National Marine Fisheries Service

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

August 31, 2010

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Agency:	United States Environmental Protection Agency
Activities Considered:	Authorization of pesticide products containing the active ingredients azinphos methyl, bensulide dimethoate, disulfoton, ethoprop, fenamiphos, methamidophos, methidathion, methyl parathion, naled, phorate, and phosmet, 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: Helen Golde for J. H. Lecky

Date: August 31, 2010

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, that agency is required to consult formally with the National Marine Fisheries Service (NMFS) or the U.S. Fish and Wildlife Service (USFWS), 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 proposal 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.s) of azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, methamidophos, methidathion, methyl parathion, naled, phorate, and phosmet from November 29, 2002 through July 29, 2004. EPA authorization of pesticide uses are categorized as FIFRA sections 3 (new product registrations), 4 (reregistrations and special review), 18 (emergency use), or 24(c) [Special Local Needs (SLN)]. At that time, EPA determined that uses of pesticide products containing these ingredients “may affect” most of the 26 Evolutionarily Significant Units (ESUs) of Pacific salmonids listed as endangered or threatened and designated critical habitat for the ESUs. This document represents NMFS’ biological opinion (Opinion) on the impacts of EPA’s authorization of pesticide products containing the above-mentioned a.i.s on the listed ESUs, plus on two newly listed salmonids. This is a partial consultation because pursuant to the court’s order, EPA sought consultation on only this group of listed species under NMFS’ jurisdiction. However, even though the court’s order did not address the two more recently listed salmonids, NMFS analyzed the impacts of EPA’s action to them because they belong to the same taxon. NMFS analysis requires consideration of the same information. Consultation with NMFS will be completed when EPA makes effect determinations on all remaining species 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 §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 2002, 2003, and 2004 requests for formal consultation on the proposed authorization of the above a.i.s. It also includes our review of recovery plans for listed Pacific salmonids, past and current research and population dynamics modeling efforts, monitoring reports from prior research, Opinions on similar research, 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 a.i.s for azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, methamidophos, methidathion, methyl parathion, naled, phorate, and phosmet. NMFS also considered information and comments provided by EPA and by the registrants identified as applicants by EPA.

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 to 26 ESUs of listed Pacific salmonids of its continuing approval of 54 pesticide a.i.s.

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 to the salmonids on all 54 a.i.s by December 2004.

In December 2002, EPA and the USFWS and NMFS (referred to as the Services) began interagency discussions for streamlining EPA's court ordered consultations.

On January 24, 2003, EPA and the Services published an Advance Notice of Proposed Rulemaking seeking public comment on improving the process by which EPA and the Services work together to protect listed species and critical habitat (68 FR 3785).

Between May and December 2003, EPA and the Services reviewed EPA's ecological risk assessment methodology and earlier drafts of EPA's "Overview of the Ecological Risk Assessment Process in the Office of Pesticide Programs, U.S. Environmental Protection Agency (Overview Document)". EPA and the Services also developed counterpart regulations to streamline the consultation process.

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, 357 F.Supp. 2d 1266 (W.D. Wash. 2004). The court imposed several additional restrictions on pesticide use in specific settings.

On January 23, 2004, EPA finalized its Overview Document which specified EPA's conduct of ecological risk assessment on pesticide registrations.

On January 26, 2004, the Services approved EPA's procedures and methods for conducting ecological risk assessments and approved interagency counterpart regulations for EPA's pesticide registration program.

On January 30, 2004, the Services published in the Federal Register (69 FR 4465) proposed joint counterpart regulations for consultation under the ESA for regulatory actions under the FIFRA, codified at 50 C.F.R. Part 402 Subpart D.

On August 5, 2004, the Services promulgated final joint counterpart regulations for EPA's ESA-related actions taken pursuant to FIFRA. These regulations and the Alternative Conservation Agreement (ACA) under the regulations allowed EPA to conduct independent analyses of potential impacts of pesticide registration on listed species and their designated critical habitats. The ACA outlined procedures to ensure EPA's risk assessment approach will produce effect determinations that reliably assess the effects of pesticides on listed species and designated critical habitat. Additionally, EPA and the Services agreed to meet annually, or more frequently as may be deemed appropriate. The intention of these meetings was to identify new research and other activities that may improve EPA's current approach for assessing the potential ecological risks posed by use of a pesticide to listed species or designated critical habitat.

On September 23, 2004, the Washington Toxics Coalition and others challenged the counterpart regulations in the U.S. District Court for the Western District of Washington, Civ. No. 04-1998, alleging that the regulations were not authorized by the ESA and that the Services had not complied with the Administrative Procedure Act and the National Environmental Policy Act (NEPA) in promulgating these counterpart regulations.

In January 2006, EPA and the Services developed a draft joint interagency research agenda to address several critical areas of scientific and procedural uncertainties in EPA's current effects determination process. The jointly developed document identified eight areas of risk assessment and research uncertainties.

On August 24, 2006, the court determined the Services did not implement NEPA procedures properly during their promulgation of the joint counterpart regulations for EPA actions under FIFRA. Additionally, the court determined that the "not likely to adversely affect" and emergency consultation provisions of the counterpart regulations were arbitrary and capricious and contrary to the substantive requirements of ESA section 7(a)(2). The court determined that EPA may conduct its own formal consultation with the Services' involvement. <u>Washington Toxics Coalition v. Department of the Interior, 457 F.Supp. 2d 1158 (W.D.Wash. 2006)</u>.

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. This third consultation evaluates 12 organophosphate insecticides: azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, methamidophos, methidathion, methyl parathion, naled, phorate, and phosmet. EPA consultations on pesticide products currently focus on their effects to listed Pacific salmonids. EPA consultations remain incomplete until all protected species under NMFS' jurisdiction are covered.

Consultation History

On November 29, 2002, the EPA sent a letter to NMFS' Office of Protected Resources (OPR) requesting section 7 consultation for the registration of the a.i. bensulide and detailing its effects determinations on 26 ESUs of Pacific salmonids listed at that time. In that same letter, EPA's Office of Pesticide Programs (OPP) determined that the use of bensulide will have "no effect" for 7 ESUs, "may affect but is not likely to adversely affect" 2 ESUs, and "may affect" 17 ESUs of listed salmonids. EPA's "no effect" determinations for bensulide applied to California Coastal Chinook salmon, Oregon Coast coho salmon, Hood Canal Summer-run chum salmon, Columbia River chum salmon, Ozette Lake sockeye salmon, Snake River sockeye salmon, and Northern California steelhead.

On July 31, 2003, the EPA sent a letter to NMFS' OPR requesting section 7 consultation for the registration of the a.i. azinphos methyl and detailing its effects determinations on 26 ESUs of Pacific salmonids listed at that time. In that same letter, EPA's OPP determined that the use of azinphos methyl will have "no effect" for 1 ESU and "may affect 25 ESUs of listed salmonids. EPA's "no effect" determinations for azinphos methyl applied to Ozette Lake sockeye salmon.

On August 1, 2003, the EPA sent a letter to NMFS' OPR requesting section 7 consultation for the registration of the a.i. phorate and detailing its effects determinations on 26 ESUs of Pacific salmonids listed at that time. In that same letter, EPA's OPP determined that the use of phorate will have "no effect" for 4 ESUs, "may affect but is not likely to adversely affect" 2 ESUs, and "may affect" 19 ESUs of listed salmonids. EPA's "no effect" determinations for phorate applied to California Coastal Chinook salmon, Central California coho salmon, Southern California steelhead, and Northern California steelhead.

On December 1, 2003, the EPA sent letters to NMFS' OPR requesting section 7 consultation for the registration of the a.i.s disulfoton, fenamiphos, phosmet, and ethoprop, and detailing the effects determinations on 26 ESUs of Pacific salmonids listed at that time. In those same letters, EPA's OPP determined that the use of disulfoton will have "no effect" for 11 ESUs, "may affect but is not likely to adversely affect" 9 ESUs, and "may affect 6 ESUs of listed salmonids. EPA's "no effect" determinations for disulfoton applied to the California Coastal Chinook salmon, Lower Columbia River Chinook salmon, Columbia River chum salmon, Hood Canal Summer-run chum salmon, Central California Coast coho salmon, Southern Oregon and Northern California Coast coho salmon, Oregon Coast coho salmon, Ozette Lake sockeye salmon, Snake River sockeye salmon, Lower Columbia River steelhead, and Northern California steelhead.

EPA's OPP also determined that the use of fenamiphos will have "no effect" for 15 ESUs, "may affect but is not likely to adversely affect" 8 ESUs, and "may affect" 2 ESUs of listed salmonids. EPA's "no effect" determinations for fenamiphos applied to California Coastal Chinook salmon, Upper Columbia River Spring-run Chinook salmon, Puget Sound Chinook salmon, Snake River Fall-run Chinook salmon, Snake River Spring/Summer-run Chinook salmon, Columbia River chum salmon, Hood Canal Summer-run chum salmon, Southern Oregon and Northern California Coast coho salmon, Oregon Coast coho salmon, Ozette Lake sockeye salmon, Snake River sockeye salmon, Middle Columbia River steelhead, Northern California steelhead, Snake River steelhead, and Upper Columbia River steelhead.

EPA's OPP also determined that the use of phosmet will have "no effect" for 13 ESUs and "may affect but is not likely to adversely affect" 13 ESUs of listed salmonids. EPA's "no effect" determinations for phosmet applied to Southern California steelhead, South-Central California Coast steelhead, Central California Coast steelhead, Central Valley California steelhead, Northern California steelhead, Sacramento River Winter-run Chinook salmon, Central Valley Spring-run Chinook salmon, California Coastal Chinook salmon, Central California coho salmon, Oregon Coast coho salmon, Hood Canal Summer-run chum salmon, Ozette Lake sockeye salmon, and Snake River sockeye salmon.

EPA's OPP also determined that the use of ethoprop will have "no effect" for 8 ESUs, "may affect but is not likely to adversely affect" 12 ESUs, and "may affect" 6 ESUs of listed salmonids. EPA's "no effect" determinations for ethoprop applied to Northern California steelhead, Central California Coast steelhead, California Coastal Chinook salmon, Sacramento River Winter-run Chinook salmon, Central California Coast coho salmon, Hood Canal Summer-run chum salmon, Columbia River chum salmon, and Ozette Lake sockeye salmon.

On March 31, 2004, the EPA sent a letter to NMFS' OPR requesting section 7 consultation for the registration of the a.i. methamidophos and detailing its effects determinations on 26 ESUs of Pacific salmonids listed at that time. In that same letter, the EPA's OPP determined that the use of methamidophos will have "no effect" for 23 ESUs and "may affect but is not likely to adversely affect" 3 ESUs of listed salmonids. EPA's "no effect" determinations for methamidophos applied to the California Coastal Chinook salmon, Central Valley Spring-run Chinook salmon, Lower Columbia River Chinook salmon, Puget Sound Chinook salmon, Sacramento River Winter-run Chinook salmon, Snake River Fall-run Chinook salmon, Snake River Spring/Summer-run Chinook salmon, Upper Willamette River Chinook salmon, Columbia River chum salmon, Hood Canal Summer-run chum salmon, Central California Coast coho salmon, Lower Columbia River coho salmon, Southern Oregon and Northern California Coast coho salmon, Oregon Coast coho salmon, Ozette Lake sockeye salmon, Snake River sockeye salmon, Central California Coast steelhead, California Central Valley steelhead, Lower Columbia River steelhead, Northern California steelhead, Snake River steelhead, South-Central California Coast steelhead, Southern California steelhead, and Upper Willamette River steelhead.

On April 1, 2004, the EPA sent letters to NMFS' OPR requesting section 7 consultation for the registration of the a.i.s methidathion, methyl parathion, and naled, and detailing the effects determinations on 26 ESUs of Pacific salmonids listed at that time. In those same letters, the EPA's OPP determined that the use of methidathion will have "no effect" for 7 ESUs, "may affect but is not likely to adversely affect" 9 ESUs, and "may affect" 10 ESUs of listed salmonids. EPA's "no effect" determinations for methidathion applied to the California Coastal Chinook salmon, Central California coho salmon, Hood Canal Summer-run chum salmon, Columbia River chum salmon, Ozette Lake sockeye salmon, Northern California steelhead, and Southern California steelhead.

EPA's OPP also determined that the use of methyl parathion will have "no effect" for 5 ESUs, "may affect but is not likely to adversely affect" 12 ESUs, and "may affect" 9 ESUs of listed salmonids. EPA's "no effect" determinations for methyl parathion applied to the California Coastal Chinook salmon, Central California coho salmon, South Central California steelhead, Southern California steelhead, and Northern California steelhead.

EPA's OPP also determined that the use of naled will have "no effect" for 6 ESUs, "may affect but is not likely to adversely affect" 12 ESUs, and "may affect" 9 ESUs of listed salmonids. EPA's "no effect" determinations for naled applied to the California Coastal Chinook salmon, Central Valley Spring-run Chinook salmon, Sacramento River Winter-run Chinook salmon, Central California coho salmon, Ozette Lake sockeye salmon, and Northern California steelhead.

On July 29, 2004, the EPA sent a letter to NMFS' OPR requesting section 7 consultation for the registration of the a.i. dimethoate and detailing its effects determinations on 19 ESUs of Pacific salmonids listed at that time. In that same letter, EPA's OPP determined that the use of dimethoate will have "no effect" for 6 ESUs; "may affect but is not likely to adversely affect" 5 ESUs, and "may affect" 14 ESUs of listed salmonids. EPA's "no effect" determinations applied to the California Coastal Chinook salmon, Hood Canal Summer-run chum salmon, Central California Coast coho salmon, Ozette Lake sockeye salmon, Northern California steelhead, South-Central California Coast steelhead, and Southern California steelhead.

On June 28, 2005, NMFS listed the Lower Columbia River coho salmon ESU as endangered. Given this recent listing, EPA's 2002, 2003, and 2004 effects determinations for azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, methamidophos, methidathion, methyl parathion, naled, phorate, and phosmet on listed Pacific salmonids lack an effects determination for the Lower Columbia River coho salmon.

On May 22, 2007, NMFS listed the Puget Sound Steelhead Distinct Population Segment (DPS) as threatened. Given this recent listing, EPA's 2002, 2003, and 2004 effects determinations for azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, methamidophos, methidathion, methyl parathion, naled, phorate, and phosmet on listed Pacific salmonids lack an effects determination for the Puget Sound steelhead.

On December 10-12, 2007, EPA and the Services met and discussed approaches for moving forward with ESA consultations and pesticide registrations. The agencies agreed to develop methodologies for filling existing data gaps. In the interim, the Services will develop approaches within their Opinions to address these gaps. 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 ESU as threatened. EPA's 2002, 2003, and 2004 initiation packages for azinphos methyl, dimethoate, ethoprop, methidathion, methyl parathion, naled, and phorate provided an effects determination for the Oregon Coast coho salmon ESU. This ESU was previously listed in 1998 and its ESA status was in flux until 2008.

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

On August 29, 2008, NMFS met with EPA and the applicants for chlorpyrifos, diazinon, and malathion. At that meeting, NMFS asked 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 may 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 will 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 identified applicants by such agency during formal consultation. NMFS also requested incident reports and label information for subsequent pesticide consultations from EPA. The specified timeline for NMFS' receipt of incident report and label information for 12 organophosphate insecticides was April 1, 2009.

On March 24, 2009, NMFS requested information from EPA via e-mail on the 12 a.i.s for this consultation by May 1, 2009. They include any termination, cancellation, or mitigation that may be planned for any of the 12 a.i.s, the most current Science Chapters for these compounds, and a hard copy or pdf files from the OPPIN database for NMFS' assessment of potential use sites and exposure of listed salmonids to those chemicals. NMFS also requested a representative sub-sample of labels showing major product types and/or use sites for chemicals with a large number of registrations. NMFS further requested that applicants provide information they would like considered in the consultation by June 30, 2009.

On March 30, 2009, EPA sent formal correspondence to eight technical registrants for the a.i.s under consultation. EPA's letter requested confirmation on these registrants' desire to have applicant status and for parties to submit data not already provided with EPA's consultations that may inform the outcome of the consultation. That information includes any toxicity data, field studies or mesocosm studies not part of the consultation package, or EPA's Interim Registration Eligibility Decision (IRED) or Registration Eligibility Decision (RED) documents for the pesticide a.i.; and current labels for end use products or if available; and a master label that includes all use instructions for all products containing the a.i.s. These data would be submitted to NMFS and EPA.

On March 31, 2009, EPA provided e-mail correspondence to NMFS identifying EPA's primary staff contact for this consultation and notified NMFS of eight companies who are primary applicants for one or more of the 12 a.i.s. EPA also provided NMFS electronic files of EPA's letters sent to these eight companies. In that same letter, EPA informed NMFS that there are no applicants for the a.i. fenamiphos as registrations containing fenamiphos have been cancelled. EPA also referred NMFS to the IRED, RED, and California red-legged frog documents for any changes to the 12 subject a.i.s since consultation was initiated in 2002 through 2004. Furthermore, EPA will mail incident data on the 12 a.i.s. to NMFS.

On April 1, 2009, NMFS received incident data on the 12 a.is. under consultation.
On April 9, 2009, EPA and NMFS coordinated via e-mail on NMFS receipt of the Special Local Need (SLN)/representative active labels for the 12 a.i.s.

On April 22, 2009, NMFS requested (via e-mail and phone) of EPA tables containing information for the registered uses of the 12 a.i.s under consultation for California, Idaho, Oregon, and Washington by June 1, 2009. They include maximum single application rate, number of applications, interval between applications, and maximum application rate/year. NMFS also provided an example table for the requested information to EPA in that same message.

On April 24, 2009, EPA informed NMFS via e-mail that it had forwarded NMFS' request for use table information to the appropriate chemical teams at EPA in order to meet the June 1, 2009, deadline.

On April 28, 2009, EPA e-mailed NMFS section 3 and 24(c) active label information for 11 a.i.s. EPA did not provide information on fenamiphos as there are no active labels for this a.i. On that same date, EPA and NMFS exchanged e-mail communications on the status of label information for fenamiphos. EPA informed NMFS of a December 10, 2003, Use Deletion and Product

Cancellation Order, published in the Federal Register (FRL-7332-5). This order specified that EPA would grant a request from the chemical's sole registrant to voluntarily cancel all registrations for products containing fenamiphos. The order also stated that the registrant would cease sale and distribution of fenamiphos products by May 31, 2007. Persons other than the registrant were required to halt sale and distribution of products by May 31, 2008. A subsequent order extended this deadline to November 30, 2008, for two fenamiphos products (EPA Reg. Nos. 432-1291 and 264-731).

On April 30, 2009, EPA confirmed via e-mail that fenamiphos can no longer be sold in the U.S. and the only EPA authorized use of fenamiphos is for products that were sold on or before November 30, 2008. For most products, it would include those products that were sold on or before May 31, 2008, and there have been caps put on production of fenamiphos since 2003.

On May 11, 2009, NMFS requested of EPA via e-mail for estimates on the amount of existing stocks of fenamiphos and when those stocks are likely to be exhausted. On that same date, EPA informed NMFS that it would follow up on the query. EPA also informed NMFS that the requested use table of information from NMFS is under development.

On May 13, 2009, EPA provided use tables for the 12 a.i.s under consultation. All use tables (with the exception of the ethoprop use table) are from the California Red-legged frog assessment on each of the chemicals. EPA also stated that it did not track existing stocks of fenamiphos directly and that EPA's existing Stocks Policy generally allows registrants to sell and distribute existing stocks for one year after the cancellation date.

On that same date, NMFS responded to EPA via e-mail stating that the provided use tables were not what NMFS had requested. NMFS informed EPA that defining the action is not as simple as adding the 24(c) labels from California, Idaho, Oregon, and Washington specific information. Information on emergency uses is also lacking. NMFS requested of EPA of an efficient way for identifying restrictions than for NMFS to piece together EPA's action by reviewing the information in the 129 product labels EPA had sent.

On May 26, 2009, NMFS e-mailed EPA and requested greater assistance from EPA in defining its actions for the 12 a.i.s under consultation. NMFS also requested comprehensive tables of registered uses for these same ingredients if EPA is unable to summarize label restrictions for California, Idaho, Oregon, and Washington.

