmonid habitats, the presence of other AChE-inhibiting pesticides, and whether elevated water temperatures occur.

12.1. Risk Characterization

NMFS reviews the status of designated critical habitat affected by the proposed action separate from species effects by examining the condition and trends of primary constituent elements (PCEs) throughout the action area. PCEs are the physical and biological features identified as essential to the conservation of listed species. The PCEs for salmonids are identified in The *Status of the Species* section. We determined the PCEs at risk from the use of thiobencarb are freshwater rearing, freshwater migration, and time spent in estuarine areas. Spawning areas for each of the species are upstream of current rice growing areas. Therefore, the freshwater spawning PCE is not considered at risk from this action. Also, considering the use of thiobencarb is limited to rice which is more than 40 miles from the California coast, we do not consider nearshore or offshore marine PCEs at risk from this action.

We use the toxicity information presented earlier in the *Response* section to evaluate the scientific lines of evidence that support or refute risk hypotheses developed for designated critical habitats. Freshwater rearing sites and migration corridors within designated critical habitats are likely to be exposed to the stressors of the action over the 15-year registration duration. We estimate expected concentrations and durations of exposure for these habitats based on pesticide use information, surface water monitoring data, EPA modeling estimates, and NMFS modeling estimates. For each risk hypothesis below, we qualitatively weigh the evidence to determine whether the PCE attributes of water quality and/or prey availability are affected. We ultimately determine whether the degradation of water quality, prey availability, and natural cover within freshwater rearing and migration corridors will rise to the level expected to reduce the intended conservation role of designated critical habitats, which is evaluated within the

Integration and Synthesis for Designated Critical Habitat section. The final conclusion of whether EPA's proposed action with thiobencarb containing end use products are likely to adversely modify or destroy a species' designated critical habitat is provided in the *Conclusion* section.

Risk hypothesis 1. Exposure to the stressors of the action is sufficient to degrade water quality, natural cover, and /or reduce prey availability in freshwater rearing and migration areas.

Freshwater rearing and migration areas need to provide good water quality, abundant forage, and natural cover to support juvenile development, mobility and survival. Reductions in any of these attributes from the stressors of the action can limit the existing and potential carrying capacity of rearing and migration areas and subsequently reduce their conservation value. Recovery of listed salmonid populations in the Central Valley is tied closely to the success of juveniles to fully develop, mature, and grow during freshwater residency periods. All species of Pacific salmonids spend some amount of time in freshwater feeding and rearing areas. Chinook and steelhead spend much longer periods rearing in freshwater systems with steelhead trout spending up to several years before migrating to the ocean.

Freshwater rearing areas are diverse, extensive, and complex sites that can range from small, shallow, intermittent floodplain habitats to channel edges of large river systems such as the Sacramento River. Salmon recovery plans also call for floodplain connectivity and other restoration activities to improve rearing. Many freshwater salmonid rearing sites are located in floodplains where shallow, low flow habitats are at high risk of thiobencarb drift and runoff. The Yolo Bypass, when inundated, is one example. Habitats such as the Yolo provide abundant prey and are important foraging areas for developing juveniles (Sommer et al. 2001).

Expected floodplain concentrations of thiobencarb are shown in the *Exposure and Response Integration* section above (Table 53). At these concentrations, water quality would be affected because concentrations exceed toxicity thresholds for fish and aquatic invertebrate survival as well as for plankton and aquatic macrophyte growth and survival. Additionally, we expect modification to riparian and floodplain vegetation will result in decreased prey production, decreased shading of these habitats, and an increase in the occurrence of elevated temperatures that are stressful to salmonids in freshwater rearing sites.

The habitats most likely affected would be those areas that receive primary drift from application or direct runoff from early releases of treated rice irrigation water. In these circumstances, thiobencarb may reduce forage availability because concentrations can reach levels that kill prey and affect their growth and reproduction (shown in the *Exposure and Response Integration* section above (Figure 39).

Thiobencarb is applied primarily in May, and rice discharge waters are typically released in June or July after the required holding period. As previously stated, the Yolo Bypass floodplain is an important rearing area for juvenile salmonids when inundated by flood waters. Rice planting and subsequent thiobencarb application in the Yolo occurs after the water has receded and the salmonids are gone.[24] During the rice growing period, the Yolo ceases to be an available rearing area. While drift or runoff of thiobencarb may affect some salmonid prey and natural cover on the Yolo, we do not know to what extent the vegetation and prey will rebound prior to subsequent flood events.

The extent surface waters flowing through rice growing areas are used by salmonids for rearing (*i.e.*, not merely a migratory corridor) is unknown. We presume rearing occurs within rice growing areas. Rearing in the Sacramento River for Chinook and steelhead is likely along shallow edge habitats where emergent vascular plants and/or over-hanging vegetation provide shelter and prey resources. We would expect prey to be adversely affected in shallower areas receiving direct drift or early release of rice irrigation waters. Water quality would be adversely affected at those sites where runoff and drift occur.

We also anticipate that in some situations, surface waters in the CA Central Valley receiving thiobencarb drift or early release of rice irrigation waters will alter characteristics of the plant

[24] Some Chinook may be entrained in perennial ponds within the Yolo floodplain. These fish would be exposed to excess temperatures and predation and are considered lost to the population Sommer, T., Harrell, W., and Nobriga, M. 2005a. Habitat use and stranding risk of juvenile Chinook salmon on a seasonal floodplain. *North American Journal of Fisheries Management*, 25(4): 1493-504.

communities in freshwater rearing and migration areas, thereby decreasing the availability of cover for salmonids and their prey. Aquatic plants provided natural cover for salmonids as well as prey habitat. Thiobencarb is acutely toxic to a range of terrestrial and primary producers. Aquatic primary producers showed reduced growth (EC_{50}s) at concentrations as low as 17 µg/L and as high as 4000 µg/L (Table 51). Both values were reported for green algae. No studies were available that tested aquatic community responses to thiobencarb in running water systems and none that evaluated salmonid habitats within the Central Valley. The bioassay results indicate that following 48-h continuous exposure to thiobencarb, reductions in aquatic primary producers can occur. Aquatic vegetation provides cover to salmonids from avian predation as well as protection from larger fish. In addition, a diverse vascular plant community provides important substrate for an array of insect species upon which young salmon prey. We conclude that effects to primary producers are likely, particularly in salmonid habitats receiving thiobencarb drift from aerial applications and those areas exposed to early release of rice water.

Collectively the data indicate that expected concentrations of thiobencarb from drift and early release of treated rice water are sufficient to adversely affect water quality, salmonid prey, and natural cover attributes of freshwater rearing and migration PCEs. Therefore, we evaluate these effects in order to determine whether the conservation value of species' designated critical habitats will be reduced (see *Integration and Synthesis for Designated Critical Habitat* section below). The final conclusion of whether EPA's proposed action with thiobencarb containing end use products are likely to adversely modify or destroy a species' designated critical habitat is provided in the *Conclusion* section.

Risk hypothesis 2. Exposure to the stressors of the action is sufficient to degrade riparian areas adjacent to rearing and migration corridors.

Few studies were found on the effects of thiobencarb on riparian and aquatic (non-target) vegetation. Registrant submitted data indicated that thiobencarb is highly toxic to non-target plants where EC_{25}s ranged from 0.019 – 3.1 lbs/A active ingredient. In a field study, tolerant and susceptible species of non-target bayou and levee vegetation adjacent to rice fields were recorded following treatment with Bolero 8 EC (Lauck 1979). All species of levee vegetation examined had emerged and were in a mature stage of development and new foliage of the bayou woody plants was rapidly expanding during the period of Bolero usage. The author did not evaluate long term effects to plants. All species had been directly exposed to Bolero by an aerial application at the label rate, 4 lbs/A. One plant showed symptoms of injury due to Bolero applications, reed canary grass. Growth and maturity was reduced by 55 percent relative to untreated plants (Lauck 1979). Based on these results, we conclude that grasses and sedges in riparian areas may be susceptible to adverse effects in areas receiving drift from thiobencarb applications. Vascular plants and other macrophytes are important structural elements that provide important cover to salmonids in their preferred freshwater rearing areas.

It is unclear how established woody plants in the riparian corridor of the Sacramento and San Joaquin may respond to varying concentrations of thiobencarb containing drift. We can only presume that, at a minimum, new growth may be adversely affected. Drift may also adversely affect the growth performance of seedling trees. If this were the case, long-term health and function of those riparian zones receiving drift annually may be impaired. In extreme cases, where woody vegetation is eliminated from riparian zones to the extent that bank destabilization occurs, altered stream hydrology could affect the availability of other cover including rocks, side channels, and undercut banks. One extreme example suggested that the application of an herbicide to a riparian zone caused major long-term changes to the hydrology of a stream and degraded fish habitat (http://water.epa.gov/scitech/datait/tools/warsss/streamero.cfm). Herbicide-induced changes to vegetative communities in the riparian zone and aquatic habitat also have implications for the availability of prey as salmonids consume both terrestrial and aquatic insects. As plant communities are modified in riparian zones, the species that rely on them will also be affected. Habitat responses in these environments are expected to be variable and will depend on the sensitivity of existing plants to thiobencarb in conjunction with the other stressors in the environment.

Collectively expected concentrations of thiobencarb from drift are sufficient to adversely affect riparian areas adjacent to rearing and migration corridors affecting water quality, salmonid prey, and natural cover attributes of freshwater rearing and migration PCEs. Therefore, we evaluate these effects in order to determine whether the conservation value of species' designated critical habitats will be reduced (see *Integration and Synthesis for Designated Critical Habitat* section below). The final conclusion of whether EPA's proposed action with thiobencarb-containing end use products are likely to adversely modify or destroy a species' designated critical habitat is provided in the *Conclusion* section.

Risk hypothesis 3. Exposure to the stressors of the action is sufficient to degrade water quality, natural cover, and/or reduce prey availability in estuarine areas.

Estuarine areas require good water quality to support juvenile and adult physiological transitions between fresh water and salt water as well as to provide juvenile and adult cover and prey resources sufficient to support survival, growth, and maturation. Prey resources for Pacific salmonids within estuaries include a diverse group of organisms from aquatic invertebrates to small fishes. The allowable uses of the stressors of the action overlap with estuaries designated as critical habitat (Kuivila and Jennings 2007).

Contamination of estuaries occurs via drift, runoff, and atmospheric deposition. Streams and rivers flowing into estuaries act as conveyor belts as they transport the stressors of the action from areas higher in watersheds (Johnson et al. 1997). EPA did not provide estimates for thiobencarb concentrations in estuarine habitats. A study to characterize the temporal inputs of thiobencarb (along with five other pesticides) into San Francisco Bay found thiobencarb in 28 percent of the samples. The samples were taken 8 km downstream of the confluence of the Sacramento and San Joaquin rivers beginning in mid-May, following the release of rice-field water (Kuivila and Jennings 2007). Water samples were collected daily or twice-daily from mid-January through mid-July. The maximum concentration detected was 66 ng/L in late May. It is impossible to determine how representative this value is relative to other areas in the estuary that may be occupied by listed salmonids, or relative to other values on subsequent years. However, the sample site selected receives the majority of flow from the Sacramento prior to entering the San Francisco Bay (Kuivila and Jennings 2007). The maximum value reported, 66 ng/L, is very low relative to freshwater toxicity endpoints we reviewed (Table 53). The occurrence of thiobencarb followed the application season (Figure 24), and the concentration of thiobencarb spiked following release of rice-field water.

The available toxicity information for estuarine and marine organisms for thiobencarb is presented in Table 61 as reported in EPA's Red Legged Frog BE for thiobencarb. The estuarine aquatic toxicity data are from survival assays for the Atlantic silverside (fish) and mysid (estuarine invertebrate), and chronic results were reported for the Atlantic silverside and Opossum shrimp. The available studies indicate a similar range of sensitivity to thiobencarb. The toxicity values reported are three orders of magnitude greater than concentrations found in the estuary (Kuivila and Jennings 2007). Without additional toxicity studies on specific prey organisms residing in the Sacramento-San Joaquin estuary, it is impossible to say for sure how thiobencarb exposures similar to those reported would affect salmonid prey. However, if responses are similar to those found for the opossum shrimp, concentrations would have to be 100 times greater for an observable effect. It is unlikely that concentrations reported would be exceeded elsewhere in the estuary. Collectively, data do not suggest that estuarine PCEs are adversely affected, therefore we do not carry this hypothesis forward in our analysis of impacts to the conservation value of designated critical habitat.

Table 61. Thiobencarb's assessment endpoint toxicity values (µg/L) for saltwater aquatic organisms presented in California Red-Legged Frog (CRLF) BEs.

Assessment endpoint	Test	Species	Toxicity Value (µg/L)	Citation
Estuarine/marine fish	Acute	Atlantic silverside *Menidia menidia*	96-h LC_{50} = 204 (96 h)	(EPA 2009b)
	Chronic	Atlantic silverside *Menidia menidia*	NOEC = 6 (estimated)	(EPA 2009b)
Estuarine/marine invertebrates	Acute	Mysid *Americamysis bahia*	96-h LC_{50} = 150 (110 -200)	(EPA 2009b)
	Chronic	Opossum Shrimp *Neomysis mercedis*	NOEC = 3.2 LOEC = 6.2	(EPA 2009b)

Summary of the Effects of the Action on PCEs

We conclude that the available information on exposure and response of aquatic habitats to the stressors of the action supports two of the three risk hypotheses (Table 62). We expect essential physical and biological features will be reduced in rearing and migratory habitats, and that riparian vegetation will be negatively affected. Next, within the *Integration and Synthesis of Effects to Designated Critical Habitat* section, we evaluate whether these adverse changes to PCEs affect the conservation value of designated critical habitat.

Table 62. Risk to the primary constituent elements (PCEs) from stressors of the proposed action.

PCE	Essential Physical or Biological Feature	PCE at Risk from Stressors of the Action?
Freshwater Spawning	Water Quality	No
	Water Quantity	No
	Substrate	No
Freshwater Rearing	Water Quantity	No
	Floodplain Connectivity	No
	Water Quality	Yes
	Forage	Yes
	Natural Cover	Yes
Freshwater Migration	Free of Obstructions	No
	Water Quality	Yes
	Water Quantity	No
	Natural Cover	Yes
Estuarine Areas	Free of Obstruction	No
	Water Quality	No
	Water Quantity	No
	Salinity	No
	Forage	No
	Natural Cover	No
Nearshore Marine	Free of Obstruction	No
	Water Quality	No
	Water Quantity	No
	Natural Cover	No
	Forage	No
Offshore Marine	Water Quality	No
	Forage	No

12.2. Integration and Synthesis for Designated Critical Habitat

This section describes NMFS' assessment of the likelihood that EPA's registration of thiobencarb will destroy or adversely modify designated critical habitat for three salmonids species covered in this Opinion. Each of the three species has similar PCEs, as described in the *Effects to Designated Critical Habitat section*. These PCEs are sites that support one or more life stages and include:

1. freshwater rearing sites;
2. freshwater migration corridors;
3. estuarine areas;
4. nearshore marine areas[25]; and
5. offshore marine areas.

We determined the PCEs at risk from thiobencarb are freshwater rearing and freshwater migration sites (Table 63**).** These sites contain physical or biological features essential to the conservation of the ESU/DPS. Physical features include cover, substrate, water temperature, and water quality. Biological attributes include forage i.e., prey. We do not expect freshwater spawning, estuarine areas, nearshore marine or offshore marine areas to be affected by the stressors of the action.

As noted in salmonid recovery plans, salmonids require cool water, free of contaminants. Water free of contaminants promotes normal fish behavior for successful migration, spawning, and juvenile rearing. In the juvenile life stage, salmonids also require stream habitat providing adequate forage. Sufficient forage is necessary for juveniles to maintain growth which subsequently reduces freshwater predation mortality, increases overwintering success, supports smoltification, and improves their ocean survival.

[25] Nearshore marine areas are free of obstruction and excessive predation with: (i) water quality and quantity conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation; and (ii) natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, and side channels (70 FR 52488; 73 FR 7816).

We expect thiobencarb and its degradates, other ingredients within its formulations, tank mixture chemicals to affect natural cover (riparian plants), water quality, and forage (prey abundance). Destruction or adverse modification of designated critical habitat is evaluated based on whether the stressors of the action are expected to cause appreciable reductions in natural cover, water quality, or prey abundance at juvenile freshwater rearing and migration sites. The stressors of the action include thiobencarb, degradates, inert/other ingredients in formulations, surfactants, and tank mixtures; and their individual and collective interactions when applied to rice fields in California's Central Valley. Data are not available for some of these other stressors, thus they are evaluated in a qualitative fashion.

Table 63. Key Findings from the Effects of the Proposed Action Section to Critical Habitat

Section of Biological Opinion	Summary Finding
Exposure	Habitats of the three species are expected to be exposed to thiobencarb and its associated stressors.
Response	Anticipated levels of thiobencarb and its associated stressors are sufficient to impair several biological and physical attributes of primary constituent elements.
Risk Characterization	Freshwater rearing and freshwater migration primary constituent elements are expected to be reduced.
Environmental Baseline	Pesticides, elevated temperature, water flow, other pollutants, and dams are some of the stressors affecting salmonid habitats within the Central Valley. Water quality is severely degraded throughout the Central Valley. California's current risk reduction measures for thiobencarb reduced exposure to designated critical habitats.
Cumulative Effects	Increases in human population and a continuation of high intensity agriculture are anticipated within the Central Valley. Some beneficial activities are also anticipated including river restoration projects and best management practices to improve water quality.

Responses of designated critical habitat to thiobencarb and its associated chemical stressors is a function of exposure concentrations and durations. The longer or more frequent the exposure the greater the likelihood of affecting PCEs. We found that designated critical habitats proximate to rice fields are at the greatest risk from thiobencarb containing drift. Drift is expected during application of the liquid formulations; however we expect minimal drift from granular formulations. Based on CDPR use statistics, granular applications comprise approximately 75% and liquid applications comprise the remaining 25%. In rice growing areas of the Central Valley, concentrations from spray drift in flood plain habitats and small streams may attain levels that impair swimming, impair breathing, kill salmonid prey, and in some cases kill salmonids. Additionally, we anticipate some riparian plants to die or experience reduced growth from drift. Overall, we expect these occurrences to be infrequent, although not absent. We also expect in stream effects to be magnified when other pesticides are present or when water temperatures are elevated.

The other high risk scenario involves early releases of thiobencarb-containing irrigation water. Rice water may be released into drainages prematurely during high rain events or from leaky dikes. High concentrations that would kill salmonids, their prey, inhibit aquatic primary producers, all of which degrade water quality may occur if rice water is released prior to the end of holding periods, particularly rice water released within a few days of application (Table 53). The magnitude of effects depends, in part, on how soon the water is released after application of thiobencarb, how far salmonid-bearing waters are located downstream from the release, and how long they are exposed. We anticipate these occurrences as rare given the information reviewed and the current state regulations already in place (Table 43).

We used a GIS overlay to determine potential overlap of rice growing areas and designated critical habitat distributions for the three species to (Appendix 6). This overlay is a different approach compared to earlier Opinions on EPA's registrations of pesticides where we used land use classifications i.e., agricultural, forestry, urban/developed, etc. were used rather than a specific crop. We selected the current approach because thiobencarb is used exclusively on rice in California's Central Valley. In the Effects of the Proposed Action Section for critical habitat, we found that freshwater rearing and freshwater migration PCEs for the three species were at risk, particularly water quality, forage, and natural cover (Table 38: Risk to the PCEs). The GIS overlay shows that current rice fields overlap extensively with portions of each species' designated critical habitats. The critical habitats that overlap rice areas have been ranked as having "high" conservation value (Appendix 6).[26]

[26] At this point in time, conservation values within the designated critical habitat for Sacramento River winter-run Chinook have not been assigned. For this Opinion, we assumed each watershed to have a high conservation value.

We evaluate whether anticipated effects to PCEs reduce appreciably the overall conservation value of the species' designated critical habitat given the current status, environmental baseline, and cumulative effects. All three species designated critical habitats include the Sacramento River and its tributaries that overlap with rice growing in the Central Valley. Spawning, rearing and migration areas are currently degraded by elevated temperatures and lost access to historic spawning sites. Rearing PCEs in the Sacramento River are degraded by loss of floodplain habitats, thereby reducing foraging opportunities. Migration PCEs are degraded by lack of natural cover along the migration corridors and degraded water quality from a host of chemical and non-chemical stressors. Juvenile migration is further affected by water diversions and by two large state and federal water-export facilities along the Sacramento- San Joaquin Delta. In aggregate, many of the designated critical habitats within the Central Valley are in a degraded state and expected to remain so throughout the 15 years of the action.

Waters that show elevated temperatures and/or contain other pesticides that share a mode of action with thiobencarb i.e., other AChE inhibitors, effects of the action will be more pronounced and dependent on potency of the chemicals as well as the magnitude of elevated temperature.

Water quality, forage, and riparian plants were the assessment endpoints expected to be affected by use of thiobencarb on rice. We do not anticipate these endpoints to be degraded to the degree that would reduce appreciably the high conservation value of rearing and migratory PCEs (Table 64). The high risk scenarios of spray drift and early release of thiobencarb-containing rice water are expected to be infrequent. Additionally, the toxic levels necessary to kill and impair plants are not expected in the vast majority of designated critical habitats.

Table 64. Summary findings for Designated Critical Habitats

Species	Co-occurrence of action with rice growing areas in California	Designated critical habitat	
		PCEs affected	Appreciably reduce conservation values
Central Valley Spring-run Chinook (T)	Yes	Yes	No
Sacramento River Winter-Run (E)	Yes	Yes	No
California Central Valley Steelhead (T)	Yes	Yes	No

T = Threatened; E = Endangered

13. Conclusion

After reviewing the current status of Central Valley Spring-run Chinook, Sacramento River Winter-run Chinook, and California Central Valley Steelhead, the environmental baseline for the action area, the effects of the proposed action, and the cumulative effects, it is NMFS' Opinion that the registration of thiobencarb is not likely to jeopardize the continued existence of these endangered or threatened species nor is it likely to destroy or adversely modify designated critical habitat of these three species (Table 65).

Table 65. Conclusions for EPA's Reregistration of Thiobencarb

Species	ESU	Jeopardy	Adverse Modification/Destruction of Designated Critical Habitat
Chinook	Central Valley Spring-run	No	No
	Sacramento River Winter-run	No	No
Steelhead	California Central Valley	No	No

14. Incidental Take Statement

Section 9(a)(1) of the ESA prohibits the taking of endangered species without a specific permit or exemption. Protective regulations adopted pursuant to section 4(d) of the ESA extend the prohibition to threatened species. Take is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt to engage in any such conduct (50 CFR 222.102). Harm is further defined by NMFS to include significant habitat modification or degradation that results in death or injury to listed species by significantly impairing essential behavioral patterns, including breeding, feeding, or sheltering. Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity conducted by the Federal agency or applicant (50 CFR 402.02). Under the terms of section 7(b)(4) and section 7(o)(2), taking that is incidental to and not intended as part of the agency action, whether implemented as proposed or as modified by reasonable and prudent alternatives, is not considered to be prohibited taking under the ESA provided that such taking is in compliance with the terms and conditions of this Incidental Take Statement.

14.1. Amount or Extent of Take

As described earlier in this Opinion, this is a consultation on the EPA's registration of pesticide products containing thiobencarb and their formulations as they are used in California's Central Valley on rice and the effects of these applications on three listed ESUs/DPSs of Pacific salmonids. The EPA authorizes use of thiobencarb in the western U.S. in California on rice only as described in the *Description of the Proposed Action* and elsewhere in the document. The goal of this Opinion is to evaluate the impacts to NMFS' listed resources from the EPA's broad authorization of applied pesticide products. This Opinion is a partial consultation because pursuant to the court's order, EPA sought consultation on only 26 listed Pacific salmonids under NMFS' jurisdiction. Consultation with NMFS will be completed when EPA makes effect determinations on all remaining species under NMFS' jurisdiction and consults with NMFS as necessary.

For this Opinion, NMFS anticipates the general direct and indirect effects that would occur from EPA's registration of pesticide products California's Central Valley to listed salmonids under NMFS' jurisdiction during the 15-year duration of the proposed action. Recent and historical surveys indicate that listed salmonids occur in the action area, in places where they will be exposed to the stressors of the action. The RPM below provided in this Opinion is designed to reduce this exposure but not eliminate it. Pesticide runoff and drift of thiobencarb is likely to reach streams and other aquatic sites when they are applied to rice located adjacent to wetlands, riparian areas, ditches, off-channel habitats, perennial, intermittent, and ephemeral streams. These inputs into aquatic habitats are especially high during spray drift of liquid formulations and following early releases of thiobencarb-containing rice water.

The range of effects of thiobencarb on salmonids includes direct and indirect toxicological effects. Within this range, effects include death of fish, impaired respiration and swimming. Adverse impacts to riparian vegetation could lead to increased water temperature, increased sedimentation from bank instability, reductions in cover, alterations to or decreases in prey production, and reduction in chemical and nutrient filtering from upland sources. Impacts to aquatic vegetation would reduce dissolved oxygen, natural cover, alter or reduce the prey base, affect growth, and lead to an increased susceptibility to predation. These results are not the purpose of the proposed action. Therefore, incidental take of listed salmonids is reasonably certain to occur over the 15-year duration of the proposed action.

Given the variability of real-life conditions, the broad nature and scope of the proposed action, and the migratory nature of salmon, the best scientific and commercial data available are not sufficient to enable NMFS to estimate a specific amount of incidental take associated with the proposed action. As explained in the *Description of the Proposed Action* and the *Effects of the Proposed Action* sections, NMFS identified multiple uncertainties associated with the proposed action. Areas of uncertainty include:

1. Unable to quantify effect of herbicides on salmon habitat due to variability in plant susceptibility to the herbicides and variability in species composition and density in the various locations.

2. Minimal information on exposure and toxicity for pesticide formulations, adjuvants, and other/inert ingredients within registered formulations;
3. Little information on permitted tank mixtures and associated exposure estimates;
4. Limited data on toxicity of environmental mixtures;
5. Responses from exposure to combinations of thiobencarb and other stressors in the baseline;
6. Variable conditions of water bodies in which salmonids live.

NMFS therefore identifies, as a surrogate for the allowable extent of take, the ability of this action to proceed without any fish kills attributed to the legal use of thiobencarb, or any compounds, degradates, or mixtures in aquatic habitats containing individuals from any ESU/DPS. Because of the difficulty of detecting salmonid deaths, the fishes killed do not have to be listed salmonids. In general, salmonids appear to be more sensitive to thiobencarb than many other species of fish, so that if there are kills of other freshwater fishes attributed to use of pesticide products containing thiobencarb [or “to use of thiobencarb”], it is likely that salmonids have also died, even if no dead salmonids can be located. In addition, if stream conditions due to pesticide use kill less sensitive fishes in certain areas, the potential for lethal and non-lethal takes in downstream areas increases. A fish kill is considered attributable to thiobencarb, its metabolites, or degradates, if thiobencarb is known to have been applied in the vicinity, may reasonably be supposed to have run off or drifted into the affected area, and if surface water samples, or pathology indicate lethal levels of thiobencarb. NMFS notes that with increased monitoring and study of the impact of thiobencarb on water quality, particularly water quality in floodplain habitats, NMFS will be able to refine this incidental take statement, and future incidental take statements, to allow other measures of the extent of take.

14.2. Reasonable and Prudent Measures

The measures described below are non-discretionary measures to avoid or minimize take that must be undertaken by the EPA so that they become binding conditions of any grant or permit issued to the applicant(s), as appropriate, for the exemption in section 7(o)(2) to apply. The EPA has a continuing duty to regulate the activity covered by this incidental take statement. If the EPA (1) fails to assume and implement the terms and conditions or (2) fails to require the applicant(s) to adhere to the terms and conditions of the incidental take statement through enforceable terms that are added to the FIFRA label, the protective coverage of section 7(o)(2) may lapse. In order to monitor the impact of incidental take, the EPA must report a) the progress of the action and b) its impact on the species to NMFS OPR as specified in the incidental take statement [50 CFR§402.14(i)(3)].

To satisfy its obligations pursuant to section 7(a)(2) of the ESA, the EPA must monitor (a) the direct, indirect, and aggregate impacts to listed Pacific salmonids under NMFS' jurisdiction of its long-term registration of pesticide products containing thiobencarb and (b) the direct, indirect, or aggregate impacts of pesticide misapplications in the aquatic habitats in which they occur. The purpose of the monitoring program is for the EPA to use the results of the monitoring data and modify the registration process in order to reduce exposure and minimize the effect of exposure where pesticides will occur in salmonid habitat. NMFS believes all measures described as part of the proposed action, together with use of the Reasonable and Prudent Measures and Terms and Conditions described below, are necessary and appropriate to minimize the likelihood of incidental take of listed species due to implementation of the proposed action.

The EPA shall:

1. Minimize the amount and extent of incidental take to the three species, CV spring run Chinook, Sacramento River winter run Chinook, and CV steelhead from use of pesticide products containing thiobencarb by reducing the potential of thiobencarb to reach salmon-bearing waters;
2. Monitor any incidental take or surrogate measure of take that occurs from the action.

14.3. Terms and Conditions

To be exempt from the prohibitions of section 9 of the ESA, within nine months following the date of issuance of this Opinion, the EPA must comply with the following terms and conditions. These terms and conditions implement the reasonable and prudent measures described above. These terms and conditions are non-discretionary.

1. To implement RPM 1, EPA shall include the following risk reduction measures on all thiobencarb labels registered in California or in the appropriate bulletins once the label refers to compliance with the bulletin. These reductions have a proven track record for reducing thiobencarb loading into surface waters, which is essential to minimize incidental take. The pesticide use limitations are adapted from California Department of Pesticide Regulation's Prescribe Database and Pesticide Use Enforcement Standards Compendium, Volume 3 (Rev. 2-11), Appendix C, Sections C.1 and C.2.

Mandatory language follows:

Drift:

- For sprayable or dust formulations: when the air is calm or moving away from salmon habitat, commence applications on the side nearest the habitat and proceed away from the habitat. When air currents are moving toward habitat, do not make applications within 200 yards by air or 40 yards by ground upwind from occupied habitat. The appropriate California county agricultural commissioner may reduce or waive buffer zones following a site inspection, if there is an adequate hedgerow, windbreak, riparian corridor or other physical barrier that substantially reduces the probability of drift.

- Aircraft application equipment used to apply a pesticide spray solution shall be configured as follows:
  1. Functional boom length, measured from outboard nozzle to outboard nozzle, shall not exceed 75% of the overall wing span or rotor length.
  2. Boom pressure shall not exceed 40 pounds per square inch for the nozzles being used.
  3. The flow of liquid from each nozzle shall be controlled by a positive shutoff system.
  4. Nozzle orifices shall be directed backward, neutral to the airstream.
  5. Aircraft shall be equipped with:
     (a) Jet nozzles having an orifice of not less than one-sixteenth of an inch in diameter. Nozzles shall not be equipped with any device or mechanism which would cause a sheet, cone, fan, or similar type dispersion of the discharged material, except helicopters operating at 60 miles per hour or less may add a number 46 (or equivalent) or larger whirlplate;
     (b) Fan nozzles with a fan angle number not larger than 80 degrees and a flow rate not less than one gallon per minute at 40 pounds per square inch pressure (or equivalent), as an alternative to (a) for helicopters operating at 60 miles per hour or less; or
     (c) Other nozzles for aircraft use, as authorized by the director of the California Department of Pesticide Regulation, after evaluation.

- Aerial applications of a pesticide spray solution shall meet the following requirements:
  1. Apply only when there is a positive air flow. Wind speed shall not be more than ten miles per hour at the application site, as measured by an anemometer positioned four feet above the ground.
  2. Discharge shall start after entering the target site; discharge height shall not exceed ten feet above the crop or target; discharge shall be shut off whenever necessary to raise the equipment over obstacles; discharge shall be shut off before exiting the target site.

- Vehicle-mounted or towed ground equipment, other than handguns, used to make applications shall be equipped with:
  1. Nozzles having an orifice not less than one-sixteenth of an inch in diameter (or equivalent) and operated at a boom pressure not to exceed the manufacturer's recommended pressure for the nozzles being used; or
  2. Low-pressure fan nozzles with a fan angle number not larger than 80 degrees and nozzle orifice not less than 0.2 gallon per minute flow rate (or equivalent) and operated at a boom pressure not to exceed 15 pounds per square inch.

- Applications of a pesticide spray solution made by vehicle-mounted or towed ground equipment shall meet the following requirements:
  1. Apply only when wind speed is ten miles per hour or less at the application site, as measured by an anemometer positioned four feet above the ground.
  2. Discharge shall start after entering the target site; discharge shall be shut off before exiting the target site.

- The use of Bolero 10G formulation is prohibited in the Sacramento Valley rice growing counties of Butte, Colusa, Glenn, Placer, Sacramento, Sutter, Tehama, Yolo, and Yuba.

- In Butte, Colusa, Glenn, Placer, Sacramento, Sutter, Tehama, Yolo, and Yuba Counties, no aerial applications shall be made or continued within ½ mile of the Sacramento or Feather Rivers unless there is a continuous positive airflow away from the river.

- In Butte, Colusa, Glenn, Placer, Sacramento, Sutter, Tehama, Yolo, and Yuba Counties, no aerial application shall be made or continued within ½ mile of the Sacramento or Feather Rivers when the wind speed exceeds seven miles per hour.

- In Sacramento and Yolo Counties, no aerial applications shall be made or continued within ¼ mile of the Sacramento River unless they are made under the direct supervision of the appropriate California county agricultural commissioner or representative.

- In Sacramento and Yolo Counties, the maximum acres treated by air each day within ¼ mile of the Sacramento River shall not exceed 33 percent of the average acres treated per day by air within this area in each county during 2002.

General Water-Holding:

- Do not release water from the treated field during the water-holding period (**Error! Reference source not found.**).
- Prevent seepage[27]from moving offsite during the water-holding period.

1. Thiobencarb shall not be applied to rice fields exhibiting visible water seepage that moves offsite into drains that are considered state waters.
2. Borders surrounding each rice field shall be compacted before water is allowed to fill the field; the degree of compaction shall be sufficient to prevent water from seeping through the border. For example, compaction may be achieved by driving the tires or tracks of a tractor, or other heavy vehicle, on one side of the border.
3. This requirement (2) applies to new or reworked existing borders for the current rice season.
4. A common border between two existing rice fields does not need to be compacted.

[27] Seepage is lateral movement of irrigation water through a rice field levee or border to an area outside the normally flooded production area. Seepage can occur through levees into adjacent dry fields or into adjacent drains and canals.

Rice pesticides water management requirments summary. Rice Pesticides Water Management Requirements Summary

Water must be held for the indicated number of 24-hour periods on site or containment before release to State Waters.	Granular (e.g., Bolero 15G and UltraMax)	Liquid (e.g., Abolish 8EC)
	Hold (days after application)	Hold (days after application)
Single Field(d)	30	19
Single field Southern areas only (a).	19	
Single permitted release into tailwater recovery system or pond onto fallow field [Except Southern area (a)].	14(b)	14(b)
Multi-growers & district release onto closed recirculating systems.	6	6
Multi-growers & district release onto closed recirculating systems in Southern area(a).	6	
Release from closed recirculating system.	19	19
Release into area that discharge negligible amount into perennial streams	19	6(c)
Emergency Release of tailwater	19	19
Commissioner verifies the hydrologic isolation of the fields	6	6

a – Sacramento/San Joaquin Valley defined as: South of the line defined by Roads E10 and 116 in Yolo County and the American River in Sacramento County.
b – Thiobencarb permit condition allows Bolereo 15G and Bolero UltraMax label hold periodd of 14 days.
c – See hydrologic isolation fields.
d – When drainage begins after 30 day hold, discharge must not exceed two inches of water over a drain box weir for seven additional days. Unregulated discharges from these fields may then begin after 37 days.

- On rice fields treated with thiobencarb in the Sacramento Valley (north of the line defined by Roads E10 and 116 in Yolo County and the American River in Sacramento County), except those treated with liquid formulations e.g., Abolish® 8EC, all water must be retained on the treated fields for at least 30 days following application, except as described below. When drainage begins, discharge must not exceed two inches of water over a drain box weir for seven additional days. Unregulated discharges from these fields may then begin after 37 days.

  When water is contained within a tailwater recovery system, ponded on fallow land, or contained in other systems appropriate for preventing discharge, the water must be retained in the system for 19 days, unless:

(a) The system is under the control of one permittee, then water may be discharged from the application after a 14-day water hold period.
(b) The system includes drainage from more than one permittee, then water must be retained on the site of application for six days before being discharged from the application site into the system.
(c) Water is on fields within the bounds of areas that discharge negligible amounts of rice field drainage into perennial streams until fields are drained for harvest.
(d) Water-hold may be reduced to six days if the appropriate California county agricultural commissioner evaluates such sites and verifies the hydrologic isolation of the fields.

- On rice fields treated with thiobencarb in the Sacramento/San Joaquin Valley (south of the line defined by Roads E10 and 116 in Yolo County and the American River in Sacramento County), except those treated with liquid formulations e.g., Abolish® 8EC, all water must be retained on the treated fields for at least 19 days following application, except as described below. When drainage begins, water discharge must not exceed two inches of water over a drain box weir for an additional seven days. Unregulated discharges from these fields may begin after 26 days.

When water is contained within a tailwater recovery system, ponded on fallow land, or contained in other systems appropriate for preventing discharge, the system may discharge 19 days following the last application of thiobencarb within the system unless:
(a) The system is under the control of one permittee, then water may be discharged from the application after a 14-day water-hold period.
(b) The system includes drainage from more than one permittee, then water must be retained on the site of application for six days before discharged from the application site into the system.
(c) Water is on fields within the bounds of areas that discharge negligible amounts of rice field drainage into perennial streams until fields are drained for harvest. Water-hold may be reduced to six days, if the appropriate California county agricultural commissioner evaluates such sites and verifies the hydrologic isolation of the fields.

- On all areas and fields treated with liquid formulations e.g., Abolish® 8EC, all water must be retained on the treated fields for at least 19 days following application, except as described below. When drainage begins, water discharge must be released at a volume not to exceed two inches of water over a drain box weir for an additional seven days. Unregulated discharges from these fields may begin after 26 days.

For water contained within a tailwater recovery system, ponded on fallow land, or contained in other systems appropriate for preventing discharge, the system may discharge 19 days following the last application within the system unless:

(a) The system is under the control of one permittee, then water may be discharged from the application after a 14-day water-hold period.
(b) The system includes drainage from more than one permittee, then water must be retained on the site of application for six days before discharged from the application site into the system.
(c) Water is on fields within the bounds of areas that discharge negligible amounts of rice field drainage into perennial streams until fields are drained for harvest, then water-hold may be reduced to six days if the appropriate California county agricultural commissioner evaluates such sites and verifies the hydrologic isolation of the fields.

Training:

- All thiobencarb applicators shall receive mandatory thiobencarb stewardship training. This requirement is satisfied by:
    1. Attending the Preseason Thiobencarb Stewardship Meeting, or receiving certification from the County Agricultural Commissioner after viewing a video of the Preseason Thiobencarb Stewardship Meeting.
    2. Inspection of thiobencarb aerial applications by California Department of Pesticide Regulation to monitor thiobencarb drift mitigation requirements.

2. 2.1 EPA shall include the following instructions requiring reporting of fish kills either on the labels or ESPP Bulletins:

NOTICE: Incidents where salmon appear injured or killed as a result of pesticide applications shall be reported to NMFS OPR at 301-427-8400 and to the National Pesticide Information Center (NPIC) at 1-800-858-7378. The finder should leave the fish alone, make note of any circumstances likely causing the death or injury, location and number of fish involved, and take photographs, if possible. Adult fish should generally not be disturbed unless circumstances arise where an adult fish is obviously injured or killed by pesticide exposure, or some unnatural cause. The finder may be asked to carry out instructions provided by NMFS OPR to collect specimens or take other measures to ensure that evidence intrinsic to the specimen is preserved.

2.2 EPA shall report to NMFS OPR annually any incidences regarding thiobencarb effects on aquatic ecosystems added to its incident database that EPA has classified as “probable” or “highly probable.”

14.4 Conservation Recommendations

Section 7(a) (1) of the ESA directs federal agencies to use their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of endangered and threatened species. Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of a proposed action on listed species or critical habitat, to help implement recovery plans, or to develop information.

The following conservation recommendations would provide information for future consultations involving future authorizations of other pesticide a.i.s that may affect listed species:

1. Conduct mixture toxicity analysis in screening-level and endangered species biological evaluations;
2. Develop models to estimate pesticide concentrations in flood plain habitats; and
3. Collaborate with CDPR to review applicability of use limitations identified in the PRESCRIBE database to other pesticides and other listed species.
4. Develop inspection program in coordination with USDA' NRCS (or other entities) to evaluate riparian system condition and function. .This program could be used to inform future section 7 pesticide consultations and could support structuring less restrictive reasonable and prudent alternatives and reasonable and prudent measures. The program ultimately could provide incentives to land owners for creating and maintaining riparian systems in listed species habitat.

In order for NMFS to be kept informed of actions minimizing or avoiding adverse effects or benefiting listed species or their habitats, the EPA should notify NMFS OPR of any conservation recommendations it implements in the final action.

14.5 Reinitiation Notice

This concludes formal consultation on the EPA's proposed registration of pesticide products containing thiobencarb and their formulations to ESA-listed Pacific salmonids under the jurisdiction of the NMFS. 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 extent of take specified in the *Incidental Take*

Statement is exceeded; (2) new information reveals effects of this action that may affect listed species or designated critical habitat in a manner or to an extent not previously considered in this biological opinion; (3) the identified action is subsequently modified in a manner that causes an effect to the listed species or critical habitat that was not considered in this Opinion; or (4) a new species is listed or critical habitat designated that may be affected by the identified action.

For this consultation, in addition to reinitiation for authorization by EPA for uses other than rice growing in these four states, reinitiation will also be required for authorization for use of thiobencarb by state or local governments in Washington, Oregon, or Idaho, or expansion of rice-growing in California beyond the counties in which rice is now grown. Should reinitiation occur because of a change in one state or one area only, the reiniation may review the impacts only in that geographic area. If reinitiation of consultation appears warranted due to one or more of the above circumstances, EPA must contact NMFS OPR. If none of these reinitiation triggers are met within the next 15 years, then reinitiation will be required because the Opinion only covers the action for 15 years.

15. References

Allan, J. D. and Castillo, M. M. 2008. *Stream Ecology: Structure and Function of Running Waters.* The Netherlands: Springer.

Allan, J. D., Wipfli, M. S., Caouette, J. P., Prussian, A., and Rodgers, J. 2003. Influence of streamside vegetation on inputs of terrestrial invertebrates to salmonid food webs. *Canadian Journal of Fisheries and Aquatic Sciences,* 60(3): 309-20.

Altinok, I., Capkin, E., Karahan, S., and Boran, M. 2006. Effects of water quality and fish size on toxicity of methiocarb, a carbamate pesticide, to rainbow trout. *Environmental Toxicology and Pharmacology,* 22(1): 20-26.

Anderson, B. S., Phillips, B. M., Hunt, J. W., Connor, V., Richard, N., and Tjeerdema, R. S. 2006. Identifying primary stressors impacting macroinvertebrates in the Salinas River (California, USA): Relative effects of pesticides and suspended particles. *Environmental Pollution,* 141(3): 402-08.

Anderson, S. E. 1999. 'Use of off-channel freshwater wetlands by juvenile chinook and other salmonids: Potential for habitat restoration in Puget Sound.' in *Use of off-channel freshwater wetlands by juvenile chinook and other salmonids: Potential for habitat restoration in Puget Sound*, 73. The Evergreen State College.

Antolos, M., Roby, D. D., Lyone, D. E., Collis, K., Evans, A. F., Hawbecker, M., and Ryan, B. A. 2005. Caspian tern predation on juvenile salmonids in the Mid-Columbia River. *Transaction of American Fisheries Sociery,* (134): 466-80.

Arsenault, J. T. M., Fairchild, W. L., MacLatchy, D. L., Burtidge, L., Haya, K., and Brown, S. B. 2004. Effects of water-borne 4-nonylphenol and 17 beta-estradiol exposures during parr-smolt transformation on growth and plasma IGF-I of Atlantic salmon (Salmo salar L.). *Aquatic Toxicology,* 66(3): 255-65.

Arukwe, A. and Roe, K. 2008. Molecular and cellular detection of expression of vitellogenin and zona radiata protein in liver and skin of juvenile salmon (Salmo salar) exposed to nonylphenol. *Cell and Tissue Research,* 331(3): 701-12.

Aspelin, A. L. and Grube, A. H. 1999. 'Pesticides industry sales and usage-1996 and 1997 market estimates.' in *Pesticides industry sales and usage-1996 and 1997 market estimates*, 39. EPA, Biological and Economic Analysis Division.

Associates, A. 2001. 'Two-dimensional modeling and analysis of spawning bed mobilization, lower American River.' in *Two-dimensional modeling and analysis of spawning bed mobilization, lower American River*. Sacramento, California.

Austin, A. P., Harris, G. E., and Lucey, W. P. 1991. Impacts of an organophosphate herbicide (glyphosate) on periphyton communities developed in experimental streams. *Bulletin of Environmental Contamination and Toxicology,* 47(1): 29-35.

Babu, P., Reddy, G., Babu, G., and Chetty, C. 1989. RECOVERY OF BENTHIOCARB-INHIBITED ACHE IN FISH BRAIN - AN INVITRO STUDY. *Ecotoxicology and Environmental Safety,* 17(3): 317-22.

Bailey, E. D. 1954. 'Time pattern of the 1953 to 1954 migration of salmon and steelhead in the upper Sacramento River.' in *Time pattern of the 1953 to 1954 migration of salmon and*

steelhead in the upper Sacramento River, 4 pages. Inland Fisheries Division, California Department of Fish and Game.
Bailey, H. 1993. Acute and chronic toxicity of the rice herbicides thiobencarb and molinate to opossum shrimp (Neomysis mercedis). *Marine Environmental Research,* 36(4): 197-215.
Bajjaliya, F. and Vincik, R. 2008a. 'Timing, Composition, and Abundance of Juvenile Salmonid Emigration in the Sacramento River near Knights Landing October 2002 - July 2003 ' in *Timing, Composition, and Abundance of Juvenile Salmonid Emigration in the Sacramento River near Knights Landing October 2002 - July 2003* 27 pages.
--- 2008b. 'Timing, Composition, and Abundance of Juvenile Salmonid Emigration in the Sacramento River Near Knights Landing October 2002-July 2003.' in *Timing, Composition, and Abundance of Juvenile Salmonid Emigration in the Sacramento River Near Knights Landing October 2002-July 2003* ed. C. D. o. F. a. Game, 27. Sacramento, CA.
Baker, P. F., Speed, T. P., and Ligon, F. K. 1995. Estimating the influence of temperature on the survival of chinook salmon smolts (*Oncorhynchus tshawytscha*) migrating through the Sacramento-San Joaquin River Delta of California. *Canadian Journal of Fisheries and Aquatic Sciences,* 52: 855-63.
Baldwin, D., Sandahl, J., Labenia, J., and Scholz, N. 2003a. Sublethal effects of copper on coho salmon: Impacts on nonoverlapping receptor pathways in the peripheral olfactory nervous system. *Environmental Toxicology and Chemistry,* 22(10): 2266-74.
Baldwin, D., Tatara, C., and Scholz, N. 2011. Copper-induced olfactory toxicity in salmon and steelhead: Extrapolation across species and rearing environments. *Aquatic Toxicology,* 101(1): 295-97.
Baldwin, D. H., Sandahl, J. F., Labenia, J. S., and Scholz, N. L. 2003b. Sublethal effects of copper on coho salmon: Impacts on nonoverlapping receptor pathways in the peripheral olfactory nervous system. *Environmental Toxicology and Chemistry,* 22(10): 2266-74.
Beamish, R. J. and Bouillon, D. R. 1993. Pacific salmon production trends in relation to climate. *Canadian Journal of Fisheries and Aquatic Sciences,* 50: 1002-16.
Beamish, R. J. and Neville, C.-E. M. 1995. Pacific salmon and Pacific herring mortalities in the Fraser River plume caused by river lamprey (*Lampetra ayresi*). *Canadian Journal of Fisheries and Aquatic Sciences,* (52): 644-50.
Beamish, R. J., Noakes, D., Macfarlane, G., Pinnix, W., Sweeting, R., and King, J. 2000. Trends in coho marine survival in relation to the regime concept. *Fisheries Oceanography,* 9(1): 114-19.
Beamish, R. J., Sweeting, R. M., and Neville, C. M. 2009. Planning the management of Pacific salmon in a changing climate. *American Fisheries Society Symposium,* 69: 155-73.
Beamish, R. J., Thomson, B. L., and Farlane, G. A. 1992. Spiny dogfish predation on chinook and coho slamon and the potential effects on hatchery-produced salmon. *Transaction of American Fisheries Sociery,* 121(4): 444-45.
Beechie, T. and Bolton, S. 1999. An approach to restoring salmonid habitat-forming processes in Pacific Northwest watersheds. *Fisheries,* 24: 6-15.
Beechie, T. J., Liermann, M., Beamer, E. M., and Henderson, R. 2005. A classification of habitat types in a large river and their use by juvenile salmonids. *Transactions of the American Fisheries Society,* 134(3): 717-29.

Benda, L. E., Miller, D. J., Dunne, T., Reeves, G. H., and Agee, J. K. 2001. 'Dynamic landscape sysyems.' in *Dynamic landscape sysyems* eds. R. J. Naiman & R. E. Bilby, 261-88. New York: Springer-Verlag New York, Inc.

Bennett, K. P., Singhasemonon, N., Miller, N., and Gallavan, R. 1998. 'Rice Pesticide Monitoring in the Sacramento Valley, 1995.' in *Rice Pesticide Monitoring in the Sacramento Valley, 1995* ed. C. D. o. P. Regulation.

Bilby, R. E. 1984. Removal of woody debris may affect stream channel stability. *Journal of Forestry,* 82: 609-13.

Bilby, R. E. and Bisson, P. A. 2001. 'Function and distribution of large woody debris.' in *Function and distribution of large woody debris* eds. R. J. Naiman & R. E. Bilby, 324-46. New York: Springer-Verlag, Inc.

Bilby, R. E., Fransen, B. R., Walter, J. K., and Scarlett, W. J. 2001. Preliminary evaluation of the use of nitrogen stable isotope ratios to establish escapement levels for pacific salmon. *Fisheries,* 26(1): 6-14.

Binelli, A., Ricciardi, F., Riva, C., and Provini, A. 2005. Screening of POP pollution by AChE and EROD activities in Zebra mussels from the Italian Great Lakes. *Chemosphere,* 61(8): 1074-82.

Bird, S. L., Perry, S. G., Ray, S. L., and Teske, M. E. 2002. Evaluation of the AgDisp aerial spray algorithms in the AgDrift model. *Environmental Toxicology and Chemistry,* 21(3): 672–81.

Bisson, P. A. and Bilby, R. E. 2001. 'Organic matter and trophic dynamics.' in *Organic matter and trophic dynamics* eds. R. J. Naiman & R. E. Bilby, 373-98. New York: Springer-Verlag, Inc.

Bisson, P. A., Rieman, B. E., Luce, C., Hessburg, P. F., Lee, D. C., Kershner, J. L., Reeves, G. H., and Gresswell, R. E. 2003. Fire and aquatic ecosystems of the western USA: current knowledge and key questions. *Forest Ecology and Management,* (178): 213-29.

Bjorkstedt, E. P., Spence, B. C., Garza, J. C., Hankin, D. G., Fuller, D., Jones, W. E., Smith, J. J., and Macedo, R. 2005. 'An analysis of historical population structure for evolutionarily significant units of Chinook salmon, coho salmon, and steelhead in the North-Central California Coast Recovery Domain.' in *An analysis of historical population structure for evolutionarily significant units of Chinook salmon, coho salmon, and steelhead in the North-Central California Coast Recovery Domain*, 210. U.S. Department of Commerce.

Bjornn, T. C. and Reiser, D. W. 1991. 'Habitat requirements of salmonids in streams.' in *Habitat requirements of salmonids in streams* ed. M. R. Meehan, 83-138. American Fisheries Society Special Publications.

Blanchet, S., Loot, G., and Dodson, J. J. 2008. Competition, predation and flow rate as mediators of direct and indirect effects in a stream food chain. *Oecologia*, 157(1): 93-104.

Boles, G. L. 1988. 'Water temperature effects on Chinook salmon (*Oncorhynchus tshawytscha*) with emphasis on the Sacramento River: a literature review.' in *Water temperature effects on Chinook salmon (Oncorhynchus tshawytscha) with emphasis on the Sacramento River: a literature review*. Sacramento, CA: California Department of Fish and Game, Northern District.

Borthwick, P. W., Patrick, J. M., and Middaugh, D. P. 1985. COMPARATIVE ACUTE SENSITIVITIES OF EARLY LIFE STAGES OF ATHERINID FISHES TO CHLORPYRIFOS AND THIOBENCARB. *Archives of Environmental Contamination and Toxicology*, 14(4): 465-73.

Bowman, K. E. and Minshall, G. W. 2000. Assessment of short- and mid-term effects of wildlife on habitat structure of the Payette National Forest. *Prepared by the Stream Ecology Center, Idaho State University for the Payette National Forest*: 45 p.

Bradford, M. J., Taylor, G. C., and Allan, J. A. 1997. Empirical review of coho salmon smolt abundance and prediction of smolt production at the regional level. *Transactions of the American Fisheries Society,* 126: 49-64.

Brandes, P. L. and McLain, J. S. 2001. 'Juvenile Chinook salmon abundance, distribution, and survival in the Sacramento-San Joaquin Estuary.' in *Juvenile Chinook salmon abundance, distribution, and survival in the Sacramento-San Joaquin Estuary* ed. R. L. Brown, 39-136.

Bretaud, S., Saglio, P., and Toutant, J. P. 2001. Effects of carbofuran on brain acetylcholinesterase activity and swimming activity in Carassius auratus (Cyprinidae). *Cybium,* 25(1): 33-40.

Brett, J. R. 1952. Temperature tolerance in young Pacific salmon, genus *Oncorhynchus*. *Journal Fisheries Research Board of Canada,* 9(6): 265-323.

--- 1995. Energetics. *Physiological ecology of Pacific salmon. Edited by C.Groot, L. Margolis, and W.C. Clarke. Unversity of British Columbia Press, Vancouver, B.C.*: pp. 3-68.

Brock, T. C. M., Crum, S. J. H., Deneer, J. W., Heimbach, F., Roijackers, R. M. M., and Sinkeldam, J. A. 2004. Comparing aquatic risk assessment methods for the photosynthesis-inhibiting herbicides metribuzin and metamitron. *Environmental Pollution,* 130(3): 403-26.

Brock, T. C. M., Lahr, J., and Van den Brink, P. J. 2000. 'Ecological risk of pesticides in freshwater ecosystems. Part 1: herbicides.' in *Ecological risk of pesticides in freshwater ecosystems. Part 1: herbicides*. Wageningen, The Netherlands.

Brodeur, R. D., Fisher, J. P., Teel, D. J., Emmett, R. L., Casillas, E., and Miller, T. W. 2004. Juvenile salmonid distribution, growth, condition, origin, and environmental and species associations in the Northern California Current. *Fish. Bull,* 102: 25-46.

Brown, A. R., Riddle, A. M., Cunningham, N. L., Kedwards, T. J., Shillabeer, N., and Hutchinson, T. H. 2003. Predicting the effects of endocrine disrupting chemicals on fish populations. *Human and Ecological Risk Assessment,* 9(3): 761-88.

Brown, A. R., Riddle, A. M., Winfield, I. J., Fletcher, J. M., and James, J. B. 2005. Predicting the effects of endocrine disrupting chemicals on healthy and disease impacted populations of perch (perca fluviatilis). *Ecological Modelling,* 189(3-4): 377-95.

Brown, L. R., Moyle, P. B., and Yoshiyama, R. M. 1994. Historical decline and current status of coho salmon in California. *North American Journal of Fisheries Management,* 14: 237 - 61.

Brown, R. J., Conradi, M., and Depledge, M. H. 1999. Long-term exposure to 4-nonylphenol affects sexual differentiation and growth of the amphipod Corophium volutator (Pallas, 1766). *Science of the Total Environment,* 233(1-3): 77-88.

Brown, S. B. and Fairchild, W. L. 2003. Evidence for a causal link between exposure to an insecticide formulation and declines in catch of Atlantic salmon. *Human and Ecological Risk Assessment,* 9(1): 137-48.

Brust, K., Licht, O., Hultsch, V., Jungmann, D., and Nagel, R. 2001. Effects of terbutryn on aufwuchs and Lumbriculus variegatus in artificial indoor streams. *Environmental Toxicology and Chemistry,* 20(9): 2000-07.

Buchwalter, D. B., Jenkins, J. J., and Curtis, L. R. 2003. Temperature influences on water permeability and chlorpyrifos uptake in aquatic insects with differing respiratory strategies. *Environmental Toxicology and Chemistry*, 22(11): 2806-12.

Buchwalter, D. B., Sandahl, J. F., Jenkins, J. J., and Curtis, L. R. 2004. Roles of uptake, biotransformation, and target site sensitivity in determining the differential toxicity of chlorpyrifos to second to fourth instar *Chironomous riparius* (Meigen). *Aquatic Toxicology*, 66(2): 149-57.

Burdett, A. S., Stevens, M. M., and MacMillan, D. L. 2001. Laboratory and field studies on the effect of molinate, clomazone, and thiobencarb on nontarget aquatic invertebrates. *Environmental Toxicology and Chemistry*, 20(10): 2229-36.

Busby, P. J., Wainwright, T. C., Bryant, G. J., Lierheimer, L. J., Waples, R. S., Waknitz, F. W., and Lagomarsine, I. V. 1996. 'Status review of steelhead from Washington, Oregon, and California.' in *Status review of steelhead from Washington, Oregon, and California*. Seattle, Washington: U.S. Department of Commerce, Northwest Fisheries Science Center.

Caffrey, J. M. 1996. Glyphosate in fisheries management. *Hydrobiologia*, 340: 259-63.

California Commercial, I. a. R. R. E. S. D. 2002. 'California Commercial, Industrial and Residential Real Estate Services Directory.' in *California Commercial, Industrial and Residential Real Estate Services Directory*. http://www.ured.com/citysubweb.html.

California, U. o. 1993. 'Integrated Pest Management for Rice. 2nd Edition. University of California Statewide Integrated Pest Management Project, Division of Agriculture and Natural Resources. Publication 320.' in *Integrated Pest Management for Rice. 2nd Edition. University of California Statewide Integrated Pest Management Project, Division of Agriculture and Natural Resources. Publication 320.*, 99.

Carter, J. L. and Resh, V. H. 2005. 'Pacific Coast rivers of the coterminous United States.' in *Pacific Coast rivers of the coterminous United States* eds. A. C. Benke & C. E. Cushing, 541-90. Burlington, Massachusetts: Elsevier Academic Press.

CDFG 1995. Letter dated 30 March 1995 to M. Schiewe for the ESA Administrative Record for West Coast Steelhead. *available from Environmental and Technical Services Division, National Marine Fisheries Service, 525 NE Oregon St., Suite 500, Portland, Oregion 97232*.

--- 1998a. 'A status review of the spring run Chinook salmon in the Sacramento River drainage.' in *A status review of the spring run Chinook salmon in the Sacramento River drainage*, 394 pages. Sacramento, CA.

--- 1998b. 'Report to the Fish and Game Commission: a status review of the spring-run chinook salmon (*Oncorhynchus tshawytscha*) in the Sacramento River drainage.' in *Report to the Fish and Game Commission: a status review of the spring-run chinook salmon (Oncorhynchus tshawytscha) in the Sacramento River drainage*. California Department of Fish and Game.

--- 2003. September 2002 Klamath River fish kill: preliminary analysis of contributing factors. *??????*, ??????: ??-??

--- 2008. 'Anadromous fish restoration program, GRANDTAB salmon data http://www.delta.dfg.ca.gov/afrp/.' in *Anadromous fish restoration program, GRANDTAB salmon data http://www.delta.dfg.ca.gov/afrp/*. USFWS.