On May 27, 2009, NMFS contacted EPA via phone regarding pesticide use summaries for the four states. EPA informed NMFS that it will send an example Land Use Information System (LUIS) report for NMFS verification on whether this format contained the information NMFS had previously requested.

On May 29, 2009, NMFS e-mailed EPA and requested a reference for EPA's use estimate provided for existing stocks of fenamiphos. On that same date, NMFS requested clarification via e-mail for several 24(c) labels with expiration dates and whether these same labels were granted extensions.

On June 1, 2009, EPA informed NMFS via e-mail that the expired SLN labels for azinphos methyl had been extended until September 30, 2012. EPA requested additional expired SLN labels for follow-up within EPA's Special Registration Review Division.

On June 3, 2009, NMFS e-mailed EPA additional questions associated with phosmet and naled labels regarding their use precautions and expiration dates.

On June 5, 2009, EPA e-mailed NMFS LUIS reports for 11 a.i.s.

On June 9, 2009, EPA provided NMFS via e-mail with its reference for the estimate on existing stocks of fenamiphos. On that same date, NMFS also requested clarification on how EPA derived its estimate of existing stocks of fenamiphos via e-mail. NMFS and EPA further discussed questions on SLN registrations for products containing phorate via phone. Following that discussion, NMFS requested status information on SLN labels in Oregon and Washington (radishes); and in California (lilies and daffodils). NMFS also requested a copy of the Gowan assessment referenced from EPA's methidathion assessment.

On June 10, 2009, EPA and NMFS exchanged several e-mails regarding information on existing stocks of fenamiphos and its cancellation, and several SLN labels. EPA responded to NMFS' fenamiphos and SLN queries and e-mailed the two additional active SLN labels for phorate to NMFS. In that same message, EPA stated that potential future uses for the SLN labels must undergo an assessment process even if they have been mentioned in the RED. EPA further confirmed cancellation of a SLN label for Washington State. EPA requested NMFS contact information for prompt delivery of the Gowan report. NMFS provided its business mailing address on that same day.

On June 11, 2009, NMFS requested the LUIS report for azinphos methyl from EPA via e-mail. On that same date, EPA stated that the azinphos methyl table was misfiled with the bensulide files. EPA provided the missing bensulide table.

On June 12, 2009, NMFS requested clarification of EPA for several section 3 and SLN registrations via e-mail.

On June 15, 2009, EPA provided information to NMFS regarding the registration for methyl parathion in question.

On June 16, 2009, EPA -mailed NMFS current label information for the state of Idaho. On that same date, NMFS requested copies of labels for Nemacur 3 Emulsifiable System Insecticide-Nematicide (EPA Reg. No. 264-731) and Nemacur 10% Turf and Ornamental Nematicide (EPA Reg. No. 432-1291).

On June 17, 2009, EPA provided NMFS with information on the fenamiphos cancellation via e -mail. The fenamiphos cancellation deadline was extended for persons other than the registrant to sell and distribute Nemacur 3 Emulsifiable system Insecticide-Nematicide (EPA Reg. No. 264-731) until March 31, 2009 (from November 2008).

On June 23, 2009, NMFS requested clarification on methamidophos use in cauliflower and for SLN registrations for this same a.i. in Idaho, Oregon, and Washington via e-mail.

On June 24, 2009, EPA confirmed that methamidophos use on cauliflower had been cancelled and there were no SLN registrations for methamidophos in Idaho, Oregon, and Washington.

On June 26, 2009, NMFS provided a list of questions awaiting responses from EPA via e-mail. On that same date, NMFS requested EPA status information on five dimethoate labels cited in the NPIRS site via e-mail.

On June 29, 2009, EPA provided copies of the fenamiphos labels to NMFS via e-mail.

On June 30, 2009, NMFS re-sent its list of questions to EPA via e-mail.

On July 1, 2009, EPA informed NMFS via e –mail of the cancellation of five dimethoate labels and Product 34704-207 specified in the July 20, 2008 Federal Register Notice (70 FR 41714). NMFS requested clarification on an NPRS notice that the state of Oregon and the registrant may opt to submit an application to amend and/or extend two SLN labels in Oregon for another five years. EPA informed NMFS that it is unaware of such applications.

On that same date, NMFS requested clarification on the maximum use rate for naled of EPA via e-mail.

On July 6, 2009, EPA confirmed via e-mail that the maximum single application is 0.1 lb of naled per acre. EPA further stated that it is working with a registrant to revise label information for maximum single application use rate of naled in agricultural areas.

On that same date, NMFS received correspondence from Gowan stating its wish to participate in consultation with EPA and NMFS on the consultation for phosmet, bensulide, and methidathion. Gowan holds technical registrations for these three a.i.s.

On July 8, 2009, NMFS further requested clarification on the naled degradate, dichlorvos (DDVP), and its potential cumulative exposure to listed salmonids from both active ingredients. NMFS requested copies of any labels that permit DDVP use (section 3 or 24(c) to control sea lice in salmon aquaculture.

On July 16, 2009, EPA provided electronic copies of applicant letters for this consultation. Applicants include Makhteshim Chemical Works, Ltd, registrant of products containing the a.i. aiznphos methyl, and Bayer Crop Science, registrant for the a.i. ethoprop. Makhteshim Chemical Works, Ltd identified Makhteshim Agan of North America (MANA) as its representative during the consultation process.

On July 23, 2009, NMFS e-mailed a draft file of the *Description of the Proposed Action* for the 12 a.i.s under consultation to EPA. NMFS requested EPA verification on the contents of this file by August 20, 2009.

On July 28, 2009, EPA informed NMFS via e-mail that there are no active registrations for the use of DDVP on salmonids and aquaculture operations.

On August 19, 2009, EPA notified NMFS via e-mail that EPA's chemical teams' review of the draft *Description of the Proposed Action* for this Opinion will be sent in separate e-mails. EPA's e-mail included comments on azinphos methyl, bensulide, disulfoton, and fenamiphos, and disulfoton.

On August 31, 09, NMFS requested additional information (via e-mail) from EPA on its comments on the draft *Description of the Proposed Action*. NMFS also stated that EPA's electronic comment files for disulfoton and fenamiphos were not attached. On that same date, EPA responded and stated that the disulfoton and fenamiphos comments were embedded in EPA's e-mail text message. In that same e-mail message, EPA stated it would look out for NMFS' follow-up questions.

On September 18, 2009, NMFS and EPA discussed scheduling meetings with the four identified applicants for this consultation. EPA conveyed that a meeting with Cheminova is tentatively scheduled for October 23, 2009, and additional meetings with the remaining applicants have yet to be scheduled.

From September 23, 2009 through November 5, 2009, NMFS staff completed their renewal of CBI status.

On October 1, 2009, EPA e-mailed NMFS and confirmed three separate meetings with the applicants for this consultation. The agencies will meet with Cheminova on October 23, 2009, and with Gowan, MANA, and Bayer on October 28, 2009.

On October 2, 2009, EPA informed NMFS via e-mail of the proposed voluntary cancellation of disulfoton and methamidophos.

On October 14, 2009, EPA informed NMFS via e-mail on its final order approving the voluntary cancellation of disulfoton and methamidophos. For all methamidophos products and most disulfoton products, the cancellations are effective December 31, 2009; two disulfoton products will be canceled effective December 31, 2010. Uses of the disulfoton and methamidophos products canceled by this order may continue until existing stocks are exhausted, provided that use is consistent with approved product labeling.

On October 23, 2009, EPA, NMFS, and Cheminova (applicant for dimethoate and methyl parathion), met and shared information for this consultation. NMFS explained the consultation process and the rights of applicants for this consultation. Cheminova presented chemical specific information for NMFS consideration during its development of the Opinion. NMFS requested feedback from Cheminova on the draft *Description of the Proposed Action* and on nonylphenol (NP) polyethoxylates in dimethoate and methyl parathion formulations.

On October 28, 2009, EPA, NMFS, and Gowan (applicant for bensulide, methidathion, and phosmet), met and shared information for this consultation. On that same date, EPA, NMFS, Bayer, and MANA met and shared information for this consultation. Bayer and MANA are the applicants for ethoprop and azinphos methyl, respectively. At both meetings, NMFS explained the consultation process and the rights of applicants in this process. Gowan, Bayer, and MANA presented chemical specific information for NMFS' consideration during its development of the Opinion.

On November 2, 2009, EPA provided electronic copies of materials from Gowan and MANA to NMFS via e-mail. These documents were specific to bensulide and azinphos methyl.

On November 17, 2009, EPA provided NMFS an electronic copy of a Cheminova report on dimethoate dated July 17, 2009.

On December 7, 2009, EPA provided NMFS an electronic copy of the phosmet hydrolysis study from Gowan.

On that same date, EPA requested information from NMFS (via e-mail) on how it considers changes on pesticide labels in the current consultation. NMFS responded via e-mail and stated that EPA must make an affirmative decision to proceed with a label change before it becomes part of the description of the proposed action in the consultation.

January 4, 2010, NMFS mailed out to EPA computer discs containing electronic files of the salmonid life cycle population models developed by NMFS' Northwest Fishery Science Center.

January 7, 2010, EPA informed NMFS via e-mail that information from Gowan and Cheminova will be FEDEXed to NMFS. This package excludes information and comments from MANA on existing stocks of azinphos methyl.

January 12, 2010, NMFS received incoming information from EPA containing reports and other miscellaneous information from applicants for this consultation. However, the package lacked any comment on NMFS' draft *Description of the Proposed Action* that was requested at the October 28, 2009, meeting.

On that same date, NMFS contacted EPA (via e-mail) and confirmed receipt of information pertaining to the consultation. In that same message, NMFS indicated that it has yet to receive comments from the four applicants on the draft *Description of the Proposed Action*. NMFS also sought clarification from EPA on the information format for documents that were made available to the public and enclosed in the package to NMFS. The documents had attached instructions stating that review of this information requires certification of non-multinational status and CBI clearance.

On January 13, 2010, NMFS requested from EPA for a copy of reports associated with Cramer Fish Sciences presentation provided at the 2008 American Chemical Society meeting.

On January 19, 2010, NMFS provided computer disks containing electronic files on the salmonid population models to EPA.

On January 21-22, 2010, EPA and NMFS communicated via e-mail regarding NMFS receipt of information on dimethoate, methyl parathion, ethoprop, azinphos methyl, and captan. NMFS also requested applicant confirmation or revisions on NMFS' draft *Description of the Proposed Action,* and for electronic copies of received information and non–CBI materials.

On January 28, 2010, EPA and NMFS confirmed (via e-mail) provision and receipt of all materials requested from the applicants to NMFS. EPA sent NMFS computer discs containing electronic copies of documents and information and hard copies of CBI materials. NMFS also requested applicants provide simultaneous delivery of consultation-related materials to both EPA and NMFS. EPA indicated it was receptive to this process.

February 8, 2010, NMFS notified EPA (via e-mail) that EPA's list of documents sent to NMFS may have misclassified two items and did not adequately list a third document. NMFS sent EPA a tracking system file as an example for consideration to ensure accurate records are kept for the documents sent to NMFS.

On March 1, 2010, NMFS notified EPA via e-mail of a court granted extension for this consultation. The new deadline for issuance of this Opinion is August 31, 2010.

On April 13, 2010, NMFS requested via e-mail information on EPA’s decision on the July 22, 2009, proposed amendments for higher application rates for azinphos methyl.

On that same date, NMFS further requested information on several SLN registrations for dimethoate use in Idaho, Oregon, and Washington.

On April 15, 2010, EPA confirmed via e-mail to NMFS that the rate reductions as stipulated in EPA’s 2006 decision document remain unchanged for azinphos methyl. Documentation on this decision will be available soon.

On April 19, 2010, EPA provided clarification (via e-mail) on the dimethoate SLNs for Idaho, Oregon, and Washington.

On April 30, 2010, EPA informed NMFS via e-mail of its receipt of a voluntary cancellation request from registrations for product registrations containing methyl parathion. EPA’s public comment for this voluntary cancellation request closes on May 28, 2010.

On that same date, NMFS requested information on the status of the cancellation request for methyl parathion.

On May 3, 2010, EPA provided clarification (via e-mail) on the registrants’ request to cancel methyl parathion products. In that same message, EPA conveyed that it does not expect methyl parathion to be registered after June. EPA also plans to use the cancellation action as the registration review decision.

On May 28, 2010, NMFS and EPA held the first of three RPA (others held on June 3 and June 10, 2010). Prior to engaging in the RPA meetings, NMFS advised EPA that discussions continuing past the date scheduled for issuance of the Draft Opinion (June 9, 2010) would be considered part of EPA's 60-day review period. NMFS communicated to EPA the intent to suggest elements similar to those in the previous two pesticide Opinions, including a requirement for vegetative filter strips. EPA requested NMFS provide more flexibility than in previous Opinions so EPA could adapt elements of the RPAs based on feedback from stakeholders and still achieve the desired level of protection for the species. During the June 3, 2010 teleconference meeting, NMFS and EPA discussed options for a mutually agreeable approach to defining the elements of the RPA. At the June 10, 2010 teleconference meeting, NMFS presented draft RPAs incorporating the flexibility desired by EPA. The new RPAs allowed EPA greater discretion in selecting which risk reduction measures to implement for the specific a.i.s, provided predicted concentrations of the pesticides in aquatic habitats remained below specified maximum concentration limits for the a.i. EPA indicated the elements were appropriate, but requested additional time for internal review before NMFS issued the draft Opinion.

On June 2, 2010, EPA published final cancellation order for mehidathion in the *Federal Register*.

On June 15, 2010, EPA advised NMFS they were ready to receive the Draft Opinion. EPA stated they intended to obtain input from applicants, affected users and states regarding the draft RPAs. EPA also indicated they were in the process of setting up applicant meetings for NMFS to present findings.

On June 15, 2010, NMFS delivered a copy of the Draft Opinon to EPA electronically, advising EPA the 60-day review period would end on August 9, 2010, and comments would be due on that date.

On June 16, 2010, EPA acknowledged receipt of the electronic copy of the Draft Opinion.

On June 17, 2010 EPA posted the Draft Opinion to their docket, requesting applicant input on the RPAs by July 15, 2010.

Between June 24, 2010, and July 15, 2010, NMFS met with the three applicants EPA had identified for a.i.s addressed in this consultation. On June 24, 2010, NMFS and EPA met with Gowan, applicant for bensulide, nethidathion, and phosmet). On July 1, 2010, NMFS and EPA met with Cheminova, applicant for dimethoate and methyl parathion. On July 15, 2010, NMFS and EPA met jointly with applicants Bayer (disulfoton, ethoprop, and methamidophos), and MANA (azinphos methyl). Applicants generally gave presentations and/or asked questions. None provided written comments at the meetings.

Between July 12, 2010, and August 16, 2010 NMFS received comments regarding the Draft Opinon from applicants.

Only July 16, 2010, EPA published final cancellation order for methyl parathion in the *Federal Register*.

On July 19, 2010, another registrant, AMVAC, whom EPA had not previously identified as an applicant, requested a meeting with NMFS and EPA regarding naled and phorate. The meeting was scheduled for July 27, 2010.

On July 27, 2010, NMFS and EPA met with AMVAC, applicant for naled and phorate. AMVAC provided some written comments and information regarding naled at the meeting.

On August 19, 2010, EPA provided comments to NMFS regarding Draft Opinion.

On August 31, 2010, after addressing comments from EPA and applicants, NMFS issued the Final Biological Opinon on azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, methamidophos, methidathion, methyl parathion, naled, phorate, and phosmet.

Species Addressed in the BEs

EPA's BEs considered the effects of pesticides containing the 12 a.i.s to 26 species of listed Pacific salmonids and their designated critical habitat (EPA 2002, EPA 2003a, EPA 2003b, EPA 2003c, EPA 2003d, EPA 2003e, EPA 2003f, EPA 2004a, EPA 2004b, EPA 2004c, EPA 2004d, EPA 2004e). Two listed species, the Lower Columbia River coho and the Puget Sound steelhead, were not considered in the BEs. Although EPA has determined that its action of registering pesticide products containing some of the a.i.s will have no effect on some endangered or threatened Pacific salmon and steelhead, it also determined that some of the a.i.s may affect certain ESUs/DPSs (Table 1). With regard to methamidophos and phosmet, EPA determined "no effect" or "NLAA" for all of the ESUs/DPSs it evaluated. NMFS does not concur with some of these determinations. When an action agency concludes its action will not affect any listed species or critical habitat, then no section 7 consultation is necessary (USFWS, & NMFS 1998). However, consultation on the proposed action is necessary because EPA concluded for 10 of the a.i.s that the proposed action may affect some listed Pacific anadromous salmonids and their designated critical habitat, and NMFS did not concur with all of EPA's "NLAA" determinations for methamidophos and phosmet. In this Opinion, NMFS will analyze 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 salmonid listed after EPA provided its BEs to NMFS.

Table 1. Summary of EPA conclusions from BEs.

Species	ESU	AZM[3]	Bensulide	Dimethoate	Disulfoton	Ethoprop	Fenamiphos	Methamidophos	Methidathion	Methyl parathion	Naled	Phorate	Phosmet
Chinook	California Coastal	LAA[1]	No Effect	No Effect	No Effect	No Effect	No Effect	No Effect	No Effect	No Effect	No Effect	No Effect	No Effect
	Central Valley Spring - Run	LAA	LAA	NLAA[2]	NLAA	NLAA	NLAA	No Effect	NLAA	NLAA	No Effect	LAA	No Effect
	Lower Columbia River	LAA	LAA	LAA	No Effect	NLAA	NLAA	No Effect	NLAA	LAA	LAA	LAA	NLAA
	Upper Columbia River Spring - run	LAA	LAA	LAA	NLAA	LAA	No Effect	NLAA	LAA	LAA	LAA	LAA	NLAA
	Puget Sound	LAA	LAA	LAA	NLAA	NLAA	No Effect	No Effect	NLAA	NLAA	LAA	LAA	NLAA
	Sacramento River Winter - run	LAA	NLAA	NLAA	NLAA	No Effect	NLAA	No Effect	NLAA	NLAA	No Effect	LAA	No Effect
	Snake River Fall -run	LAA	LAA	LAA	LAA	NLAA	No Effect	No Effect	LAA	LAA	LAA	LAA	NLAA
	Snake River Spring/Summer -run	LAA	LAA	LAA	LAA	NLAA	No Effect	No Effect	LAA	LAA	LAA	LAA	NLAA
	Upper Williamette River	LAA	LAA	LAA	NLAA	LAA	NLAA	No Effect	LAA	LAA	LAA	LAA	NLAA
Chum	Columbia River	LAA	No Effect	NLAA	No Effect	No Effect	No Effect	No Effect	No Effect	NLAA	NLAA	LAA	NLAA
	Hood Canal Summer - run	LAA	No Effect	No Effect	No Effect	No Effect	No Effect	No Effect	No Effect	NLAA	NLAA	NLAA	No Effect
Coho	Central California Coast	LAA	LAA	No Effect	No Effect	No Effect	NLAA	No Effect	No Effect	No Effect	No Effect	No Effect	No Effect
	Lower Columbia River	*Not Evaluated*											
	Southern Oregon and Northern California Coast	LAA	NLAA	LAA	No Effect	NLAA	No Effect	No Effect	NLAA	NLAA	NLAA	NLAA	NLAA

Species	ESU	AZM[3]	Bensulide	Dimethoate	Disulfoton	Ethoprop	Fenamiphos	Methamidophos	Methidathion	Methyl parathion	Naled	Phorate	Phosmet
	Oregon Coast	LAA	No Effect	LAA	No Effect	NLAA	No Effect	No Effect	NLAA	NLAA	NLAA	LAA	No Effect
Sockeye	Ozette Lake	No Effect	No Effect	No Effect	No Effect	No Effect	No Effect	No Effect	No Effect	NLAA	No Effect	NLAA	No Effect
	Snake River	LAA	No Effect	LAA	No Effect	NLAA	No Effect	No Effect	LAA	NLAA	NLAA	LAA	No Effect
Steelhead	Central California Coast	LAA	LAA	NLAA	NLAA	No Effect	NLAA	No Effect	NLAA	NLAA	NLAA	LAA	No Effect
	California Central Valley	LAA	LAA	NLAA	NLAA	NLAA	LAA	No Effect	LAA	NLAA	NLAA	LAA	No Effect
	Lower Columbia River	LAA	LAA	LAA	No Effect	NLAA	NLAA	No Effect	NLAA	NLAA	NLAA	LAA	NLAA
	Middle Columbia River	LAA	LAA	LAA	LAA	LAA	No Effect	NLAA	LAA	LAA	LAA	LAA	NLAA
	Northern California	LAA	No Effect	No Effect	No Effect	No Effect	No Effect	No Effect	No Effect	No Effect	No Effect	No Effect	No Effect
	Puget Sound	*Not Evaluated*											
	Snake River	LAA	LAA	LAA	LAA	LAA	No Effect	No Effect	LAA	LAA	LAA	LAA	NLAA
	South-Central California Coast	LAA	LAA	No Effect	NLAA	NLAA	LAA	No Effect	NLAA	No Effect	NLAA	LAA	No Effect
	Southern California	LAA	LAA	No Effect	LAA	NLAA	NLAA	No Effect	No Effect	No Effect	NLAA	No Effect	No Effect
	Upper Columbia River	LAA	LAA	LAA	LAA	LAA	No Effect	NLAA	LAA	LAA	LAA	LAA	NLAA
	Upper Williamette River	LAA	LAA	LAA	NLAA	LAA	NLAA	No Effect	LAA	LAA	LAA	LAA	NLAA

1- May affect, Likely to Adversely Affect
2- May affect, Not Likely to Adversely Affect
3- Azinphos methyl

Description of the Proposed Action

The Federal Action

The proposed action encompasses EPA's registration of the uses (as described by product labels) of all pesticides containing azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, methamidophos, methidathion, methyl parathion, naled, phorate, or phosmet. 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 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 (reregistrations and special review), 18 (emergency use), or 24(c) Special Local Needs (SLN).