CDPR 2010a. 'Geographic Information System Mapping and Preliminary Data Analysis for Thiobencarb Use in California, 2003-2009 - Memorandum.' in *Geographic Information*

System Mapping and Preliminary Data Analysis for Thiobencarb Use in California, 2003-2009 - Memorandum. ed. D. o. P. Regulation, 11. Sacramento, California.
--- 2010b. 'Summary of pesticide use data 2009, indexed by commodity. California Department of Pesticide Regulation. Sacramento, CA.' in *Summary of pesticide use data 2009, indexed by commodity. California Department of Pesticide Regulation. Sacramento, CA.*
--- 2011. 'Pesticide Use Enforcement Program Standards Compendium at: http://www.cdpr.ca.gov/docs/enforce/compend.htm.' in *Pesticide Use Enforcement Program Standards Compendium at: http://www.cdpr.ca.gov/docs/enforce/compend.htm*.
CDWR 2002. 'Suisun Marsh Salinity Control Gates salmon passage evaluation report.' in *Suisun Marsh Salinity Control Gates salmon passage evaluation report*, 19 pages. Sacramento, CA: Environmental Services Office, California Department of Water Resources.
Census Bureau, U. S. 2010. '2010 Census.' in *2010 Census*. Quick Facts at: http://quickfacts.census.gov/qfd/states/06000lk.html.
Chapman, D. W. and Bjornn, T. C. 1969. 'Distribution of salmonids in streams, with special reference to food and feeding.' in *Distribution of salmonids in streams, with special reference to food and feeding* ed. T. G. Northcote, 153 - 76. H.R. MacMillan Lectures in Fisheries, University of British Columbia, Institute of Fisheries, Vancouver.
Chapman, P. 1996. A warning: NOECs are inappropriate for regulatory use. *Environmental Toxicology and Chemistry,* 15: 77-79.
Climate Impacts Group (CIG) 2004. 'Overview of climate change impacts in the U.S. Pacific Northwest.' in *Overview of climate change impacts in the U.S. Pacific Northwest*. Seattle, Washington: University of Washington.
Colgrove, D. J. and Wood, J. W. 1966. Occurrence and control of *Chondrococcus columnaris* as related to Fraser River sockeye salmon. *Int. Pac. Salmon Fish. Comm. Prog. Rep. No. 15.*
Collis, K. 2007. Research, monitoring, and evaluation of avian predation on salmonid smolts in the lower and mid-Columbia River - 2006 season summary. *U.S. Geological survey report to Bonneville Power Administration and the U.S. Army Corps of Engineers*.
Conomos, T., Smith, R., and Gartner, J. 1985. Environmental setting of San-Francisco Bay. *Hydrobiologia,* 129(OCT): 1-12.
Cook, D. 2008. 'Chinook salmon spawning study Russian River fall 2002-2007.' in *Chinook salmon spawning study Russian River fall 2002-2007,* 24. Santa Rosa, California: Sonoma County Water Agency.
Cordone, A. J. and Kelley, D. W. 1961. The influences of inorganic sediment on the aquatic life of streams. *California Fish and Game,* 47: 89-228.
CRA 1989. 'Upper Sacramento River fisheries and riparian management plan.' in *Upper Sacramento River fisheries and riparian management plan*, 157 pages. California Resources Agency.
CRC 2011. '2010 Annual Monitoring Report, Sacramento River Drainage Basin.' in *2010 Annual Monitoring Report, Sacramento River Drainage Basin*, 123 pages. Sacramento, CA: California Rice Commission.
CRWQCB-CVR 1998. 'Water Quality Control Plan (Basin Plan) for the Sacramento River and San Joaquin River Basins, fourth edition.' in *Water Quality Control Plan (Basin Plan) for the Sacramento River and San Joaquin River Basins, fourth edition*. California Regional Water Quality Control Board-Central Valley Region.

--- 2005. 'Staff Report on 2004 Rice Pesticide Program at: http://www.waterboards.ca.gov/centralvalley/board_decisions/tentative_orders/0503/contested/rice/staffrpt.pdf.' in *Staff Report on 2004 Rice Pesticide Program at: http://www.waterboards.ca.gov/centralvalley/board_decisions/tentative_orders/0503/contested/rice/staffrpt.pdf*. California Regional Water Quality Control Board-Central Valley Region.

--- 2010. 'Statewide 2010 Integrated Report for the Clean Water Act Section 303(d) List & 30b(b) Report.' in *Statewide 2010 Integrated Report for the Clean Water Act Section 303(d) List & 30b(b) Report*. California Regional Water Quality Control Board-Central Valley Region.

De Boeck, G., Vlaeminck, A., and Blust, R. 1997. Effects of sublethal copper exposure on copper accumulation, food consumption, growth, energy stores, and nucleic acid content in common carp. *Archives of Environmental Contamination and Toxicology,* 33(4): 415-22.

DeLorenzo, M. E., Lauth, J., Pennington, P. L., Scott, G. I., and Ross, P. E. 1999. Atrazine effects on the microbial food web in tidal creek mesocosms. *Aquatic Toxicology,* 46(3-4): 241-51.

Demko, D. B. and Cramer, S. P. 2000. Effects of pulse flows on juvenile Chinook migration in the Stanislaus River. *Report of S.P. Cramer & Associates, Inc. to South San Joaquin Irrigation District, Manteca, California and Oakdale Irrigation District, Oakdale, California*.

DeNoyelles, F., Kettle, W. D., and Sinn, D. E. 1982. The responses of plankton communities in experimental ponds to atrazine, the most heavily used pesticide in the United States. *Ecology,* 63(5): 1285-93.

Dettman, D. H., Kelley, D. W., and Mitchell, W. T. 1987. 'The influence of flow on Central Valley salmon. Prepared for the California Department of Water Resources.' in *The influence of flow on Central Valley salmon. Prepared for the California Department of Water Resources*. Available from D.W. Kelley and Associates, 8955 Langs Hill Rd., P.O. Box 634, Newcastle, CA 95658.

Dewey, S. L. 1986. Effects of the Herbicide Atrazine on Aquatic Insect Community Structure and Emergence. *Ecology,* 67(1): 148-62.

Dodson, J. and Mayfield, C. 1979. Modification of the rheotropic response of rainbow trout (*Salmon gairdneri*) by sublethal doses of the aquatic herbicides diquat and simizine. *Environmental Pollution,* 18: 147-57.

DOI 1999. 'Final Programmatic Environmental Impact Statement for the Central Valley Project Improvement Act, October 1999.' in *Final Programmatic Environmental Impact Statement for the Central Valley Project Improvement Act, October 1999*. Department of the Interior.

Dubrovsky, N. M., Knifong, D. L., Dileanis, P. D., Brown, L. R., May, J. T., Connor, V., and Alpers, C. N. 1998. Water quality in the Sacramento River basin. *U.S. Geological Survey Circular 1215*.

Edwards, G. W., Urquhart, K. A. F., and Tillman, T. L. 1996. 'Adult salmon migration monitoring, Suisun Marsh Salinity Control Gates, September-November 1994.' in *Adult salmon migration monitoring, Suisun Marsh Salinity Control Gates, September-November 1994,* 27 pages. Interagency Ecological Program for the San Francisco Bay/Delta Estuary.

EPA 1994. 'Methods for measuring the toxicity and bioaccumulation of sediment associated contaminants with freshwater invertebrates.' in *Methods for measuring the toxicity and bioaccumulation of sediment associated contaminants with freshwater invertebrates*. Duluth, MN: Environmental Protection Agency.

--- 1996. 'Ecological Effects Test Guidelines OPPTS 850.4000, Background -Nontarget Plant Testing.' in *Ecological Effects Test Guidelines OPPTS 850.4000, Background -Nontarget Plant Testing*, 15.

--- 1997a. 'Registration Eligibility Decision (RED) Thiobencarb.' in *Registration Eligibility Decision (RED) Thiobencarb* ed. P. a. T. S. Office of Prevention, 234. Washington, D.C.

--- 1997b. 'Registration Eligibility Decision - Thiobencarb.' in *Registration Eligibility Decision - Thiobencarb* ed. O. o. P. Programs, 216. Washington, D.C.

--- 1998. Guidelines for ecological risk assessment. *Federal Register,* 63(93): 26846-924.

--- 2002a. 'Effects Determination for Molinate and Thiobencarb for Pacific Anadromous Salmonids (July 23, 2002).' in *Effects Determination for Molinate and Thiobencarb for Pacific Anadromous Salmonids (July 23, 2002)*.

--- 2002b. 'Final Report of the FIFRA Scientific Advisory Panel (SAP) Meeting Held on June 27-29, 2001: Review of Non-Target Plant Toxicity Tests Under the North American Free Trade Agreement.' in *Final Report of the FIFRA Scientific Advisory Panel (SAP) Meeting Held on June 27-29, 2001: Review of Non-Target Plant Toxicity Tests Under the North American Free Trade Agreement.*, 52.

--- 2002c. 'Thiobencarb and Molinate - Analysis of Risks to Endangered and Threatened Salmon and Steelhead.' in *Thiobencarb and Molinate - Analysis of Risks to Endangered and Threatened Salmon and Steelhead* ed. O. o. P. Programs, 22. Washington, D.C.

--- 2004. 'Overview of the ecological risk assessment process in the Office of Pesticide Programs, U.S. Environmental Protection Agency - Endangered and threatened species effects determinations.' in *Overview of the ecological risk assessment process in the Office of Pesticide Programs, U.S. Environmental Protection Agency - Endangered and threatened species effects determinations*. Environmental Protection Agency - Office of Pesticide Programs.

--- 2009a. 'Effects determinations for propanil relative to the Califronia Red-legged Frog and designated critical habitat (Ocober 16, 2009).' in *Effects determinations for propanil relative to the Califronia Red-legged Frog and designated critical habitat (Ocober 16, 2009)*.

--- 2009b. 'Potential Risks of Thiobencarb Use to Federally Threatened California Red-legged Frog (*Rana aurora draytonii*) and Delta Smelt (*Hypomesus transpacificus*).' in *Potential Risks of Thiobencarb Use to Federally Threatened California Red-legged Frog (Rana aurora draytonii) and Delta Smelt (Hypomesus transpacificus)* ed. O. o. P. Programs, 168. Washington, D.C.

Everest, F. H. and Chapman, D. W. 1972. Habitat selection and spatial interaction by juvenile Chinook salmon and steelhead trout. *Journal of the Fisheries Research Board of Canada,* 29: 91 - 100.

Faggella, G. A. and Finlayson, B. J. 1988. 'Standardized testing program 1987 progress report.' in *Standardized testing program 1987 progress report* ed. C. D. o. F. a. G. E. S. Division. Elk Grove, California.

Fairchild JF, Ruessler DS, and Haverland PS 1997. Comparative sensitivity of *Selenastrum capricornutum* and *Lemna minor* to sixteen herbicides. Arch. *Archives of Environmental Contamination and Toxicology,* 32: 353-57.

Fairchild, W. L., Swansburg, E. O., Arsenault, J. T., and Brown, S. B. 1999. Does an association between pesticide use and subsequent declines in catch of Atlantic salmon (*Salmo salar*) represent a case of endocrine disruption? *Environmental Health Perspectives,* 107(5): 349-57.

Fernandes, J. C. and Henriques, F. S. 1991. BIOCHEMICAL, PHYSIOLOGICAL, AND STRUCTURAL EFFECTS OF EXCESS COPPER IN PLANTS. *Botanical Review,* 57(3): 246-73.

Fernandez-Vega, C., Sancho, E., Ferrando, D., and Andreu, E. 2002. Physiological effects of thiobencarb stress in the eel (Anguilla anguilla). *Journal of Pesticide Science,* 27(1): 31-38.

Fernandez-Vega, C., Sancho, E., Ferrando, M. D., and Andreu-Moliner, E. 1999. Thiobencarb toxicity and plasma AChE inhibition in the European eel. *Journal of Environmental Science and Health Part B-Pesticides Food Contaminants and Agricultural Wastes,* 34(1): 61-73.

Figueroa, E. B. and Woods, R. A. 2007. Industry output and employment projections to 2016. *Monthly Labor Review,* 130: 53-85.

Finlayson, B. J. and Faggella, G. A. 1986. COMPARISON OF LABORATORY AND FIELD OBSERVATIONS OF FISH EXPOSED TO THE HERBICIDES MOLINATE AND THIOBENCARB. *Transactions of the American Fisheries Society,* 115(6): 882-90.

Finney, B. P., Gregory-Eaves, I., Douglas, M. S. V., and Smol, J. P. 2002. Fisheries productivity in the northeastern Pacific Ocean over the past 2200 years. *Nature,* 416: 729–33.

Fischer, J. R., Quist, M. C., Wigen, S. L., Schaefer, A. J., Stewart, T. W., and Isenhart, T. M. 2010. Assemblage and Population-Level Responses of Stream Fish to Riparian Buffers at Multiple Spatial Scales. *Transactions of the American Fisheries Society,* 139(1): 185-200.

Fisher, F. W. 1992. 'Chinook salmon, *Onchorhynchus tshawytscha,* growth and occurence in the Sacramento-San Joaquin River system.' in *Chinook salmon, Onchorhynchus tshawytscha, growth and occurence in the Sacramento-San Joaquin River system.* ed. C. D. o. F. a. Game. Sacramento, California.

--- 1994. Past and present status of Central Valley Chinook salmon. *Conservation Biology,* 8: 870-73.

Foott, J. S., Harmon, R., and Stone, R. 2003. Ceratomyxosis resistance in juvenile Chinook salmon and steelhead trout from the Klamath River, 2002 Investigational Report. *U.S. Fish and Wildlife Service, California-Nevada Fish Health Center, Anderson, California. [768 Kb] at KRIS Web site*.

Ford, J. K. B. and Ellis, G. M. 2006. Selective foraging by fish-eating killer whales Orcinus orca in British Columbia. *Marine Ecology Progress Series,* (316): 185-99.

Francis, R. C. and Hare, S. R. 1994. Decadal-scale regime shifts in the large marine ecosystems of the North-east Pacific: A case for historical science. *Fisheries Oceanography,* 3(4): 279-91.

Fresh, K. L., Casillas, E., Johnson, L. L., and Bottom, D. L. 2005. 'Role of the estuary in the recovery of Columbia River Basin salmon and steelhead: An evaluation of the effects of selected factors on population viability.' in *Role of the estuary in the recovery of*

Columbia River Basin salmon and steelhead: An evaluation of the effects of selected factors on population viability, 155. Seattle, Washington: U.S. Department of Commerce.
Fry, D. H., Jr., 1961. King salmon spawning stocks of the California Central Valley, 1940-1959. *California Department of Fish and Game,* 47(1): 55-71.
Fujimura, R., Finlayson, B., and Chapman, G. 1991. 'Evaluation of acute and chronic toxicity tests with larval striped bass.' in *Evaluation of acute and chronic toxicity tests with larval striped bass* eds. M. A. Mayes & M. G. Barron, 193-211. Philadelphia, PA.
Garland, R., Tiffan, K., Rondorf, D., and Clark, L. 2002. Comparison of subyearling fall chinook salmon's use of riprap, revetments and unaltered habitats in Lake Wallula of the Columbia River. *North American Journal of Fisheries Management,* 22(4): 1283-89.
Geisy, J. P., Solomon, K. R., Coats, J. R., Dixon, K. R., Giddings, J. M., and Kenaga, E. E. 1999. *Chlorpyrifos: Ecological risk assessment in North American aquatic environments.* New York City: Springer-Verlag.
Giddings, J. M. 2009. 'Carbaryl exposure and potential effects on threatened and endangered salmonids.' in *Carbaryl exposure and potential effects on threatened and endangered salmonids*, 73. CSI-USA: Compliance Services International for Bayer CropSciences LP.
Gilliom, R. J. 2007. Pesticides in U.S. streams and groundwater. *Environmental Science & Technology*, 41(10): 3407-13.
Gilliom, R. J., Barbash, J. E., Crawford, C. G., Hamilton, P. A., Martin, J. D., Nakagaki, N., Nowell, L. H., Scott, J. C., Sackelberg, P. E., G.P., T., and Wolock, D. M. 2006a. 'The quality of our nation's waters-Pesticides in the nation's streams and ground water, 1992-2001.' in *The quality of our nation's waters-Pesticides in the nation's streams and ground water, 1992-2001*, 172.
Gilliom, R. J., Barbash, J. E., Crawford, C. G., Hamilton, P. A., Martin, J. D., Nakagaki, N., Nowell, L. H., Scott, J. C., Stackelberg, P. E., Thelin, G. P., and Wolock, D. M. 2006b. The quality of our nation's waters; pesticides in the nation's streams and groundwater, 1992-2001. *U.S. Geological Survey circular 1291*: 52 p.
Good, T. P., Waples, R. S., and Adams, P. 2005. 'Updated status of federally listed ESUs of West Coast salmon and steelhead.' in *Updated status of federally listed ESUs of West Coast salmon and steelhead.*, 1-598. Seattle, Washington: U.S. Department of Commerce.
Goyer, R. A. 1996. 'Toxic effects of metals.' in *Toxic effects of metals* ed. C. D. Klassen, 691-736. New York, NY: McGraw Hill.
Greene, S. 1992. 'Memorandum: Daily length tables.' in *Memorandum: Daily length tables* ed. C. D. o. W. R. E. S. Office. Sacramento, California.
Gregory, S. V. and Bisson, P. A. 1997. Degradation and loss of anadromous salmonid habitat in the Pacific Northwest. *pages 277-314 in Stouder D.J, Bisson, P.A, Naiman, R.J. (editors) Pacific salmon and their ecoystems. New York: Chapman and Hall*.
Greswell, R. E. 1999. Fire and aquatic ecosystems in forested biomes of North America. *Transactions of the American Fisheries Society,* 128(2): 193-221.
Gronwald, J. W. 1991. Lipid biosynthesis inhibitors. *Weed Science,* 32: 435-49.
Groot, C. and Margolis, L. 1991. *Pacific salmon life histories.* Vancouver, BC, Canada: Vancouver University of British Columbia Press.
Gruessner, B. and Watzin, M. C. 1996. Response of aquatic communities from a Vermont stream to environmentally realistic atrazine exposure in laboratory microcosms. *Environmental Toxicology and Chemistry,* 15(4): 410-19.

Guillen, G. 2003. Klamath River fish die-off, September 2002: causative factors of mortality. *Report number AFWO-F-02-03. U.S. Fish and Wildlife Service, Arcata Fish and Wildlife Office. Arcata, California,* 128 p.

Gustafson, R. G., Wainwright, T. C., Winans, G. A., Waknitz, F. W., Parker, L. T., and Waples, R. S. 1997. 'Status review of sockeye salmon from Washington and Oregon.' in *Status review of sockeye salmon from Washington and Oregon*, 282. U.S. Department of Commerce.

Hallock, R. J. 1989. 'Upper Sacramento River steelhead (*Oncorhynchus mykiss*) 1952-1988.' in *Upper Sacramento River steelhead (Oncorhynchus mykiss) 1952-1988*.

Hallock, R. J., Van Woert, W. F., and Shapovalov, L. 1961. An evaluation of stocking hatchery-reared steelhead rainbow trout (*Salmo gairdneri gairdneri*) in the Sacramento River System. *Fisheries Bulletin,* 114: 1-74.

Hansen, J., Rose, J., Jenkins, R., Gerow, K., and Bergman, H. 1999. Chinook salmon (Oncorhynchus tshawytscha) and rainbow trout (Oncorhynchus mykiss) exposed to copper: Neurophysiological and histological effects on the olfactory system. *Environmental Toxicology and Chemistry,* 18(9): 1979-91.

Hansen, J., Welsh, P., Lipton, J., Cacela, D., and Dailey, A. 2002. Relative sensitivity of bull trout (Salvelinus confluentus) and rainbow trout (Oncorhynchus mykiss) to acute exposures of cadmium and zinc. *Environmental Toxicology and Chemistry,* 21(1): 67-75.

Hanson, B., Baird, R. W., and Schorr, G. 2005. Focal behavioral observations and fish-eating killer whales: improving our understanding of foraging behaviour and prey selection. *Abstract at the 16th Biennial Conference on the Biology of Marine Mammals, San Diego, California*.

Hanson, C. H. 1997. 'Acute temperature tolerance of juvenile Chinook salmon from the Mokelumne River.' in *Acute temperature tolerance of juvenile Chinook salmon from the Mokelumne River*, 11. Walnut Creek, California: Hanson Environmental, Inc.

Hard, J., Jones, R. P. J., Delarm, M. R., and Waples, R. S. 1992. 'Pacific salmon and artificial propagation under the endangered species act.' in *Pacific salmon and artificial propagation under the endangered species act*. Seattle, Washington: U.S. Department of Commerce.

Hare, S. R. and Mantua., N. J. 2000. Empirical evidence for North Pacific regime shifts in 1977 and 1989. *Progress in Oceanography,* 16-Feb(4-Feb): 103-45.

Harper, D. E., Jr., Landry, A. M., Jr., and Ray, S. M. 1979. 'MRID 79986. Studies in Halls Bayou To Test the Effects of a Pre-emergent Herbicide, Bolero, on Aquatic Organisms. (Unpublished study received Dec 11, 1979 under 239-2449; prepared by Harper Environmental Consulting Co., submitted by Chevron Chemical Co., Richmond, Calif.; CDL:241495-D).' in *MRID 79986. Studies in Halls Bayou To Test the Effects of a Pre-emergent Herbicide, Bolero, on Aquatic Organisms. (Unpublished study received Dec 11, 1979 under 239-2449; prepared by Harper Environmental Consulting Co., submitted by Chevron Chemical Co., Richmond, Calif.; CDL:241495-D).*

Harrington, J. M. 1990. 'Hazard assessment of the rice herbicides molinate and thiobencarb to aquatic organisms in the Sacramento River system.' in *Hazard assessment of the rice herbicides molinate and thiobencarb to aquatic organisms in the Sacramento River system*, 97. California Department of Fish and Game.

Hartgers, E. M., Aalderink, G. H., van den Brink, P. J., Gylstra, R., Wiegman, J. W. F., and Brock, T. C. M. 1998. Ecotoxicological threshold levels of a mixture of herbicides

(atrazine, diuron and metolachlor) in freshwater microcosms. *Aquatic Ecology,* 32: 135-52.
Harvey, C. J., Peterson, B. J., Bowden, W. B., Hershey, A. E., Miller, M. C., Deegan, L. A., and Finlay, J. C. 1998. Biological Responses to Fertilization of Oksrukuyik Creek, a Tundra Stream. *Journal of the North American Benthological Society,* 17(2): 190-209.
Healey, M. C. 1982. 'Juvenile Pacific salmon in estuaries: the life support system.' in *Juvenile Pacific salmon in estuaries: the life support system* ed. V. S. Kennedy, 315-41. New York, NY: Academic Press.
--- 1991. 'Life history of Chinook salmon (*Oncorhynchus tshawytscha*).' in *Life history of Chinook salmon (Oncorhynchus tshawytscha)* eds. C. Groot & L. Margolis, 311-94. Vancouver, Canada: University of British Columbia Press.
Heath, A. G. 1995. *Water Pollution and Fish Physiology.* Boca Raton, Florida: Lewis Publishers, CRC Press.
Hemmer, M. J., Bowman, C. J., Hemmer, B. L., Friedman, S. D., Marcovich, D., Kroll, K. J., and Denslow, N. D. 2002. Vitellogenin mRNA regulation and plasma clearance in male sheepshead minnows, (Cyprinodon variegatus) after cessation of exposure to 17[beta]-estradiol and p-nonylphenol. *Aquatic Toxicology,* 58(1-2): 99-112.
Henning, J. A., Gresswell, R. E., and Fleming, I. A. 2006. Juvenile salmonid use of freshwater emergent wetlands in the floodplain and its implications for conservation management. *North American Journal of Fisheries Management,* 26: 367-76.
Herren, J. R. and Kawasaki, S. S. 2001. 'Inventory of water diversions in four geographic areas in California's Central Valley.' in *Inventory of water diversions in four geographic areas in California's Central Valley*, 343-55.
Hill, J. E., Brandon, D. M., Williams, J. F., and Canevari, W. M. 1992. 'Rice production in California.' in *Rice production in California*, 22.
Hirsch, R. M., Alley, W. M., and Wilbur, W. G. 1988. 'Concepts for a national water quality assessment program.' in *Concepts for a national water quality assessment program.*, 42. U.S. Geological Survey.
Ho, B. 1990. 'Aquatic field dissipation of BOLERO 10G in Rice. Submitted by Valent U.S.A. to EPA. MRID # 434040-05.' in *Aquatic field dissipation of BOLERO 10G in Rice. Submitted by Valent U.S.A. to EPA. MRID # 434040-05.*
Hoogeweg, C. G., Williams, W. M., Breuer, R., Denton , D., B., R., and Watry, C. 2011. 'Spatial and temporal quantification of pesticide loadings to the Sacramento River, San Joaquin River, and Bay-Delta to guide Risk assessment for sensitive species. CALFED Science Grant #1055. Nov, 2 2011. 293 pp.' in *Spatial and temporal quantification of pesticide loadings to the Sacramento River, San Joaquin River, and Bay-Delta to guide Risk assessment for sensitive species. CALFED Science Grant #1055. Nov, 2 2011. 293 pp.*
Howard, T. E. 1975. Swimming performance of juvenile coho salmon (Oncorhynchus kisutch) exposed to bleached kraft pulp mill effluent. *Journal of Fisheries Research Board of Canada,* 32: 789-93.
Huryn, A. D. 1996. An appraisal of the Allen paradox in a New Zealand trout stream. *Limnology and Oceanography,* 41(2): 243-52.
--- 1998. Ecosystem-level evidence for top-down and bottom-up control of production in a grassland stream system. *Oecologia,* 115(1-2): 173-83.

Hutchinson, T. H., Ankley, G. T., Segner, H., and Tyler, C. R. 2006. Screening and testing for endocrine disruption in fish - Biomarkers as "signposts," not "traffic lights," in risk assessment. *Environmental Health Perspectives,* 114: 106-14.
ICTRT 2007. 'Viability criteria for application to Interior Columbia Basin salmonid ESUs.' in *Viability criteria for application to Interior Columbia Basin salmonid ESUs*, 90. Seattle, Washington: NMFS, Northwest Fisheries Science Center.
Ingersoll, C. G. 1995. 'Sediment tests.' in *Sediment tests* ed. G. M. Rand, 231-55. Bristol, PA: Taylor and Francis.
IPCC 2001. 'Climate change 2001: impacts adaptation and vulnerability. Summary for policy makers and technical summary.' in *Climate change 2001: impacts adaptation and vulnerability. Summary for policy makers and technical summary*. Geneva, Switzerland: IPCC.
--- 2007. 'Contribution of working groups I, II, and III to the fourth assessment report of the intergovernmental panel on climate change.' in *Contribution of working groups I, II, and III to the fourth assessment report of the intergovernmental panel on climate change* eds. R. K. Pachauri & A. Reisinger, 104. Geneva, Switzerland: Intergovernmental Panel on Climate Change.
Ishibashi, H., Hirano, M., Matsumura, N., Watanabe, N., Takao, Y., and Arizono, K. 2006. Reproductive effects and bioconcentration of 4-nonylphenol in medaka fish (*Oryzias latipes*). *Chemosphere,* 65(6): 1019-26.
Jardine, T. D., MacLatchy, D. L., Fairchild, W. L., Chaput, G., and Brown, S. B. 2005. Development of a short-term in situ caging methodology to assess long-term effects of industrial and municipal discharges on salmon smolts. *Ecotoxicology and Environmental Safety,* 62(3): 331-40.
Jeffres, C. A., Opperman, J. J., and Moyle, P. B. 2008. Ephemeral floodplain habitats provide best growth conditions for juvenile Chinook salmon in a California river. *Environmental Biology of Fishes,* 83: 449-58.
Jenkins, D. G. and Buikema, A. L. 1998. Do similar communities develop in similar sites? A test with zooplankton structure and function. *Ecological Monographs,* 68(3): 421-43.
Johnson, B. T. 1986. Potential impact of selected agricultural chemical contaminants on a northern prairie wetland - a microcosm evaluation. *Environmental Toxicology and Chemistry,* 5(5): 473-85.
Johnson, O. W., Grant, W. S., Kope, R. G., Neely, K., Waknitz, F. W., and Waples, R. S. 1997. 'Status review of chum salmon from Washington, Oregon, and California.' in *Status review of chum salmon from Washington, Oregon, and California,* 280. Seattle, WA: U.S. Department of Commerce.
Johnson, S. W., Thedinga, J. F., and Koski, K. V. 1992. Life history of juvenile ocean-type Chinook salmon (*Oncorhynchus tshawytscha*) in the Situk River, Alaska. *Canadian Journal of Fisheries and Aquatic Sciences,* 49(12): 2621-29.
Juttner, I., Peither, A., Lay, J. P., Kettrup, A., and Ormerod, S. J. 1995. An outdood mesocosm study to assess ecotoxicological effects of atrazine on a natural plankton community. *Archives of Environmental Contamination and Toxicology,* 29(4): 435-41.
Kasai, F. and Hanazato, T. 1995. Effects of the triazine herbicide, Simetryn, on freshwater plankton communities in experimental ponds. *Environmental Pollution,* 89(2): 197-202.
Kawakami, T., Eun, H., Arao, T., Endo, S., Ueji, M., Tamura, K., and Higashi, T. 2006. Concentration and loading of several pesticides in water, suspended solids and sediment

during ordinary water discharge in Sugao marsh, Ibaraki Prefecture, Japan. *Journal of Pesticide Science,* 31(1): 6-13.

Keller, E. A. and Swanson, F. J. 1979. Effects of large organic material on channel form and fluvial processes. *Earth Surface Processes,* 4: 361-80.

Kelley, K. 2000. 'Memorandum: Results of thiobencarb monitoring at seepage sites in Colusa and Glenn Counties. December 7, 2000.' in *Memorandum: Results of thiobencarb monitoring at seepage sites in Colusa and Glenn Counties. December 7, 2000.* ed. C. E. D. o. P. Regulation.

Kiely, T., Donaldson, D., and Grube, A. 2004. 'Pesticides industry sales and usage 2000 and 2001 market estimates.' in *Pesticides industry sales and usage 2000 and 2001 market estimates,* 33. U.S. Environmental Protection Agency, Biological and Economic Analysis Division.

Kier Associates 1991. Long range plan for the Klamath River Basin Conservatin Area Fishery Restoration Program. *U.S. Fish and Wildlife Service Klamath River Fishery Resource Office. Yreka, California. [8.5 MB] at KRIS Web site*: 403 p.

Koehler, M. E., Fresh, K. L., Beauchamp, D. A., Cordell, J. R., Simenstad, C. A., and Seiler, D. E. 2006. Diet and bioenergetics of lake-rearing juvenile Chinook salmon in Lake Washington. *Transaction of American Fisheries Sociery,* 135: 1580-91.

Kooijman, S. 1996. An alternative for the NOEC exists, but the standard model has to be abandoned first. *Oikos,* 75: 310-16.

Koplin, D. W., Furlong, E. T., M eyer, M. T., Thurman, E. M., Zaugg, S. D., Barber, L. B., and Buxton, H. T. 2002. Pharmaceuticals, hormones, and other organic wastewater contaminants in U.S. Streams,1999-2000: A national reconnaissance. *Environmental Science and Technology,* 36(6): 1202-11.

Kuivila, K. M. and Jennings, B. E. 2007. Input, flux, and persistence of six select pesticides in San Francisco Bay. *International Journal of Environmental Analytical Chemistry,* 87: 897-911.

Lai, J. C. 1991. 'Field dissipation of BOLERO 8EC in Rice. Submitted by Valent U.S.A. Corporation to EPA in support of U.S. EPA Registration No. 62499-18. MRID #420034-04.' in *Field dissipation of BOLERO 8EC in Rice. Submitted by Valent U.S.A. Corporation to EPA in support of U.S. EPA Registration No. 62499-18. MRID #420034-04.*

Lande, R. 1993. Risks of population extiction from demographic and environmental stochasticity and random catastrophs. *The American Naturalist,* 142: 911-27.

Landis, W. and Chapman, P. 2011. Well past time to stop using LOELs and LOELs. *Integrated Environmental Assessment and Management,* 7(4): vi-viii.

Laskowski, R. 1995. Some good reasons to ban the use of the NOEC, LOEC, and related concepts in ecotoxicology. *Oikos,* 73: 140-44.

Lauck, J. E. 1979. 'Final report of field study: Ortho Bolero 8EC--Rice, 1979. (Unpublished study received by EPA Dec. 11, 1979 under 239-2450; submitted by Chevron Chemical Co., Richmond, California.; CDL:241490-F). MRID 25179.' in *Final report of field study: Ortho Bolero 8EC--Rice, 1979. (Unpublished study received by EPA Dec. 11, 1979 under 239-2450; submitted by Chevron Chemical Co., Richmond, California.; CDL:241490-F). MRID 25179.*

Leidy, R. A. 1984. Distribution and ecology of stream fishes in the San Francisco Bay drainage. *Hilgardia,* 52: 1-175.

Lerner, D. T., Bjornsson, B. T., and McCormick, S. D. 2007a. Aqueous exposure to 4-nonylphenol and 17 beta-estradiol increases stress sensitivity and disrupts ion regulatory ability of juvenile Atlantic salmon. *Environmental Toxicology and Chemistry,* 26(7): 1433-40.

--- 2007b. Larval exposure to 4-nonylphenol and 17 beta-estradiol affects physiological and behavioral development of seawater adaptation in atlantic salmon smolts. *Environmental Science & Technology,* 41(12): 4479-85.

Levin, P. S. 2003. Regional differences in responses of Chinook salmon populations to large-scale climatic patterns. *Journal of Biogeography,* 30(5): 711-17.

Liebig, M., Schmidt, G., Bontje, D., Kooi, B. W., Streck, G., Traunspurger, W., and Knacker, T. 2008. Direct and indirect effects of pollutants on algae and algivorous ciliates in an aquatic indoor microcosm. *Aquatic Toxicology,* 88(2): 102-10.

Liermann, M. and Hilborn, R. 2001. Depensation: evidence, models, and implications. *Fish and Fisheries,* 2: 33-58.

Likens, G. E., Bormann, F. H., Johnson, N. M., Fisher, D. W., and Pierce, R. S. 1970. Effects of forest cutting and herbicide treatment on nutrient budgets in Hubbard Brook watershed ecosystem. *Ecological Monographs,* 40(1): 23-&.

Limm, M. P. and Marchetti, M. P. 2003. 'Contrasting patterns of juvenile chinook salmon (*Oncorhynchus tshawytschaw*) growth, diet, and prey densities in off-channel and main stem habitats on Sacramento River.' in *Contrasting patterns of juvenile chinook salmon (Oncorhynchus tshawytschaw) growth, diet, and prey densities in off-channel and main stem habitats on Sacramento River,* 35. Chico, California: The Nature Conservancy.

--- 2009. Juvenile Chinook salmon (*Oncorhynchus tshawytscha*) growth in off-channel and main-channel habitats on the Sacramento River, CA using otolith increment widths. *Environmental Biology of Fishes,* 85: 141-51.

Linbo, T., Baldwin, D., McIntyre, J., and Scholz, N. 2009. EFFECTS OF WATER HARDNESS, ALKALINITY, AND DISSOLVED ORGANIC CARBON ON THE TOXICITY OF COPPER TO THE LATERAL LINE OF DEVELOPING FISH. *Environmental Toxicology and Chemistry,* 28(7): 1455-61.

Linbo, T., Stehr, C., Incardona, J., and Scholz, N. 2006. Dissolved copper triggers cell death in the peripheral mechanosensory system of larval fish. *Environmental Toxicology and Chemistry,* 25(2): 597-603.

Lindley, S. T., Grimes, C. B., Mohr, M. S., Peterson, W., Stein, J., Anderson, J. T., Botsford, L. W., Bottom, D. L., Busack, C. A., Collier, T. K., Ferguson, J., Garza, J. C., Grover, A. M., Hankin, D. G., Kope, R. G., Lawson, P. W., Low, A., MacFarlane, R. B., Moore, K., Palmer-Zwahlen, M., Schwing, F. B., Smith, J., Tracy, C., Webb, R., Wells, B. K., and Williams, T. H. 2009. 'What caused the Sacramento River fall Chinook stock collapse?' in *What caused the Sacramento River fall Chinook stock collapse?* , 57 pages plus a 61-page appendix.

Lindley, S. T., Schick, R., Agrawal, A., Goslin, M., Pearson, T., Mora, E., Anderson, J. J., May, B., Greene, S., Hanson, C., Low, A., McEwan, D., MacFarlane, R. B., Swanson, C., and Williams, J. G. 2006a. Historical population structure of Central Valley steelhead and its alteration by dams. *San Francisco Estuary and Watershed Science,* 4(1): 1-19.

Lindley, S. T., Schick, R., May, B. P., Anderson, J. J., Greene, S., Hanson, C., Low, A., McEwan, D., MacFarlane, R. B., Swanson, C., and Williams, J. G. 2004a. 'Population structure of threatened and endangered Chinook salmon ESU in California's Central

Valley basin.' in *Population structure of threatened and endangered Chinook salmon ESU in California's Central Valley basin*. Santa Cruz, CA: NMFS Southwest Science Center NOAA-TM-NMFS-SWFSC-360.

--- 2004b. 'Population Structure of Threatened and Endangered Chinook Salmon ESUs in California's Central Valley Basin.' in *Population Structure of Threatened and Endangered Chinook Salmon ESUs in California's Central Valley Basin* ed. N. Commerce, NMFS, 56. Silver Spring, Maryland: NOAA/National Marine Fisheries Service.

Lindley, S. T., Schick, R. S., Agrawal, A., Goslin, M. N., Pearson, T. E., Mora, E., Anderson, J. J., May, B. P., Green, S., Hanson, C. H., Low, A., McEwan, D., MacFarlane, R. B., Swanson, C., and Williams, J. G. 2006b. Historical population structure of Central Valley steelhead and its alterations by dams. *San Francisco Estuary & Watershed Science,* 4(1): 1-19.

Lindley, S. T., Schick, R. S., Mora, E., Adams, P. B., Anderson, J. J., Greene, S., Hanson, C., May, B., McEwan, D., MacFarlane, R. B., Swanson, C., and Williams, J. G. 2007. Framework for assessing viability of threatened and endangered Chinook salmon and steelhead in the Sacramento - San Joaquin Basin. *San Francisco Estuary and Watershed Science,* 5(4).

Linley, T. J. 2001. A comparison of first feeding characteristics in two populations of Chinook salmon. *Transaction of American Fisheries Sociery,* 130: 519-25.

Little, E. E. and Finger, S. 1990. Swimming behavior as an indicator of sublethal toxicity in fish. *Environmental Toxicology and Chemistry,* 9: 13-19.

Luo, Q., Ban, M., Ando, H., Kitahashi, T., Bhandari, R. K., McCormick, S. D., and Urano, C. 2005. Distinct effects of 4-nonylphenol and estrogen-17 beta on expression of estrogen receptor alpha gene in smolting sockeye salmon. *Comparative Biochemistry and Physiology C-Toxicology & Pharmacology,* 140(1): 123-30.

Madsen, S. S., Skovbolling, S., Nielsen, C., and Korsgaard, B. 2004. 17-beta estradiol and 4-nonylphenol delay smolt development and downstream migration in Atlantic salmon, *Salmo salar*. *Aquatic Toxicology,* 68(2): 109-20.

Magnusson, A. and Hilborn, R. 2003. Estuarine Influence on Survival Rates of Coho (*Oncorhynchus kisutch*) and Chinook Salmon (*Oncorhynchus tshawytscha*) Released from Hatcheries on the U. S. Pacific Coast. *Estuaries,* 26(4): 1094-103.

Mangel, M. 1994. Climate change and salmonid life history variation. *Deep-Sea Research,* 41(1): 75-106.

Mantua, N. J., Hare, S. R., Zhang, Y., Wallace, J. M., and Francis, R. C. 1997. A pacific interdecadal climate oscillation with impacts on salmon production. *Bulletin of the American Meteorological Society,* (78): 1069-79.

Marcotte, B. D. 1984a. 'Life history, status and habitat requirements of spring-run Chinook salmon in California.' in *Life history, status and habitat requirements of spring-run Chinook salmon in California* ed. U. S. F. S. C. Lassen National Forest, California, 34.

--- 1984b. 'Life History, Status and Habitat Requirements of Spring-run Chinook Salmon in California.' in *Life History, Status and Habitat Requirements of Spring-run Chinook Salmon in California* ed. L. N. Forest, 36. Chester, CA: U.S. National Forest Service.

Marshall, D. E. and Britton, E. W. 1990. Carrying capacity of coho salmon streams. *Canadian Reportns on Fisheries and Aquatic Science,* (2058): 1-32.

Martin, C. D., Gaines, P. D., and Johnson, R. R. 2001. 'Estimating the abundance of Sacramento River juvenile winter Chinook salmon with comparisons to adult escapement.' in *Estimating the abundance of Sacramento River juvenile winter Chinook salmon with comparisons to adult escapement.* ed. U. S. F. a. W. Service. Red Bluff, California.

Maund, S., Biggs, J., Williams, P., Whitfield, M., Sherratt, T., Powley, W., Heneghan, P., Jepson, P., and Shillabeer, N. 2009. The influence of simulated immigration and chemical persistence on recovery of macroinvertebrates from cypermethrin and 3,4-dichloroaniline exposure in aquatic microcosms. *Pest Management Science,* 65(6): 678-87.

Mayer, F. L. and Ellersieck, M. R. 1986. 'Manual of acute toxicity: Interpretation and data base for 410 chemicals and 66 species of freshwater animals.' in *Manual of acute toxicity: Interpretation and data base for 410 chemicals and 66 species of freshwater animals,* 579. USFWS.

McClure, M. M., Holmes, E. E., Sanderson, B. L., and Jordan, C. E. 2003. A large-scale, multispecies status assessment: Anadromous salmonids in the Columbia River basin. *Ecological Applications,* 13(4): 964-89.

McCormick, S. D., Lerner, D. T., Monette, M. Y., Nieves-Puigdoller, K., Kelly, J. T., and Bjornsson, B. 2009. Taking it with you when you go: How pertubations to the freshwater enviornment, including temperature, dams, and contaminants, affect marine survival of salmon. *American Fisheries Society Symposium,* (69): 195-214.

McCormick, S. D., O'Dea, M. F., Moeckel, A. M., Lerner, D. T., and Bjornsson, B. T. 2005. Endocrine disruption of parr-smolt transformation and seawater tolerance of Atlantic salmon by 4-nonylphenol and 17 beta-estradiol. *General and Comparative Endocrinology,* 142(3): 280-88.

McCullough, D. A. 1999. 'A review and synthesis of effects of alterations to the water temperature regime on freshwater life stages of salmonids, with special reference to Chinook salmon.' in *A review and synthesis of effects of alterations to the water temperature regime on freshwater life stages of salmonids, with special reference to Chinook salmon,* 279. Portland, Oregon: Columbia River Inter-Tribal Fish Commission.

McElhany, P., Ruckleshaus, M. H., Ford, M. J., Wainwright, T. C., and Bjorkstedt, E. P. 2000. 'Viable salmonid populations and the recovery of evolutionarily significant units.' in *Viable salmonid populations and the recovery of evolutionarily significant units*, 156. Seattle, Washington: U.S. Department of Commerce.

McEwan, D. 2001a. 'Central Valley steelhead.' in *Central Valley steelhead* ed. R. L. Brown, 1-44. California Department of Fish and Game.

McEwan, D. and Jackson, T. A. 1996a. Steelhead restoration and management plan for California. *California Department of Fish and Game, Sacramento, California*.

--- 1996b. 'Steelhead restoration and management plan for California.' in *Steelhead restoration and management plan for California*. Sacramento, CA: California Department of Fish and Game.

McEwan, D. R. 2001b. 'Central Valley steelhead.' in *Central Valley steelhead* ed. R. L. Brown, 1-43. Sacramento, California: California Department of Fish and Game.

McGill, R. R. J. 1987. 'Land use changes in the Sacramento River riparian zone, Redding to Colusa. A third update: 1982-1987.' in *Land use changes in the Sacramento River riparian zone, Redding to Colusa. A third update: 1982-1987,* 19 pages. Department of Water Resources, Northern District.

McIntyre, J., Baldwin, D., Meador, J., and Scholz, N. 2008. Chemosensory deprivation in juvenile coho salmon exposed to dissolved copper under varying water chemistry conditions. *Environmental Science & Technology,* 42(4): 1352-58.

McReynolds, T. R., Garman, C. E., Ward, P. D., and Schommer, M. C. 2005. 'Butte and Big Chico Creeks spring-run Chinook salmon, *Oncorhynchus tshawytscha* life history investigation, 2003-2004.' in *Butte and Big Chico Creeks spring-run Chinook salmon, Oncorhynchus tshawytscha life history investigation, 2003-2004*. California Department of Fish and Game.

Mebane, C. and Arthaud, D. 2010. Extrapolating Growth Reductions in Fish to Changes in Population Extinction Risks: Copper and Chinook Salmon. *Human and Ecological Risk Assessment,* 16(5): 1026-65.

Meehan, W. R. 1991. 'Introduction and overview.' in *Introduction and overview* ed. W. R. Meehan, 1-16. Bethesda, MD: American Fisheries Society.

Meehan, W. R. and Bjornn, T. C. 1991. 'Salmonid distributions and life histories.' in *Salmonid distributions and life histories* ed. W. R. Meehan, 47-82. Bethesda, MD: American Fisheries Society.

Michael, J. L. 2003. Environmental fate and impacts of sulfometuron on watersheds in the southern United States. *Journal of Environmental Quality,* 32(2): 456-65.

Michael, J. L., Batzer, D. P., Fischer, J. B., and Gibbs, H. L. 2006. Fate of the herbicide sulfometuron methyl (Oust((R))) and effects on invertebrates in drainages of an intensively managed plantation. *Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere,* 36(10): 2497-504.

Michael, J. L., Webber, E. C., Bayne, D. R., Fischer, J. B., Gibbs, H. L., and Seesock, W. C. 1999. Hexazinone dissipation in forest ecosystems and impacts on aquatic communities. *Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere,* 29(7): 1170-81.

Miller, J. A. and Simenstad, C. A. 1997. A comperative assessment of a natural and created estuarine slough as rearing habitat for juvenile Chinook and coho salmon. *Estuaries,* 20(4): 792-806.

Mills, S. K. and Beatty, J. H. 1979. The propensity interpretation of fitness. *Philosophy of Science,* 46: 263-86.

Minshall, G. W., Royer, T. V., and Robinson, C. T. 2001. Response of the Cache Creek marcroinvertebrates during the first 10 years following disturbance by the 1988 Yellowstone wildfires. *Canadian Journal of Fisheries and Aquatic Sciences,* 58(6): 1077-88.

Monroe, M., Kelly, J., and Lisowski, N. 1992. 'State of the estuary, a report of the conditions and problems in the San Francisco Bay/Sacramento-San Joaquin Delta Estuary.' in *State of the estuary, a report of the conditions and problems in the San Francisco Bay/Sacramento-San Joaquin Delta Estuary*, 269 pages.

Montgomery, D. R., E. M. Beamer, G. R. Pess, and T. P. Quinn 1999. Channel type and salmonid spawning distribution and abundance. *Canadian Journal of Fisheries and Aquatic Sciences,* 56: 377-87.

Moore, D. and Caux, P. 1997. Estimating low toxic effects. *Environmental Toxicology and Chemistry,* 16: 794-801.

Moran, K. D. 2012. 'Thiobencarb monitoring data evaluation. Memorandum to Elissa Callman. TDC Environmental.' in *Thiobencarb monitoring data evaluation. Memorandum to Elissa Callman. TDC Environmental.*

Morley, S. A., Garcia, P. S., Bennett, T. R., and Roni, R. 2005. Juvenile salmonid (*Oncorhynchus* spp.) use of constructed and natural side channels in Pacific Northwest rivers. *Canadian Journal of Fisheries and Aquatic Sciences*, 62: 2811-21.

Morris, W. F. and Doak, D. F. 2002. *Quantitative conservation biology. Theory and practice of population viability analysis.* Sunderland, MA: Sinauer Associates Inc.

Mount, J. F. 1995. 'California rivers and streams: The conflict between fluvial process and land use.' in *California rivers and streams: The conflict between fluvial process and land use*. Berkeley, CA: University California Press.

Moyle, P. B. 2002. *Inland fish of California.* Berkeley, California: University Press.

Mundie, J. H., Simpson, K. S., and Perrin, C. J. 1991. Responses of stream periphyton and benthic insects to increases in dissolved inorganic phosphorus in a mesocosm. *Canadian Journal of Fisheries and Aquatic Sciences,* 48(11): 2061-72.

Myers, J. M., Kope, R. G., Bryant, G. J., Teel, D., Lierheimer, L. J., Wainwright, T. C., Lindley, S. T., and Waples, R. S. 1998. 'Status review of chinook salmon from Washington, Idaho, Oregon, and California.' in *Status review of chinook salmon from Washington, Idaho, Oregon, and California*, 443. Seattle, Washington: U.S. Department of Commerce.

Myrick, C. A. and Cech, J. J. J. 2005. Effects of temperature on the growth, food consumption, and termal tolerance of age-0 Nimbus-strain steelhead. *North American Journal of Aquaculture*, 67: 324-30.

Neary, D. G., Bush, P. B., and Michael, J. L. 1993. Fate, Dissipation and Environmental-Effects of Pesticides in Southern Forests - a Review of a Decade of Research Progress. *Environmental Toxicology and Chemistry*, 12(3): 411-28.

Nichols, F., Cloern, J., Luoma, S., and Peterson, D. 1986. The modification of an estuary. *Science*, 231(4738): 567-73.

Nielsen, L. W. and Dahllof, I. 2007. Direct and indirect effects of the herbicides Glyphosate, Bentazone and MCPA on eelgrass (Zostera marina). *Aquatic Toxicology,* 82(1): 47-54.

NMFS 1996a. 'Factors for decline: a supplement to the notice of determination for west coast steelhead under the Endangered Species Act.' in *Factors for decline: a supplement to the notice of determination for west coast steelhead under the Endangered Species Act*. Portland, OR and Long Beach, CA: National Marine Fisheries Service, Protected Resources Division.

--- 1996b. 'Making Endangered Species Act determinations of effect for individual or group actions at the watershed scale.' in *Making Endangered Species Act determinations of effect for individual or group actions at the watershed scale*. Portland, OR: National Marine Fisheries Service, Environmental and Technical Services Branch, Habitat Conservation Branch.

--- 1997. 'National Marine Fisheries Service Proposed Recovery Plan for the Sacramento River Winter-run Chinook Salmon.' in *National Marine Fisheries Service Proposed Recovery Plan for the Sacramento River Winter-run Chinook Salmon.* ed. S. R. National Marine Fisheries Service, 217. Long Beach, California.

--- 1998. 'Factors Contributing to the Decline of Chinook Salmon: An Addendum to the 1996 West Coast Steelhead Factors For Decline Report.' in *Factors Contributing to the Decline of Chinook Salmon: An Addendum to the 1996 West Coast Steelhead Factors For*

Decline Report. Portland, OR: National Marine Fisheries Service, Protected Resource Division.

--- 2005. 'Final assessment of the National Marine Fisheries Service's critical habitat analytical review teams (CHARTs) for seven salmon and steelhead evolutionarily significant units (ESUs) in California.' in *Final assessment of the National Marine Fisheries Service's critical habitat analytical review teams (CHARTs) for seven salmon and steelhead evolutionarily significant units (ESUs) in California*, 23 + appendices. Long Beach, California: Protected Resources, National Marine Fisheries Service.

--- 2006. 'Draft Columbia River estuary recovery plan module.' in *Draft Columbia River estuary recovery plan module*. Lower Columbia River Estuary Partnership under contract to NOAA Fisheries.

--- 2008a. 'Endangered Species Act section 7 consultation: biological opinion on Environmental Protection Agency registration of pesticides containing chlorpyrifos, diazinon, and malathion.' in *Endangered Species Act section 7 consultation: biological opinion on Environmental Protection Agency registration of pesticides containing chlorpyrifos, diazinon, and malathion*, 482. Silver Spring, Maryland: U.S. Department of Commerce.

--- 2008b. 'Endangered Species Act Section 7(a)(2) Consultation Biological Opinion and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Consultation on remand for operaiton of the Federal Columbia River Power System, 11 Bureau of Reclamation Projects in the Columbia Basin and ESA Section 10(a)(1)(A) permit for juvenile fish transportation program.' in *Endangered Species Act Section 7(a)(2) Consultation Biological Opinion and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Consultation on remand for operaiton of the Federal Columbia River Power System, 11 Bureau of Reclamation Projects in the Columbia Basin and ESA Section 10(a)(1)(A) permit for juvenile fish transportation program*. U.S. Department of Commerce.

--- 2008c. 'National Marine Fisheries Service endangered species act section 7 consultation: Environmental Protection Agency registration of pesticides containing chlorpyrifos, diazinon, and malathion.' in *National Marine Fisheries Service endangered species act section 7 consultation: Environmental Protection Agency registration of pesticides containing chlorpyrifos, diazinon, and malathion.*, 482. Silver Spring, Maryland: U.S. Department of Commerce.

--- 2009a. 'Biological Opinion and Conference Opinion on the long-term operations of the Central Valley Project and State Water Project.' in *Biological Opinion and Conference Opinion on the long-term operations of the Central Valley Project and State Water Project*, 844. Sacramento, California: U.S. Department of Commerce.

--- 2009b. 'Endangered Species Act section 7 consultation: biological opinion on Environmental Protection Agency registration of pesticides containing carbaryl, carbofuran, and methomyl.' in *Endangered Species Act section 7 consultation: biological opinion on Environmental Protection Agency registration of pesticides containing carbaryl, carbofuran, and methomyl.*, 607. Silver Spring, Maryland: U.S. Department of Commerce.

--- 2009c. 'Endangered Species Act Section 7(a)(2) Consultation Biological Opinion and Conference Opinion on the Long-Term Operations of the Central Valley Project (CVP) and State Water Project (SWP).' in *Endangered Species Act Section 7(a)(2) Consultation Biological Opinion and Conference Opinion on the Long-Term Operations of the Central*

Valley Project (CVP) and State Water Project (SWP), 844 pages. Long Beach, CA: National Marine Fisheries Service, Southwest Region.

--- 2009d. 'Endangered Species Act – Section 7 Consultation Biological Opinion & Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Consultation Nez Perce National Forest Noxious Weeds Programmatic, Middle Salmon River 17090207, Lower Salmon River 17060209, Main Salmon River 17060209, Little Salmon River 17060210, Selway River 17060302 and 17060302, Middle Fork Clearwater River 17060304, and South Fork Clearwater River 17060305, Idaho County, Idaho.' in *Endangered Species Act – Section 7 Consultation Biological Opinion & Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Consultation Nez Perce National Forest Noxious Weeds Programmatic, Middle Salmon River 17090207, Lower Salmon River 17060209, Main Salmon River 17060209, Little Salmon River 17060210, Selway River 17060302 and 17060302, Middle Fork Clearwater River 17060304, and South Fork Clearwater River 17060305, Idaho County, Idaho*.

--- 2009e. 'NMFS Endangered Species Act Section 7 Consultation, Biological Opinion: Environmental Protection Agency Registration of Pesticides Containing Carbaryl, Carbofuran, and Methomyl.' in *NMFS Endangered Species Act Section 7 Consultation, Biological Opinion: Environmental Protection Agency Registration of Pesticides Containing Carbaryl, Carbofuran, and Methomyl*. Washington, D.C.: U.S. Department of Commerce.

--- 2010a. 'NMFS Endangered Species Act Section 7 Consultation, Biological Opinion: Environmental Protection Agency registration of pesticides containing azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, naled, methamidophos, methidathion, methyl parathion, phorate, and phosmet.' in *NMFS Endangered Species Act Section 7 Consultation, Biological Opinion: Environmental Protection Agency registration of pesticides containing azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, naled, methamidophos, methidathion, methyl parathion, phorate, and phosmet*. Silver Spring, MD: U.S. Department of Commerce.

--- 2010b. 'NMFS Endangered Species Act Section 7 Consultation, Biological Opinion: Environmental Protection Agency registration of pesticides containing azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, naled, methamidophos, methidathion, methyl parathion, phorate and phosmet.' in *NMFS Endangered Species Act Section 7 Consultation, Biological Opinion: Environmental Protection Agency registration of pesticides containing azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, naled, methamidophos, methidathion, methyl parathion, phorate and phosmet*. Washington, D.C.: U.S. Department of Commerce.

--- 2011a. 'Central Valley Recovery Domain, 5-Year Review: Summary and Evaluation of Central Valley Spring-run Chinook Salmon ESU.' in *Central Valley Recovery Domain, 5-Year Review: Summary and Evaluation of Central Valley Spring-run Chinook Salmon ESU*, 34 pages. Long Beach, CA: National Marine Fisheries Service, Southwest Region.

--- 2011b. 'NMFS Endangered Species Act Consultation, Biological Opinion: Environmental Protection Agency's Issuance of a Pesticides General Permit to Authorize Pesticide Discharges to Waters of the US.' in *NMFS Endangered Species Act Consultation, Biological Opinion: Environmental Protection Agency's Issuance of a Pesticides General Permit to Authorize Pesticide Discharges to Waters of the US*. Silver Spring,

MD: National Marine Fisheries Service, Office of Protected Resources: Interagency Cooperation Division.
--- 2011c. 'Toxicity of eleven pesticides to embryonic zebrafish.' in *Toxicity of eleven pesticides to embryonic zebrafish*, 16. Northwest Fisheries Science Center.
--- 2011d. 'Toxicity of six herbicides and fungicides to embryonic zebrafish.' in *Toxicity of six herbicides and fungicides to embryonic zebrafish*, 13. Seattle: Northwest Fisheries Science Center.
NMFS/CDFG 2001. 'Final report on anadromous salmon fish hatcheries in California.' in *Final report on anadromous salmon fish hatcheries in California* ed. P. b. J. H. R. Committee.
Nobriga, M. L. and Cadrett, P. 2003. Differences among hatchery and wild steelhead: evidence from Delta fish monitoring programs. *Interagency Ecological Program for the San Francisco Estuary Newsletter,* 14(3): 30-38.
Norberg-King, T. J., Durhan, E. J., Ankley, G. T., and Robert, E. 1991. Application of toxicity identification evaluation procedures to the ambient waters of the Colusa Basin Drain, Califronia. *Environmental Toxicology and Chemistry,* 10(7): 891-900.
Opperman, J. J. and Merenlender, A. M. 2004. The effectiveness of riparian restoration for improving instream fish habitat in four hardwood-dominated California streams. *North American Journal of Fisheries Management,* 24: 822-34.
Orlando, J. L. and Kuivila, K. M. 2004. 'Changes in rice pesticide use and surface water concentrations in the Sacramento River watershed, California:U.S. Geological Survey Scientific Investigations Report 2004-5097, 28p.' in *Changes in rice pesticide use and surface water concentrations in the Sacramento River watershed, California:U.S. Geological Survey Scientific Investigations Report 2004-5097, 28p.*
Orr, J. C., Fabry, V. J., Aumont, O., Bopp, L., Doney, S. C., Feely, R. A., Gnanadesikan, A., Gruber, N., Ishida, A., Joos, F., Key, R. M., Linsay, K., Maier-Reimer, E., Matear, R., Monfray, P., Mouchet, A., Najjar, R. G., Plattner, G., Rodgers, K. B., Sabine, C. L., Sarmiento, J. L., Schlitzer, R., Slater, R. D., Totterdell, I. J., Weirig, M., Yamanka, Y., and Yool, A. 2005. Anthropogenic ocean acidification over the twenty-first century and its impact on calcifying organisms. *Nature,* 43(681-686).
Parveen, S., Kohguchi, T., Biswas, M., and Nakagoshi, N. 2005. Predicting herbicides concentrations in paddy water and runoff to the river basin. *Journal of Environmental Sciences-China,* 17(4): 631-36.
Pearcy, W. G. 1992. Ocean ecology of North Pacific salmonids. *University of Washington. University of Washington Press, Seattle, Washington*: 179 p.
--- 1997. What have we learned in the last decade? What are research priorities? *in Emmett, R.L. and M.H. Schiewe (editors). NOAA-NMFS-NWFSC TM-29: Estuarine and Ocean Survival of Northeastern Pacific Salmon: Proceedings of the workshop. U.S. Dep. Commer.*: 5 p.
Pentyala, S. N. and Chetty, C. S. 1993. Comparative study on the changes in AChE and ATPase activities in neonate and adult rat brains under thiobencarb stress. *Journal of Applied Toxicology,* 13: 39-42.
Perez, G. L., Torremorell, A., Mugni, H., Rodriguez, P., Vera, M. S., Do Nascimento, M., Allende, L., Bustingorry, J., Escaray, R., Ferraro, M., Izaguirre, I., Pizarro, H., Bonetto, C., Morris, D. P., and Zagarese, H. 2007. Effects of the herbicide roundup on freshwater microbial communities: A mesocosm study. *Ecological Applications,* 17(8): 2310-22.