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.s azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, methamidophos, methidathion, methyl parathion, naled, phorate, or phosmet; their 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 12 BEs indicate that the subject a.i.s are labeled for a variety of uses including applications to residential areas, pastures, forested areas, and crop lands (EPA 2002, EPA 2003a, EPA 2003b, EPA 2003c, EPA 2003d, EPA 2003e, EPA 2003f, EPA 2004a, EPA 2004b, EPA 2004c, EPA 2004d, EPA 2004e)

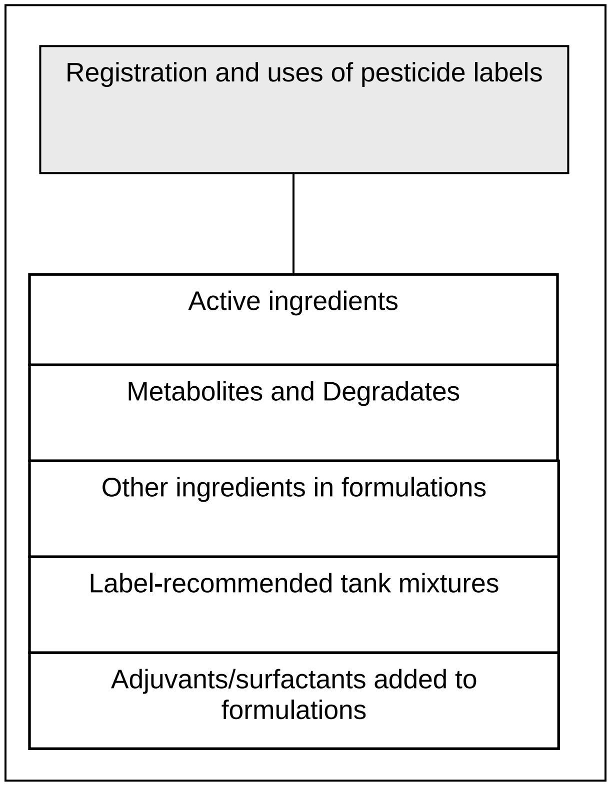


Figure 1. Stressors of the Action

Mode of Action of Organophosphorus (OP) Insecticides. Azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, methamidophos, methidathion, methyl parathion, naled, phorate, and phosmet share the same mechanism of action. They are neurotoxicants to the central and peripheral nervous systems of animals. Azinphos methyl, bensulide, dimethoate, disulfoton, methidathion, methyl parathion, phorate, and phosmet are parent OPs that are metabolized and degraded to toxic oxygen analogues, or oxons. Ethoprop, fenamiphos, methamidophos, and naled do not form oxons. The a.i.s and their oxon metabolites inhibit the enzyme acetylcholinesterase found in brain and muscle tissue of invertebrates and vertebrates. Thus, OPs belong to a class of insecticides known as acetylcholinesterase (AChE) inhibitors. Inhibition of AChE results in a build-up of the neurotransmitter, acetylcholine, which can lead to continued stimulation. Normally, acetylcholine is broken down rapidly in the nerve synapse by AChE. Chemical neurotransmission and communication are impaired when acetylcholine is not quickly degraded in animals which ultimately may result in a number of adverse responses from behavioral modification to death. NMFS batched the consultations on these 12 a.i.s into one Opinion because these compounds all inhibit AChE. Additionally, cumulative exposure to different combinations of the 12 a.i.s and other cholinesterase inhibitors is expected as they have overlapping uses and occur together in surface water samples.

Active and Other Ingredients. Azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, methamidophos, methidathion, methyl parathion, naled, phorate, and phosmet are the a.i.s that kill or otherwise affect targeted organisms (listed on the label). However, pesticide products that contain these a.i.s also contain inert ingredients. Inert ingredients are ingredients which 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 RED documents, EPA's cumulative assessment on organophosphorous compounds, and federal register publications (EPA 2006b, EPA 2006d, EPA 2006e, EPA 2006f, EPA 2006g, EPA 2006h, EPA 2006i, EPA 2006j, EPA 2006k, EPA 2006l, EPA 2006m, EPA 2006n, EPA 2006o).

Registrants can submit applications for the registration of new products and new uses following reregistration of an a.i. 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. The EPA may also cancel product registrations. EPA typically 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 the 12 a.i.s 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 proposed action is also 15 years.

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

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 azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, methamidophos, methidathion, methyl parathion, naled, phorate, or phosmet. EPA provided copies of all active product labels containing these a.i.s. The following descriptions represent information acquired from review of these labels as well as information conveyed in the EPA BEs, REDs, and other documents.

Azinphos methyl

Azinphos methyl was first registered in the U.S. in 1959 for use as an insecticide and has been registered on a variety of crops (EPA 2006d). In 2006, EPA issued a determination that due to farm worker and ecological risks all remaining uses of azinphos methyl will be phased out by 2012 (EPA 2006a, EPA 2006d). The registrants for azinphos methyl submitted registration amendments and voluntary cancellations requests to terminate certain uses in 2007 and 2009, and cancel all remaining registrations in 2012. EPA published a product cancellation order in a February 20, 2008 Federal Register notice (73 FR 9328) that specified all registrations of azinphos methyl will be canceled effective September 30, 2012. The notice also required further limitations including implementation of application rate reductions and buffer zones around water bodies during the phase out period. There are currently no residential or public health uses of azinphos methyl (EPA 2006d). Active labels for azinphos methyl include EPA Reg. No. 10163-78, 66330-233, and 66222-11 (formerly EPA Reg. No. 264-733). Additionally, there are two SLN registrations in California, Idaho, Oregon, and Washington (EPA Reg. No. OR-040020 and WA-030025). There are no emergency use registrations (section 18) for azinphos methyl in California, Idaho, Oregon, or Washington.

Usage Information.

Azinphos methyl usage averaged 2.2 million lbs per year in the U.S. from 1987 through 1997, with approximately 41% applied to apples (EPA 2003a). Agriculture use of azinphos-methyl in California declined from over 300,000 lbs in 1997 to a reported 25,000 lbs in 2007 (CDPR 2008b). Recent usage information for Washington, Oregon, and Idaho is not available. EPA indicated 90% of azinphos methyl used in Washington is primarily used on apples (EPA 2003a). The maximum use on apples in Washington between 1990 and 2001 was 474,400 lb in 1995. In Oregon, azinphos methyl is used mostly on pears and apples, with some use on sweet cherries, pears, and blueberries. Potato is the only crop on which usage was reported in Idaho, where an average of 6% of the crop was treated with azinphos-methyl during this period (EPA 2003a). Azinphos methyl is no longer approved for use in potato. According to MANA, one of the registrants of the azinphos methyl products, in 2008 Washington, Oregon and Idaho accounted for 55% of the total usage of azinphos methyl in the United States. Additionally, 78 % of the total usage was on apples (Gur 2009).

Agricultural Uses. Azinphos methyl may be applied to apples, cherries, pears, and border treatments around alkali bee beds.

Non-agricultural Uses. Non-agricultural uses of azinphos methyl are not permitted.

Registered Formulation Types. Azinphos methyl products are formulated as either an emulsifiable concentrate or as a wettable powder in water soluble bags (EPA 2003a). Active labels for azinphos methyl products do not include any other a.i.

Methods and Rates of Application.

Methods. Azinphos methyl can be applied by ground application methods. Aerial applications of azinphos methyl are not permitted in California, Idaho, Oregon, or Washington. Labeling indicates azinphos methyl can be applied in tank mixes and is compatible with summer oils, many registered pesticides, and liquid fertilizers. There are crop-specific requirements for "no spray" buffers and vegetative filter-strip to reduce drift and runoff of pesticide ingredients to aquatic habitats (EPA Reg. No. 10163-78, 66222-11, 264-733, and 66330-233).

Application Rates. The maximum single application rate for crop sites is 3 lbs azinphos methyl/acre with up to 6 lbs a.i./acre annually (Table 2). Multiple applications are allowed on all approved use sites. The minimum application interval is not specified for several crops. Active labels allow for a maximum single application rate of up to more than 16 lbs azinphos methyl/acre for in-ditch applications to control beetles around alkali bee beds. Broadcast applications can be made at a rate of 10 lbs a.i./acre. Up to 10 applications are allowed around bee beds at a recommended interval of 4 - 5 days.

Table 2. Summary of all authorized use sites and application restrictions for active azinphos methyl products registered in California, Idaho, Oregon, and Washington.

Use(s)	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
Apples and crabapples[5]	2[1]	2	4[1]	7	Ground boom
Cherries[5]	0.75[2]	2	1.5[2]	14	Air blast and other ground applications
Pears[5]	2[3]	2[3]	4[3]	7	Air blast and other

					ground applications
24(c) WA: Border treatment around alkali bee beds[5]	16.3 in ditch, 10 broadcast[4]	10	NS	4 - 5	Ground spray in ditch or broadcast
24(c) OR: Border treatment around alkali bee beds[5]	16.3 in ditch, 10 broadcast[4]	NS	NS	4 - 5	Ground spray in ditch or broadcast
NS = not specified 1 – Maximum allowed single and annual application rate in 2010 is 2 and 4 lbs a.i./A, respectively; in 2011 and 2012 maximum single and annual application rate restricted to 1.5 and 3 lbs a.i./A, respectively. 2 - In 2010-2012 maximum single and annual application rate restricted to 0.75 and 1.5 lbs a.i./A, respectively. 3 – In 2010 maximum single and annual application rate restricted to 2 and 4 lbs a.i./A, respectively; in 2011-2012 maximum single and annual application rate restricted to 1.5 and 3 lbs a.i./A, respectively. 4 – broadcast application permitted if ditch application is not feasible. 5- product use prohibited after September 30, 2012.					

Metabolites and Degradates.

Azinphos methyl has been documented to degrade to its oxygen analog (azinphos methyl oxon) in drinking water and in aerobic soil metabolism and aqueous photolysis studies (EPA 2007d).

Additionally, several other degradates have been identified including, anthranilic acid, methyl anthranilate, mercaptomethyl benzazimide, hydroxymethyl benzazimide, benzazamide, and *bis*-methyl benzazamide sulfide, and methyl benzazimide sulfonic acid (EPA 2007d).

Bensulide

Bensulide is an organophosphate herbicide first registered in 1964 for pre-emergence control of crabgrass and annual bluegrass in turf. It has been registered for use in a variety of food crops since 1968 (EPA 2002). In plants, bensulide's mode of action is through the inhibition of cell division in the roots and shoots. It is applied directly to the soil and has no foliar activity. Bensulide must be incorporated into the soil by cultivation if applied at or before planting time or watered in through irrigation if applied after planting (EPA 2002). It may be applied through irrigation systems in California, but not in Oregon, Washington, or Idaho. Aerial application is prohibited (EPA 2002). There are currently 15 active labels for end use products containing bensulide (EPA Reg. No. 10163-196, 10163-198, 10163-199, 10163-200, 10163-204, 10163-205, 10163-222, 2217-696, 2217-778, 2217-838, 538-26, 538-155, 538-164, 9198-172, and 9198-176). Additionally, there is one SLN registration (CA-960003). There are no section 18 registrations for use of bensulide products in California, Idaho, Oregon, or Washington.

Usage Information.

EPA estimates approximately 700,000 lbs a.i. of bensulide are applied in the U.S. per year based on data from 1996 (EPA 2007e). California use reports indicate bensulide use has increased by 250 - 300% in recent years (CDPR 2008b). In Oregon, Washington, and Idaho, information on the actual amount of bensulide was not reported. EPA provided census information on crops to estimate bensulide use in these three states (EPA 2002). For the Northwest, EPA's Qualitative Usage Analysis (QUA) evaluated sugar beets in Oregon with average annual use of 5,000 lbs, "other crops" in Oregon with average annual use of 1,000 lbs, and onions in Idaho, Oregon, and Texas with average annual use of 99,000 lbs (EPA 2002). More recent analysis by EPA suggests the greatest use of bensulide is on lettuce where approximately 300,000 lbs are applied annually in the U.S. Other crops receiving relatively high use of bensulide include onions, broccoli, and cantaloupe (70,000 lbs/year). EPA estimates 10,000 lbs of bensulide are applied annually to sod (EPA 2007e).

Agricultural Uses. Bensulide is used for pre-emergent control of annual grasses and broadleaf weeds in agricultural crops (60 - 65% of all use). Current registered uses include: leafy vegetables (mostly head lettuce), dry bulb vegetables (onions), cucurbits (mostly melons), and cole crops (cauliflower, cabbage, broccoli, broccolini, broccoflower). It is also used on field grown herbaceous plants and field grown bulbs (EPA 2002).

Non-agricultural Uses. Non-agricultural uses of bensulide includes application on residential lawns and use on golf course turf. Use is also allowed on field grown (commercial) ornamental herbaceous plants and bulbs (EPA 2002). Although not listed as an approved use site on active labels, California pesticide use reports also indicate bensulide is used for maintenance of right-of-ways and for structural pest control (CDPR 2008b).

Registered Formulation Types. Bensulide products are available in liquid and granular formulations. Two granular products also contain a second a.i., oxadiazon, (538-164 and 9198-176). Oxadiazon is a oxidiazole herbicide registered by EPA for use on a variety of turf and ornamental use sites. Many of the bensulide formulations also contain petroleum distillates (EPA Reg.No.1063-196, 1063-200, 1063-205, and 2217-696). Home and garden formulations of bensulide also contain fertilizers.

Methods and Rates of Application.

Methods. Bensulide may only be applied by ground application methods such as spray boom, tractor-drawn spreaders, in irrigation water, and by homeowner push-spreader and garden hose and other hand-held equipment. It must be soil incorporated or "watered-in" to get below the soil surface (EPA 2002). Two active bensulide labels recommend tank mixing the bensulide product with another herbicide, ALANAP, for a broader spectrum of weed control (EPA Reg. No. 10163-200 and 10163-202). ALANAP contains the a.i. naptalam and is recommended for use in combination with bensulide products in cantaloupes, cucumbers, musk melons, and watermelons.

Application Rates. Active labels allow a maximum single application rate of up to 16 lbs bensulide/acre to golf courses (Table 3). Agricultural applications are limited to one per season. Multiple applications are allowed to non-agricultural sites.

Table 3. Summary of all authorized use sites and application restrictions for active bensulide products registered in California, Idaho, Oregon, and Washington.

Use(s)	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
Cucurbit Vegetable Group	6	1[1]	6	-	Ground, Chemigation[2]
Brassica (Cole) Leafy Vegetable Group	6	1	6	-	Ground, Chemigation
Leafy Vegetable Group (not inc.	6	1	6	-	Ground, Chemigation

Use(s)	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
Brassica)					
Fruiting Vegetables	6	1	6	-	Ground, Chemigation
Dry Bulb Vegetables	6	1	6	-	Ground, Chemigation
Cantaloupes, Cucumbers, Muskmelons, Watermelons[3]	6	1	6	-	Ground, Chemigation
Field grown Flowers, Bulbs, and Ornamentals	9	NS[4]	NS	-	Ground, Chemigation
Turf Grass	12.6	3	25	4 months	Ground, Chemigation
Residential Lawn	12.6	2 or more	25	4 months	Ground, Chemigation
Golf Course	12.5 16[5]	2	NS 32[5]	4 months NS[6]	Ground, Chemigation
24(c) CA: Cucurbits	9	NS	NS	NS	Ground

- Not Applicable
1. Pre-plant or pre-emergence only
2. Chemigation is only permitted in CA for all use sites
3. This group is registered for application of a tank mix with ALANAP
4. NS = not specified
5. These rates are for a product that also includes oxadiazon (single = 4 lbs a.i., annual = 8 lbs a.i.)
6. Not specified for 2 normal applications; 6 weeks between a normal and a half-strength application

Metabolites and Degradates.

Bensulide has two major degradates, bensulide oxon and benzenesulphonamide. These are products of aerobic soil metabolism and range from mobile to highly mobile in laboratory soil tests (EPA 2006e). Other degradate products of bensulide were not identified (EPA 2002).

Dimethoate

Dimethoate is classified as a general use pesticide. It is a systemic organophosphate used on a large variety of field grown agricultural crops, tree crops, and ornamentals. It was first registered in the U.S. in 1962. All non-agricultural uses, including residential uses, were cancelled in 2000 (EPA 2006f). In addition, seven crops that were identified as significant dietary risk contributors (apples, broccoli raab, cabbage, collards, grapes, head lettuce, and spinach), along with four crops for which there were no field trial data to support tolerances (fennel, lespedeza, tomatillo, and trefoil) were canceled in 2005 (Federal Register Notice/Vol. 70, No. 138/Wednesday, July 20, 2005/Notices/41714). Ten companies currently hold active registrations for 18 products containing dimethoate including four technical products (EPA Reg. No. 4787-7, 19713-209, 19713-525 and 66330-271), one manufacturing use concentrate (EPA Reg. No. 7969-32), and 13 end – use products (EPA Reg. No. 769-948, 5905-497, 9779-273, 10163-56, 19713-231, 19719-232, 34704-207, 34704-489, 66330-223, 66330-237, 66330-244, 66330-245, and 67760-44). There are six SLN registrations in California, Idaho, Oregon, and Washington (EPA Reg. No. CA-97003, ID-980006, ID-970011,OR-050019, OR-050020, and WA-970029). There are no emergency use registrations for dimethoate in California, Idaho, Oregon, or Washington.

Usage Information.

According to EPA, about 1.8 million lbs of dimethoate are used annually, with the largest use occurring on alfalfa (EPA 2006f). Cheminova, the primary technical registrant of dimethoate, estimates the current use of dimethoate is significantly less based on sales data (Whatling 2009). Use on alfalfa, wheat, cotton, and corn accounts for more than 64% of total dimethoate use (EPA 2006f). Agriculture use of dimethoate in California has generally declined over the last decade from 516,000 lbs in 1997 to a reported 314,000 lbs in 2007(CDPR 2008b). EPA provided estimates of dimethoate use within the distribution of listed salmon that occur within Idaho, Oregon, and Washington (EPA 2004a). The use estimates were based on USDA crop census data for 1997 or 2002.

Agricultural Uses. Dimethoate may be applied on alfalfa, grass, a variety of row crops, fruit and nut trees, conifer seed orchards, ornamentals, and cropland areas adjacent to vineyards.

Non-agricultural Uses. Non-agricultural uses of dimethoate are not permitted (EPA 2006f).