Perry, R. W., Bradford, M. J., and Grout, J. A. 2003. Effects of disturbance on contribution of energy sources to growth of juvenile chinook salmon (Oncorhynchus tshawytscha) in boreal streams. *Canadian Journal of Fisheries and Aquatic Sciences*, 60(4): 390-400.
Persichino, J., Christian, E., and Carter, R. 1998. Short-term effects of Bolero on the gill apparatus of a small number of mosquitofish (*Gambusia affinis*). *Bulletin of Environmental Contamination and Toxicology*, 61: 162-68.
Pesce, S., Fajon, C., Bardot, C., Bonnemoy, F., Portelli, C., and Bohatier, J. 2006. Effects of the phenylurea herbicide diuron on natural riverine microbial communities in an experimental study. *Aquatic Toxicology*, 78(4): 303-14.
Phillips, R. W. and Campbell, H. J. 1961. The embryonic survival of coho salmon and steelhead trout as influenced by some environmental conditions in gravel beds. *Annual Report to Pacific Marine Fisheries Commission*, 14: 60-73.
PNERC 2002. 'Willamette River Basin Planning Atlas: Trajectories of Environmental and Ecological Change. .' in *Willamette River Basin Planning Atlas: Trajectories of Environmental and Ecological Change.* ed. D. Hulse, Gregory, S., & Baker, J. Eugene, OR: Institute for a Sustainable Environment, University of Oregon.
Poleksic, V. and Mitrovic-Tutundzic, V. 1994. 'Fish gills as a monitor of sublethal and chronic effects of pollutants on freshwater fish.' in *Fish gills as a monitor of sublethal and chronic effects of pollutants on freshwater fish* eds. R. Muller & R. Lloyd, 339-53. Oxford, UK: Fishing News Books.
Pratt, J. R., Melendez, A. E., Barreiro, R., and Bowers, N. J. 1997. Predicting the ecological effects of herbicides. *Ecological Applications*, 7(4): 1117-24.
Project, G. 1999. 'Baylands ecosystem habitat goals: A report of habitat recommendations prepared by the San Francisco Bay Area Wetlands Ecosystem Goals Project.' in *Baylands ecosystem habitat goals: A report of habitat recommendations prepared by the San Francisco Bay Area Wetlands Ecosystem Goals Project* ed. S. F. U.S. Environmental Protection Agency. Oakland, CA: San Francisco Bay Regional Water Quality Control Board.
Raimondo, S. and McKenney, C. J. 2006. From organisms to populations: Modeling aquatic toxicity data across two levels of biological organization. *Environmental Toxicology and Chemistry*, 25: 589-96.
Rand, G. M., P.G. Wells, P. G., and McCarty, L. S. 1995. 'Introduction to aquatic toxicology.' in *Introduction to aquatic toxicology* ed. G. M. Rand, 3-66. Bristol, PA: Taylor and Francis.
Reimers, P. E. 1973. The length of residence of fall Chinook in Sixes River, Oregon. *Research Report of the Fisheries Commission of Oregon*, 4(2): 43.
Relyea, R. A. 2009. A cocktail of contaminants: how mixtures of pesticides at low concentrations affect aquatic communities. *Oecologia*, 159(2): 363-76.
Reynolds, F. L., Mills, T. J., Benthin, R., and Low, A. 1993. 'Restoring Central Valley streams: a plan for action.' in *Restoring Central Valley streams: a plan for action*. Sacramento, CA: California Department of Fish and Game, Inland Fisheries Division.
Richardson, J. S., Taylor, E., Schluter, D., Pearson, M., and Hatfield, T. 2010. Do riparian zones qualify as critical habitat for endangered freshwater fishes? *Canadian Journal of Fisheries and Aquatic Sciences*, 67: 1197-204.
Rinne, J. N. 2004. Forests, fish and fire: relationships and management implications for fishes in the southwestern USA. *pages 151-156 in G.J. Scrimgeour, G.Eisler, B. McCulloch, U. Silins, and M. Monita (editors). Forest Land - Fish Conference II - Ecosystem*

Stewardship through Collaboration. Proceedings of the Forest-Land-Fish Conference II, April 26-28, 2004. Edmonton, Alberta, Canada.

Robison, G. E. and Beschta, R. L. 1990. Identifying trees in riparian areas that can provide coarse woody debris to streams. *Forest Service*, 36: 790-801.

Roby, D. D., Colis, K., Lyons, D. E., Suzuki, Y., Adkins, J. Y., Reinalda, L., Hostetter, N., and Adrean, L. 2008. 'Research, monitoring, and evaluation of avian predation on salmonid smolts in the lower and Mid-Columbia River.' in *Research, monitoring, and evaluation of avian predation on salmonid smolts in the lower and Mid-Columbia River*, 85. Corvallis, OR: USGS - Oregon Cooperative Fish and Wildlife Research Unit Department of Fisheries and Wildlife.

Roby, D. D., Collis, K., Adkins, J. Y., Couch, C., Courtot, B., Lord, R., Lyons, D. E., Suzuki, Y., Evans, A., and Hawbecker, M. 2006. 'Research, monitoring, and evaluation of avian predation on salmonid smolts in the lower and mid-Columbia River.' in *Research, monitoring, and evaluation of avian predation on salmonid smolts in the lower and mid-Columbia River*, 130 p. Portland, Oregon.

Rohr, J. R. and Crumrine, P. W. 2005. Effects of an herbicide and an insecticide on pond community structure and processes. *Ecological Applications*, 15(4): 1135-47.

Rohr, J. R., Elskus, A. A., Shepherd, B. S., Crowley, P. H., McCarthy, T. M., Niedzwiecki, J. H., Sager, T., Sih, A., and Palmer, B. D. 2004. Multiple stressors and salamanders: Effects of an herbicide, food limitation, and hydroperiod. *Ecological Applications,* 14(4): 1028-40.

Rondorf, D. W., Gray, G. A., and Fairley, R. B. 1990. Feeding ecology of subyearling Chinook salmon in riverine and reservoir habitats of the Columbia River. *Transaction of American Fisheries Sociery,* 119: 16-24.

Roni, P. 2002. Habitat use by fishes and pacific giant salamanders in small western Oregon and Washington streams. *Transactions of the American Fisheries Society,* 131(4): 743-61.

Rosemond, A. D., Mulholland, P. J., and Brawley, S. H. 2000. Seasonally shifting limitation of stream periphyton: response of algal populations and assemblage biomass and productivity to variation in light, nutrients, and herbivores. *Canadian Journal of Fisheries and Aquatic Sciences*, 57(1): 66-75.

Ross, L. J. and Sava, R. J. 1986. Fate of thiobencarb and molinate in rice fields. *Journal of Environmental Quality*, 15: 220-25.

Ruggerone, G. T. and Goetz, F. A. 2004. Survival of Puget Sound Chinook salmon (*Oncorhynchus tshawytscha*) in resposne to climate-induced competition with pink slamon (*Oncorhynchus gorbuscha*). *Canadian Journal of Fisheries and Aquatic Sciences,* 61: 1756-70.

Ryan, B. A., Carper, M., Marsh, D., Elliott, D., Murray, T., Applegate, L., McKibben, C., and Mosterd, S. 2006. Alternative barging strategies to improve survival of transported juvenile salmonids. *Draft report sent to U.S. Army Corps of Engineers, Walla Walla, Washington*.

Sabater, C. and Carrasco, J. 1996. Effects of thiobencarb on the growth of three species of phytoplankton. *Bulletin of Environmental Contamination and Toxicology,* 56(6): 977-84.

Sancho, E., Fernandez-Vega, C., Sanchez, M., Ferrando, M. D., and Andreu-Moliner, E. 2000. Alterations on AChE activity of the fish Anguilla anguilla as response to herbicide-contaminated water. *Ecotoxicology and Environmental Safety,* 46(1): 57-63.

Sandahl, J. F., Baldwin, D. H., Jenkins, J. J., and Scholz, N. L. 2004. Odor-evoked field potentials as indicators of sublethal neurotoxicity in juvenile coho salmon (*Oncorhynchus*

kisutch) exposed to copper, chlorpyrifos, or esfenvalerate. *Canadian Journal of Fisheries Aquatic Sciences,* 64: 404-13.
Sanderson, B. L., Barnas, K. A., and Wargo Rub, A. M. 2009. Nonindigenous species of the Pacific Northwest: An overlooked risk to endangered salmon? *Bioscience,* 59(3): 245-56.
Sazaki, S. 1966. 'Distribution and food habits of king salmon, *Oncorhynchus tshawytscha*, and steelhead rainbow trout, *Salmo gairdnerii*, in the Sacramento-San Joaquin delta.' in *Distribution and food habits of king salmon, Oncorhynchus tshawytscha, and steelhead rainbow trout, Salmo gairdnerii, in the Sacramento-San Joaquin delta* eds. J. L. Turner & D. W. Kelly.
Schafer, R. B., Caquet, T., Siimes, K., Mueller, R., Lagadic, L., and Liess, M. 2007. Effects of pesticides on community structure and ecosystem functions in agricultural streams of three biogeographical regions in Europe. *Science of the Total Environment,* 382(2-3): 272-85.
Schaffter, R. 1980. 'Fish occurrence, size, and distribution in the Sacramento River near Hood, California during 1973 and 1974.' in *Fish occurrence, size, and distribution in the Sacramento River near Hood, California during 1973 and 1974*. California Department of Fish and Game.
--- 1997. White sturgeon spawning migrations and location of spawning habitat in the Sacramento River, California. *California Department of Fish and Game,* 83: 1-20.
Schmetterling, D., Clancy, C., and Brandt, T. 2001. Effects of riprap bank reinforcement on stream salmonids in the western United States. *Fisheries*, 26(7): 6-13.
Schoenfuss, H. L., Baftell, S. E., Bistodeau, T. B., Cediel, R. A., Grove, K. J., Zintek, L., Lee, K. E., and Barber, L. B. 2008a. Impairment of the reproductive potential of male fathead minnows by environmentally relevant exposures to 4-nonylphenolf. *Aquatic Toxicology,* 86(1): 91-98.
Schoenfuss, H. L., Bartell, S. E., Bistodeau, T. B., Cediel, R. A., Grove, K. J., Zintek, L., Lee, K. E., and Barber, L. B. 2008b. Impairment of the reproductive potential of male fathead minnows by environmentally relevant exposures to 4-nonylphenol. *Aquatic Toxicology,* 86(1): 91-98.
Segner, H. 2005. Developmental, reproductive, and demographic alterations in aquatic wildlife: Establishing causality between exposure to endocrine-active compounds (EACs) and effects. *Acta Hydrochimica Et Hydrobiologica,* 33(1): 17-26.
Sergeant, C. J. and Beauchamp, D. A. 2006. Effects of physical habitat ontogony on lentic habitat preference of juvenile Chinook salmon. *Transaction of American Fisheries Sociery,* 135: 1191-204.
Shapovalov, L. and Taft, A. C. 1954. 'The life histories of the steelhead rainbow trout (*Salmo gairdneri gairdneri*) and silver salmon (*Oncorhynchus kisutch*) with special reference to Waddell Creek, California, and recommendations regarding their management.' in *The life histories of the steelhead rainbow trout (Salmo gairdneri gairdneri) and silver salmon (Oncorhynchus kisutch) with special reference to Waddell Creek, California, and recommendations regarding their management,* 1-375. California Department of Fish and Game.
Slater, D. W. 1963a. Winter-run Chinook salmon in the Sacramento River, California, with notes on water temperature requirements at spawning. *U.S. Fish and Wildlife Service, Special Science Report Fisheries,* 461: 9.

--- 1963b. 'Winter-run Chinook salmon in the Sacramento River, California, with notes on water temperature requirements at spawning.' in *Winter-run Chinook salmon in the Sacramento River, California, with notes on water temperature requirements at spawning*, 9. U.S. Fish and Wildlife Service.

Smith, C. J. 2005. 'Salmon habitat limiting factors in Washington State.' in *Salmon habitat limiting factors in Washington State*, 222. Olympia, Washington: Washington State Conservation Commission.

Snider, B. and Titus, R. G. 2000a. 'Timing, Composition and Abundance of Juvenile Anadromous Salmonid Emigration in the Sacramento River near Knights Landing October 1996 - September 1997 ' in *Timing, Composition and Abundance of Juvenile Anadromous Salmonid Emigration in the Sacramento River near Knights Landing October 1996 - September 1997* 68 pages. California Department of Fish and Game, Habitat Conservation Division, Native Anadromous Fish and Watershed Branch, Stream Evaluation Program Technical Report No. 00-4.

--- 2000b. 'Timing, composition, and abundance of juvenile anadromous salmonid emigration in the Sacramento River near Knights Landing, October 1996-September 1997.' in *Timing, composition, and abundance of juvenile anadromous salmonid emigration in the Sacramento River near Knights Landing, October 1996-September 1997.* ed. H. C. D. California Department of Fish and Game.

Sommer, T., Harrell, W., and Nobriga, M. 2005a. Habitat use and stranding risk of juvenile Chinook salmon on a seasonal floodplain. *North American Journal of Fisheries Management,* 25(4): 1493-504.

Sommer, T., Harrell, W., Solger, A., Tom, B., and Kimmerer, W. 2004. Effects of flow variation on channel and floodplain biota and habitats of the Sacramento River, California, USA. *Aquatic Conservation-Marine and Freshwater Ecosystems,* 14(3): 247-61.

Sommer, T. R., Harrell, W. C., and Nobriga, M. L. 2005b. Habitat use and stranding risk of juvenile Chinook salmon on a seasonal floodplain. *North American Journal of Fisheries Management,* 25: 1493-504.

Sommer, T. R., Nobriga, M. L., Harrell, W. C., Batham, W., and Kimmerer, W. J. 2001. Floodplain rearing of juvenile chinook salmon: evidence of enhanced growth and survival. *Canadian Journal of Fisheries and Aquatic Sciences,* 58: 325-33.

Spence, B. C., Bjorkstedt, E. P., Garza, J. C., Smith, J. J., Hankin, D. G., Fuller, D., Jones, W. E., Macedo, R., Williams, T. H., and Mora, E. 2008. A framework for assessing the viability of threatened and endangered salmon and steelhead in the nort-central California Coast Recovery Domain. *NOAA Technical Memorandum NMFS,* NOAA-TM-NMFS-SWFSC-423: 173.

Spence, B. C., Lomnicky, G. A., Hughes, R. M., and Novitzki, R. P. 1996a. *An Ecosystem Approach to Salmonid Conservation.* Corvallis, OR: ManTech Environmental Research Corp.

Spence, B. C., Lomnicky, G. A., Hughs, R. M., and Novitzki, R. P. 1996b. An ecosystem approach to salmonid conservation. *Man Tech Environmental Research Services Corps, TR-4501-96-6057, Corvallis, Oregon.*

Stearns, S. C. 1982. The evolution of life histories. *New York, New York, Oxford University Press*.

Sudo, M., Kunimatsu, T., and Okubo, T. 2002. Concentration and loading of pesticide residues in Lake Biwa basin (Japan). *Water Research,* 36(1): 315-29.

Suter, G. I. 1996. Abuse of hypothesis testing statistics in ecological risk assessment. *Human and ecological risk assessment,* 2: 331-47.

Sweeney, B. W., Bott, T. L., Jackson, J. K., Kaplan, L. A., Newbold, J. D., Standley, L. J., Hession, W. C., and Horwitz, R. J. 2004. Riparian deforestation, stream narrowing, and loss of stream ecosystem services. *Proc Natl Acad Sci,* 101(39): 14132–37.

Swift III, C. H. 1979. 'Preferred stream discharges for salmon spawning and rearing in Washington.USGS Open-File Report 77-422 ' in *Preferred stream discharges for salmon spawning and rearing in Washington.USGS Open-File Report 77-422* 51p. USGS.

Tabor, R. A., Gearns, H. A., McCoy III, C. M., and Camacho, S. 2006. 'Nearshore habitat use by Chinook salmon in lentic systems of the Lake Washington basin.' in *Nearshore habitat use by Chinook salmon in lentic systems of the Lake Washington basin*, 94. Lacy, Washington: U.S. Fish and Wildlife Service.

Teel, D. J., Baker, C., Kuligowski, D. R., Friesen, T. A., and Shields, B. 2009. Genetic stock compostition of subyearling Chinook salmon in seasonal floodplains wetlands of the Lower Willamette River, Oregon. *Transaction of American Fisheries Sociery,* 138: 211-17.

Teske, M. E. 2001. 'A user's guide for AgDrift 2.04: A tiered approach for the assessment of spray drift of pesticides.' in *A user's guide for AgDrift 2.04: A tiered approach for the assessment of spray drift of pesticides*, 128. Spray Drift Task Force.

Thorpe, J. E. 1994. Salmonid fishes and the estuarine environment. *Estuaries,* 17(1A): 76 - 93.

Tierney, K., Ross, P., Jarrard, H., Delaney, K., and Kennedy, C. 2006. Changes in juvenile coho salmon electro-olfactogram during and after short-term exposure to current-use pesticides. *Environmental Toxicology and Chemistry,* 25(10): 2809-17.

Tiffan, K. F., Clark, L. O., Garland, R. D., and Rondorf, D. W. 2006. Variables influencing the presence of subyearlings fall Chinook salmon in shoreline habitats of the Hanford Reach, Columbia River. *North American Journal of Fisheries Management,* 26: 351-60.

Tillman, T. L., Edwards, G. W., and Urquhart, K. A. F. 1996. 'Adult salmon migration during the various operational phases of Suisun Marsh Salinity Control Gates in Montezuma Slough: August-October 1993.' in *Adult salmon migration during the various operational phases of Suisun Marsh Salinity Control Gates in Montezuma Slough: August-October 1993*, 25 pages.

Torres-Ruiz, M., Wehr, J. D., and Perrone, A. A. 2007. Trophic relations in a stream food web: importance of fatty acids for macroinvertebrate consumers. *Journal of the North American Benthological Society,* 26(3): 509-22.

USCB 2005. 'Interim State Population Projections, 2005.' in *Interim State Population Projections, 2005.* ed. P. Division. U.S. Census Bureau.

USDA 2012. 'National Agriculture Statistics Service (NASS).' in *National Agriculture Statistics Service (NASS).* http://usda.mannlib.cornell.edu/MannUsda/viewDocumentInfo.do?documentID=1141.

USEPA 2001. 'Thiocarbamates: A Determination of the Existence of a Common Mechanism of Toxicity and A Screening Level Cumulative Food Risk Assessment .' in *Thiocarbamates: A Determination of the Existence of a Common Mechanism of Toxicity and A Screening Level Cumulative Food Risk Assessment* .

USFS 2000. 'Fire recharges native fisheries.' in *Fire recharges native fisheries,* 3. U.S. Department of Agriculture, Forest Service - Northern Region.

USFWS 2000. 'Impacts of riprapping to ecosystem functioning, lower Sacramento River, California.' in *Impacts of riprapping to ecosystem functioning, lower Sacramento River, California*. Sacramento, California: U.S. Fish and Wildlife Service, Sacramento Field Office.

--- 2001. 'Abundance and seasonal, spatial, and diel distribution patterns of juvenile salmonids passing the Red Bluff Diversion Dam, Sacramento River. Draft Progress Report for Red Bluff Research Pumping Plant, Vol. 14.' in *Abundance and seasonal, spatial, and diel distribution patterns of juvenile salmonids passing the Red Bluff Diversion Dam, Sacramento River. Draft Progress Report for Red Bluff Research Pumping Plant, Vol. 14*. Red Bluff, CA.

--- 2008. 'Informal consultation on the effects of atrazine re-registration on the endangered Alabama sturgeon and the endangered dwarf wedge-mussel.' in *Informal consultation on the effects of atrazine re-registration on the endangered Alabama sturgeon and the endangered dwarf wedge-mussel*.

--- 2010a. 'Juvenile fish monitoring and abundance and distribution of Chinook in the Sacrament-San Joaquin Estuary 2007 - 2008 Annual Report.' in *Juvenile fish monitoring and abundance and distribution of Chinook in the Sacrament-San Joaquin Estuary 2007 - 2008 Annual Report* ed. U. S. F. a. W. Service. Stockton, California.

--- 2010b. 'Juvenile fish monitoring and abundance and distribution of Chinook salmon in the Sacramento-San Joaquin Estuary. 2007-2010 Annual Report. Stockton, California.' in *Juvenile fish monitoring and abundance and distribution of Chinook salmon in the Sacramento-San Joaquin Estuary. 2007-2010 Annual Report. Stockton, California*, 111. Stockton Fish and Wildlife Office.

--- 2012a. 'Abundance and distribution of Chinook salmon and other catch in the Sacramento - San Joaquin Estuary. 2009 annual report. Stockton, California.' in *Abundance and distribution of Chinook salmon and other catch in the Sacramento -San Joaquin Estuary. 2009 annual report. Stockton, California*, 74. Stockton Fish and Wildlife Office.

--- 2012b. 'Abundance and distribution of Chinook Salmon and other catch in the Sacramento-San Joaquin Estuary.' in *Abundance and distribution of Chinook Salmon and other catch in the Sacramento-San Joaquin Estuary* ed. U. S. F. a. W. Service, 74. Stockton, California.

USFWS and Reclamation 2007. 'A compilation and analysis of anadromous fish monitoring data from the Central Valley of California 1992-2006.' in *A compilation and analysis of anadromous fish monitoring data from the Central Valley of California 1992-2006*, 99. Sacramento, California: U.S. Department of the Interior.

Van den Brink, P. J., Blake, N., Brock, T. C. M., and Maltby, L. 2006. Predictive value of species sensitivity distributions for effects of herbicides in freshwater ecosystems. *Human and Ecological Risk Assessment*, 12(4): 645-74.

Van den Brink, P. J., Crum, S. J. H., Gylstra, R., Bransen, F., Cuppen, J. G. M., and Brock, T. C. M. 2009. Effects of a herbicide-insecticide mixture in freshwater microcosms: Risk assessment and ecological effect chain. *Environmental Pollution*, 157(1): 237-49.

van der Geest, H. G., Greve, G. D., Kroon, A., Kuijl, S., Kraak, M. H. S., and Admiraal, W. 2000. Sensitivity of characteristic riverine insects, the caddisfly Cyrnus trimaculatus and the mayfly Ephoron virgo, to copper and diazinon. *Environmental Pollution*, 109(2): 177-82.

van der Geest, H. G., Soppe, W. J., Greve, G. D., Kroon, A., and Kraak, M. H. S. 2002. Combined effects of lowered oxygen and toxicants (copper and diazinon) on the mayfly Ephoron virgo. *Environmental Toxicology and Chemistry,* 21(2): 431-36.

Vogel, D. A. 2008. 'Pilot study to evaluate acoustic-tagged juvenile Chinook salmon smolt migration in the Northern Sacramento-San Joaquin Delta 2006-2007.' in *Pilot study to evaluate acoustic-tagged juvenile Chinook salmon smolt migration in the Northern Sacramento-San Joaquin Delta 2006-2007,* 43 pages. Natural Resource Scientists, Inc.

Vogel, D. A. and Marine, K. R. 1991. 'Guide to upper Sacramento River Chinook salmon life history.' in *Guide to upper Sacramento River Chinook salmon life history.* ed. C. V. P. U.S. Bureau of Reclamation, 55.

Vogel, D. A., Marine, K. R., and Smith, J. G. 1988. 'Fish passage action program for Red Bluff Diversion Dam. Final report on fishery investigations.' in *Fish passage action program for Red Bluff Diversion Dam. Final report on fishery investigations.* Red Bluff, CA: U.S. Fish and Wildlife Service, Northern Central Valley Fishery Resource Office.

Walker, R. L. and Foott, J. S. 1993. Disease survey of Klamath River salmonid smolt populations, 1992. *U.S. Fish and Wildlife Service, California-Nevada Fish Health Center, Anderson, California. [150 Kb] at KRIS Website*: 46 p.

Wallace, J. B., Eggert, S. L., Meyer, J. L., and Webster, J. R. 1999. Effects of resource limitation on a detrital-based ecosystem. *Ecological Monographs,* 69(4): 409-42.

Waples, R. S. 1991. Pacific Salmon, *Oncorhynchus spp*., and the definition of “species” under the Endangered Species Act. *Marine Fisheries Review,* 53: 11-21.

Ward, P. D., McReynolds, T. R., and Garman, C. E. 2002. 'Butte and Big Chico Creeks spring-run Chinook salmon, *Oncorhynchus tshawytscha* life history investigation, 2000-2001.' in *Butte and Big Chico Creeks spring-run Chinook salmon, Oncorhynchus tshawytscha life history investigation, 2000-2001*. California Department of Fish and Game, Inland Fisheries Administrative Report.

--- 2003. 'Butte and Big Chico Creeks spring-run Chinook salmon, *Oncorhynchus tshawytscha* life history investigation, 2001-2002.' in *Butte and Big Chico Creeks spring-run Chinook salmon, Oncorhynchus tshawytscha life history investigation, 2001-2002*. California Department of Fish and Game, Inland Fisheries Administrative Report.

Watanabe, H., Nguyen, M. H. T., Souphasay, K., Vu, S. H., Phong, T. K., Tournebize, J., and Ishihara, S. 2007. Effect of water management practice on pesticide behavior in paddy water. *Agricultural Water Management,* 88(1-3): 132-40.

Waters, T. F. 1995. Sediment in streams: sources, biological effects, and control. *American Fisheries Society Monograph,* 7.

Weed Science Society of America, W. 2007. *Herbicide Handbook.* Lawrence, Kansas: Weed Science Society of America.

Wendt-Rasch, L., Pirzadeh, P., and Woin, P. 2003. Effects of metsulfuron methyl and cypermethrin exposure on freshwater model ecosystems. *Aquatic Toxicology,* 63(3): 243-56.

Williams, T. H., Lindley, S. T., Spence, B. C., and Boughton, D. A. 2011. 'Status review update for Pacific salmon and steelhead listed under the Endangered Species Act: Southwest.' in *Status review update for Pacific salmon and steelhead listed under the Endangered Species Act: Southwest* ed. N. M. F. Service. Santa Cruz, California.

Wright, D. and Phillips, D. 1988. Chesapeake and San-Francisco Bays - A study in contrasts and parallels. *Marine Pollution Bulletin,* 19(9): 405-13.

Xie, L. T., Thrippleton, K., Irwin, M. A., Siemering, G. S., Mekebri, A., Crane, D., Berry, K., and Schlenk, D. 2005. Evaluation of estrogenic activities of aquatic herbicides and surfactants using an rainbow trout vitellogenin assay. *Toxicological Sciences*, 87(2): 391-98.

Yates, D., Galbraith, H., Purkey, D., Huber-Lee, A., Sieber, J., West, J., Herrod-Julius, S., and Joyce, B. 2008. Climate warming, water storage, and Chinook salmon in California's Sacramento Valley. *Climate Change*, 91(3-4): 335-50.

Yoshiyama, R. M., Gerstung, E. R., Fisher, F. W., and Moyle, P. B. 1996. 'Historical and present distribution of Chinook salmon in the Central Valley drainage of California, Sierra Nevada Ecosystem Project: final report to Congress.' in *Historical and present distribution of Chinook salmon in the Central Valley drainage of California, Sierra Nevada Ecosystem Project: final report to Congress*, 309-62. Davis, CA: University of California, Center for Water and Wildland Resources.

--- 1998. Historical abundance and decline of Chinook salmon in the Central Valley region of California. *North American Journal of Fisheries Management*, 18: 487-521.

Yoshiyama, R. M., Gertung, E. R., Fisher, F. W., and Moyle, P. B. **2001**. 'Historical and Present Distribution of Chinook Salmon in the Central Valley Drainage of California.' in *Historical and Present Distribution of Chinook Salmon in the Central Valley Drainage of California* ed. R. L. Brown, 71-176. U.C. San Diego, Scripps Institute of Oceanograpy: California Department of Fish and Game.