Registered Formulation Types. End use products containing dimethoate are formulated in wettable powders and emulsifiable concentrates (EPA 2004a). Several formulations contain xylene range aromatic solvents (EPA Reg. No. 66330-244, 66330-245, 19713-232) or petroleum distillates (EPA Reg. No. 9779-273).

Methods and Rates of Application.

Methods. Dimethoate is applied by aircraft, ground spray, and backpack sprayers (EPA 2004a). Several labels indicate dimethoate is compatible with other registered pesticides in tank mix applications (EPA Reg. No. 19713-232, 66330-226). Some labels make specific recommendations for tank mixtures. For example, one product label indicates dimethoate is compatible with several other cholinesterase-inhibiting insecticides including azinphos methyl, malathion, parathion, carbaryl, and diazinon; other neurotoxic insecticides (pyrethriods and dicofo, an organochlorine insecticide); and the fungicides captan, thiram, zineb, and dodine (EPA Reg. No.19713-231). Another label indicates dimethoate provides a compatible tank mix with endosulfan, malathion, and parathion for control of cabbage worm and cabbage loopers in vegetable crops (EPA Reg. No. 9779-273). Several active labels provide recommendations for vegetative filter strips to reduce pesticide runoff and require specific drift reduction methods (EPA Reg. No. 9779-273, 10163-56, 19713-232, 66330-226).

Application Rates. Active labels allow for a maximum single application rate of up to 4.15 lbs of dimethoate/acre and a maximum annual application rate of up to 6 lbs a.i./acre. Most field crops allow ≤ 0.5 lbs a.i./acre for a single application and ≤ 1.5 lbs a.i./acre annually(Table 4.

Table 4. Summary of all authorized use sites and application restrictions for active dimethoate products registered in California, Idaho, Oregon, and Washington.

Use(s)	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
Alfalfa (incl. for seed)[5]	0.5	NS	0.5[2]	NA	Aerial, ground, chemigation
Asparagus (not CA)	0.5	NS	1	14	Aerial, ground, chemigation

Use(s)	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
Beans	0.5	2	1[3]	14	Aerial, ground, chemigation
Broccoli/Cauliflower	0.5	NS	1.5	7	Aerial, ground, chemigation
Brussels sprouts	0.5	NS	1.5	7	Aerial, ground, chemigation
Celery	0.5	NS	1.5	7	Aerial, ground, chemigation
Cherries – preharvest (OR, WA, ID)	1.33	1	1.33	NS	Aerial, ground, chemigation
Cherries – postharvest (OR, WA, ID)	1.33	1	1.33	NS	Aerial, ground, chemigation
Citrus	1	1	1	NA	Ground - Foliar spray, soil drench
Conifer seed orchard	1	NS	1	NA	Aerial, ground, chemigation
Cotton	0.5	NS	1[3]	14	Aerial, ground, chemigation
Cottonwood, poplar for pulp (WA, OR)	2	NS	6	NS	Aerial, ground, chemigation
Douglas fir orchard for seed (WA, OR)	4.15	NS	4.15	NS	Airblast application
Field corn, popcorn	0.5	NS	0.5	NA	Aerial, ground, chemigation
Garbanzo beans	0.5	NS	1	14	Aerial, ground, chemigation
Grass (incl. for seed)	0.5	NS	1	90	Aerial, ground, chemigation
Herbaceous ornamentals	0.25	NS	0.25	NA	Aerial, ground, chemigation
Kale	0.25	NS	0.5	15	Aerial, ground, chemigation
Lentils	0.5	NS	1	7	Aerial, ground, chemigation
Leaf lettuce	0.25	NS	0.75	7	Aerial, ground, chemigation
Lupine	0.5	2[3]	1	14	Aerial, ground, chemigation
Melon (not water)	0.5	NS	1	7	Aerial, ground, chemigation
Mustard greens	0.25	NS	0.5	9	Aerial, ground, chemigation
Pears	1	1	1	NA	Aerial, ground, chemigation
Peas	0.16	1	0.16	NA	
Peas – succulent (WA, OR, ID)	0.33	NS	0.5[3]	7	Aerial, ground
Peas – succulent	0.16	3	0.5[3]	14	Aerial, ground,

Use(s)	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
(CA)					chemigation
Pecans	0.33	1	0.33	NA	Aerial, ground, chemigation
Peppers	0.33	NS	1.65	7	Aerial, ground, chemigation
Potatoes	0.5	NS	1	7	Aerial, ground, chemigation
Safflower	0.5	NS	0.5	14	Aerial, ground, chemigation
Sorghum	0.5	NS	1	7	Aerial, ground, chemigation
Soybean	0.5	NS	1	7	Aerial, ground, chemigation
Swiss chard, endive	0.25	NS	0.75	7	Aerial, ground, chemigation
Tomatoes	0.5	NS	1	6	Aerial, ground, chemigation
Turnips	0.25	NS	1.75	3	Aerial, ground, chemigation
Watermelon	0.5	NS	1	7	Aerial, ground, chemigation
Wheat	0.5	NS	0.5	NS	Aerial, ground, chemigation
Woody Ornamentals and Christmas trees nurseries	1	NS	3	14	Aerial, ground, chemigation
24(c) OR: Meadowfoam[4]	0.5	NS	NS	NS	Aerial, ground
24(c) CA: Non-cropland areas adjacent to vineyards	2.0	2	4.0	NS	Ground-rig hand application
24(c) OR: Peas	0.32	NS	0.5	7	Aerial, ground, chemigation
24(c) WA: Peas	0.33	3	0.5	7	Aerial, ground, chemigation
24(c) ID: Peas	0.5	NS	0.5	7	Aerial, ground

1. NS = not specified
2. Per crop cycle or cutting
3. Per season
4. Will expire on December 31, 2009
5. The technical registrants of dimethoate have requested that EPA require all end-use registrants to amend their labels to limit the number of applications in alfalfa to 3/year, one application per cutting, and a maximum annual use of 1.5 lbs a.i./A/year (Whatling 2009). Additionally, the registrant indicates only one application would occur in alfalfa seed crops because they are only grown once per season (Cheminova 2010).

Metabolites and Degradates.

Dimethoate degrades to dimethoxon which is also called omethoate. Omethoate is an oxygen analog of dimethoate. Other known degrades of dimethoate include dimethyldimethoate and dimethylthiophosphoric acid.

Disulfoton

Disulfoton was first registered in 1961 for use as an insecticide and has been authorized for use on a variety of agricultural crops and domestic outdoor uses on potted plants and ornamentals, including herbaceous plants, flowers, woody shrubs, and trees (EPA 2006g). During the public participation process for the reregistration of disulfoton, Bayer Corporation, the technical registrant, proposed several changes to their disulfoton registrations that were accepted by EPA as interim risk mitigation measures (EPA 2006g). These changes included use deletions, voluntary cancellations, rate reductions, and reduction in the number of applications of disulfoton allowed per year (EPA 2006g). In addition, various disulfoton end-use registrants voluntarily canceled products and/or deleted uses that were no longer supported by Bayer. There are five active labels for end use products containing disulfoton (EPA Reg. No. 264-723, 264-734, 432-1286, 5481-8989, and 72155-49) and seven SLN registrations in California, Idaho, Oregon, and Washington (EPA Reg. No. CA-050010, CA-810044, CA-840192, OR-040030, OR-050024, WA-040015, and WA-920026). There are no emergency use registrations for disulfoton in California, Idaho, Oregon, or Washington. On September 23, 2009 EPA issued a cancellation order for all remaining disulfoton products registered in the U.S. in response to an earlier request for voluntary cancellations by product registrants (74 FR 48551). Under the terms of the order, the effective cancellation date for EPA Reg. Nos. 264-723, 264-734, 5481-8989, and 432-1286 was December 31, 2009. Cancellations for EPA Reg. Nos. 264—725 and 72155-49 are effective December 31, 2010. EPA authorized product registrants to sell and distribute Reg. Nos. 264-725, 264-734, 432-1286, and 5481-8989 until December 31, 2010. EPA Reg. Nos. 264-723 and 72155-49 may be sold and distributed until June 30, 2011. Persons other than the registrants may sell and distribute existing stocks of these products until they are exhausted. Use of these canceled products may continue until the existing stocks are exhausted.

Usage Information

Disulfoton usage ranged from 1.2 – 1.8 million lbs per year in the U.S. from 1987 through 1997 with the largest use occurring in the Southeast and Northwest (EPA 2003b). The major crops in the Northwest and California for disulfoton use are asparagus, broccoli, peppers, barley, potatoes, and wheat (EPA 2003b). However, disulfoton uses have since been voluntarily canceled on peppers, barley, potatoes, and wheat. Recent data from California indicate agriculture use of disulfoton has declined from over 100,000 lbs to approximately 24,000 lbs during the last decade (CDPR 2008b). Recent usage information for Washington, Oregon, and Idaho is not available. EPA estimates from crops census data suggest 50,000 – 100,000 lbs of disulfoton may be applied in Oregon and Idaho. The BE indicates that useage in Washington may exceed 800,000 lbs annually (EPA 2003b). However, this is presumed to be an overestimate of current use because it assumes application on several crops (*e.g.*, barley and potato) that are not represented on active pesticide labels.

Agricultural Uses. Disulfoton is approved for a variety of food (*e.g.*, asparagus, beans, broccoli, Brussels sprouts, cabbage, cauliflower, cotton, lettuce, pepper, radish, clover) and non-food crops (radish and clover seed crops, Christmas trees) (EPA 2006g).

Non-agricultural Uses. Active labels allow disulfoton use on home gardens, ornamental gardens or parks, ornamental shrubs, bedding plants, interior plantscapes, and home greenhouses (*e.g.*, EPA Reg. No. 432-1286) (EPA 2006g).

Registered Formulation Types. Disulfoton end use products include granular and emulsifiable concentrate formulations. Disulfoton is a common component of multiple active ingredient formulations, where it provides the insecticidal elements of products marketed as fungicides and/or fertilizer. The most common a.i.s added are the fungicides pentachloronitrobenzene (PCNB) and etridiazole (EPA 2003b). There is one active label that contains disulfoton, PCNB, and etridiazole (EPA Reg. No. 400-408); Another product is formulated with disulfoton and fertilizers (EPA Reg. No. 432-1286); and a third contains disulfoton and petroleum distillates (EPA Reg. No. 264-734).

Methods and Rates of Application.

Methods. Aerial applications of disulfoton are prohibited in all crops except asparagus. Ground application methods include broadcast, soil incorporation, and foliar treatments. Disultofon is applied as a seed treatment, pre-plant, or post-emergence (EPA 2003b). Most of the active labels do not specify products that disulfoton may be tank mixed with. One label indicates that many registered pesticides and liquid fertilizers provide compatible tank mixtures (EPA Reg. No. 264-734). Products registered for agricultural uses indicate that "a well maintained 25 ft vegetative buffer strip between areas to which this product is applied and permanent surface water features" will reduce the potential for contamination of surface waters (EPA Reg. No. 264-723, 264-734, and 400-408). Products registered for outdoor residential uses do not require a vegetative buffer strips (EPA Reg. No. 432-1286 and 72155-49).

Application Rates. Active labels allow for a maximum single application rate of 1 - 2 lbs disulfoton/acre in most agricultural crops (Table 5). However, up to 4.5 lbs disulfoton/acre can be applied in Christmas trees and approximately 9 lbs/acre are allowed in flower beds and bedding plants. Disulfoton is limited to one or two applications in most agricultural crops. Labels do not specify limits on the number of applications, maximum seasonal rate, or application intervals for residential use on ornamental flowers, roses, shrubs, and trees (EPA 2006g).

Table 5. Summary of all authorized use sites and application restrictions for active disulfoton products registered in California, Idaho, Oregon, and Washington.

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)[1]	App. Interval (days)	App. Method
Azaleas, camellias, rhododendrons	0.0025 lb. a.i./foot plant height	NS	NS	42	Ground - apply granules to soil around plant
Beans	1	1	1	NA	Ground, chemigation
Broccoli (CA)	1	1	1	NA	Soil (shank) injection only
Brussels sprouts	1	1	1	NA	Soil (shank) injection only
Cabbage	2	1	2	NA	Ground
Cauliflower	1	1	1	NA	Ground – soil

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)[1]	App. Interval (days)	App. Method
					(shank) injection only
Cotton – unspecified	1 (0.0075 lb a.i./1000 ft of row)	NS	NS	NS	Ground – spray in furrow, soil injection on side of furrow
Cotton-drill planting	0.975 (2.2[2])	NS	NS	NS	Ground – soil in-furrow
Cotton – hill-drop planting	0.325 (0.735[2])	NS	NS	NS	Ground – soil in-furrow
Easter lilies	NS	1	NS	NS	Ground, chemigation
Fir Christmas trees	4.5	1	4.5	NA	Ground - broadcast
Flowers, bulbs, and bedding plants	8.7 (0.02 lb a.i./100 sq ft)	NS	NS	42	Ground – hand application to soil
Flower beds	9.1 (0.0025 lb. a.i./12 sq ft.)	NS	NS	42	Ground - apply granules to soil
Lettuce	2	1[3]	NS	NA	Ground, chemigation
Ornamental shrub	0.0025 lb. a.i./foot shrub height	NS	NS	42	Add to soil – work in
Roses	0.0013 lb a.i./plant	NS	NS	42	Ground - apply granules to soil around plant
24(c) CA, OR[4], WA: Asparagus	1	2	2	NS	Aerial and Ground (not chemigation)
24(c) CA, OR[5]: Easter lilies	(0.069 lb a.i./100 ft of row)	1	NS	NA	Ground (not chemigation)
24(c) CA: Lettuce	2	1	2	NA	Ground – sprinkler irrigation, soil in-furrow treatment
24(c) WA[6]: Radish for seed	2	1	2	NA	Ground – soil injection

1. NS = not specified
2. All three a.i.s considered
3. Applications per crop season
4. SLN label valid until December 31, 2012
5. Only within Curry County, OR
6. Only for members of the Columbian Basin Vegetable Seed Association

Metabolites and Degradates

Known toxic degradates of disulfoton include disulfoton sulfoxide and disulfoton sulfone. Disulfoton, sulfoxide, and sulfone also form oxygen analogs which are presumed to be toxic.

Ethoprop

Ethoprop is an insecticide and nematicide first registered in the U.S. in 1967 (EPA 2006h). Ethoprop is a restricted use pesticide that has been used on a variety of sites, including agricultural crops, field grown ornamentals, and golf course turf. Golf course turf applications were canceled in 2002 and there are no registered home and garden uses (EPA 2003c). There are three active labels for end-use products containing ethoprop (EPA Reg. No. 264-457, 264-458, and 264-459) and five SLN registrations in California, Idaho, Oregon, and Washington (EPA Reg. No. CA-0600007, OR-060010, OR-060024, OR-070021, and OR-090003). There are no emergency use registrations (section 18) for ethoprop in California, Idaho, Oregon, or Washington.

Usage Information.

EPA reported the average annual domestic use of ethoprop was 691,000 lbs for the period 1987 to 1996 (EPA 2003c). EPA also reported that the main usage areas are in the Southeast and the Pacific Northwest. In the Northwest and California, ethoprop is most commonly used on potatoes, corn, and sweet potatoes (EPA 2003c). Agriculture use of ethoprop in California has remained relatively stable over the last decade ranging from 16,000 – 28,000 lbs/year. Approximately 24,000 lbs were applied in 2007, the most recent data available (CDPR 2008b). EPA estimates for use of ethoprop suggest more than 100,000 lbs of ethoprop may be applied annually in each of the three Northwestern states (EPA 2003c).

Agricultural Uses. Ethoprop is approved for use on a variety of food crops and field nursery stock for ornamental plants (EPA 2006h).

Non-agricultural Uses. Non-agricultural uses of ethoprop are not permitted (EPA 2006h).

Registered Formulation Types. End use products containing ethoprop are available as granular formulations and emulsifiable concentrate sprays (EPA 2003c). Emulsifiable concentrates contain petroleum distillates (EPA Reg. No. 264-458).

Methods and Rates of Application.

Methods. Ethoprop may be applied only by direct ground application to the soil and current usage restrictions require immediate soil incorporation by mechanical methods or by watering in at the time of application (EPA 2003c). The active label for liquid formulation specifies the product cannot be applied within 140 feet of inland freshwater habitats (EPA Reg. No. 264-458). However, the granular formulations do not require setbacks from these habitats (EPA Reg. No. 264-457 and 264-469). Other required drift reduction measures for the liquid formulation include a maximum nozzle height of 4 ft, a wind speed restriction of 10 mph or less, and use of droplet size distribution that is greater than or equal to a "medium" on the American Society of Agricultural Engineers (ASAE) droplet spectrum.

Application Rates. Active labels allow for a maximum single application rate of up to 12 lbs of ethoprop/acre (Table 6). Multiple applications of ethoprop are not permitted, except for bananas and plantains which can be retreated every six months. Otherwise ethoprop applications are limited to once per season (EPA 2006h).

Table 6. Summary of all authorized use sites and application restrictions for active ethoprop products registered in California, Idaho, Oregon, and Washington.

Use(s)	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
Bananas	0.0132 per stem	2	NS	6 months	Ground
Beans	8.1	1 per season	NS	NS[1]	Ground
Cabbage	5.1	1 per season	NS	NS	Ground
Corn	6	1 per season	NS	NS	Ground

Use(s)	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
Cucumber	1.95	1 per season	NS	NS	Ground
Hops	3	1 per season	NS	NS	Ground
Mint	6	1 per season	NS	NS	Ground
Plantains	0.0132 per stem	2	NS	6 months	Ground
Potatoes	12	1 per season	NS	NS	Ground
Sugarcane	0.56[2]	1 per season	NS	NS	Ground
Sweet Potatoes	3.9	1 per season	NS	NS	Ground
Ornamentals (CA, OR, WA)	3	1 per season	NS	NS	Ground
Tobacco	6	1 per season	NS	NS	Ground
24(c) OR: Sugar Beets grown for seed[3]	3	1 per season	NS	NS	Ground
24(c) OR: Easter Lilies[4]	6	1 per season	NS	NS	Ground
1. NS = not specified 2. Banded application per 1,000 row feet 3. West of the Cascade Mountains only 4. Only in Curry County (within SONCC Coho ESU)					

Metabolites and Degradates.

Ethoprop degradates identified by EPA include S,S-dipropylphosphorodithioate, O-ethyl-S-methyl-S-propylphosphorodithioate, O-ethyl-O-methyl-S-propylphosphorodithioate, and O-ethyl-S-propylphosphorodithioate (EPA 2003c).

Fenamiphos

Fenamiphos is a restricted use pesticide first registered in the U.S. in 1972 for use on a variety of agricultural crops (EPA 2006i). A December 10, 2003, Use Deletion and Product Cancellation Order, published in the Federal Register (73 FR 21942), specified that the EPA would grant a request from the chemical's sole registrant to voluntarily cancel all registrations for products containing fenamiphos. The Order provided that the registrant would cease sale and distribution of fenamiphos products by May 31, 2007. Persons other than the registrant were required to halt sale and distribution of products by

May 31, 2008. A subsequent order extended this deadline to November 30, 2008 for two fenamiphos products (EPA Reg. No. 432-1291 and 264-731). There was an additional amendment to the fenamiphos cancellation that extended the deadline for persons other than the registrant to sell and distribute fenamiphos until March 31, 2009 (73 FR 75097). Currently, there are no active labels for fenamiphos products. There is one SLN registration for California, Idaho, Oregon, and Washington (EPA Reg. No. WA 090006). This registration permits use of fenamiphos on iris and narcissus bulbs in Washington State. It expires on December 31, 2010. Although there are no active labels for fenamiphos products, according to the terms of the cancellation order, use of existing stocks of fenamiphos products are permitted until they are fully depleted (73 FR 21942). Historically, about 780,000 lbs of fenamiphos was applied annually in the U.S. to a variety of field and orchard crops (EPA 2006h). Use of fenamiphos has declined substantially over the last decade given product cancellations and required caps on sales and production. The most recent data available show that 39,677 lbs of fenamiphos were used on agricultural sites in California during 2007, primarily on grapes (CDPR 2008b). Future use of fenamiphos is expected to be minimal given EPA estimates suggesting less than 25,000 lbs of fenamiphos is available in existing stocks for future applications (EPA 2009a).

Methamidophos

Methamidophos is an organophosphate insecticide, first registered in the U.S. in 1972 for agricultural crops (EPA 2004b). Methamidophos is also a degradate of acephate, another organophosphate pesticide registered by EPA (EPA 2004b). Acephate is used on a number of food crops, and is also approved for residential, public health, and other non-agricultural uses (golf course turf, field borders, fence rows, roadsides, horticultural nursery floral and foliage plants) (EPA 2006c). In 1997, all methamidophos uses were cancelled except for applications to cotton, potatoes, and tomatoes. On September 30, 2009 all methamidophos uses on cotton were canceled. Existing stocks labeled for use on cotton may be used until September 2010. Applications of methamidophos are also currently approved for alfalfa seed crops in California. There are currently two active labels for end use products containing methamidophos (EPA Reg. No. 264-1020 and 264-

729) and two SLN registrations in California (EPA Reg. No. CA-980013, and CA-780163). There are no other SLN registrations for methamidophos products in California, Idaho, Oregon, or Washington. Additionally, there are no registered residential or public health uses for methamidophos and no emergency use registrations for methamidophos in California, Idaho, Oregon, or Washington. On September 23, 2009 EPA issued a cancellation order for all remaining methamidophos products registered in the U.S. in response to an earlier request for voluntary cancelations by product registrants (74 FR 48551). Under the terms of the order, the effective cancellation date for all methamidophos products is December 31, 2009. EPA authorized product registrants to sell and distribute methamidophos products until December 31, 2010. Persons other than the registrants may sell and distribute existing stocks of these products until they are exhausted. Use of these canceled products may continue until the existing stocks are exhausted.

Usage Information.

Use data from 2001 - 2006 indicate an average annual domestic use of approximately 288,000 lbs of active ingredient (EPA 2008b). The majority of methamidophos used during that period was applied to potatoes (69%), followed by tomatoes (14%), and alfalfa (7%) (EPA 2008b). Agriculture use of methamidophos in California has declined over the last decade, with a high of over 300,000 lbs applied in 1997 and approximately 19,000 lbs applied in 2007 (CDPR2008b). There have also been changes in the dominant use of methamidophos in California. Approximately 66% of the methamidophos used in 2007 was applied to alfalfa crops (CDPR 2008b).

Agricultural Uses. Methamidophos is registered for use on four agricultural sites: cotton, potatoes, alfalfa grown for seed (California only), and tomatoes (California only).

*Non-agricultural Uses***.** Non-agricultural uses of methamidophos products are not permitted.

Registered Formulation Types. Registered formulations include emulsifiable concentrate end use products.

Methods and Rates of Application.

Methods. Methamidophos can be applied by chemigation and by spray through ground and aerial application methods. Active labels for end use products specify drift reduction methods for application including a wind speed restriction of 15 mph, a boom length restriction of 75% wingspan or rotor diameter, and other guidance to reduce drift.

Application Rates. Active labels allow a maximum single application rate of one lb methamidophos/acre and an annual application rate of up to 4 lbs methamidophos/acre (Table 7).

Table 7. Summary of all authorized use sites and application restrictions for active methamidophos products registered in California, Idaho, Oregon, and Washington.

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)[1]	App. Method
Cotton[2]	1	2[3]	2	NS	Aerial, ground, chemigation
Potatoes	1	4	4	7 to 10 days	Aerial, ground, chemigation
24(c) CA: Tomato	1	4	4	7 - 10	Aerial, ground
24(c) CA: Alfalfa for seed	1	1	1	NA	Aerial, ground
24 (c)[2] CA: Cotton	1	2	2	NS	Aerial, ground, chemigation

1. NS = not specified
2. Sale and distribution of products allowing this use expires September 30, 2009
3. Per crop cycle

Metabolites and Degradates.

In addition to being a registered pesticide methamidophos is also a degradate of acephate, another registered pesticide, and environmentally occurring concentrations are indistinguishable as to source. Methamidophos does not form an oxon. The identified major degradates of methamidophos are S-methyl phosphoramidothioate, O,S-dimethyl phosphorothioate, methyl mercaptan, dimethyl disulfide, and methyl disulfide (EPA 2007f).

Methidathion

Methidathion is a restricted use compound first registered in the U.S. in 1972. It is a non-systemic organophosphate insecticide/acaricide registered for use to control a wide range of sucking, leaf-eating, and scale insects. It has been used on a variety of food and feed crops that include alfalfa (grown for seed), almonds, apples, apricots, artichokes, cherries, clover (grown for seed), cotton, grapefruit, hay-grass, kiwi fruit, lemons, mandarins, mangos, nectarines, olives, oranges, peaches, pears, pecans, plums, prunes, safflower, sorghum, sugar apple, sunflower, timothy, and walnuts. Methidathion has also been used on terrestrial non-food crops such as tobacco and nursery stock (EPA 2004c). There are currently three active labels for end use products containing methidathion (EPA Reg. No. 10163-236, 10163-238, and 10163-244) and 10 SLN registrations in California, Idaho, Oregon, and Washington (EPA Reg. No. CA-010002, CA-010009, CA-010011, CA-020002, CA-040007, ID-000005, ID-040007, OR-000010, OR-020018, and WA-000006). There are no emergency use registrations for methidathion in California, Idaho, Oregon, or Washington.

On April 7, 2010 EPA issued a notice of receipt by the registrants of methidathion to cancel all remaining uses methidathion products in the United States (75 FR 17735). EPA proposed to include the following provisions for the treatment of existing stocks of methidathion products:

- After December 31, 2010 registrants are prohibited from selling or distributing existing stocks of products containing methidathion.
- After December 31, 2014, persons other than registrants are prohibited from selling or distributing existing stocks of products containing methidathion.
- After December 31, 2014, existing stocks of products containing methidathion can continue to be used legally until they are exhausted.

Usage Information.

Based on 1987 through 1997 usage information, an estimate of methidathion's total domestic annual usage averaged approximately 241,000 lbs a.i (EPA 2004c). However, the reported use of methidathion in California exceeded 300,000 lbs in 1997

(CDPR2008b). Agriculture use of methidathion in California has declined over the last decade to a low of approximately 46,000 lbs applied in 2007 (CDPR2008b). EPA estimates the largest markets in terms of total lbs methidathion used are almonds (18%), oranges (17%), plums and prunes (15%), and walnuts (13%)(EPA 2004c). However, the most recent use statistics from California indicate that a majority of the methidathion was applied to artichokes (30%), almonds (18%), and alfalfa (11%) (CDPRb).

Agricultural Uses. Methidathion is registered for use on alfalfa, almonds, apples, artichokes, clover (grown for seed), citrus, cotton, kiwi fruit, lemons, mandarins, mangos, nectarines, olives, oranges, peaches, pears, pecans, plums, prunes, safflower, sunflower, timothy, and walnuts.

Non-agricultural Uses. Methidation is registered for use on nursery stock.

Registered Formulation Types. End use products include wettable powders in water-soluble bags (25% a.i.) and emulsifiable concentrates (22 - 24% a.i.) (EPA 2004c). Active labels indicate petroleum distillates and a xylene range aromatic solvent are also ingredients in current formulations (EPA Reg. No. 10163-236 and 10163-238).

Methods and Rates of Application.

Methods. Methidathion can be applied by fixed wing aircraft, groundboom, airblast, low-pressure handwand or backpack sprayer. Active labels indicate methidathion can be mixed and applied in oil (EPA Reg. No. 10163-236, 10163-238, and 10163-244). Active labels specify not to apply methidathion within 25 ft of lakes, reservoirs, permanent streams, natural ponds, marshes, and estuaries. The setback to these aquatic features is increased to 50 ft for all applications greater than 3.0 lbs a.i./acre and 150 ft for all aerial applications. Ground boom applications are restricted to a height of ≤ 4 ft above the ground or crop canopy, wind speeds of ≤ 12 mph, and a droplet size distribution that is medium or coarser "according to the ASAE 572 definition." Air blast and aerial applications are restricted to wind speeds of 3 - 10 mph.

Application Rates. Active labels allow a maximum single application rate of up to 10 lbs methidathion/acre and an annual application rate of up to 20 lbs methidathion/acre. In some cases the number of applications and/or application interval is not specified.

Table 8. Summary of all authorized use sites and application restrictions for active methidathion products registered in California, Idaho, Oregon, and Washington.

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A/yr)	App. Interval (days)	App. Method
Almond	3	2[3]	6	14	Aerial, ground
Apple	3	1[3]	3	NA	Aerial, ground
Apricots	3	1[3]	3	NA	Aerial, ground
Artichokes	1	8[2]	8[6]	14	Aerial, ground
Cherries	3	1[3]	3	NA	Aerial, ground
Citrus	10	2[3]	20	45	Aerial, ground
Clover for seeds	1	2[2]	2	NS	Aerial, ground
Cotton	1	4[2]	4	5-7	Aerial, ground
Deciduous fruit	3	1	3	NA	Aerial, ground
Mango	0.25	5[3]	2.25	21	Ground
Nectarines	3	1[3]	3	NA	Aerial, ground
Nursery woody ornamental and herbaceous plants (not Christmas trees)	0.5 lb a.i./100 gallons	1[3]	NS	NS	NS
Olives	3	1[2]	3	NA	Ground
Pears	3	1[3]	3	NA	Aerial, Ground
Peaches	3	1[3]	3	NA	Aerial, ground
Plumes	3	1[3]	3	NA	Aerial, ground
Prunes	3	1[3]	3	NA	Aerial, ground
Kiwi	2	1	2	NA	Ground
Safflower	0.5	3	1.5	7-14	Aerial, ground
Sunflower	0.5	3	1.5	7	Aerial, ground
Walnut	2	3[3]	2	NS	Aerial, ground
24(c) CA: Alfalfa	1	NS	1[2]	NS	Aerial, ground
24(c) ID: Alfalfa	1	NS	1[2]	NS	Aerial, ground
24(c) OR[5]: Alfalfa	1	NS	1[3]	NS	Aerial, ground
24(c) WA: Alfalfa	1	NS	5[4]	NS	Aerial, ground
24(c) CA: Alfalfa for seed	1	1[3]	1[3]	NA	Aerial, ground
24(c) ID, OR: Alfalfa for seed	1	NS	1 (5[4])	NS	Aerial, ground
24(c) CA: Citrus	4	1[3]	NS	NS	Ground
24(c) CA: Clover for seed	1	2	2	NS	Aerial, ground
24(c) CA: Kiwifruit	2	1	2	NA	Ground
1. NS = not specified					

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A/yr)	App. Interval (days)	App. Method
2. Per crop cycle or cutting 3. Per season 4. Maximum annual application rate not specified by SLN label but section 3 label allows for up to 5 lbs. a.i./acre/year 5. Label expires December 31, 2013 6. EPA estimates one crop per year					

Metabolites and Degradates.

Several degradates of methidathion have been identified in soil or water, including methidathion oxon, 5-methyl-1-3,4-thiadazol-2 (3H)-one , des-methyl S-[(5-methoxy-2-oxo-1, 3, 4-thiadiazol-(2-1)-yl-methyl O,O-dimethylphosphorothioate], phosphorthioic acid, 4-(mercaptomethy1)-2-meth-oxy-A2-,13 ,4-thiadiazolin-5-one, and S-[(5-methione-2-oxo-1,3,4-thiodiazol-3(2H)-yl methyl o,o dimethyl phosphorothioate.

Methyl parathion

Methyl parathion is a broad spectrum insecticide/miticide, first registered in 1954. It has registered uses on terrestrial food and feed crops such as alfalfa, almonds, barley, dried beans, cabbage, corn, cotton, grass forage/fodder/hay, hops, lentils, oats, onion, pastures, dried peas, pecans, rangeland, rape seed (canola), rice, rye, soybeans, sugar beets, sunflower, sweet potatoes, walnuts, wheat, white potatoes, and yams (EPA 2004d). Methyl parathion is a restricted use compound used to control pests like mites, thrips, weevils, aphids and leafhoppers (EPA 2004d).

There are currently three active labels for end products containing methyl parathion, including one technical methyl parathion registration (EPA Reg. No. 4787-48), and two end-use products (EPA Reg. No. 70506-193 and 67760-43). There is also one SLN registration in California, Idaho, Oregon, and Washington (EPA Reg. No. CA-000001). However, on July 16, 2010 EPA issued a cancellation order for all remaining uses of methyl parathion. A rescission to that order, and a revised cancellation order was published in the Federal Register on July 27, 2010 (75 FR 43981). The revised order cancels all remaining uses of methyl parathion products, effective December 31, 2012.

Except for export, no end-use products containing methyl parathion will be sold after August 31, 2013, and end-use products cannot legally be used in the United States after December 31, 2013. All end use product labels will be amended to reflect the last legal use date. There are no emergency use registrations for methyl parathion in California, Idaho, Oregon, or Washington.

Usage Information.

Based on 1987 through 1997 usage information, EPA estimates that approximately four million lbs of methyl parathion are applied to five million acres annually in the U.S. The largest uses for methyl parathion in terms of lbs a.i. were: cotton, corn, wheat, soybeans, and rice (EPA 2004d). Agriculture use of methyl parathion in California has been variable with a high of 158,000 lbs applied in 1998 and a low of 54,000 lbs applied in 2002. The most recent pesticide use report for California indicates 75,000 lbs of methyl parathion were applied in 2007, with approximately 99% applied to walnuts (CDPR 2008b). Millions of acres could potentially be treated with methyl parathion in Idaho, Oregon, and Washington given registered uses (*e.g.*, alfalfa, forage grass, onions, potato, and cereal grains). The Washington Department of Agriculture estimated that a total of 73,000 lbs of methyl parathion are annually applied in alfalfa, pastures, and wheat in Washington (EPA 2004d). However, use data for other major crops in Washington, and reliable data reporting recent totals of methyl parathion used in the Northwestern states are not available.

Agricultural Uses. Methyl parathion may be used on alfalfa, canola, cereal grains, corn, cotton, forage grass, onions, rice, soybean, sunflower, potato, and walnuts.

Non-agricultural Uses. Non-agricultural uses of methyl parathion are not permitted.

Registered Formulation Types. Methyl parathion is formulated as a microencapsulate (ME) (20.9% a.i.) and as an emulsifiable concentrate (EC) (ranges from 27.59 to 52.7% a.i.). Methyl parathion has been formulated with other a.i.s, including malathion (EPA 2004d). Although there may be existing stocks of methyl parathion products that contain more than one a.i., all methyl parathion products with active labels contain a single a.i. Some products with active labels list petroleum distillates as an ingredient in the formulation (Reg. No. 4787-48, 70506-193, and 67760-43).

Methods and Rates of Application.

Methods. Methyl parathion can be applied by fixed wing aircraft, power ground equipment, and chemigation. One active label recommends the methyl parathion product be tank-mixed with a pyrethroid or other non-organophosphate insecticide to control whitefly in cotton (EPA Reg. No. 4581-393). Tank mixing with other products is also recommended for control of *Heliothis* species (EPA Reg. No. 4581-393). Active labels provide restrictions to control drift including using the largest droplet size consistent with good pest control, not applying at wind speeds greater than 10 mph or release heights greater than 10 ft for aerial applications and 4 ft for ground application, and boom lengths are not to exceed 75% of wingspan or the rotor diameter.

Application Rates. Active labels allow for a maximum single application rate of up to 2 lbs methyl parathion/acre and an annual application rate of up to 8 lbs methyl parathion/acre (Table 9). Multiple applications are allowed on all approved use sites. The minimum application interval is not specified for several crops.

Table 9. Summary of all authorized use sites and application restrictions for active methyl parathion products registered in California, Idaho, Oregon, and Washington.

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)[1]	App. Method
Alfalfa	1	6	6	NS	Aerial, ground
Corn (field, popcorn[2], and specialty[2])	1	3	3	14-NS	Aerial, ground
Corn-sweet	0.75 (0.5[3])	4 (2[3])	3 (1[3])	14-NS	Aerial, ground, chemigation[2]
Cotton	1.0	5	4	4-5	Aerial, ground
Grass (forage)	0.75	4	3	NS	Aerial, ground
Onions	0.5	4	2	7-NS	Aerial, ground, chemigation[2]
Rapeseed/Canola	0.5	2	1	NS	Aerial, ground
Rice	0.75	2	1.5	NS	Aerial, ground
Soybeans	0.75	2	1.5	7-NS	Aerial, ground,

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)[1]	App. Method
					chemigation[2]
Sunflower	1	2	2	5	Aerial, ground
Wheat, oats, rye, and barley	0.75	2	1.5	7-NS	Aerial, ground, chemigation[2]
White potatoes	1.5	4	6	7-NS	Aerial, ground, chemigation[2]
24(c) CA: Walnut	2	4	8	21	Aerial
1. NS = not specified 2. Not in CA 3. Maximum allowable use in California					

Metabolites and Degradates.

Methyl paraoxon forms the toxic degradates methyl paraoxon and 4-nitrophenol. Other degradates identified in environmental fate studies include monodesmethyl parathion, phosphorothioic acid, O,S-dimethyl o-(4-nitrophenyl)ester, nitrophenyl phosphoric acid, mono (4-nitrophenyl) ester and CO_2 (EPA 2006l).

Naled

Naled is a general use pesticide first registered in 1959 for use as an insecticide-acaricide. It is used primarily to control adult mosquitos, but it is also used to control leaf-eating insects on a variety of fruits, nuts, vegetables and field crops such as cucurbit vegetables, citrus, brassica and leafy vegetables, cotton, alfalfa, safflower, sugar beets, soybeans, peaches, grapes, strawberries, and dried and succulent beans and peas. Other uses include control of pest insects such as blackflies, horn flies, and stable flies in woodlands, swamps, corrals, holding pens, feedlots, pastureland and rangeland for public pest control programs and for areas containing dairy and beef cattle, hogs, horses and sheep. Other non-food uses include treatments in and around food processing plants, loading docks, cull piles, refuse areas, in greenhouses and on outdoor-grown ornamentals. Its use in pet flea collars was canceled prior to issuance of the BE (EPA 2004e). There are currently five active labels for end use products containing naled (EPA Reg. No. 5481-479, 5481-480, 5481-481, 5481-482, and 10163-46). Additionally, there are six SLN registrations in California, Idaho, Oregon, and Washington (CA-000006, CA-050011, CA-860005, ID-010017, OR-990032, and WA-990028). There are no section 18 registrations for use of naled products in these states.

Usage Information.

EPA estimates one million lbs of naled are applied annually in the U.S. (EPA 2004e). Approximately 70% is used for mosquito control. The greatest use of naled in crops occurs in cotton, alfalfa, and safflower (EPA 2004e). Agriculture use of naled in California has declined over the last decade, with a high of over 616,000 lbs applied in 1997 and approximately 132,000 lbs applied in 2007 (CDPR 2008b). EPA indicated 1,000 - 23,000 lbs of naled are applied within the freshwater distribution of listed salmonids in California (EPA 2004e). Use estimates for states in the Pacific Northwest suggest much greater application of naled is possible, although actual use in Idaho, Oregon, and Washington is unknown.

Agricultural Uses. Examples of registered use sites include alfalfa, almonds, beans, broccoli, cabbage, cantaloupes, hops, melons, celery, cotton, eggplant, peppers, grapes, citrus, peaches, safflower, strawberries, sugar beets, squash, and walnuts.

Non-agricultural Uses. Naled is registered for mosquito and fly control on a number of use sites include swamps, tidal marshes, pastures, residential, agricultural, woodlands, in and around food processing plants, loading docks, refuse areas, etc. It is also registered for use on ornamentals, flowering plants, and trees in greenhouses and outside.

Registered Formulation Types. Naled is formulated as an emulsifiable concentrate (36% to 85 % a.i.) and soluble concentrate liquid (87.4% a.i.) (EPA 2004e). Products with active labels contain a single a.i. and petroleum distillates.

Methods and Rates of Application.

Methods. Naled is applied by air and ground equipment, and in greenhouses, via hot plate/hot pan equipment. It cannot be applied through any type of irrigation system for any use, nor through backpack spray equipment on agricultural crops (EPA 2004e).