16. Appendix 1 Abbreviations / Acronyms

7-DADMax	7-day average of the daily maximum
ACA	Alternative Conservation Agreement
AChE	acetylcholinesterase
a.i.	active ingredient
APEs	alkylphenol ethoxylates
APHIS	U.S. Department of Agriculture Animal Plant and Health Inspection Service
BE	Biological Evaluation
BEAD	Biological and Economic Analysis Division
BLM	Bureau of Land Management
BMP	Best Management Practices
BOR	Bureau of Reclamation
BOR	Bureau of Reclamation
BPA	Bonneville Power Administration
BRT	Biological Review Team (NOAA Fisheries)
BY	Brood Years
CAISMP	California Aquatic Invasive Species Management Plan
CALFED	CALFED Bay-Delta Program (California Resource Agency)
CBFWA	Columbia Basin Fish and Wildlife Authority
CBI	Confidential Business Information
CC	California Coastal
CCC	Central California Coast
CCV	Central California Valley
CDPR	California Department of Pesticide Regulation
CHART	Critical Habitat Assessment Review Team
CIDMP	Comprehensive Irrigation District Management Plan
CFR	Code of Federal Regulations
cfs	cubic feet per second
CDFG	California Department of Fish and Game
Corps	U.S. Department of the Army Corps of Engineers

CSOs	combined sewer/stormwater overflows
CSWP	California State Water Project
CURES	Coalition for Urban/Rural Environmental Stewardship
CVP	Central Valley Projects
CVRWQCB	Central Valley Regional Water Quality Control Board
CWA	Clean Water Act
d	day
DCI	Date Call-Ins
DDD	*Dichloro* Diphenyl Dichloroethane
DDE	Diphenyl Dichlorethylene
DDT	*Dichloro Diphenyl Trichloroethane*
DER	*Data Evaluation Review*
DEQ	Oregon Department of Environmental Quality
DIP	Demographically Independent Population
DOE	Washington State Department of Ecology
DPS	Distinct Population Segment
EC	Emulsifiable Concentrate Pesticide Formulation
EC_{50}	Median Effect Concentration
EEC	Estimated Environmental Concentration
EFED	Environmental Fate and Effects Division
EIM	Environmental Information Management
EPA	U.S. Environmental Protection Agency
ESPP	Endangered Species Protection Program
ESA	Endangered Species Act
ESU	Evolutionarily Significant Unit
EU	European Union
EXAMS	Tier II Surface Water Computer Model
FERC	Federal Energy Regulatory Commission
FCRPS	Federal Columbia River Power System
FFDCA	Federal Food and Drug Cosmetic Act
FIFRA	Federal Insecticide, Fungicide, and Rodenticide Act

FQPA	Food Quality Protection Act
ft	feet
GENEEC	Generic Estimated Exposure Concentration
h	hour
HCP	Habitat Conservation Plan
HSRG	Hatchery Scientific Review Group
HUC	Hydrological Unit Code
IBI	Indices of Biological Integrity
ICTRT	Interior Columbia Technical Recovery Team
ILWP	Irrigated Lands Waiver Program
IPCC	Intergovernmental Panel on Climate Change
IRED	Interim Re-registration Decision
LCFRB	Lower Columbia Fish Recovery Board
ISG	Independent Science Group
ITS	Incidental Take Statement
km	kilometer
Lbs	Pounds
LC_{50}	Median Lethal Concentration.
LCR	Lower Columbia River
LOAEC	Lowest Observed Adverse Effect Concentration.
LOEL	Lowest Observed Adverse Effect level
LOC	Level of Concern
LOEC	Lowest Observed Effect Concentration
LOQ	Limit of Quantification
LWD	Large Woody Debris
m	meter
MCR	Middle Columbia River
mg/L	milligrams per liter
MOA	Memorandum of Agreement
MPG	Major Population Group
MRID	Master Record Identification Number

MTBE	Methyl tert-butyl ether
NASA	National Aeronautics and Space Administration
NAWQA	U.S. Geological Survey National Water-Quality Assessment
NC	Northern California
NEPA	National Environmental Protection Agency
NLCD	Natural Land Cover Data
NP	Nonylphenol
NPDES	National Pollutant Discharge Elimination System
NPS	National Parks Services
NRCS	Natural Resources Conservation Service
NWS	National Weather Service
NEPA	National Environmental Policy Act
NMA	National Mining Association
NMC	*N*-methyl carbamates
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
NOAEC	No Observed Adverse Effect Concentration
NPDES	National Pollution Discharge Eliminating System
NPIRS	National Pesticide Information Retrieval System
NRC	National Research Council
OC	Oregon Coast
ODFW	Oregon Division of Fish and Wildlife
OP	Organophosphates
Opinion	Biological Opinion
OPP	EPA Office of Pesticide Program
PAH	polyaromatic hydrocarbons
PBDEs	polybrominated diphenyl ethers
PCBs	polychlorinated biphenyls
PCEs	primary constituent elements
POP	Persistent Organic Pollutants
ppb	Parts Per Billion

PPE	Personal Protection Equipment
PSP	Pesticide Stewardship Partnerships
PSAMP	Puget Sound Assessment and Monitoring Program
PSAT	Puget Sound Action Team
PRIA	Pesticide Registration Improvement Act
PRZM	Pesticide Root Zone Model
PUR	Pesticide Use Reporting
QA/QC	Quality Assurance/Quality Control
RED	Re-registration Eligibility Decision
REI	Restricted Entry Interval
RPA	Reasonable and Prudent Alternatives
RPM	reasonable and prudent measures
RQ	Risk Quotient
SAP	Scientific Advisory Panel
SAR	smolt-to-adult return rate
SASSI	Salmon and Steelhead Stock Inventory
SC	Southern California
S-CCC	South-Central California Coast
SONCC	Southern Oregon Northern California Coast
SLN	Special Local Need (Registrations under Section 24(c) of FIFRA)
SR	Snake River
TCE	Trichloroethylene
TCP	3,5,6-trichloro-2-pyridinal
TGAI	Technical Grade Active Ingredient
TIE	Toxicity Identification Evaluation
TMDL	Total Maximum Daily Load
TRT	Technical Recovery Team
UCR	Upper Columbia River
USFS	United States Forest Service
USC	United States Code
USFWS	United States Fish and Wildlife Service

USGS	United States Geological Survey
UWR	Upper Willamette River
VOC	Volatile Organic Compounds
VSP	Viable Salmonid Population
WDFW	Washington Department of Fish and Wildlife
WLCRTRT	Willamette/Lower Columbia River Technical Review Team
WQS	Water Quality Standards
WWTIT	Western Washington Treaty Indian Tribes
WWTP	Wastewater Treatment Plant
YOY	Young of year

17. Appendix 2: Glossary

303(d) waters Section 303 of the federal Clean Water Act requires states to prepare a list of all surface waters in the state for which beneficial uses – such as drinking, recreation, aquatic habitat, and industrial use - are impaired by pollutants. These are water quality limited estuaries, lakes, and streams that do not meet the state's surface water quality standards and are not expected to improve within the next two years. After water bodies are put on the 303(d) list they enter into a Total Maximum Daily Load Clean Up Plan.

Active ingredient	The component(s) that kills or otherwise affects the pest. A.i.s are always listed on the label (FIFRA 2(a)).
Adulticide	A compound that kills the adult life stage of the pest insect.
Anadromous Fish	Species that are hatched in freshwater migrate to and mature in salt water and return to freshwater to spawn.
Adjuvant	A compound that aides the operation or improves the effectiveness of a pesticide.
Alevin	Life history stage of a salmonid immediately after hatching and before the yolk-sac is absorbed. Alevins usually remain buried in the gravel in or near the egg nest (redd) until their yolk sac is absorbed when they swim up and enter the water column.
Anadromy	The life history pattern that features egg incubation and early juvenile development in freshwater migration to sea water for adult development, and a return to freshwater for spawning.
Assessment Endpoint	Explicit expression of the actual ecological value that is to be protected (*e.g.*, growth of juvenile salmonids).

Bioaccumulation	Accumulation through the food chain (*i.e.*, consumption of food, water/sediment) or direct water and/or sediment exposure.
Bioconcentration	Uptake of a chemical across membranes, generally used in reference to waterborne exposures.
Biomagnification	Transfer of chemicals via the food chain through two or more trophic levels as a result of bioconcentration and bioaccumulation.
Degradates	New compounds formed by the transformation of a pesticide by chemical or biological reactions.
Distinct Population Segment	A listable entity under the ESA that meets tests of discreteness and significance according to USFWS and NMFS policy. A population is considered distinct (and hence a "species" for purposes of conservation under the ESA) if it is discrete from and significant to the remainder of its species based on factors such as physical, behavioral, or genetic characteristics, it occupies an unusual or unique ecological setting, or its loss would represent a significant gap in the species' range.
Escapement	The number of fish that survive to reach the spawning grounds or hatcheries. The escapement plus the number of fish removed by harvest form the total run size.
Evolutionarily Significant Unit	A group of Pacific salmon or steelhead trout that is (1) substantially reproductively isolated from other conspecific units and (2) represent an important component of the evolutionary legacy of the species.
Fall Chinook	This salmon stock returns from the ocean in late summer and early

Salmon	fall to head upriver to its spawning grounds, distinguishing it from other stocks which migrate in different seasons.
Fate	Dispersal of a material in various environmental compartments (sediment, water air, biota) as a result of transport, transformation, and degradation.
Flowable	A pesticide formulation that can be mixed with water to form a suspension in a spray tank.
Fry	Stage in salmonid life history when the juvenile has absorbed its yolk sac and leaves the gravel of the redd to swim up into the water column. The fry stage follows the alevin stage and in most salmonid species is followed by the parr, fingerling, and smolt stages. However, chum salmon juveniles share characteristics of both the fry and smolt stages and can enter sea water almost immediately after becoming fry.
Half-pounder	A life history trait of steelhead exhibited in the Rogue, Klamath, Mad, and Eel Rivers of southern Oregon and northern California. Following smoltification, half-pounders spend only 2-4 months in the ocean, then return to fresh water. They overwinter in fresh water and emigrate to salt water again the following spring. This is often termed a false spawning migration, as few half-pounders are sexually mature.
Hatchery	Salmon hatcheries use artificial procedures to spawn adults and raise the resulting progeny in fresh water for release into the natural environment, either directly from the hatchery or by transfer into another area. In some cases, fertilized eggs are outplanted (usually in “hatch-boxes”), but it is more common to release fry or smolts.
Inert ingredients	“an ingredient which is not active” (FIFRA 2(m)). It may be toxic or enhance the toxicity of the active ingredient.

Iteroparous	Capable of spawning more than once before death
Jacks	Male salmon that return from the ocean to spawn one or more years before full-sized adults return. For coho salmon in California, Oregon, Washington, and southern British Columbia, jacks are 2 years old, having spent only 6 months in the ocean, in contrast to adults, which are 3 years old after spending 1 ½ years in the ocean.
Jills	Female salmon that return from the ocean to spawn one or more years before full-sized adult returns. For sockeye salmon in Oregon, Washington, and southern British Columbia, jills are 3 years old (age 1.1), having spent only one winter in the ocean in contrast to more typical sockeye salmon that are age 1.2, 1.32.2, or 2.3 on return.
Kokanee	The self-perpetuating, non-anadromous form of *O. nerka* that occurs in balanced sex ration populations and whose parents, for several generations back, have spent their whole lives in freshwater.
Lambda	Also known as Population growth rate, or the rate at which the abundance of fish in a population increases or decreases.
LRL	Laboratory Reporting Level (USGS NAWQA data)- Generally equal to twice the yearly determined LT-MDL. The LRL controls false negative error. The probability of falsely reporting a non-detection for a sample that contained an analyte at a concentration equal to or greater that the LRL is predicted to be less than or equal to 1 percent.
Major Population Group (MPG)	A group of salmonid populations that are geographically and genetically cohesive. The MPG is a level of organization between demographically independent populations and the ESU.

Main channel	The stream channel that includes the thalweg (longitudinal continuous deepest portion of the channel.
Metabolite	A transformation product resulting from metabolism.
Mode of Action	A series of key processes that begins with the interaction of a pesticide with a receptor site and proceeds through operational and anatomical changes in an organisms that result in sublethal or lethal effects.
Natural fish	A fish that is produced by parents spawning in a stream or lake bed, as opposed to a controlled environment such as a hatchery.
Nonylphenols	A type of APE and is an example of an adjuvant that may be present as an ingredient of a formulated product or added to a tank mix prior to application.
Off-channel habitat	Water bodies and/or inundated areas that are connected (accessible to salmonid juveniles) seasonally or annually to the main channel of a stream including but not limited to features such as side channels, alcoves, ox bows, ditches, and floodplains.
Parr	The stage in anadromous salmonid development between absorption of the yolk sac and transformation to smolt before migration seaward.
Persistence	The tendency of a compound to remain in its original chemical form in the environment.
Pesticide	Any substance or mixture of substances intended for preventing, destroying, repelling or mitigating any pest.

Reasonable and Prudent Alternative (RPA)	Recommended alternative actins identified during formal consultation that can be implemented in a manner consistent with the scope of the Federal agency's legal authority an jurisdiction, that are economically and technologically feasible, and that the Services believes would avoid the likelihood of jeopardizing the continued existence of the listed species or the destruction or adverse modification of designated critical habitat.
Redd	A nest constructed by female salmonids in streambed gravels where eggs are deposited and fertilization occurs.
Riparian area	Area with distinctive soils and vegetation between a stream or other body of water and the adjacent upland. It includes wetlands and those portions of flood plains an valley bottoms that support riparian vegetation.
Risk	The probability of harm from actual or predicted concentrations of a chemical in the aquatic environment – a scientific judgment.
Salmonid	Fish of the family *Salmonidae*, including salmon, trout, chars, grayling, and whitefish. In general usage, the term usually refers to salmon, trout, and chars.
SASSI	A cooperative program by WDFW and WWTIT to inventory and evaluate the status of Pacific salmonids in Washington State. The SASSI report is a series of publications from this program.
Semelparous	The condition in an individual organism of reproducing only once in a lifetime.

Smolt	A juvenile salmon or steelhead migrating to the ocean and undergoing physiological changes to adapt from freshwater to a saltwater environment.
Sublethal	Below the concentration that directly causes death. Exposure to sublethal concentrations of a material may produce less obvious effect on behavior, biochemical, and/or physiological function of the organism often leading to indirect death.
Surfactant	A substance that reduces the interfacial or surface tension of a system or a surface-active substance.
Synergism	A phenomenon in which the toxicity of a mixture of chemicals is greater than that which would be expected from a simple summation of the toxicities of the individual chemicals present in the mixture.
Technical Grade Active Ingredient (TGAI)	Pure or almost pure active ingredient. Available to formulators. Most toxicology data are developed with the TGAI. The percent AI is listed on all labels.
Technical Recovery Teams (TRT)	Teams convened by NOAA Fisheries to develop technical products related to recovery planning. TRTs are complemented by planning forums unique to specific states, tribes, or reigns, which use TRT and other technical products to identify recovery actions.
Teratogenic	Effects produced during gestation that evidence themselves as altered structural or functional processes in offspring.
Total Maximum	defines how much of a pollutant a water body can tolerate (absorb)

Daily Load (TMDL)	daily and remain compliant with applicable water quality standards. All pollutant sources in the watershed combined, including non-point sources, are limited to discharging no more than the TMDL.
Viable Salmonid Population	An independent population of Pacific salmon or steelhead trout that has a negligible risk of extinction over a 100-year time frame. Viability at the independent population scale is evaluated based on the parameters of abundance, productivity, spatial structure, and diversity.
VSP Parameters	Abundance, productivity, spatial structure, and diversity. These describe characteristics of salmonid populations that are useful in evaluating population viability. See NOAA Technical Memorandum NMFS-NWFSC-, "Viable salmonid populations and the recovery of evolutionarily significant units," McElhany et al., June 2000.
WDFW	Washington Department of Fish and Wildlife is a co-manager of salmonids and salmonid fisheries in Washington State with WWTIT and other fisheries groups. The agency was formed in the early 1990s by the combination of the Washington Department of Fisheries and the Washington Department of Wildlife.
WWTIT	Western Washington Treaty Indian Tribes is an organization of Native American tribes with treaty fishing rights recognized by the U.S. government. WWTIT is a co-manager of salmonids and salmonid fisheries in western Washington in cooperation with the WDFW and other fisheries groups.
WQS	"A water quality standard defines the water quality goals of a waterbody, or portion thereof, by designating the use or uses to be made of the water and by setting criteria necessary to protect public health or welfare, enhance the quality of water and serve the purposes of the Clean Water

18. Appendix 3: Median and 95th percentile rate of thiobencarb application to rice in California (1980-2010)

Median and 95th percentile rate of thiobencarb application to rice in California (1980 - 2010). Source: California Department of Pesticide Regulation, Pesticide Use Report (PUR) Database.

Year	Median Rate (lbs a.i./A)	95th percentile Rate (lbs a.i./A)
1980	4.00	4.03
1981	4.00	4.12
1982	4.00	4.08
1983	4.00	4.11
1984	4.00	4.08
1985	4.00	4.07
1986	4.00	4.08
1987	4.00	4.07
1988	4.00	4.08
1989	4.00	4.06
1990	4.00	4.07
1991	4.00	4.06
1992	4.00	4.05
1993	4.00	4.06
1994	4.00	4.03
1995	4.00	4.03
1996	4.00	4.02
1997	4.00	4.01
1998	4.00	4.01
1999	4.00	4.02
2000	4.00	4.02
2001	4.00	4.01
2002	4.00	4.03
2003	4.01	4.05
2004	4.01	4.05
2005	4.01	4.03
2006	4.01	4.02
2007	4.01	4.01
2008	3.99	4.01
2009	3.95	4.01
2010	3.50	3.95

19. Appendix 4: Co-Application of Thiobencarb with Other Pesticides in California (1999-2010)

Co-application of thiobencarb with other pesticides in California.

This following tables summarize instances of co-application of thiobencarb with other active ingredients on rice in California from 1999 to 2010. These can include either tank mixture applications, or separate applications of different pesticides to the same field, on the same day. Either situation suggest the likely presence of environmental mixtures of pesticides. The statistics were provided by Larry Wilhoit, California Department of Pesticide Regulation (CDPR), and obtained from the CDPR Pesticide Use Report (PUR) Database. To obtain the information from the database, a co-application was defined as the use of more than one active ingredient (AI) on an agricultural field in the same day. In the PUR a field is identified by the combination of grower_id and site_loc_id after some cleaning of the data. The site_loc_id is cleaned by removing all spaces and assuming that site_loc_ids that are the same except for characters that could be confused (such as O and 0 and 2 and Z) and have the same MTRS refer to the same field. Each field is given a unique field_id. There are a few records that were not included in the summary information because of errors or missing data for the site_loc_id, application date, or some other field which made it impossible to determine the specific date and field.

Summary indicating yearly frequency of tank mixture applications of thiobencarb in California

YEAR	NUMBER OF THIOBENCARB APPLICATIONS WITH FIELD_ID	NUMBER OF CO-APPLICATIONS	PERCENT CO-APPLICATIONS	LBS THIOBENCARB	LBS OTHER AIS
1999	2,189	490	22	731,717	158,922
2000	2,989	620	21	1,006,965	267,879
2001	2,070	435	21	645,914	208,611
2002	2,656	582	22	843,773	274,187
2003	1,816	439	24	587,156	227,529
2004	1,476	359	24	521,556	157,158
2005	1,359	359	26	448,182	122,010
2006	908	180	20	310,346	73,926
2007	794	230	29	289,032	58,066
2008	820	155	19	263,499	37,479
2009	822	162	20	272,080	46,770
2010	834	237	28	258,402	16,467
Total	**18,733**	**4,248**	**23**	**6,178,623**	**1,649,004**

Summary of tank mixture application that included thiobencarb (California, 1999-2010)

LIST OF ACTIVE INGREDIENTS APPLIED WITH THIOBENCARB	NUMBER APPLICATIONS	LBS THIOBENCARB	LBS OTHER AIs
THIOBENCARB ALONE	14,474	4,828,157	
COPPER SULFATE (PENTAHYDRATE)	1,128	393,884	1,278,042
BENSULFURON METHYL	660	219,889	3,417
LAMBDA-CYHALOTHRIN	470	147,278	483
PENOXSULAM	445	132,366	1,364
BISPYRIBAC-SODIUM	217	57,420	6,842
CARFENTRAZONE-ETHYL	201	78,718	3,943
MOLINATE	166	50,332	38,876
PROPANIL	136	34,605	43,839
PENDIMETHALIN	107	17,659	5,231
(S)-CYPERMETHRIN	66	20,497	82
CARBOFURAN	62	16,510	873
COPPER SULFATE (PENTAHYDRATE); BENSULFURON METHYL	43	16,904	53,050
PROPANIL; PENDIMETHALIN	30	5,408	8,048
BENSULFURON METHYL; LAMBDA-CYHALOTHRIN	29	8,013	159
COPPER SULFATE (PENTAHYDRATE); PENOXSULAM	29	11,378	35,647
METHYLATED SOYBEAN OIL; LAMBDA-CYHALOTHRIN; KEROSENE; DIMETHYL ALKYL TERTIARY AMINES; BENZOIC ACID	25	10,279	218
COPPER SULFATE (PENTAHYDRATE); MOLINATE	24	8,073	39,934
POLY-I-PARA-MENTHENE	21	6,278	1,454
PROPANIL; MOLINATE	21	2,016	6,577
MCPA, DIMETHYLAMINE SALT	18	6,186	7,486
COPPER SULFATE (PENTAHYDRATE); LAMBDA-CYHALOTHRIN	15	5,082	15,966
BENSULFURON METHYL; POLYACRYLAMIDE POLYMER	15	2,318	37
KEROSENE; LAMBDA-CYHALOTHRIN; SOYBEAN FATTY ACIDS, DIMETHYLAMINE SALT; BENZOIC ACID	14	5,030	123
ORTHOSULFAMURON	14	3,534	553
METHYLATED SOYBEAN OIL; (S)-CYPERMETHRIN; KEROSENE; DIMETHYL ALKYL TERTIARY AMINES; BENZOIC ACID	13	5,082	124
METHYLATED SOYBEAN OIL; BENSULFURON METHYL; KEROSENE; DIMETHYL ALKYL TERTIARY AMINES; BENZOIC ACID	12	4,935	315

LIST OF ACTIVE INGREDIENTS APPLIED WITH THIOBENCARB	NUMBER APPLICATIONS	LBS THIOBENCARB	LBS OTHER AIs
COPPER SULFATE (PENTAHYDRATE); CARFENTRAZONE-ETHYL	12	3,673	11,326
METHYLATED SOYBEAN OIL; KEROSENE; DIMETHYL ALKYL TERTIARY AMINES; BENZOIC ACID	11	3,292	325
MOLINATE; BENSULFURON METHYL	11	2,198	3,275
PENOXSULAM; (S)-CYPERMETHRIN	9	2,343	37
AZOXYSTROBIN; OLEIC ACID, METHYL ESTER; 2-(3-HYDROXYPROPYL)-HEPTA-METHYL TRISILOXANE, ETHOXYLATED, ACETATE; BENSULFURON METHYL; BISPYRIBAC-SODIUM	8	555	84
CARFENTRAZONE-ETHYL; BENSULFURON METHYL	7	1,591	86
PROPANIL; LAMBDA-CYHALOTHRIN	7	2,388	3,543
PROPANIL; MOLINATE; CARBOFURAN	6	281	1,974
PENOXSULAM; LAMBDA-CYHALOTHRIN	6	3,128	49
BENSULFURON METHYL; POLYACRYLAMIDE POLYMER; POLYSACCHARIDE POLYMER	6	2,904	48
PROPANIL; PETROLEUM OIL, PARAFFIN BASED; POLYOXYETHYLENE SORBITAN MIXED FATTY ACID ESTERS; SORBITAN FATTY ACID ESTERS; ALPHA-(PARA-NONYLPHENYL)-OMEGA-HYDROXYPOLY (OXYETHYLENE) SULFATE, AMMONIUM SALT	5	990	1,547
COPPER SULFATE (PENTAHYDRATE); POLYSILOXANE; LAMBDA-CYHALOTHRIN; DERIVATED NATURAL POLYMERS	5	298	996
MINERAL OIL; PETROLEUM DISTILLATES; LAMBDA-CYHALOTHRIN; TALL OIL FATTY ACIDS; DIMETHYL SOYA AMINE; BENZOIC ACID	5	2,173	43
SODIUM CARBONATE PEROXYHYDRATE	5	1,782	3,876
GLYPHOSATE, ISOPROPYLAMINE SALT	5	1,192	14
LAMBDA-CYHALOTHRIN; ALPHA-(PARA-NONYLPHENYL)-OMEGA-HYDROXYPOLY(OXYETHYLENE); PHOSPHORIC ACID; ALPHA-ALKYLARYL-OMEGA-HYDROXYPOLY(OXYETHYLENE); ISOPROPYL ALCOHOL; DIMETHYLPOLYSILOXANE	4	1,449	27
ALKYL (C8,C10) POLYGLYCOSIDE; LAMBDA-CYHALOTHRIN	4	3,036	26
CARFENTRAZONE-ETHYL; (S)-CYPERMETHRIN	4	1,164	59
PROPANIL; VEGETABLE OIL; PENDIMETHALIN;	3	295	1,040

LIST OF ACTIVE INGREDIENTS APPLIED WITH THIOBENCARB	NUMBER APPLICATIONS	LBS THIOBENCARB	LBS OTHER AIs
FATTY ACIDS, MIXED			
DIFLUBENZURON; (S)-CYPERMETHRIN; POLYSILOXANE; DERIVATED NATURAL POLYMERS	3	1,105	22
PROPANIL; CYHALOFOP-BUTYL; TRICLOPYR, TRIETHYLAMINE SALT; BISPYRIBAC-SODIUM	3	1	3
PROPANIL; PETROLEUM DISTILLATES, ALIPHATIC; PENDIMETHALIN; CYHALOFOP-BUTYL	3	1,486	2,552
(S)-CYPERMETHRIN; PENOXSULAM	3	373	11
COPPER SULFATE (PENTAHYDRATE); PROPANIL	3	35	515
VINYL POLYMER	3	846	2
CARFENTRAZONE-ETHYL; BENSULFURON METHYL; LAMBDA-CYHALOTHRIN	3	720	38
COPPER SULFATE (PENTAHYDRATE); PENOXSULAM; LAMBDA-CYHALOTHRIN	3	1,134	5,107
CYHALOFOP-BUTYL	3	1,076	70
KEROSENE; GLYPHOSATE, ISOPROPYLAMINE SALT; SOYBEAN FATTY ACIDS, DIMETHYLAMINE SALT; BENZOIC ACID	3	621	349
LAMBDA-CYHALOTHRIN; POLYACRYLAMIDE POLYMER; POLYSACCHARIDE POLYMER	3	1,040	2
PENDIMETHALIN; ALPHA-PINENE BETA-PINENE COPOLYMER; MINERAL OIL; N,N-BIS-(2-OMEGA-HYDROXYPOLY(OXYETHYLENE)ETHYL)ALKYLAMINE, ALKYL DERIVED FROM TALLOW FATTY ACIDS; TALL OIL FATTY ACIDS	3	1,352	533
PROPANIL; PENDIMETHALIN; POLY-I-PARA-MENTHENE	3	633	1,278
CLOMAZONE	3	380	29
COPPER SULFATE (PENTAHYDRATE); BENSULFURON METHYL; POLYACRYLAMIDE POLYMER; POLYSACCHARIDE POLYMER	2	960	1,720
METHYLATED SOYBEAN OIL; BISPYRIBAC-SODIUM; KEROSENE; DIMETHYL ALKYL TERTIARY AMINES; BENZOIC ACID	2	788	34
OLEIC ACID, METHYL ESTER; BENSULFURON METHYL; 2-(3-HYDROXYPROPYL)-HEPTA-METHYL TRISILOXANE, ETHOXYLATED, ACETATE; BISPYRIBAC-SODIUM	2	89	8

LIST OF ACTIVE INGREDIENTS APPLIED WITH THIOBENCARB	NUMBER APPLICATIONS	LBS THIOBENCARB	LBS OTHER AIs
METHYLATED SOYBEAN OIL; BISPYRIBAC-SODIUM; ALPHA-(PARA-NONYLPHENYL)-OMEGA-HYDROXYPOLY(OXYETHYLENE); POLYALKENE OXIDE MODIFIED HEPTAMETHYL TRISILOXANE	2	237	31
PENDIMETHALIN; BISPYRIBAC-SODIUM	2	306	91
PROPANIL; BISPYRIBAC-SODIUM	2	496	497
DIFLUBENZURON; LAMBDA-CYHALOTHRIN	2	500	5
MINERAL OIL; (S)-CYPERMETHRIN; PETROLEUM DISTILLATES; TALL OIL FATTY ACIDS; DIMETHYL SOYA AMINE; BENZOIC ACID	2	752	17
DIFLUBENZURON	2	525	6
LAMBDA-CYHALOTHRIN; VINYL POLYMER	2	1,324	4
BENSULFURON METHYL; METHYLATED SOYBEAN OIL; LAMBDA-CYHALOTHRIN; KEROSENE; DIMETHYL ALKYL TERTIARY AMINES; BENZOIC ACID	2	620	16
METHYLATED SOYBEAN OIL; PENOXSULAM; (S)-CYPERMETHRIN; KEROSENE; DIMETHYL ALKYL TERTIARY AMINES; BENZOIC ACID	2	630	27
BENSULFURON METHYL; METHYLATED SOYBEAN OIL; ALPHA-(PARA-NONYLPHENYL)-OMEGA-HYDROXYPOLY(OXYETHYLENE); LAMBDA-CYHALOTHRIN; POLYALKENE OXIDE MODIFIED HEPTAMETHYL TRISILOXANE	2	420	9
MOLINATE; LAMBDA-CYHALOTHRIN	2	590	257
VINYL POLYMER; BISPYRIBAC-SODIUM	2	120	9
PROPANIL; POLY-I-PARA-MENTHENE	2	389	868
CARBOFURAN; VINYL POLYMER	2	633	42
PENDIMETHALIN; BENSULFURON METHYL	2	415	106
CARFENTRAZONE-ETHYL; LAMBDA-CYHALOTHRIN	2	643	28
PENDIMETHALIN; VINYL POLYMER	2	506	164
PROPANIL; ORCHEX 796 OIL; OLEIC ACID; ALPHA-ALKYL (C9-C11)-OMEGA-HYDROXYPOLY(OXYETHYLENE)	2	395	321
BISPYRIBAC-SODIUM; VINYL POLYMER	1	419	28,044
MOLINATE; PROPANIL; PETROLEUM OIL, PARAFFIN BASED; TRICLOPYR, TRIETHYLAMINE SALT; POLYOXYETHYLENE SORBITAN MIXED FATTY ACID ESTERS; SORBITAN FATTY ACID ESTERS; ALPHA-	1	176	98