Labels for agricultural uses of naled contain risk reduction measures such as the requirement to leave an uncultivated, 10 ft area adjacent to aquatic habitats to serve as a vegetative filter strip, and droplet size, release height, and temperature specifications (EPA Reg. No. 10163-46 and 5481-479). Two vector control products specify restrictions on parameters that influence drift including droplet size, nozzle type, nozzle positioning, wind speed, and temperature (EPA Reg. No. 5481-480 and 5481-481). A third product for use in vector control programs lacked specific restrictions to control drift and runoff to aquatic habitats (EPA Reg. No. 5481-482).

Application Rates. Active labels allow a maximum single application rate of 1.88 lbs naled/acre in several crops (Table 10). In some cases the active pesticide labels do not restrict the total number of applications allowed, number of days between applications, or the maximum amount that can be applied to a site annually. Prior to finalization of the RED document in 2006, EPA labels approved other uses of naled, such as use on rice. Additionally, use rates were greater for some crops than active labels (*e.g.*, the maximum application rate for peaches and almonds was 2.8 lbs/acre). The continued use of existing stocks of these products is authorized by EPA. The amount of existing stocks available for future applications is unknown.

Table 10. Summary of all authorized use sites and application restrictions for active naled products registered in California, Idaho, Oregon, and Washington.

Use(s)	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
Almonds	1.88	1	1.9	-	Ground
Beans, Lima Beans, and Peas (dry, succulent)	1.4	5	4.2	7	Ground (aerial in CA only)
Broccoli, Cabbage, Cauliflower, Brussel Sprouts, Kale, Collards	1.88	5 Per season	9.4	7	Aerial, ground
Cantaloupes, Muskmelons	0.94	2	1.9	7	Aerial, ground
Hops	0.94	5	NS	14	Aerial, ground
Melons (for seed)	0.94	2	1.9	7	Aerial, ground
Celery	1.4	5	7	7	Aerial, ground
Cotton	0.94	5	4.7	7	Aerial, ground
Eggplant, Peppers	1.88	5	5.64	7	Aerial, ground
Grapes	0.63 (CA: 0.94)	NS	5.64	NS	Aerial, ground (CA: airblast)
Oranges, Lemons, Grapefruit, Tangerines	1.88	5	5.64	7	Aerial, ground
Peaches	1.88 dormant spray	1	1.88	-	Ground
Safflower (CA only)	2.12	2	2.12	7	Aerial, ground
Strawberries	0.94	5	4.7	NS	Aerial, ground
Sugar Beets	0.94	5	4.7	7	Aerial, ground
Summer Squash	1.88	5	5.64	7	Aerial, ground
Swiss Chard	0.94	7	7.05	NS	Ground
Walnuts	1.9	4	3.8	7	Ground (CA: aerial)
Forest and Shade Trees, Ornamental Shrubs, Flowering Plants	.94	NS	NS	NS	Aerial, ground
Greenhouse: Roses and Other Ornamental Plants	0.06 lbs a.i./ 10,000 cu. ft	2 or 3-4	NS	7 or 3-4	Vapor Treatment
In and Around Food Processing Plants, Loading Docks, Cull Piles, Refuse Areas	0.06 lbs a.i./gallon	NS	NS	5	Ground Spray
Swamps and Pastures	1.25	NS	NS	NS	Aerial, ground
Corrals, Holding Pens, Feedlots (cattle, hogs, sheep, horses)	0.25	NS	NS	7	Aerial, ground
Rangeland (cattle)	0.1	NS	NS	7	Aerial, ground
Wide Area	0.25	NS	10.73[3]	24 hours	Aerial, ground

Use(s)	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
Gnat / Fly Control: Agricultural Areas, Woodlands			(limit of 0.2 per week)		
Mosquito Control: Livestock pastures	0.1	NS	10.73[3] (limit of .2 per week)	10	Aerial, ground
Mosquito Control: Residential, Agricultural, Woodlands	0.1	NS	10.73[3] (limit of 0.2 per week)	24 hours	Aerial, ground
Mosquito Control: Tidal Marshes Municipalities	0.1	NS	10.73[3] (limit of .1 per 7 day period)	24 hours	Aerial, ground
Mosquito/Fly Control: Poultry Houses, Garbage Dumps, Meat Packing Establishments, Docks, Ramps, Disposal Areas, Cider Mills	0.024 lb a.i./gal	NS	NS	NS	Ground spray
24(C) CA: fruit fly bait	1.96 lb a.i./gal Bait station = 6 sq. in. Minimum of 600 stations / sq. mile	NS	NS	14	Bait station
24(C) CA: Cotton	1.41	NS	4.7 per season	NS	Aerial, ground
24(C) CA, ID, OR[2], WA: Alfalfa Grown for seed	1.41	3	NS	7	Aerial, ground
24(C) ID: Carrots grown for seed	1.41	NS	2.81	NS	Aerial, ground

1. NS = not specified
2. These SLN registrations have an expiration date of December 31, 2009
3. Waiting for EPA clarification

Metabolites and Degradates.

Naled forms two degradates of toxic concern, dichloroacetic acid (DCAA) and dichlorovos (DDVP). Dichlorvos is a major degradate of naled (occurs at > 20% of applied naled) and is also a registered organophosphate insecticide.

Phorate

Phorate is registered in the U.S. as an organophosphate soil and systemic insecticide, and miticide used on a variety of agricultural crops and classified as a restricted use pesticide

(EPA 2003e). Several restrictions have been placed on the use of phorate products since the issuance of the BE. For example, the RED reduced the number of use sites eligible for reregistration. Additionally, only granular end use products are approved for registration and these may be applied only once per year and must be soil incorporated (EPA 2006n). EPA indicated there are currently six active labels for end use pesticide products containing phorate (EPA Reg. No. 34704-259, 9779-293, 5481-526, 5481-527, 5481-530, and 5481-8980) and two SLN registrations in California, Idaho, Oregon, and Washington (EPA Reg. No. CA-870069 and WA-910013). There are no section 18 registrations for phorate products permitted for use in California, Idaho, Oregon, or Washington.

Usage Information.

EPA estimates 3 million lbs of phorate are produced annually for pesticide products used in the U.S. (EPA 2003e). Crops with the highest usage with reference to lbs produced are corn (46%), potatoes (21%) and cotton (13%). Almost 2.5 million acres are treated with phorate products annually. Crops with the highest percentage of acres treated include potatoes (20%), fresh sweet corn (10%) and peanuts (9%) (EPA 2003e). The most recent pesticide use report available for California indicates approximately 115,000 lbs of phorate was applied in 1997, and that use declined annually through 2007 when about 34,000 lbs of phorate was applied (CDPR 2008b). Reported use of phorate within the freshwater distribution of listed salmon ranged from zero (Northern California steelhead) to over 20,000 lbs (California Central Valley steelhead) based on county totals reported in California during 2001 (EPA 2003e). The BE also provided estimates of areas that might be treated with phorate in Idaho, Oregon, and Washington given acreage historically planted in crops where phorate can be used. Estimates of acreage potentially treated were provided for migration corridors, and spawning and rearing habitat. EPA estimates for areas potentially treated with phorate ranged from 7 acres in the spawning and rearing habitat of Upper Columbia River Spring-run Chinook salmon to approximately 300,000 acres for the migration corridor of the same species (EPA 2003e).

Agricultural Uses. Historically phorate has been used on a variety of orchard and vegetable crops (EPA 2006n). Active labels restrict phorate use to beans, sweet corn, field corn, cotton, lilies, peanuts, potatoes, radishes, sorghum, soybeans, and sugar beets. Examples of product uses canceled through the reregistration process and historically used in areas where listed salmon reside include alfalfa, oats, and wheat. . The use of phorate on cotton was canceled in 2005. However, EPA continues to authorize the use of phorate in cotton through several active product labels that have not been revised (*e.g.*, EPA Reg. No. 10163-215, 10163-175, and 10163-184).

Non-agricultural Uses. The BE indicates that phorate has been used for structural pest control in California. Active pesticide labels do not permit the use of phorate for non-agricultural uses.

Registered Formulation Types. Phorate is formulated as a granular for all end use products.

Methods and Rates of Application.

Methods. Active labels authorize ground application only and require soil incorporation of granules. Additionally, active labels specify to use BMPs to minimize runoff and that "where highly erodible land (HEL) is adjacent to aquatic bodies, a 66 ft buffer/setback area should be left in grass or other natural vegetation."

Application Rates. The maximum single application rate allowed on active labels in California, Idaho, Oregon, and Washington is 8 lbs phorate/acre (Table 11).

Table 11. Summary of all authorized use sites and application restrictions for active phorate products registered in California, Idaho, Oregon, and Washington.

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year[1]	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method
Beans	2.04	1	2.04	NS	Ground
Corn, sweet	1.31	1	1.30	NS	Ground
Corn, field	1.31	1	1.30	NS	Ground
Cotton	2.18	1	2.18	NS	Ground

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year[1]	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method
Peanuts	1.50	1	1.48	NS	Ground
Potatoes	3.54	1	3.54	NS	Ground
Sorghum	1.31	1	1.31	NS	Ground
Soybeans	2.00	1	2.00	NS	Ground
Sugar beets	1.50	1	1.50	NS	Ground
Washington - Section 24C					
Radishes	3.0	1	3.0	NS	Ground
California - Section 24C					
Lillies and Daffodils	8.0	1	8.0	-	Ground
1. NA = Not Applicable 2. Label specifies one application per crop. One crop per year assumed					

Metabolites and Degradates.

Toxic metabolites and degradates of phorate formed in the environment include phorate sulfoxide and phorate sulfone, both of which are more persistent and mobile than phorate. Additionally, the sulfoxide, sulfone, and parent phorate all form oxons.

Phosmet

Phosmet is a broad-spectrum insecticide/acaricide in the phosphorothioate group of organophosphates. It is registered for control of insects on a variety of crops, mainly fruits and nuts. In addition, phosmet is registered for direct animal treatments to control fleas, lice, hornflies, sarcoptic mange, and ticks on cattle, swine, and dogs. There are also registered uses for Christmas trees, forestry (seed orchards and seedling transplants), and ornamentals, including residential sites treated by professional applicators. Phosmet can be used by homeowners to treat trees, shrubs, ornamental plants, pets (dogs only), and home gardens. It can also be used for fire ant control by professional applicators (EPA 2003f). There are currently eight active labels for end use pesticide products containing phosmet (EPA Reg. No. 2724-262, 10163-184, 10163-215, 10163-168, 10163-169, 10163-171, 10163-174 and 10163-175). Additionally, phosmet has five SLN registrations in California, Idaho, Oregon, and Washington (CA-060002, ID-990024, OR-940049, WA-010019, and WA-030031). There are no section 18 registrations for use of phosmet products in these states.

Usage Information.

EPA estimates over one million lbs of phosmet are applied annually to agricultural crops in the U.S. (EPA 2008c). The greatest use of phosmet occurs in apples (600,000 lbs/year), peaches (300,000 lbs/year), and almonds (200,000 lbs/year). Twelve other crops are estimated to receive more than 20,000 lbs of phosmet per year (EPA 2008c). The most recent pesticide use report available for California indicates that phosmet use between 1997 and 2007 was variable and ranged from a low of 342,000 lbs in 2003 to a high of 658,000 lbs in 2004. In 2007 over 421,000 lbs of phosmet was applied for agricultural uses in California (CDPR 2008b). Reported use of phosmet within the freshwater distribution of listed salmon ranged from 3,000 lbs (Northern California steelhead) to over 100,000 lbs (California Central Valley steelhead) based on county totals reported in California during 2001. Use estimates were not provided for other states although the BE provided estimates of total areas that might be treated with phosmet within the distribution of listed salmonids in Idaho, Oregon, and Washington.

Agricultural Uses. Examples of registered use sites include fruit trees (apple, pear, peach, nectarine, plum, apricot, tart cherry), nut trees (almond, beechnut, brazil nut, butternut, cashew, chestnut, filbert, macadamia, pecan, pistachio, walnut), grapes, kiwi, blueberries, cranberries, peas (succulent and dried), potato, sweet potato (foliar and post-harvest), field margins, ornamental nurseries, trees (including Christmas trees), cattle, and swine.

Non-agricultural Uses. Non-agricultural uses of phosmet by professional applicators include trees, shrubs, ornamentals (nurseries and ornamental landscape plantings), and fire ant mounds. Non-food registrations also exist for forestry and animal treatments (cattle and swine).

Registered Formulation Types. Phosmet products are available in emulsifiable concentrates and wettable-powders in water-soluble bags or packets. There is also one dust product (EPA Reg. No. 10163-168) for post-harvest use on stored sweet potatoes.

Active labels contain single a.i.s only, but some also contain aromatic solvents and petroleum distillate (EPA Reg. No. 10163-215 and 2724-262).

Methods and Rates of Application.

Methods. Active labels allow for chemigation, ground application by boom and air blast, aerial application, dips, dusting equipment, and spray and back rubber applications for livestock. Tank mixture recommendations include mixing phosmet products with dimethoate products for pest control in alfalfa (EPA Reg. No. 10163-175 and 10163-215) and mixing phosmet products with various adjuvants such as stickers, extenders, and dormant spray oils (EPA Reg. No. 10163-171, 10163-184, and 10163-215. Two product labels specify restrictions to limit spray drift that depend on the application method and include wind speed, release height, and droplet size restrictions (EPA Reg. No. 10163-169, and 10163-184). The majority of phosmet labels do not identify specific drift reduction requirements. Three labels indicate that when using the product on cotton, do not apply within one mile of coastal or estuarine water or within 100 ft of aquatic habitat. However, those setbacks are not required for other use sites listed on the labels (EPA Reg. No. 10163-125, 10163-175 and 10163-184). One SLN registration (WA-030031) includes a restriction to address endangered aquatic organisms. This label specifies that unless there is a sustained wind blowing away from fish bearing waters, an untreated buffer of 25, 50, and 150 ft must be maintained between the application site and the aquatic habitat for ground, chemigation, and aerial applications, respectively.

Application Rates. Active labels allow a maximum single application rate of 6 lbs phosmet/acre in nut trees (Table 12). Up to 21 lbs/acre of phosmet can be applied annually in apple crops. In many cases active pesticide labels do not restrict the total number of applications allowed, number of days between applications, or the maximum amount that can be applied to a site annually.

Table 12. Summary of all authorized use sites and application restrictions for active phosmet products registered in California, Idaho, Oregon, and Washington.

Use(s)	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

Use(s)	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
Alfalfa (except CA)	1.02	Once per cutting	NS[1]	NS	Aerial, ground, chemigation
Alfalfa (CA)	0.75	Once per cutting	NS	NS	Aerial, ground, chemigation
Almonds	3.75	2 foliar, 1 dormant spray	NS	NS	Aerial, ground, airblast
Walnuts, Filberts, Other tree nuts[6]	5.95	5	12	NS	Aerial, ground, airblast
Pecans	2.5	NS	7	7	Aerial, ground, airblast
Pistachios (CA only)	3.96	NS	11.98 (total) 3.96 (foliar per season)	NS	Aerial, ground, airblast
Apples	4	NS	21[4]	NS	Aerial, ground, airblast
Crab apples (CA only)	3.73	NS	21[4]	NS	Aerial, ground, airblast
Apricots	3	NS	9.1	NS	Aerial, ground, airblast
Nectarines	3	NS	9.1	NS	Aerial, ground, airblast
Peaches	3	NS	11.9	NS	Aerial, ground, airblast
Pears[2]	5.0	NS	11.2	NS	Aerial, ground, airblast
Cherries (tart)	1.75	NS	5.25	NS	Aerial, ground, airblast
Plums, Prunes	3	NS	9.1	NS	Aerial, ground, airblast
Grapes[3]	1.5	NS	4.55	10	Aerial, ground
Christmas Tree, Conifer trees, deciduous trees[4,]	1.05 Individual tree: 4% dip solution, 1 lb a.i./ 100 gallons	NS-3	NS	NS	Aerial, ground, airblast
Potato, sweet potato	1 0.2 oz / 50 lbs stored sweet potatoes	NS-5	4.66	10	Aerial, ground, chemigation
Peas	1	3	2.9	NS	Aerial, ground
Blueberries	1	5	5.0	NS	Aerial, ground
Cranberries (not CA)	2.8	NS	10.92	10	Aerial, ground, chemigation
Cattle and Swine	Spray: 1 lb a.i. in 100 gal. water Backrubber: 1 lb a.i. in 50 gal.	NS	NS	Cattle spray: 7 -10 days Swine spray : 14 days	Spray, treatment of backrubber

Use(s)	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
	water				
Fire Ants[7]	0.0087 lb a.i. / sq. ft. (~379 lbs a.i./Acre)	NS	NS	NS	ground
Ornamentals (nurseries and established ornamental landscape plantings)	Spray: 0.75 lb a.i. in 100 gallons	3	NS	NS	Aerial, ground
Field Margins[5]	2	NS	NS	NS	NS
24(c) WA: Grapes	2.1	3 times /season – pre-bloom only	6.5 lbs/year maximum	NS	Ground, aerial
24(c) CA: Citrus	2.1	2	4.2	NS	Ground, air blast
24(c) OR: Sweet Cherries	0.931	NS	NS	NS	Aerial, ground, chemigation
24(c) ID: Clover grown for seed	0.931	1	0.931	Once per season	Aerial, ground, chemigation
24(c) WA: Potato	1.75	3	4.67	10-21	Aerial, ground, chemigation

1. NS = not specified
2. The maximum use pattern for this group is based on pears (10163 - 175).
3. EPA intends for the mitigated label to limit the maximum seasonal application rate.
4. EPA intends for the mitigated label to limit the number of application per year.
5. Of all section 3 registrations (*i.e.*, not 24(c) registrations)
6. Annual use is limited to 12 lbs or 5 applications *(i.e.*, growers can apply twice at the max. rate, or up to 5 times at lower rates)
7. This product is only permitted for use within the quarantine zone for red imported fire ants. This includes three counties in southern California, within the range of Southern California steelhead.

Metabolites and Degradates.

Phosmet oxon is a toxic degradate of concern formed by soil metabolism. Phthalamic acid, phthalic acide, and pthalimid are also formed through hydrolysis. Pthalamic acid, n-hydroxymethyl phthalimide,n-methoxymethyl phthalimide are formed through soil metabolism.

Approach to this Assessment

Overview of NMFS' Assessment Framework

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 this step of our analysis, we try to identify the number, age (or life stage), gender, and life history of the individuals that are likely to be exposed to an action's effects and the populations or subpopulations those individuals represent.

Once we identify which listed 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 to assess the risk to listed individuals and their habitat from the stressors of the action. At this point in the analysis, 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. It is here where spatial analyses are used. Each species range is overlaid with land types (agriculture, urban/residential, and forested) to evaluate exposure to the stressors of the action and also to determine overall risk to listed resources.

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 measuring the individual's "fitness" defined as changes in an individual's growth, survival, annual reproductive success, or lifetime reproductive success. In particular, we examine the scientific and commercial data available to determine if an individual's probable response to an action's effect on the environment (which we identify in our *Response Analyses*) are 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 2006b, Mills and Beatty 1979, Stearns 1982. If we conclude that 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.

The critical habitat analysis focuses on reductions in the quality, quantity, or availability of 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 and water quality are the primary assessment endpoints addressed when evaluating the effects of the action 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 anticipated exposure concentrations leading to toxic responses within aquatic organisms (including salmonids and their prey) as well as instances of water bodies not meeting local, state, or federal water quality criteria.

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
- Population models run by NMFS
- Information and data provided by the registrants identified as applicants
- Comments on the draft Opinion from EPA and any applicants
- Incident reports

Collectively, this information provided the basis for our determination as to whether and to what degree listed resources under our jurisdiction are likely to be 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.

Application of Approach in this Consultation

For this consultation, we adapt our general approach to incorporate elements of EPA's ecological risk assessment framework. This risk assessment framework that organizes the available information in three phases: problem formulation, analysis, 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 each phase in the general framework.

Problem Formulation

The first phase of the framework is problem formulation, which is presented in this section. In this phase we generate conceptual models from our initial evaluation of the relationships between stressors of the action (pesticides and identified chemical stressors) and potential receptors (listed species, habitat). These relationships are presented in conceptual model diagrams (Figures 3 and 4, Table 13) and written risk hypotheses (Species Risk Hypotheses and Critical Habitat Risk Hypotheses). Conceptual model diagrams are constructed to illustrate potential pesticide exposure pathways and associated listed resources' responses. The conceptual model for Pacific salmonids is presented in Figure 2. In it, we illustrate where the pesticides generally reside in the environment following application, how pesticides may co-occur with listed species and their habitats, and how the individuals/habitat may respond upon exposure. In the case of Pacific salmonids, we ascribe exposure and response to specific life stages of individuals and then assess individual fitness endpoints sensitive to the action's stressors.

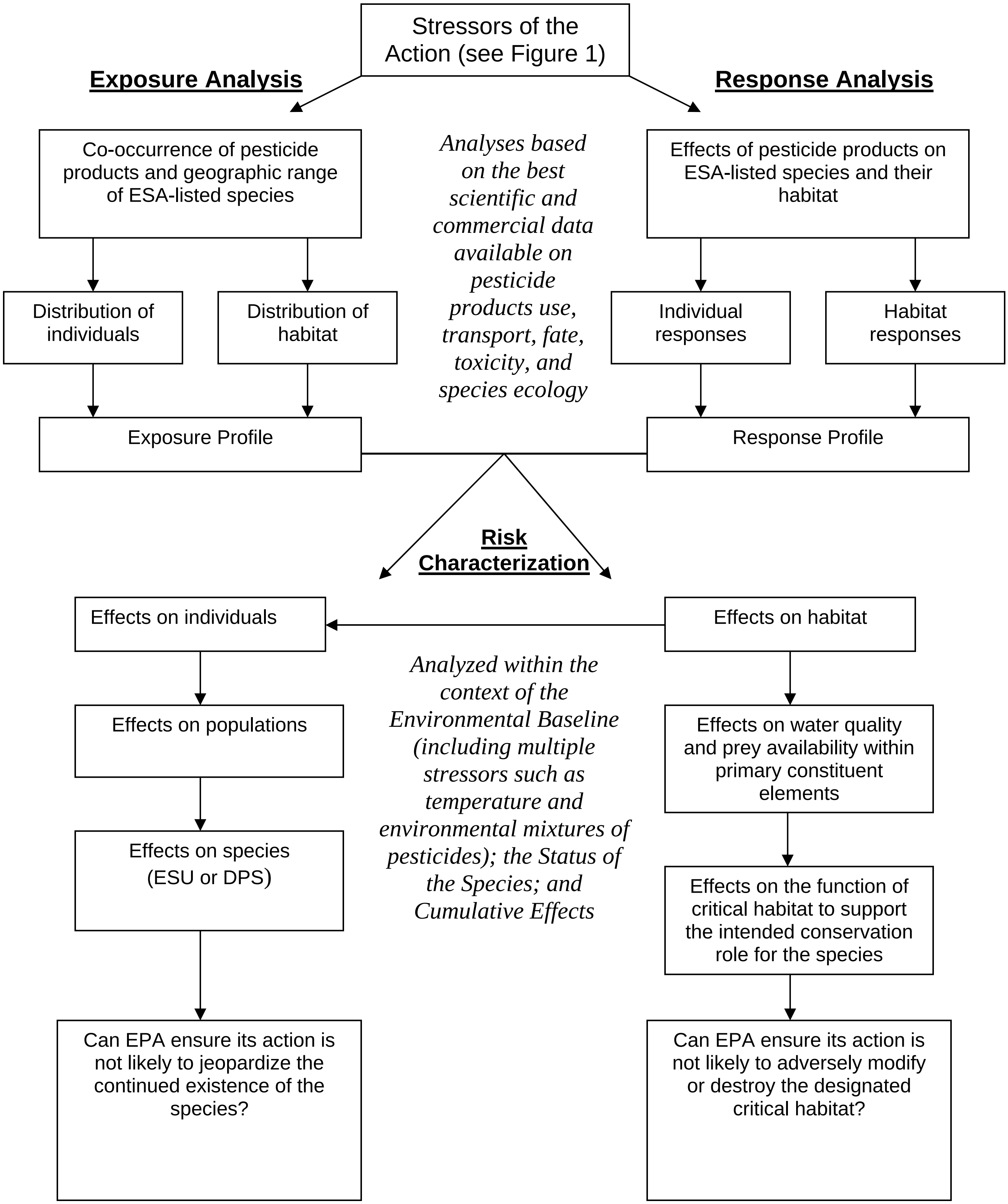


Figure 2 Conceptual framework for assessing risks of EPA's action to listed resources

In the problem formulation phase, we also identify the toxic mode and mechanism of action of chemical stressors, particularly for the pesticide a.i.s. (Figure 4). This information helps us understand what an organism's physiological consequences may be following exposure. It also helps us evaluate whether mixture toxicity occurs because we identify other pesticides that share similar modes of action and the likelihood for co-occurrence in listed species habitats. A similar mode of action with other pesticides is a key determinant of the likelihood of mixture toxicity. With vertebrates (fish and mammals) and invertebrates, the 12 a.i.s share a common mode and mechanism of action, AChE inhibition. From this mode of action, a range of potential adverse responses are possible (Figure 4). Based on this problem formulation, we then search, compile, and review the available toxicity information for the identified stressors to ascertain which physiological systems are known to be affected and to what degree. This analysis is contained in the *Response Analysis* in the *Effects of the Proposed Action to Threatened and Endangered Salmonids* section.

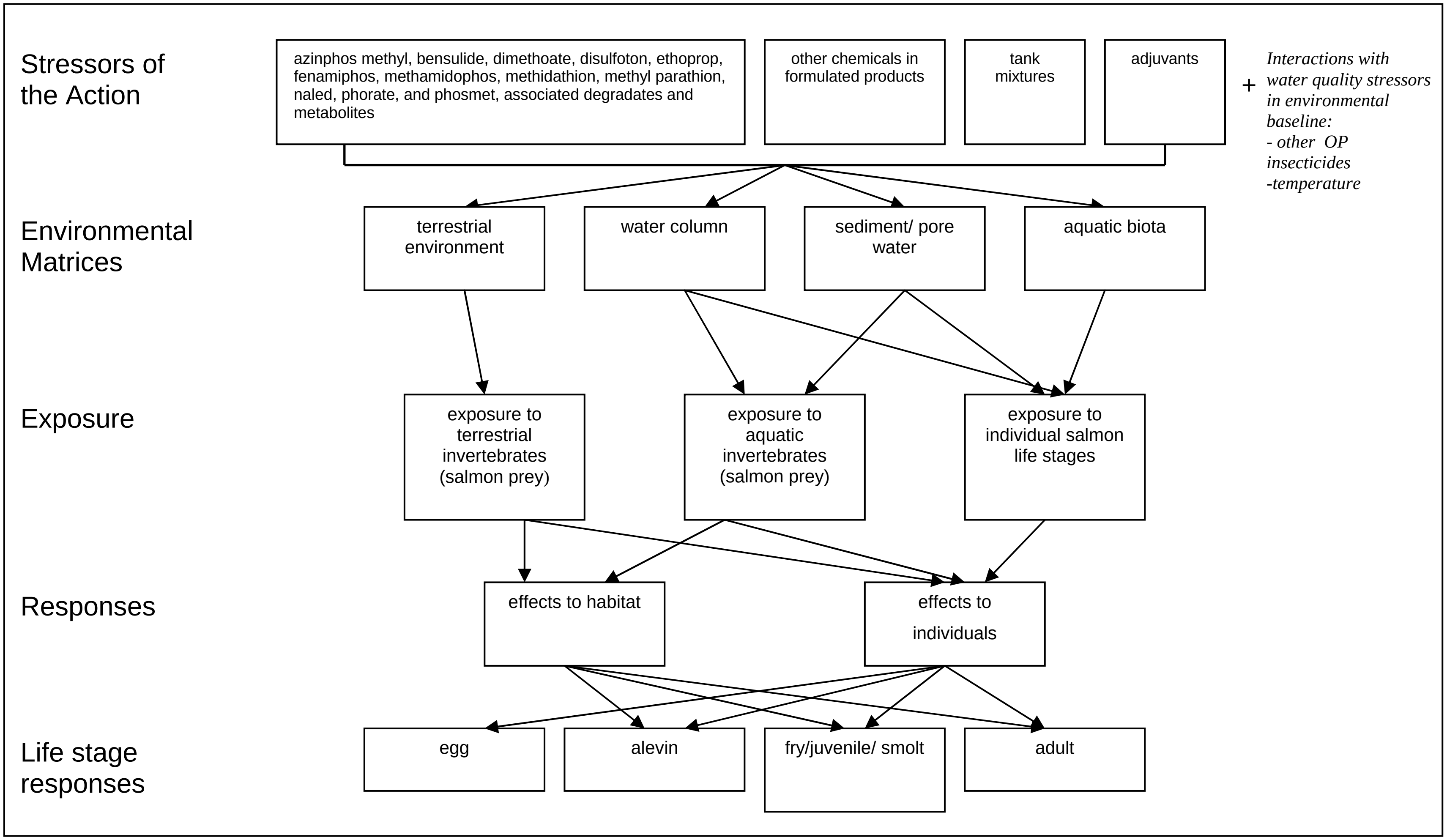
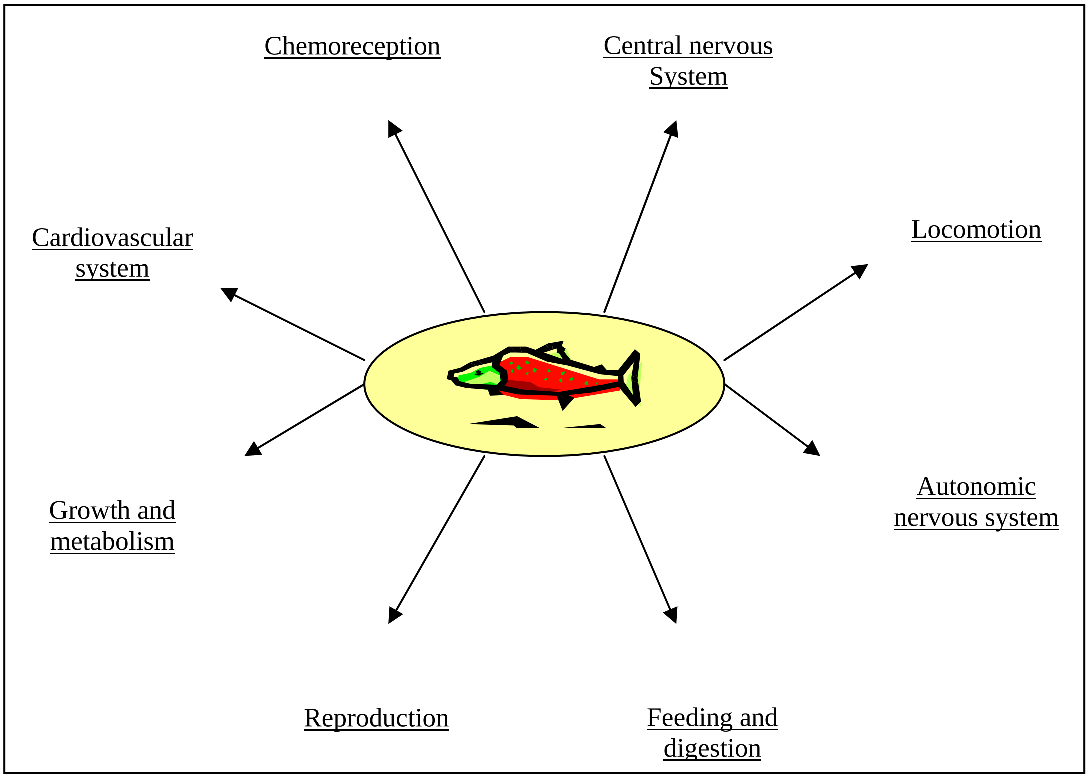


Figure 3 Exposure pathways of the stressors of the action and general responses of Pacific salmonids and habitat

Figure 4 Physiological systems potentially affected by acetylcholinesterase inhibition



Species Risk Hypotheses

We developed risk hypotheses by identifying biological requirements or assessment endpoints for listed resources in the action area that are potentially affected by the stressors of the action. We designate assessment endpoints as those biological properties of species and their habitat essential for successful completion of an individual's life cycle. We integrate the listed resources information with what is known about the stressors of the action, including their physical properties, use, presence in aquatic habitats, and their toxicity. We then evaluate how listed salmonids and their habitat are potentially affected by the stressors of the action and integrate this information with exposure information to develop risk hypotheses. Below are the risk hypotheses (written as affirmative statements) we evaluate in the *Effects of the Proposed Action to Threatened and Endangered Salmonids* section:

1. Exposure to azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, methamidophos, methidathion, methyl parathion, naled, phorate, and phosmet is sufficient to:
    a. Kill salmonids from direct, acute exposure;
    b. Reduce salmonid survival through impacts to growth;
    c. Reduce salmonid growth through impacts on the availability and quantity of salmonid prey;
    d. Impair swimming which leads to reduced growth (via reductions in feeding), delayed and interrupted migration patterns, survival (via reduced predator avoidance), and reproduction (reduced spawning success); and
    e. Reduce olfactory-mediated behaviors resulting in consequences to survival, migration, and reproduction.
2. Exposure to mixtures of the 12 a.i.s can act in combination to increase adverse effects to salmonids and salmonid habitat.
3. Exposure to other stressors of the action including degradates, adjuvants, tank mixtures, and other active and other ingredients in pesticide products containing the 12 a.i.s cause adverse effects to salmonids and their habitat.
4. Exposure to other pesticides present in the action area can act in combination with the 12 a.i.s to increase effects to salmonids and their habitat.
5. Exposure to elevated temperatures can enhance the toxicity of the stressors of the action.

Risk hypotheses are evaluated and discussed in the Risk Characterization section of Effects of the Proposed Action to Threatened and Endangered Pacific Salmonids. For example,to show the relationship between assessment endpoints and measures with species responses. In risk hypothesis 1 (d), aquatic exposure to the an aAChE-inhibiting a.i. can impair a salmonid’s nervous system and consequently affect its swimming ability. Behavioral modifications, such as changes in swimming performance, are regularly considered in NMFS’ Opinions. Swimming performance therefore is an assessment endpoint. Measurable changes in swimming speed are the assessment measure used to evaluate this endpoint. Reductions in swimming performance could also affect other assessment endpoints such as migration and predator avoidance. Behavioral modifications, such as changes in swimming performance, are regularly considered in NMFS' Opinions. This consideration of behavioral modifications and other sublethal effects is a significant difference from EPA's assessment metholodolgy, which focuses on the endpoints of survival, growth, and reproduction. We may or may not have empirical data that address these all assessment endpoints, resulting in a recognized data gap and associated uncertainty. Uncertainties pertaining to toxicity information, and exposure assessment, and the effects of these uncertainties on the ultimate conclusions in the Opinon are discussed in relevant sections.

Critical Habitat Risk Hypotheses:

To determine potential effects to designated critical habitat, NMFS evaluates the effects of the action by first looking at whether PCEs of critical habitat are 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. The stressors of the action for this Opinion are chemicals introduced into the environment by application of pesticide products. As such, the key PCEs that are potentially affected are freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, estuarine areas, and nearshore marine areas where exposure is anticipated. Below are the risk hypotheses (written as affirmative statements) which we evaluate in the *Effects of the Proposed Action to Designated Critical Habitat* section:

1. Exposure to the stressors of the action is sufficient to degrade water quality in freshwater spawning sites;
2. Exposure to the stressors of the action is sufficient to degrade water quality and/or reduce prey availability in freshwater spawning sites;
3. Exposure to the stressors of the action is sufficient to degrade water quality and/or reduce prey availability in freshwater migration corridors;
4. Exposure to the stressors of the action is sufficient to degrade water quality and/or reduce prey availability in estuarine areas;

5. Exposure to the stressors of the action is sufficient to degrade water quality and/or reduce prey availability in nearshore marine areas;

These hypotheses are evaluated using the best scientific and commercial data available, and are presented in the *Risk Characterization*section of the *Effects of the Proposed Action to Designated Critical Habitat.* Examples of assessment endpoints evaluated include prey survival, prey growth, prey drift, prey reproduction, abundance of prey, health of invertebrate aquatic communities, recovery of aquatic communities following pesticide exposure, etc. If the available evidence supports the risk hypotheses, then NMFS evaluates whether the potential reductions in PCEs overlap with habitats rated as high, medium, or low conservation value. Conservation values were determined by Biological Review Teams (BRTs). This portion of the analysis is conducted in the *Integration and Synthesis* section.

Table 13 Examples of salmonid lifestage assessment endpoints and measures

Salmonid Life Stage	Assessment Endpoint (individual fitness)	Assessment Measure (measures of changes in individual fitness)
Egg* * Is the egg permeable to pesticides (measured by pesticide concentrations in eggs)?	Development Survival	size, hatching success, morphological deformities viability (percent survival)
Alevin (yolk-sac fry)	Respiration Swimming: predator avoidance site fidelity Yolk-sac utilization: growth rate size at first feeding Development Survival	gas exchange, respiration rate swimming speed, orientation, burst speed predator avoidance assays rate of absorption, growth weight and length weight and length morphology, histology LC50 (dose-response slope). Percent dead at a given concentration
Fry, Juvenile, Smolt	First exogenous feeding (fry)– post yolk-sac absorption Survival Growth Feeding Swimming: predator avoidance behavior migration use of shelter Olfaction: kin recognition predator avoidance imprinting feeding Smoltification (smolt) Development	time to first feeding, starvation LC50 (dose-response slope). Percent dead at a given concentration weight, length stomach contents, weight, length, starvation, prey capture rates swimming speed, orientation, burst swimming speed predator avoidance assays swimming rate, downstream migration fish monitoring, bioassays electro-olfactogram measurements, behavioral assays behavioral assays behavioral assays behavioral assays Na/K ATPase activity, sea water challenge tests length, weight, malformations
Returning adult	Survival Feeding Swimming: predator avoidance migration spawning feeding Sexual development Olfaction: Predator avoidance Homing Spawning	LC50 (dose-response slope). Percent dead at a given concentration stomach contents behavioral assays numbers of adult returns, behavioral assays numbers of eggs fertilized stomach contents histological assessment of ovaries/testis electro-olfactogram measurements, measurements of intersex behavioral assays behavioral assays behavioral assays

Analysis Plan

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.

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.

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. The *Response* section details toxicity 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* also includes population-level analyses to determine if effects on an individual fitness are sufficiently large to affect population parameters. Finally, we conclude the Effects of the Proposed Action to Threatened and Endangered Pacific Salmonids with a summary of risk associated with each of the a.i.s. This summary presumes use sites are proximate to salmon populations and habitat. Note that an ESU/DPS specific co-occurrence analysis to show where use sites for the pesticide products overlap with salmon populations is done in the *Integration and Synthesis* sections for listed species and designated critical habitat.

Integration and Synthesis

In separate sections for listed species and critical habitat, we combine risk conclusions regarding the effects of the proposed action with information in the *Status of the Species* and *Environmental Baseline* to determine potential effects on populations and species.

Conclusion

Based on the potential effects for each species, we determine if the effects of the proposed action is likely to jeopardize the survival and recovery of the species or cause destruction or adverse modification of designated critical habitat.

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.

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 (Geisy et al., 1999; Giddings, 2009). 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 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. 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 pesticide exposure in salmon, NMFS selected the lowest reported salmonid LC50 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 (Figure 1) in California, Idaho, Oregon, and Washington, 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 existing health and baseline conditions of the environment 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.

Status of Listed Resources

The purpose of this section is to characterize the condition of the 28 salmonid species[1] 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 azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, methamidophos, methidathion, methyl parathion, naled, phorate, or phosmet - containing products (Table 14). 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.go/pr/species/.