LIST OF ACTIVE INGREDIENTS APPLIED WITH THIOBENCARB	NUMBER APPLICATIONS	LBS THIOBENCARB	LBS OTHER AIs
(PARA-NONYLPHENYL)-OMEGA-HYDROXYPOLY (OXYETHYLENE) SULFATE, AMMONIUM SALT			
PROPANIL; MOLINATE; ISOPARAFFINIC HYDROCARBONS; ORCHEX 796 OIL; ALPHA-(PARA-NONYLPHENYL)-OMEGA-HYDROXYPOLY(OXYETHYLENE); TALL OIL FATTY ACIDS; ALPHA-(PARA-NONYLPHENYL)-OMEGA-HYDROXYPOLY(OXYETHYLENE), PHOSPHATE ESTER; N,N-BIS-(2-OMEGA-HYDROXYPOLY(OXYETHYLENE)ETHYL)ALKYLAMINE, ALKYL DERIVED FROM TALLOW FATTY ACIDS	1	239	895
PETROLEUM OIL, PARAFFIN BASED; POLYOXYETHYLENE SORBITAN MIXED FATTY ACID ESTERS; BENSULFURON METHYL; SORBITAN FATTY ACID ESTERS; ALPHA-(PARA-NONYLPHENYL)-OMEGA-HYDROXYPOLY (OXYETHYLENE) SULFATE, AMMONIUM SALT	1	260	113
PROPANIL; HYDROTREATED PARAFFINIC SOLVENT; TALL OIL FATTY ACIDS; ALPHA-(PARA-NONYLPHENYL)-OMEGA-HYDROXYPOLY(OXYETHYLENE)	1	119	181
METHYLATED SOYBEAN OIL; KEROSENE; DIMETHYL ALKYL TERTIARY AMINES; BENZOIC ACID; LAMBDA-CYHALOTHRIN	1	278	5
PROPANIL; PHOSPHORIC ACID; ZINC SULFATE; MANGANESE SULFATE; FERROUS SULFATE; COCONUT IMIDAZOLINE SODIUM CARBOXYLATE	1	356	553
PROPANIL; MOLINATE; PETROLEUM OIL, PARAFFIN BASED; TRICLOPYR, TRIETHYLAMINE SALT; POLYOXYETHYLENE SORBITAN MIXED FATTY ACID ESTERS; SORBITAN FATTY ACID ESTERS; ALPHA-(PARA-NONYLPHENYL)-OMEGA-HYDROXYPOLY (OXYETHYLENE) SULFATE, AMMONIUM SALT	1	265	685
PENDIMETHALIN; OLEIC ACID, METHYL ESTER; TRICLOPYR, TRIETHYLAMINE SALT; 2-(3-HYDROXYPROPYL)-HEPTA-METHYL TRISILOXANE, ETHOXYLATED, ACETATE; BISPYRIBAC-SODIUM; CITRIC ACID; CALCIUM CHLORIDE	1	72	28

LIST OF ACTIVE INGREDIENTS APPLIED WITH THIOBENCARB	NUMBER APPLICATIONS	LBS THIOBENCARB	LBS OTHER AIs
BISPYRIBAC-SODIUM; ALPHA-OCTYLPHENYL-OMEGA-HYDROXYPOLY(OXYETHYLENE); POLYACRYLAMIDE, POLYETHYLENE GLYCOL MIXTURE; PHOSPHORIC ACID	1	119	3
PROPANIL; BENSULFURON METHYL	1	474	605
ORTHOSULFAMURON; (S)-CYPERMETHRIN	1	390	623
COPPER SULFATE (PENTAHYDRATE); MOLINATE; BENSULFURON METHYL	1	400	2,882
PROPANIL; PENDIMETHALIN; CYHALOFOP-BUTYL; BISPYRIBAC-SODIUM; BENSULFURON METHYL	1	0	1
KEROSENE; BENSULFURON METHYL; LAMBDA-CYHALOTHRIN; SOYBEAN FATTY ACIDS, DIMETHYLAMINE SALT; BENZOIC ACID	1	290	12
COPPER SULFATE (PENTAHYDRATE); METHYLATED SOYBEAN OIL; DIFLUBENZURON; KEROSENE; DIMETHYL ALKYL TERTIARY AMINES; BENZOIC ACID	1	960	3,612
BENSULFURON METHYL; DIFLUBENZURON	1	808	24
METHYLATED SOYBEAN OIL; PENOXSULAM; LAMBDA-CYHALOTHRIN; KEROSENE; DIMETHYL ALKYL TERTIARY AMINES; BENZOIC ACID	1	428	16
MCPA, DIMETHYLAMINE SALT; BISPYRIBAC-SODIUM	1	573	73
DIFLUBENZURON; LAMBDA-CYHALOTHRIN; POLYSILOXANE; DERIVATED NATURAL POLYMERS	1	320	4
DERIVATED NATURAL POLYMERS; POLYSILOXANE; BENSULFURON METHYL; LAMBDA-CYHALOTHRIN	1	290	6
PETROLEUM DISTILLATES, ALIPHATIC; BISPYRIBAC-SODIUM	1	1,190	521
MINERAL OIL; BENSULFURON METHYL; PETROLEUM DISTILLATES; TALL OIL FATTY ACIDS; DIMETHYL SOYA AMINE; BENZOIC ACID	1	401	18
COPPER SULFATE (PENTAHYDRATE); DIFLUBENZURON; (S)-CYPERMETHRIN	1	405	353
METHYL PARATHION; BENSULFURON METHYL; METHYL PARATHION, OTHER RELATED; VINYL POLYMER	1	480	84
CITRIC ACID; PENOXSULAM; LAMBDA-CYHALOTHRIN; CALCIUM CHLORIDE	1	506	22
COPPER SULFATE (PENTAHYDRATE);	1	1,223	1,488

LIST OF ACTIVE INGREDIENTS APPLIED WITH THIOBENCARB	NUMBER APPLICATIONS	LBS THIOBENCARB	LBS OTHER AIs
ORTHOSULFAMURON			
HYDROTREATED PARAFFINIC SOLVENT; CYHALOFOP-BUTYL; TALL OIL FATTY ACIDS; BISPYRIBAC-SODIUM; ALPHA-(PARA-NONYLPHENYL)-OMEGA-HYDROXYPOLY(OXYETHYLENE); PENOXSULAM	1	119	16
CYHALOFOP-BUTYL; PROPANIL; VEGETABLE OIL; PENDIMETHALIN; FATTY ACIDS, MIXED	1	91	255
MINERAL OIL; PENOXSULAM; PETROLEUM DISTILLATES; LAMBDA-CYHALOTHRIN; TALL OIL FATTY ACIDS; DIMETHYL SOYA AMINE; BENZOIC ACID	1	177	7
MOLINATE; DIFLUBENZURON; BENSULFURON METHYL; LAMBDA-CYHALOTHRIN	1	390	92
CARFENTRAZONE-ETHYL; PENDIMETHALIN; PROPANIL	1	0	1
MOLINATE; PENOXSULAM; CARFENTRAZONE-ETHYL; BENSULFURON METHYL	1	7	0
BENSULFURON METHYL; (S)-CYPERMETHRIN	1	447	14
DIFLUBENZURON; POLYSILOXANE; (S)-CYPERMETHRIN; DERIVATED NATURAL POLYMERS	1	476	4
AZOXYSTROBIN; FATTY ACIDS, C16-C18 AND C18-UNSATURATED, METHYL ESTERS; 4-NONYLPHENOL, FORMALDEHYDE RESIN, PROPOXYLATED; OLEIC ACID, METHYL ESTER; N,N-BIS-(2-OMEGA-HYDROXYPOLY(OXYETHYLENE)ETHYL)ALKYLAMINE, ALKYL DERIVED FROM TALLOW FATTY ACIDS; POLYBUTENES; 2-(3-HYDROXYPROPYL)-HEPTA-METHYL TRISILOXANE, ETHOXYLATED, ACETATE; BISPYRIBAC-SODIUM	1	24	14
METHYL PARATHION; KEROSENE; SOYBEAN FATTY ACIDS, DIMETHYLAMINE SALT; BENZOIC ACID	1	620	103
ALPHA-(PARA-NONYLPHENYL)-OMEGA-HYDROXYPOLY(OXYETHYLENE); PHOSPHORIC ACID; ALPHA-ALKYLARYL-OMEGA-HYDROXYPOLY(OXYETHYLENE); LAMBDA-CYHALOTHRIN; ISOPROPYL ALCOHOL; DIMETHYLPOLYSILOXANE	1	565	15
PROPANIL; PETROLEUM DISTILLATES, ALIPHATIC;	1	621	889

LIST OF ACTIVE INGREDIENTS APPLIED WITH THIOBENCARB	NUMBER APPLICATIONS	LBS THIOBENCARB	LBS OTHER AIs
PENDIMETHALIN			
COPPER SULFATE (PENTAHYDRATE); (S)-CYPERMETHRIN	1	531	1,970
PROPANIL; MINERAL OIL; TALL OIL FATTY ACIDS; ALPHA-(PARA-NONYLPHENYL)-OMEGA-HYDROXYPOLY(OXYETHYLENE)	1	403	430
BISPYRIBAC-SODIUM; ALPHA-2,6,8-TRIMETHYL-4-NONYLOXY-OMEGA-HYDROXYPOLY(OXYETHYLENE); POLYETHYLENE GLYCOL MONO(3-(TETRAMETHYL-1-(TRIMETHYLSILOXY)DISILOXANYL)PROPYL)ETHER; PROPYLENE GLYCOL; PHOSPHORIC ACID; TRISODIUM PHOSPHATE; DIMETHYLPOLYSILOXANE	1	119	5
PROPANIL; ORCHEX 796 OIL; OLEIC ACID; AZOXYSTROBIN; ALPHA-ALKYL (C9-C11)-OMEGA-HYDROXYPOLY(OXYETHYLENE); BENSULFURON METHYL; CARFENTRAZONE-ETHYL	1	375	559
MOLINATE; PROPANIL	1	40	106
MOLINATE; POLY-I-PARA-MENTHENE	1	439	228
CYHALOFOP-BUTYL; VEGETABLE OIL; PENDIMETHALIN; FATTY ACIDS, MIXED	1	49	73
BENSULFURON METHYL; DIFLUBENZURON; LAMBDA-CYHALOTHRIN	1	465	19
MOLINATE; CARFENTRAZONE-ETHYL	1	600	218
PROPANIL; CYHALOFOP-BUTYL; BISPYRIBAC-SODIUM; FENOXAPROP-P-ETHYL	1	0	0
PETROLEUM OIL, PARAFFIN BASED; CYHALOFOP-BUTYL; POLYOXYETHYLENE SORBITAN MIXED FATTY ACID ESTERS; AMMONIUM SULFATE; SORBITAN FATTY ACID ESTERS; PHOSPHORIC ACID; ALPHA-(PARA-NONYLPHENYL)-OMEGA-HYDROXYPOLY (OXYETHYLENE) SULFATE, AMMONIUM SALT; CITRIC ACID; POLYACRYLIC POLYMER	1	847	326
PROPANIL; CYHALOFOP-BUTYL; PENOXSULAM; 2,4-D; BISPYRIBAC-SODIUM	1	0	1
PROPANIL; ORCHEX 796 OIL; OLEIC ACID; 2,4-D, DIMETHYLAMINE SALT; ALPHA-ALKYL (C9-C11)-OMEGA-HYDROXYPOLY(OXYETHYLENE);	1	59	677

LIST OF ACTIVE INGREDIENTS APPLIED WITH THIOBENCARB	NUMBER APPLICATIONS	LBS THIOBENCARB	LBS OTHER AIs
BISPYRIBAC-SODIUM			
MINERAL OIL; CYHALOFOP-BUTYL; TALL OIL FATTY ACIDS; ALPHA-(PARA-NONYLPHENYL)-OMEGA-HYDROXYPOLY(OXYETHYLENE); ALPHA-2,6,8-TRIMETHYL-4-NONYLOXY-OMEGA-HYDROXYPOLY(OXYETHYLENE); POLYETHYLENE GLYCOL MONO(3-(TETRAMETHYL-1-(TRIMETHYLSILOXY)DISILOXANYL)PROPYL)ETHER; BISPYRIBAC-SODIUM; PROPYLENE GLYCOL; PENOXSULAM; CARFENTRAZONE-ETHYL; DIMETHYLPOLYSILOXANE	1	15	55
BENSULFURON METHYL; LAMBDA-CYHALOTHRIN; POLYSILOXANE; DERIVATED NATURAL POLYMERS	1	142	3
COPPER SULFATE (PENTAHYDRATE); LAMBDA-CYHALOTHRIN; POLYACRYLAMIDE POLYMER; POLYSACCHARIDE POLYMER	1	308	1,535
POLYACRYLAMIDE POLYMER; POLYSACCHARIDE POLYMER	1	119	0
PENDIMETHALIN; OLEIC ACID, METHYL ESTER; BISPYRIBAC-SODIUM; 2-(3-HYDROXYPROPYL)-HEPTA-METHYL TRISILOXANE, ETHOXYLATED, ACETATE	1	118	26
PETROLEUM DISTILLATES; ALPHA-(PARA-NONYLPHENYL)-OMEGA-HYDROXYPOLY(OXYETHYLENE); CYHALOFOP-BUTYL	1	237	455
BENSULFURON METHYL; PROPANIL; PETROLEUM OIL, PARAFFIN BASED; POLYOXYETHYLENE SORBITAN MIXED FATTY ACID ESTERS; SORBITAN FATTY ACID ESTERS; ALPHA-(PARA-NONYLPHENYL)-OMEGA-HYDROXYPOLY (OXYETHYLENE) SULFATE, AMMONIUM SALT	1	1	3
LAMBDA-CYHALOTHRIN; POLYACRYLAMIDE POLYMER	1	182	2
DIFLUBENZURON; (S)-CYPERMETHRIN	1	320	2
HYDROTREATED PARAFFINIC SOLVENT; CYHALOFOP-BUTYL; TALL OIL FATTY ACIDS; ALPHA-(PARA-NONYLPHENYL)-OMEGA-HYDROXYPOLY(OXYETHYLENE); CITRIC ACID;	1	705	384

LIST OF ACTIVE INGREDIENTS APPLIED WITH THIOBENCARB	NUMBER APPLICATIONS	LBS THIOBENCARB	LBS OTHER AIs
CALCIUM CHLORIDE			
MOLINATE; METHYLATED SOYBEAN OIL; LAMBDA-CYHALOTHRIN; KEROSENE; DIMETHYL ALKYL TERTIARY AMINES; BENZOIC ACID	1	456	108
COPPER SULFATE (PENTAHYDRATE); BENSULFURON METHYL; LAMBDA-CYHALOTHRIN; POLYSILOXANE; DERIVATED NATURAL POLYMERS	1	377	751
PROPANIL; ISOPARAFFINIC HYDROCARBONS; ORCHEX 796 OIL; ALPHA-(PARA-NONYLPHENYL)-OMEGA-HYDROXYPOLY(OXYETHYLENE); TALL OIL FATTY ACIDS; ALPHA-(PARA-NONYLPHENYL)-OMEGA-HYDROXYPOLY(OXYETHYLENE), PHOSPHATE ESTER; N,N-BIS-(2-OMEGA-HYDROXYPOLY(OXYETHYLENE)ETHYL)ALKYLAMINE, ALKYL DERIVED FROM TALLOW FATTY ACIDS	1	623	934
OLEIC ACID, METHYL ESTER; 2-(3-HYDROXYPROPYL)-HEPTA-METHYL TRISILOXANE, ETHOXYLATED, ACETATE; BISPYRIBAC-SODIUM	1	0	2
POLYALKENE OXIDE MODIFIED HEPTAMETHYL TRISILOXANE; BISPYRIBAC-SODIUM; PROPYLENE GLYCOL	1	15	2
COPPER SULFATE (PENTAHYDRATE); KEROSENE; LAMBDA-CYHALOTHRIN; SOYBEAN FATTY ACIDS, DIMETHYLAMINE SALT; BENZOIC ACID	1	173	647
MOLINATE; PROPANIL; BENSULFURON METHYL	1	1	1
SODIUM CARBONATE PEROXYHYDRATE; ORTHOSULFAMURON	1	1,080	1,278
CARFENTRAZONE-ETHYL; PENOXSULAM	1	141	11
TOTAL	**18,733**	**6,178,623**	**1,649,004**

20. Appendix 5: Toxicity of Eleven Pesticides to Embryonic Zebrafish

Toxicity of Eleven Pesticides to Embryonic Zebrafish

November 2011

Project Summary

The Northwest Fisheries Science Center conducted an experiment requested by NOAA's Office of Protected Resources in support of a Biological Opinion regarding the toxicity of various pesticides to endangered salmon species. The experiment detailed here investigated the effects of eleven pesticides on developing zebrafish (*Danio rerio)*, a species that is widely used as a toxicological model for other fish species. Zebrafish are a useful model species because the early ontogeny of zebrafish is rapid and well documented (Kimmel et al., 1995) and their features are easily observed through translucent chorions and bodies. In this experiment, embryonic zebrafish were exposed to oryzalin, trifluralin, prometryn, pendimethalin, fenbutatin oxide, thiobencarb, propargite, metolachlor, 1,3-dichloropropene, bromoxynil and diflubenzuron in 5-day static-renewal exposures. Toxicity endpoints included mortality, developmental abnormalities, and body length on the final day of the experiment. Three of the chemicals tested, prometryn, fenbutatin oxide, and diflubenzuron, did not produce an adverse effect on zebrafish survival, morphology or length at the tested concentrations. The pesticides trifluralin, pendimethalin and thiobencarb increased the rate of abnormality in developing zebrafish without appreciably increasing the rate of mortality at the concentrations tested. Fish lengths were significantly smaller following exposure to oryzalin, bromoxynil, trifluralin, pendimethalin, thiobencarb, propargite, metolachlor and 1,3-dichloropropene.

Methods

Fish: Zebrafish (*D. rerio*) embryos were obtained from a colony maintained at the Northwest Fisheries Science Center according to standard operating procedures (Linbo, 2009). Male and female zebrafish were combined in spawning tanks and eggs were collected at the beginning of the next light cycle, approximately one hour after the spawning event. Embryos were housed in a temperature-controlled incubator at 28.5 °C for the duration of the experiment.

Pesticide stock solutions: Pesticides were obtained in pure form from Chem Service, Inc. (West Chester, Pennsylvania). Pesticide stock solutions were made in acetone and stored under dark conditions at 4 °C. A working solution composed of stock solution and water from the zebrafish colony (system water) was mixed fresh at the start of each day, and subsequent exposure concentrations serially diluted. The maximum acetone concentration for any exposure was 0.1%. The highest pesticide concentration of each compound tested was generally the reported rainbow trout or zebrafish 96-hr LC_{50} value (the concentration lethal to 50% of the test organisms). The highest exposure concentration of 1,3-dichloropropene was 100 times lower than the reported LC_{50} value because of observed developmental effects, while exposure concentrations of diflubenzuron were lower due to low solubility in acetone.

Pesticide exposures: Normally developing zebrafish embryos at 1.5-2.5 hpf (hours post-fertilization) were selected and placed in 60 mm acetone-washed glass Petri dishes with 10 ml of pesticide solution. Individual dishes contained 15 embryos and each exposure concentration was tested in triplicate. Exposures were conducted in batches comprised of one or two pesticides, water controls, and 0.1% acetone controls. Exposure solutions were renewed every 24 hours. Dead embryos were removed from the dishes each day to prevent fungal growth and contamination.

Anatomical screening and measurement of fish body length: Embryos were scored every 24 hr for mortality and abnormalities through 5 dpf (days post-fertilization). See Table 2 for a description of the observed developmental abnormalities. Daily anatomical screenings were performed using a Nikon-SMZ-800 stereomicroscope with a diascopic base (Meridian Instruments, Seattle, Washington). Only surviving fish were screened for anatomical abnormalities. At 5 dpf, the embryos were anesthetized with tricaine methanesulfonate (MS-222; Sigma-Aldrich, St. Louis, Missouri) to measure body length. All surviving embryos from each exposure dish were simultaneously photographed using a Spot RT digital camera (Diagnostic Instruments, Inc., Sterling Heights, Michigan) mounted on a stereomicroscope. Length was measured from the anterior tip of the mouth along the notochord to the posterior tip of the notochord, and quantified using ImageJ software (available online at http://rsbweb.nih.gov/ij/).

Statistical tests: Length was the only parameter explicitly tested. Lengths of control fish were compared using a two-factor ANOVA comparing type (water and acetone) and batch, and showed a significant result of batch only. Subsequent analyses of exposures compared the average of three dishes (n = 3) to their corresponding batch controls. Differences in embryo lengths between concentrations of a given pesticide were tested using one-way ANOVAs with a Tukey HSD post hoc (Tables 3-13).

Results

Chemical-specific mortality and abnormality data, as well as their respective controls, are presented in Figures 1-11. Both water and acetone controls showed consistently low rates of both mortality and abnormality. We found that 3 pesticides (prometryn, fenbutatin oxide and diflubenzuron) showed no increases in mortality or abnormality as well as no significant differences in embryo length. Three additional chemicals (trifluralin, pendimethalin and thiobencarb) produced higher rates of abnormalities and significantly shorter embryos at the highest exposure concentration without increasing mortality. While the remaining pesticides (oryzalin, bromoxynil, propargite, metolachlor, and 1,3-dichloropropene) produced significantly shorter embryos at various exposure concentrations with no effect on mortality or abnormality, there was no clear dose-dependent trend. Whether there is a biological consequence to these shorter lengths at the concentrations tested here is a subject for further investigation.

Table 1. Nominal concentrations of pesticides used in exposures and rainbow trout LC_{50} values.

Compound Name	Type	Exposure Concentrations (μg/l)	Rainbow Trout LC_{50} values (μg/l)
Oryzalin	Herbicide	3, 30, 300, 3000	3260
Trifluralin	Herbicide	0.05, 0.5, 5, 50	50
Prometryn	Herbicide	0.9, 9, 90, 900	2900
Pendimethalin	Herbicide	0.15, 1.5, 15, 150	138
Fenbutatin oxide	Insecticide	0.01, 0.1, 1, 10	10
Thiobencarb	Herbicide	0.8, 8, 80, 800	790
Propargite	Insecticide	0.15, 1.5, 15, 150	<168
Metolachlor	Herbicide	0.3, 3, 30, 300	300
1,3-Dichloropropene	Insecticide	0.03, 0.3, 0.3, 3	270
Bromoxynil	Herbicide	0.05, 0.5, 5, 50	41
Diflubenzuron	Insecticide/Fungicide	2, 20, 200, 2000	72000

Table 2. Abnormalities observed during zebrafish embryo exposures.

Abnormality	Description
Edema	Accumulation of excess fluid in any one of the following cavities: heart, yolk sac, yolk extension, eyes.
Unhatched	Failure to hatch at 5 dpf.
Curved	Curvature of the tail dorsally in the sagittal plane such that a line drawn from the posterior tip of the notochord to the mouth of the fish would yield a gap between the line and body.
Lethargic	An inability to maintain an upright posture and/or inactivity.
Deformed fins	The absence or improper formation of fin tissue.
Deformed tail	A notable shortening of the tail or improper notochord development.
Bent	A bend in the body or tail of the embryo in the coronal plane.

Oryzalin

Oryzalin exposure did not impact developing zebrafish in a dose-dependent manner. Mortality was the highest (20%) at 30 μg/l, but declined to 8.9% at 3000 μg/l. Abnormality was the highest at 3000 μg/l (17.1%), but was also elevated at 3 μg/l (16.2%). The most common abnormality observed was edema.

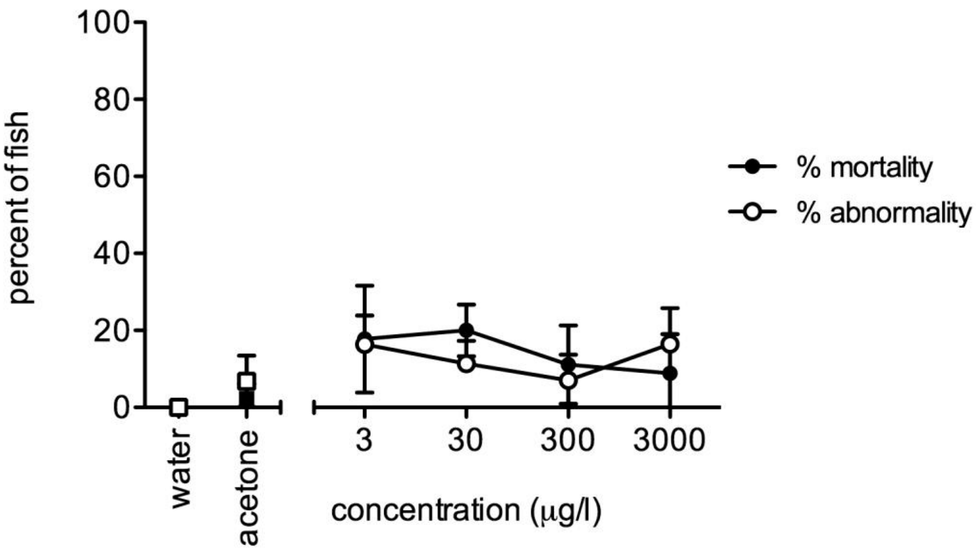


Figure 1: Percent mortality and abnormality observed in control and oryzalin-exposed zebrafish. Symbols are means (n = 3) ± SD.

Table 3: Average length of fish exposed to oryzalin and controls (n = 3 dishes). There was a significant effect of oryzalin (One-way ANOVA, $p < 0.0001$). * Indicates treatment significantly different than controls (Tukey HSD, $p < 0.01$).

Treatment (μg/l)	Average length ± SD (mm)
Water control	4.49 ± 0.02
0.1% acetone	4.50 ± 0.02
3	4.53 ± 0.05
30	4.48 ± 0.02
300	4.51 ± 0.05
3000	4.27 ± 0.02*

Bromoxynil

Bromoxynil exposure did not cause an increase in mortality or abnormality in developing zebrafish. The highest rate of abnormality (6.7%) was observed at 0.05 μg/l and 50 μg/l. Mortality occurred the most frequently at 0.5 μg/l and 5 μg/l at a rate of 2.2%.

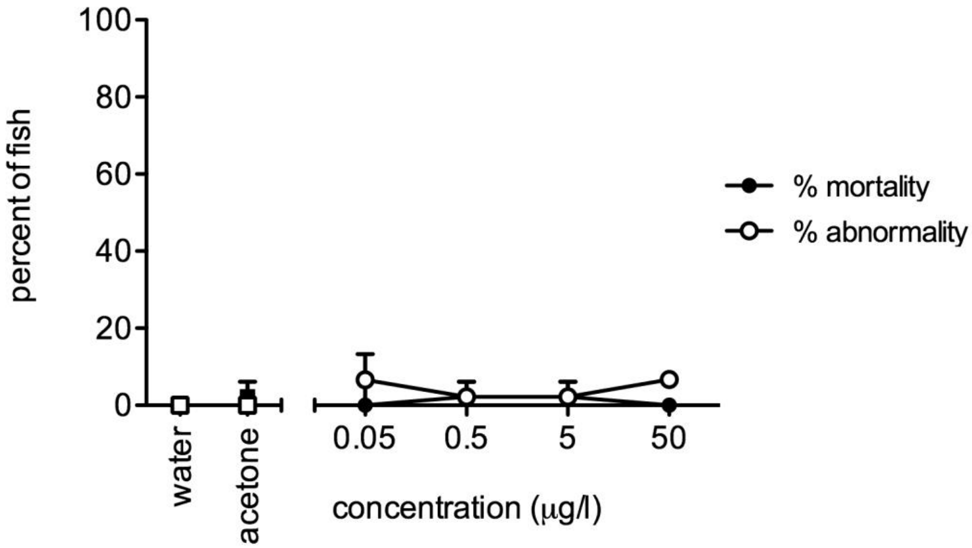


Figure 2. Percent mortality and abnormality in controls and zebrafish exposed to bromoxynil. Symbols are means (n = 3) ± SD.

Table 4: Average length of fish exposed to bromoxynil and controls (n = 3 dishes). There was a significant effect of bromoxynil (One-way ANOVA, *p* < 0.0001). * Indicates treatment significantly different than controls (Tukey HSD, *p* < 0.05).

Treatment (μg/l)	Average length ± SD (mm)
Water control	4.20 ± 0.04
0.1% acetone	4.06 ± 0.02
0.05	3.97 ± 0.03*
0.5	4.08 ± 0.01
5	4.05 ± 0.04
50	4.13 ± 0.06

Trifluralin

Exposure to trifluralin caused significant abnormalities at the highest dose tested (50 μg/l). The rate of abnormality at this dose was 95.3%, and the most common abnormality noted was lethargy, characterized by the absence of active swimming and a tendency to lose upright posture. Mortality was the greatest (22.2%) at 0.5 μg/l.

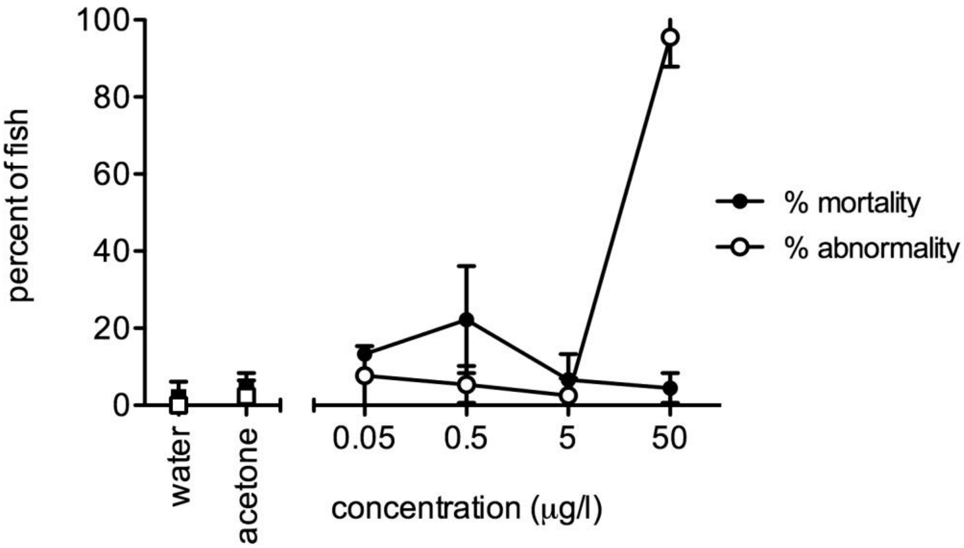


Figure 3. Percent mortality and abnormality of controls and zebrafish exposed to trifluralin. Symbols are means (n = 3) ± SD.

Table 5: Average lengths of fish exposed to trifluralin and controls (n = 3 dishes). There was a significant effect of trifluralin (One-way ANOVA, $p < 0.0001$). * Indicates treatment significantly different than controls (Tukey HSD, $p < 0.01$).

Treatment (μg/l)	Average length ± 1 SD (mm)
Water control	4.02 ± 0.01
0.1% acetone	4.11 ± 0.07
0.05	4.05 ± 0.02
0.5	4.11 ± 0.07
5	4.01 ± 0.07
50	3.59 ± 0.03*

Prometryn

Prometryn exposure did not adversely affect either the rate of abnormality or mortality in developing zebrafish. The highest rate of morality observed was at 9 μg/l (4.4%), and the highest rate of abnormality was at 0.9 μg/l and 900 μg/l (2.3%).

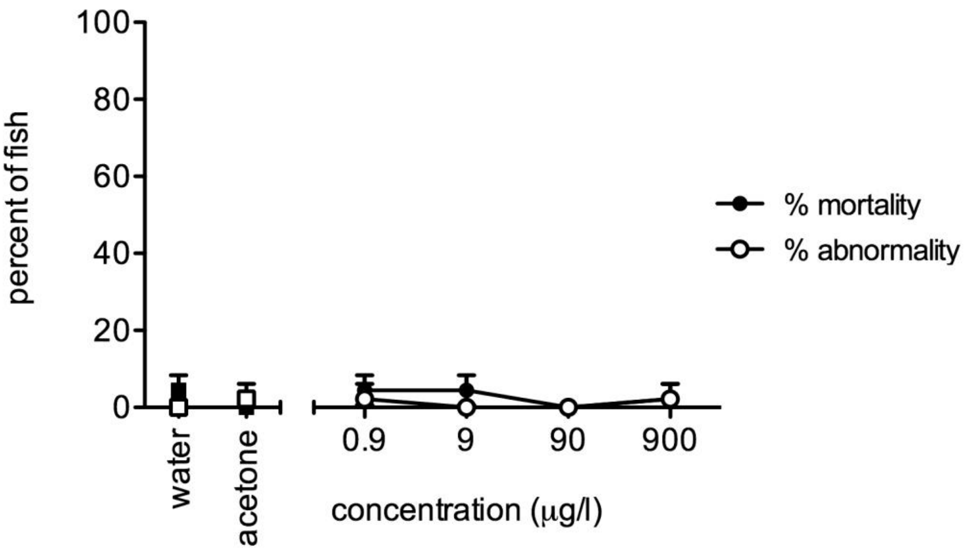


Figure 4. Percent mortality and abnormality of controls and prometryn exposed fish. Symbols are means (n = 3) ± SD.

Table 6: Average lengths of fish exposed to prometryn and controls (n = 3 dishes). Exposure to prometryn did not significantly affect fish length (One-way ANOVA, p > 0.05).

Treatment (μg/l)	Average length ± SD (mm)
Water control	3.85 ± 0.06
0.1% acetone	3.96 ± 0.03
0.9	3.96 ± 0.06
9	3.95 ± 0.01
90	3.97 ± 0.03
900	3.85 ± 0.02

Pendimethalin

Embryos exposed to 150 μg/l of pendimethalin developed a significant amount (100%) of abnormalities. Abnormal embryos were lethargic and struggled to swim. The highest rate of mortality (11.1%) was noted at 15 μg/l.

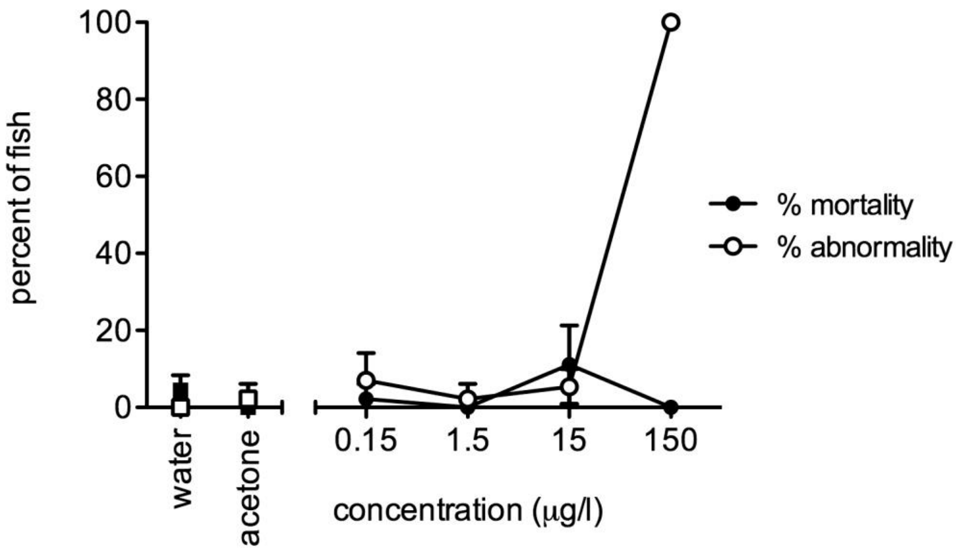


Figure 5. Percent mortality and abnormality of controls and fish exposed to pendimethalin. Symbols are means (n = 3) ± SD.

Table 7: Average lengths of fish exposed to pendimethalin and controls (n = 3 dishes). Pendimethalin exposure significantly impacted the length of larvae (One-way ANOVA, *p* <0.001). * Indicates treatment significantly different than controls (Tukey HSD, *p* < 0.01).

Treatment (μg/l)	Average length ± SD (mm)
Water control	3.85 ± 0.06
0.1% acetone	3.96 ± 0.03
0.15	3.98 ± 0.04
1.5	3.97 ± 0.03
15	3.94 ± 0.06
150	3.59 ± 0.03*

Fenbutatin oxide

Fenbutatin oxide did not cause a dose-dependent change in mortality or abnormality. Mortality occurred the most frequently at 10 μg/l (28.9%). Abnormality on the other hand was highest at 0.1 μg/l (26.3%), and declined at higher concentrations.

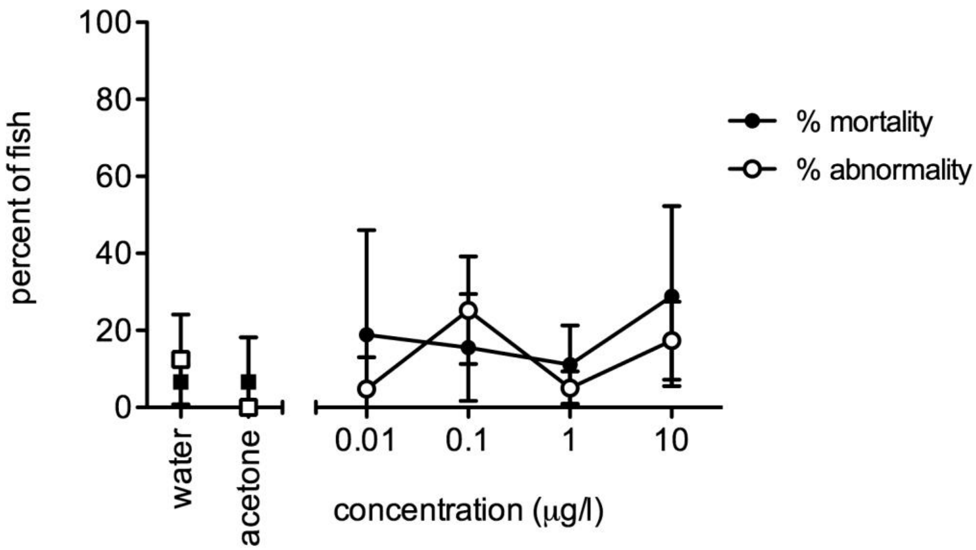


Figure 6. Percent mortality and abnormality of controls and fish exposed to fenbutatin oxide. Symbols are means (n = 3) ± SD.

Table 8: Average lengths of fish exposed to fenbutatin oxide and controls (n = 3 dishes). Fenbutatin oxide exposure did not affect the length of fish (One-way ANOVA, p > 0.05).

Treatment (μg/l)	Average length ± SD (mm)
Water control	3.90 ± 0.06
0.1% acetone	3.93 ± 0.01
0.01	3.91 ± 0.03
0.1	3.88 ± 0.06
1	3.91 ± 0.04
10	3.87 ± 0.02

Thiobencarb

Exposing developing zebrafish to thiobencarb produced abnormalities in 100% of the embryos at 800 μg/l. The 5-dpf larvae behaved abnormally with erratic swimming patterns. Mortality at 800 μg/l was 13.3%.

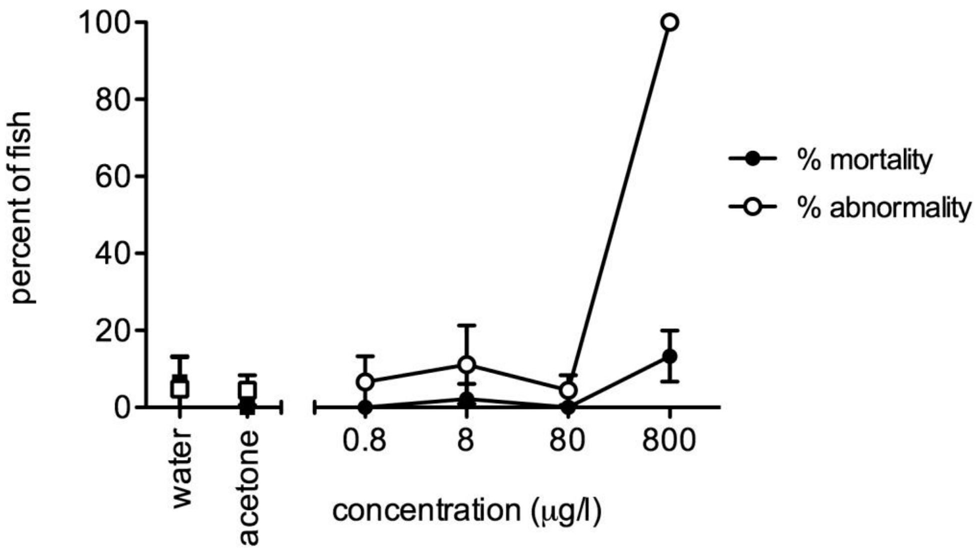


Figure 7. Percent mortality and abnormality observed in controls and fish exposed to thiobencarb. Symbols are means (n = 3) ± SD.

Table 9: Average lengths of fish exposed to thiobencarb and controls (n = 3 dishes). There was a significant effect of thiobencarb (One-way ANOVA, *p*<0.0001). * Indicates treatment significantly different than controls (Tukey HSD, *p* < 0.01).

Treatment (μg/l)	Average length ± SD (mm)
Water control	3.92 ± 0.04
0.1% acetone	3.99 ± 0.03
0.8	3.91 ± 0.03
8	3.87 ±0.04
80	3.91 ± 0.03
800	3.69 ± 0.07*

Propargite

Zebrafish embryos exposed to propargite did not show increased rates of mortality or abnormality. The highest rate of mortality (4.4 %) was observed at 0.15 μg/l and 1.5 μg/l. Embryos had the greatest number of abnormalities (13.6%) at 150 μg/l.

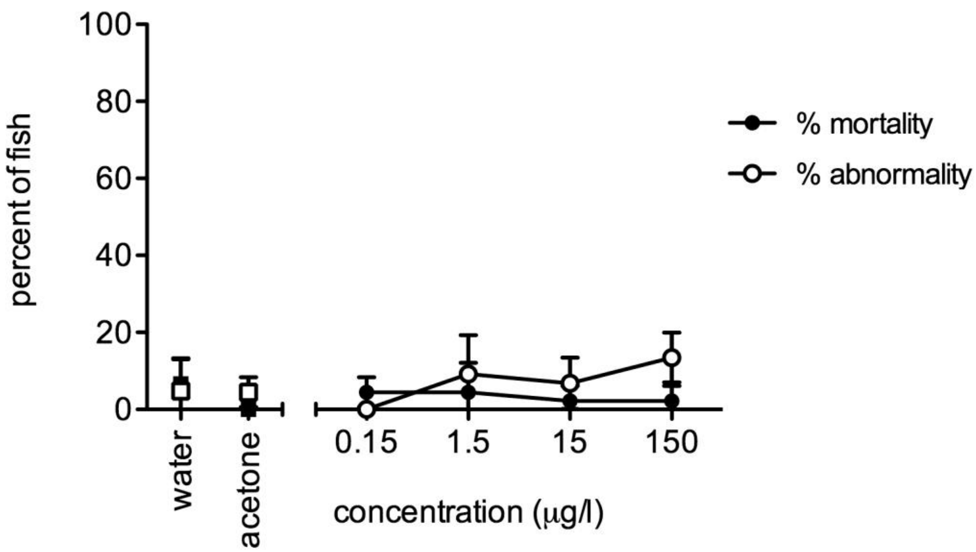


Figure 8. Percent mortality and abnormality in controls and fish exposed to propargite. Symbols are means (n = 3) ± SD.

Table 10: Average lengths of fish exposed to propargite and controls (n = 3 dishes). Propargite produced significant effects (One-way ANOVA, *p*= 0.005). * Indicates treatment significantly different than controls (Tukey HSD, *p* < 0.01).

Treatment (μg/l)	Average length ± SD (mm)
Water control	3.92 ± 0.04
0.1% acetone	3.99 ± 0.03
0.15	3.95 ± 0.04
1.5	3.92 ± 0.04
15	3.94 ± 0.02
150	3.83 ± 0.01*

Metolachlor

Exposure to metolachlor did not alter zebrafish mortality, although a higher rate (28.6%) of abnormality was observed at 300 μg/l. The most frequent abnormality noted was a failure to hatch by 5 dpf.

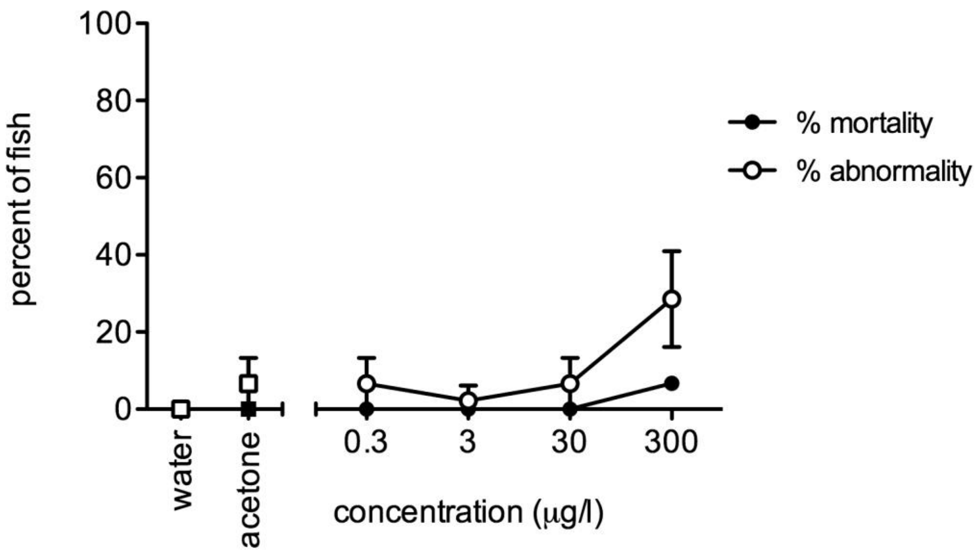


Figure 9. Percent mortality and abnormality of zebrafish exposed to metolachlor and controls. Symbols are means (n = 3) ± SD.

Table 11: **Average lengths of fish exposed to metolachlor and controls (n = 3 dishes). There was a significant effect of metolachlor (One-way ANOVA, *p* < 0.0001). * Indicates treatment significantly different than controls (Tukey HSD, *p* < 0.01).**

Treatment (μg/l)	Average length ± SD (mm)
Water control	4.42 ± 0.03
0.1% acetone	4.37 ± 0.03
0.3	4.24 ± 0.06*
3	4.40 ± 0.05
30	4.23 ± 0.05*
300	4.18 ± 0.05*

1,3-Dichloropropene

Exposure to 1,3-dichloropropene caused an increase in abnormality and mortality in developing zebrafish, but not in a dose dependent manner. The highest rate of mortality (28.9%) occurred at 0.3 μg/l, and declined at higher concentrations. The highest rate of abnormality (37.5%) was observed at 3 μg/l. The rate of abnormality remained between 28.1% and 37.5% for all exposure concentrations and the most commonly observed abnormality was failure to hatch by 5dpf.

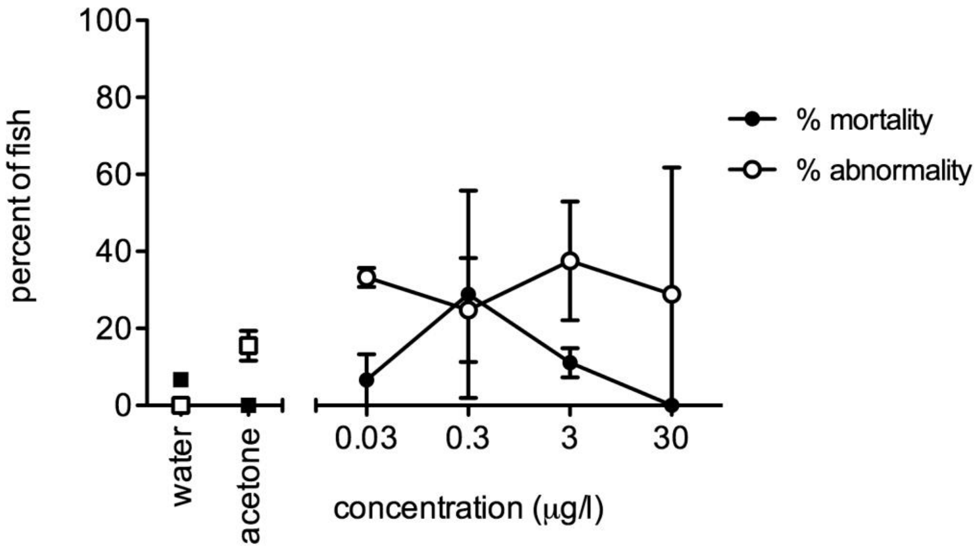


Figure 10. Percent mortality and abnormality observed in fish exposed to 1,3-dichloropropene and controls. Symbols are means (n = 3) ± SD.

Table 12. Average lengths of fish exposed to 1,3-dichloropropene and controls (n = 3 dishes). There was a significant effect of 1,3-dichloropropene (One-way ANOVA, *p* <0.001). * Indicates treatment significantly different than controls (Tukey HSD, *p* < 0.01).

Treatment (μg/l)	Average length ± SD (mm)
Water control	4.46 ± 0.03
0.1% acetone	4.34 ± 0.03
0.03	4.32 ± 0.04
0.3	4.14 ± 0.05*
3	4.28 ± 0.08
30	4.27 ± 0.06

Diflubenzuron

Diflubenzuron did not influence zebrafish mortality or abnormality. The highest rate of abnormality (6.8%) was observed at 20 μg/l, and the highest rate of morality (4.4%) was observed at 2 μg/l. However, it is important to note that diflubenzuron was difficult to work with because of its low solubility in acetone (6.5 g/l). The most concentrated stock solution of diflubenzuron we were able to make was 2 g/l. Diflubenzuron appeared to remain in solution after dosing the exposure dishes, however after 24hrs, the highest exposure concentration dishes (2000 μg/l) had visible floating particles. Thus, without using alternative methodologies (e.g. DMSO as the carrier), we are not confident about accurate dosing for this compound.

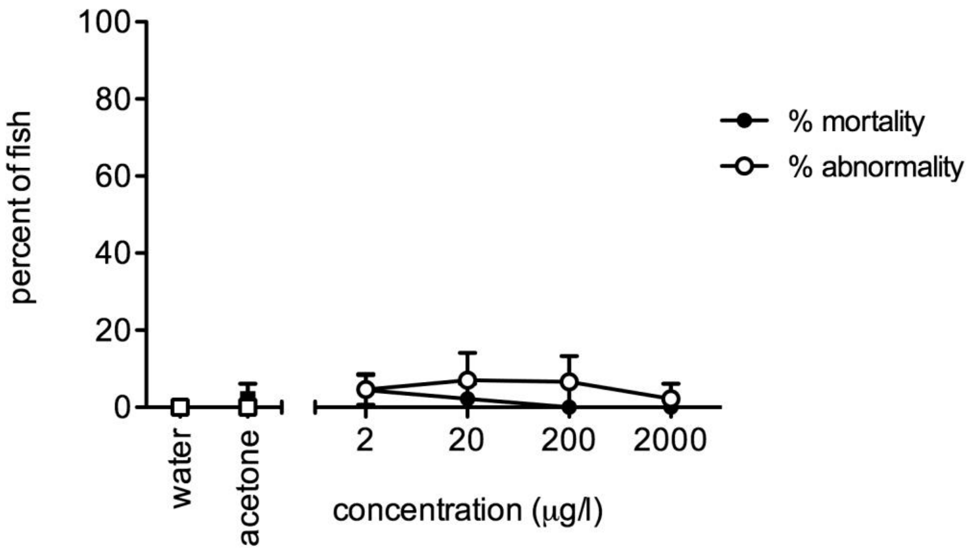


Figure 11. Percent mortality and abnormality observed in control fish and fish exposed to diflubenzuron. Symbols are means (n = 3) ± SD.

Table 13: Average lengths of fish exposed to diflubenzuron and controls (n = 3 dishes). There was not a significant effect of diflubenzuron on fish length (One-way ANOVA, p > 0.05).

Treatment (μg/l)	Average length ± SD (mm)
Water control	4.20 ± 0.04
0.1% acetone	4.06 ± 0.02
2	4.06 ± 0.03
20	4.88 ± 0.06
200	4.98 ± 0.03
2000	4.08 ± 0.03

References

Kimmel, CB, WW Ballard, SR Kimmel, B Ullman and TF Schilling. 1995. Stages of embryonic development of the zebrafish. *Developmental Dynamics* 203(3):253-310.

Linbo, TL. 2009. Zebrafish (*Danio rerio*) husbandry and colony maintenance at the Northwest Fisheries Science Center. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-NWFSC-100, 62p.

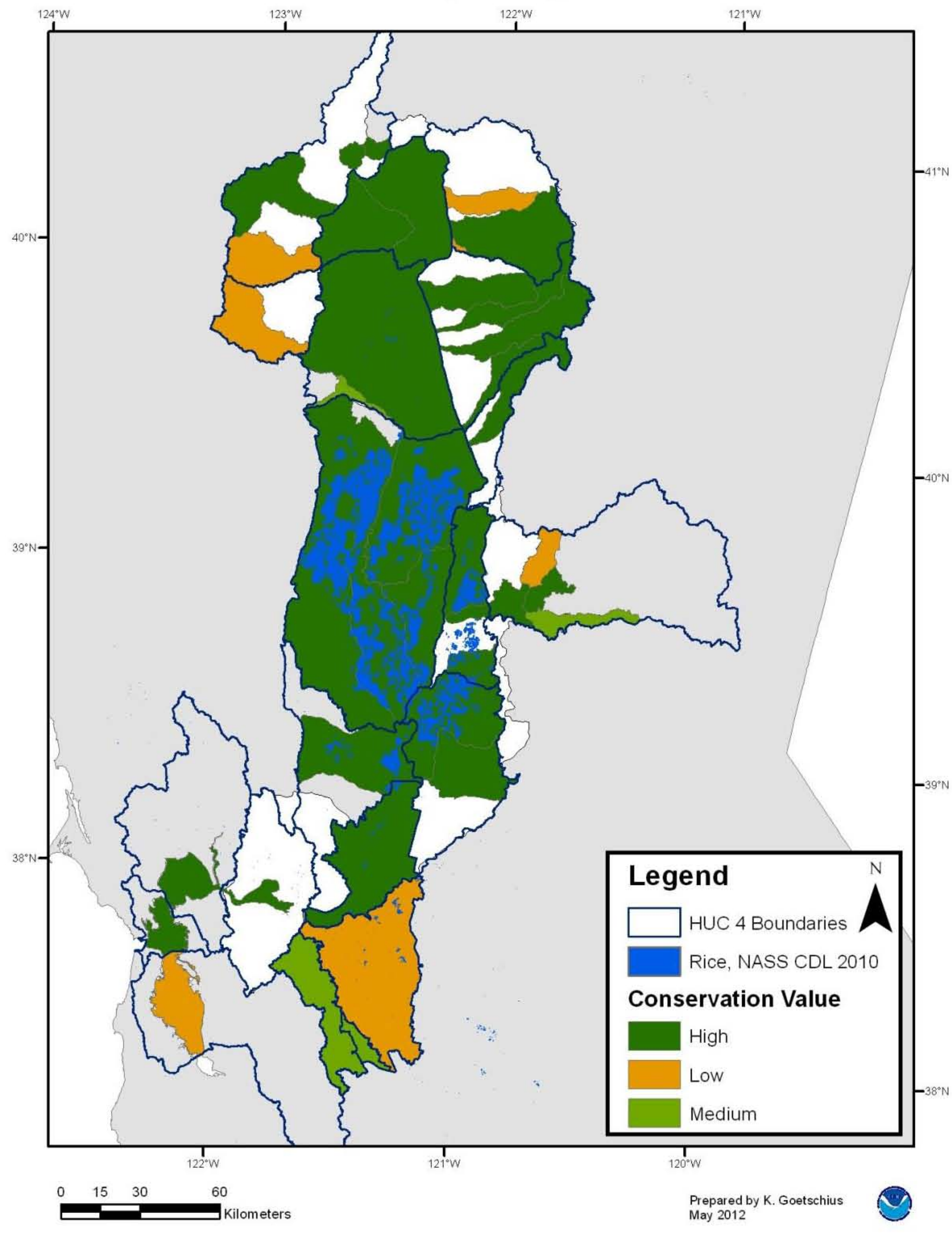
21. Appendix 6: Conservation Values of Designated Critical Habitat for Listed Salmonids in California's Central Valley

Rice Areas and Conservation Values of Designated Critical Habitat for Listed Salmonids in California's Central Valley

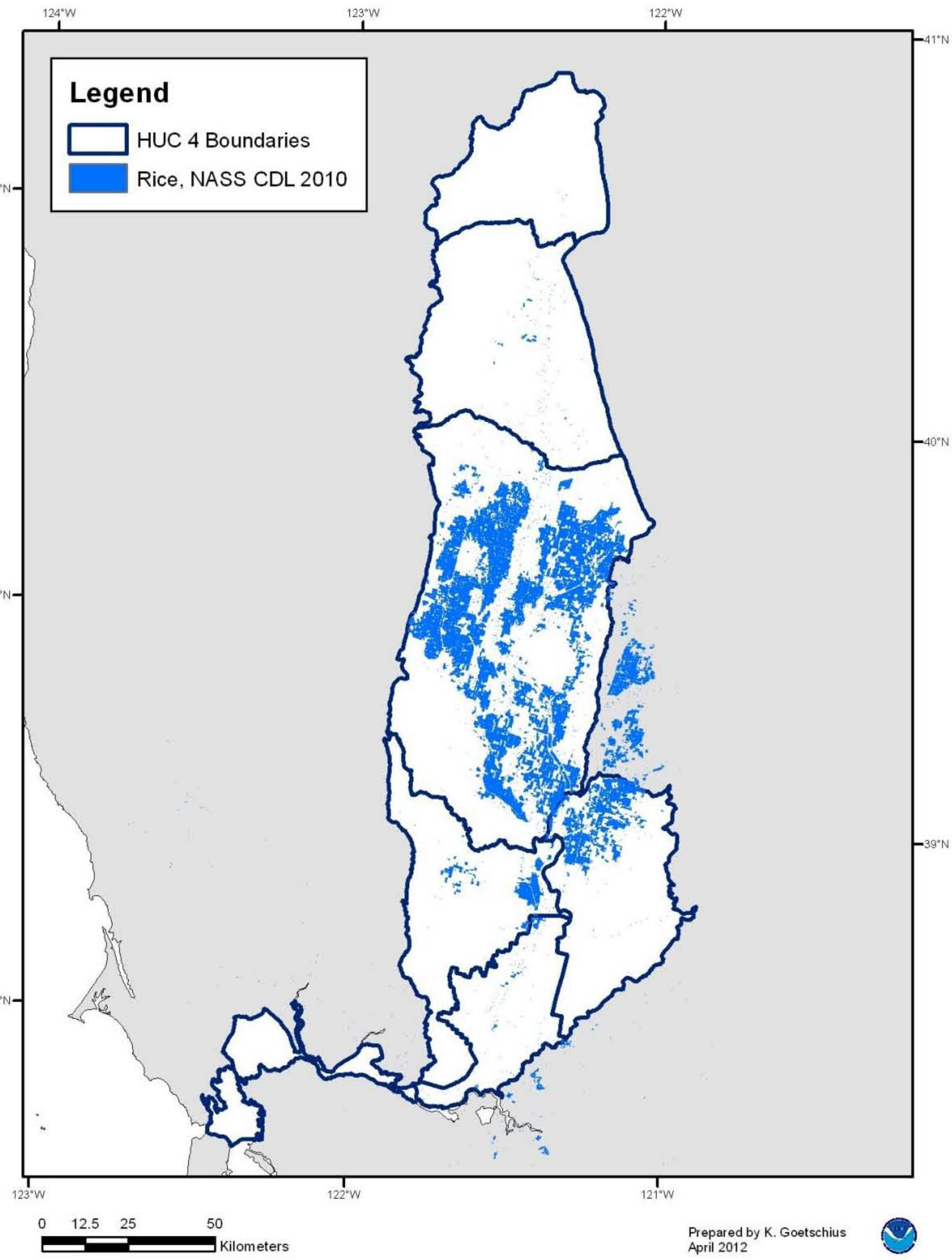
The maps below provide conservation values (High, Medium, Low) for each of the watershed units within the designated critical habitats for the three California Central Valley listed salmonids. Not all habitats have the same conservation value for an ESU. In determining what areas are critical habitat for listed salmonids, NMFS had to consider those physical or biological features that are essential to the conservation of a given species, called the primary constituent elements (PCEs). Designation of the conservation values required taking into account the quality, quantity, and distribution of PCEs within each watershed (50 CFR Part 226, September 2, 2005). Using GIS, we overlaid the NASS CDL 2010 data layer for rice to get a spatially relevant distribution of rice growing areas with the watersheds and their conservation values. We evaluated the overlap between rice growing areas and designated critical habitat.

At this point in time, conservation values within the designated critical habitat for Sacramento River winter-run Chinook have not been assigned. For this Opinion, we assumed each watershed to have a high conservation value.

Central Valley Spring-Run Chinook ESU
Conservation Value of Hydrologic Sub-Areas



Sacramento River Winter Run Chinook ESU
Critical Habitat



California Central Valley Steelhead DPS
Conservation Value of Hydrologic Sub-Areas