Table 14. Listed Species and Critical Habitat (denoted by asterisk) in the Action Area

Common Name (Distinct Population Segment or Evolutionarily Significant Unit)	*Scientific Name*	Status
Chinook salmon (Puget Sound*)	*Oncorhynchus tshawytscha*	Threatened
Chinook salmon (Lower Columbia River*)		Threatened
Chinook salmon (Upper Columbia River Spring-run*)		Endangered
Chinook salmon (Snake River Fall-run*)		Threatened
Chinook salmon (Snake River Spring/Summer-run*)		Threatened
Chinook salmon (Upper Willamette River*)		Threatened
Chinook salmon (California Coastal*)		Threatened
Chinook salmon (Central Valley Spring-run*)		Threatened
Chinook salmon (Sacramento River Winter-run*)		Endangered
Chum salmon (Hood Canal Summer-run*)	*Oncorhynchus keta*	Threatened
Chum salmon (Columbia River*)		Threatened
Coho salmon (Lower Columbia River)	*Oncorhynchus kisutch*	Threatened
Coho salmon (Oregon Coast*)		Threatened
Coho salmon (Southern Oregon & Northern California Coast*)		Threatened
Coho salmon (Central California Coast*)		Endangered
Sockeye salmon (Ozette Lake*)	*Oncorhynchus nerka*	Threatened
Sockeye salmon (Snake River*)		Endangered

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

Common Name (Distinct Population Segment or Evolutionarily Significant Unit)	*Scientific Name*	Status
Steelhead (Puget Sound)	*Oncorhynchus mykiss*	Threatened
Steelhead (Lower Columbia River*)		Threatened
Steelhead (Upper Willamette River*)		Threatened
Steelhead (Middle Columbia River*)		Threatened
Steelhead (Upper Columbia River*)		Threatened
Steelhead (Snake River*)		Threatened
Steelhead (Northern California*)		Threatened
Steelhead (Central California Coast*)		Threatened
Steelhead (California Central Valley*)		Threatened
Steelhead (South-Central California Coast*)		Threatened
Steelhead (Southern California*)		Endangered

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.

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, McElhaney et al 2007, 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[2] 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.

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; abundance; (3) annual growth rate, including trends and variability of annual growth rates; and (4) diversity (McElhany et al 2000).

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

A population's tendency to increase in abundance 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 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 and extirpated 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 presented in these papers are summarized in Appendix 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, at the time it is listed, 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 at the time it is listed 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, increase overwintering success, initiate smoltification, and increase 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 and steelhead (70 FR 52488), 12 ESUs of Oregon, Washington, and Idaho salmon (chum, sockeye, Chinook) and steelhead (70 FR 52630), and for the Oregon Coast coho salmon (73 FR 7816). 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.

NMFS similarly developed the following list of species habitat requirements and PCEs for coho salmon ESUs (64 FR 24049). They are:

1. Juvenile summer and winter rearing areas,
2. Juvenile migration corridors,

3. Areas for growth and development to adulthood,
4. Adult migration corridors, and
5. Spawning areas.

Within these areas, essential habitat attributes of coho salmon critical habitat include adequate: (1) substrate, (2) water quality, (3) water quantity, (4) water temperatures, (5) water velocity, (6) cover/shelter, (7) food, (8) riparian vegetation, (9) space, and (10) safe passage conditions. Riparian vegetation refers to its role in providing essential habitat for coho salmon such as instream woody debris and submerged vegetation for holding and shelter, low water temperature through shading, functional channel bottom substrate for development of eggs and alevins by stabilizing stream banks and capturing fine sediment in runoff, and food by providing nutrients to streams and production of terrestrial insects.

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) considerations 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.

Chinook Salmon

Description of the Species

Chinook salmon are the largest of the Pacific salmon and historically ranged from the Ventura River in California to Point Hope, Alaska in North America, and in northeastern Asia from Hokkaido, Japan to the Anadyr River in Russia (Healey 1991). 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 nine species[2] 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). 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 (McCollough 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 (McCollough 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 (McCollough 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 eta l 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, 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 dn aBjornn 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 both the Columbia River and Sacramento River, California, fry and juveniles move into seasonally inundated floodplains 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 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 2009, 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.

Status and Trends

Chinook salmon face natural threats from flooding, changes in ocean productivity, and predation. Chinook salmon have declined from overharvests, loss of genetic integrity by mixing with hatchery reared fish, retracted distribution by migration barriers such as dams, mortality and 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.

Puget Sound Chinook Salmon

The Puget Sound ESU includes all runs of Chinook salmon in the Puget Sound region from the North Fork Nooksack River to the Elwha River on the Olympic Peninsula

Thirty-six hatchery populations were included as part of the ESU and five were considered essential for recovery and listed (Table 15). They were spring Chinook salmon from Kendall Creek, the North Fork Stillaguamish River, White River, and Dungeness River, and fall run fish from the Elwha River. These artificially propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within the ESU.

Table 15. Puget Sound Chinook salmon - preliminary population structure, abundances, and hatchery contributions (Good et al 2005)

Independent Populations	Historical Abundance	Mean Number of Spawners	Hatchery Abundance Contributions
Nooksack-North Fork	26,000	1,538	91%
Nooksack-South Fork	13,000	338	40%
Lower Skagit	22,000	2,527	0.2%
Upper Skagit	35,000	9,489	2%
Upper Cascade	1,700	274	0.3%
Lower Sauk	7,800	601	0%
Upper Sauk	4,200	324	0%
Suiattle	830	365	0%
Stillaguamish-North Fork	24,000	1,154	40%
Stillaguamish-South Fork	20,000	270	Unknown
Skykomish	51,000	4,262	40%
Snoqualmie	33,000	2,067	16%
Sammamish	Unknown	Unknown	Unknown
Cedar	Unknown	327	Unknown
Duwamish/Green			
Green	Unknown	8,884	83%
White	Unknown	844	Unknown
Puyallup	33,000	1,653	Unknown
Nisqually	18,000	1,195	Unknown
Skokomish	Unknown	1,392	Unknown
Mid Hood Canal Rivers			
Dosewallips	4,700	48	Unknown
Duckabush	Unknown	43	Unknown
Hamma Hamma	Unknown	196	Unknown
Mid Hood Canal	Unknown	311	Unknown
Dungeness	8,100	222	Unknown
Elwha	Unknown	688	Unknown

Life History

Puget Sound Chinook salmon populations exhibit both early-returning (August) and late-returning (mid-September and October) Chinook salmon spawners (Healey 1991).

Juvenile Chinook salmon within the Puget Sound generally exhibit an "ocean-type" life history. However, substantial variation occurs with regard to juvenile residence time in freshwater and estuarine environments. Hayman (1996) described three juvenile life histories for Chinook salmon with varying freshwater and estuarine residency times in the Skagit River system in northern Puget Sound. In this system, 20% to 60% of sub-yearling migrants rear for several months in freshwater habitats while the remaining fry migrate to rear in the Skagit River estuary and delta (Beamer et al 2005). Juveniles in tributaries to Lake Washington exhibit both a stream rearing and a lake rearing strategy. Lake rearing fry are found in highest densities in nearshore shallow (<1 m) habitat adjacent to the opening of tributaries or at the mouth of tributaries where they empty into the lake (Tabor et al 2006). Puget Sound Chinook salmon also has several estuarine rearing juvenile life history types that are highly dependent on estuarine areas for rearing (Beamer et al 2005). In the estuaries, fry use tidal marshes and connected tidal channels including dikes and ditches developed to protect and drain agricultural land. During their first ocean year, immature Chinook salmon use nearshore areas of Puget Sound during all seasons and can be found long distances from their natal river systems (Brennan et al 2004).

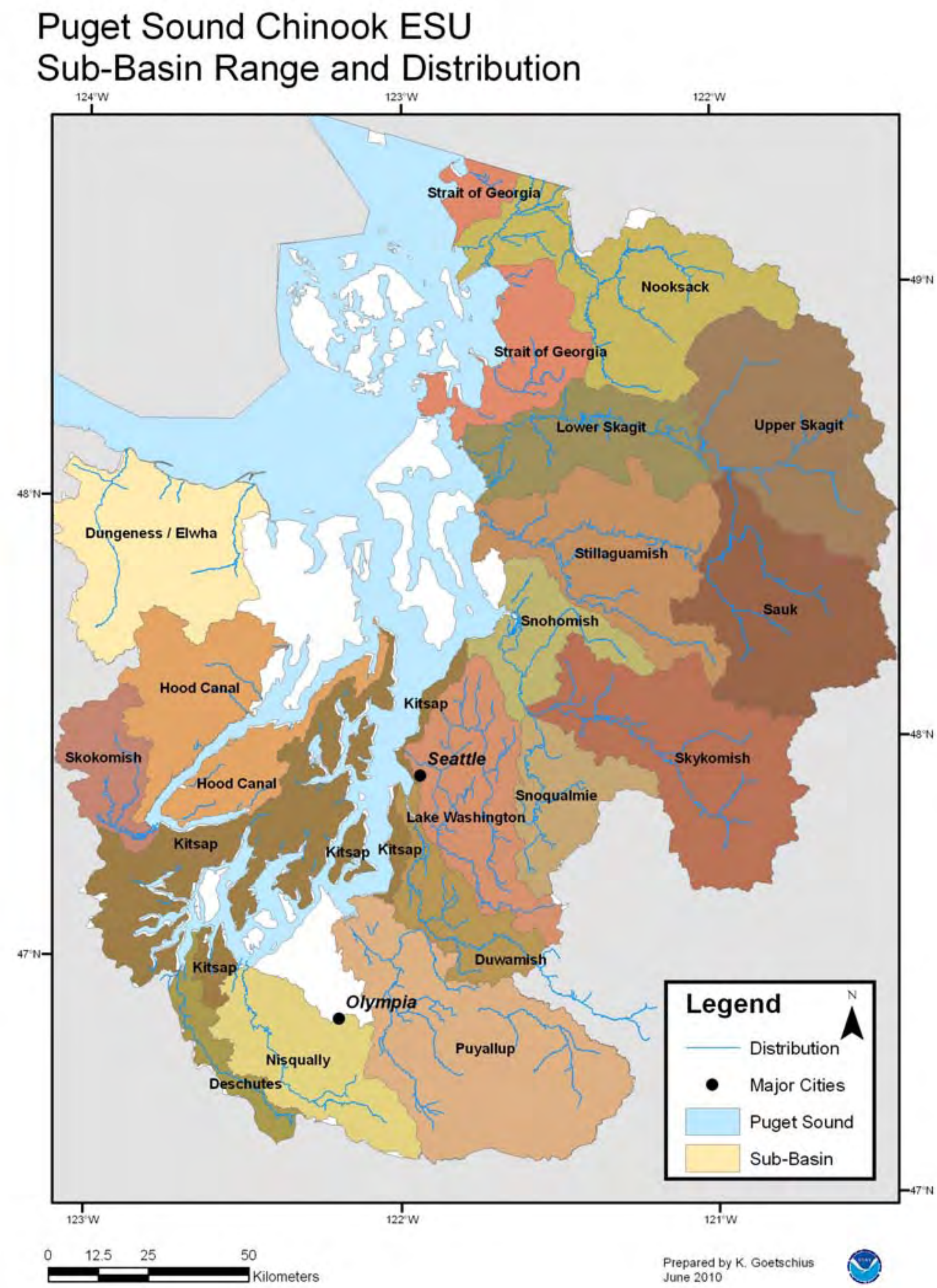


Figure 5. Puget Sound Chinook salmon distribution.

Status and Trends

NMFS listed Puget Sound Chinook salmon as threatened in 1999 (64 FR 14308) and reaffirmed its status as threatened on June 28, 2005 (70 FR 37160). Historically, the ESU included 31 rivers or river systems that supported historic independent populations. Of the historic populations, only 22 are extant (Ruckelshaus et al 2006) (Table 15). A disproportionate loss of an early-run life history represents a significant loss of the evolutionary legacy of the ESU (Ruckelshaus et al 2006).

The spatial structure of the ESU is compromised by extinct and weak populations being disproportionably distributed to the mid- to southern Puget Sound and the Strait of Juan de Fuca. A large portion (at least 11) of the extant runs is sustained, in part, through artificial propagation. Of the populations with greater than 1,000 natural spawners, only two have a low fraction of hatchery fish. Populations known to contain significant natural production are found in the northwest Puget Sound.

Estimates of the historic abundance range from 1,700 to 51,000 potential Puget Sound Chinook salmon spawners per population. During the period from 1996 to 2001, the geometric mean of natural spawners in populations of Puget Sound Chinook salmon ranged from 222 to just over 9,489 fish. Thus, the historical estimates of spawner capacity are several orders of magnitude higher than spawner abundances currently observed throughout the ESU (Good et al 2005). Long-term trends in abundance and median population growth rates for naturally spawning populations indicate that approximately half of the populations are declining and the other half are increasing in abundance over the length of available time series. However, the median overall long-term trend in abundance is close to 1 for most populations that have a lambda exceeding 1, indicating that most of these populations are barely replacing themselves. Eight of 22 populations are declining over the short-term, compared to 11 or 12 populations that have long-term declines (Good et al 2005). Populations with the greatest long-term population growth rates are the North Fork Nooksack and White rivers.

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52630). It includes 1,683 km of stream channels, 41 square km of lakes, and 3,512 km of nearshore marine habitat. Of 61 watersheds (5th field Hydrological Units or HUC 5) reviewed in NMFS' assessment of critical habitat for the Puget Sound ESU, 9 watersheds were rated as having a medium conservation value, 12 were rated as low, and the remaining watersheds (40), where the bulk of federal lands overlap with this ESU, were rated as having a high conservation value for Puget Sound Chinook salmon. The 19 nearshore marine areas were all given a high conservation value rating. (Table 16).

Table 16. Puget Sound Chinook salmon 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]
Strait of Georgia	0		0		3	(3, 1, 2)
Nooksack	4	(1, 3, 2)	1	(3, 1)	0	
Upper Skagit	4	(1, <3)	1	(3)	0	
Sauk	4	(1, 2, 3)	0		0	
Lower Skagit	2	(3, 1, 2)	0		0	
Stillaguamish	3	(1, 3)	0		0	
Skykomish	5	(1, 3)	0		0	
Snoqualmie	2	(1, 3, 2)	0		0	
Snohomish	1	(1,2,3)	1	(1, 2, 3)		
Lake Washington	1	(1)	3	(1, 3, <2)	0	
Duwamish	2	(3, 1, 2)	1	(3)	0	
Puyallup	5	(3, 2, 1)	0		0	
Nisqually	2	(1, <3)	0		0	
Deschutes	0		0		2	(1, 3)
Skokomish	1	(1, 3)	0		0	
Hood Canal	2	(1)	1	(1)	3	(1, <3,<2)
Kitsap	0		0		4	(3, 1)
Dungeness/Elwha	2	(1)	1	(3, 1)	0	
Totals	40		9		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.

Forestry practices have heavily impacted migration, spawning, and rearing PCEs in the upper watersheds of most rivers systems within critical habitat designated for the Puget Sound Chinook salmon. Degraded PCEs include reduced conditions of substrate supporting spawning, incubation and larval development caused by siltation of gravel; and degraded rearing habitat by removal of cover and reduction in channel complexity. Urbanization and agriculture in the lower alluvial valleys of mid- to southern Puget Sound and the Strait of Juan de Fuca have reduced channel function and connectivity, reduced available floodplain habitat, and affected water quality. Thus, these areas have degraded spawning, rearing, and migration PCEs. Hydroelectric development and flood control also obstruct Puget Sound Chinook salmon migration in several basins. The most functional PCEs are found in northwest Puget Sound: the Skagit River basin, parts of the Stillaguamish River basin, and the Snohomish River basin where federal land overlap with critical habitat designated for the Puget Sound Chinook salmon. However, estuary PCEs are degraded in these areas by reduction in the water quality from contaminants, altered salinity conditions, lack of natural cover, and modification and lack of access to tidal marshes and their channels.

Lower Columbia River Chinook Salmon

The Lower Columbia River (LCR) Chinook salmon ESU includes all naturally-spawned populations of fall-run and spring-run Chinook salmon from the Columbia River and its tributaries from its mouth at the Pacific Ocean upstream to a transitional point between Oregon and Washington, east of the Hood River and the White Salmon River. The eastern boundary for this species occurs at Celilo Falls, which corresponds to the edge of the drier Columbia Basin Ecosystem. It also includes the Willamette River to Willamette Falls, Oregon, exclusive of spring-run Chinook salmon in the Clackamas River. Seventeen artificial propagation programs are included in the ESU (70 FR 37160). These artificially propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within this ESU.

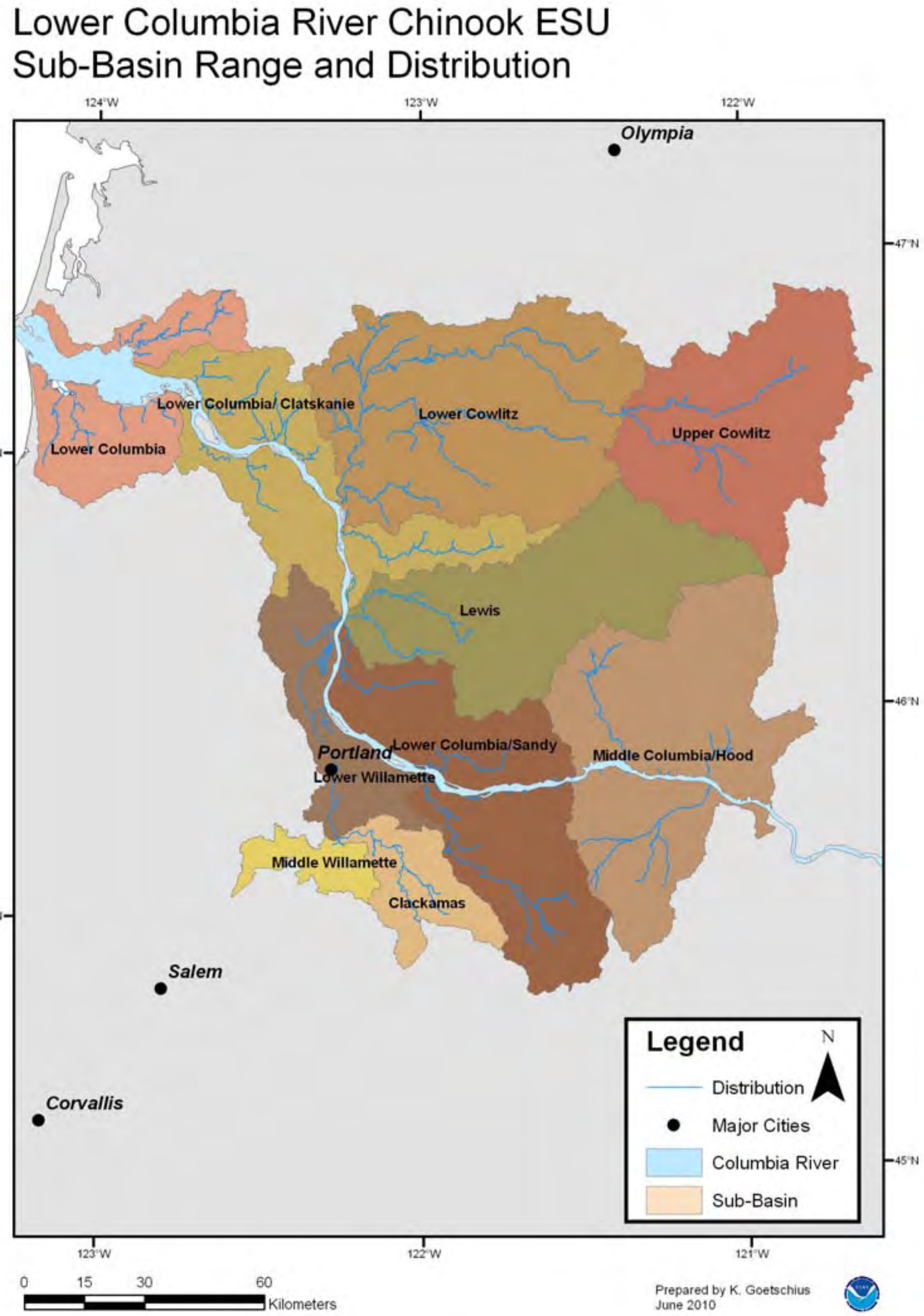


Figure 6. Lower Columbia River Chinook salmon distribution.

Life History

LCR Chinook salmon display three run types including early fall-runs, late fall-runs, and spring-runs. Presently, the fall-run is the predominant life history type. Spring-run Chinook salmon were numerous historically. Fall-run Chinook salmon enter fresh water typically in August through October. Early fall-run spawn within a few weeks in large river mainstems. The late fall-run enters in immature conditions, has a delayed entry to spawning grounds, and resides in the river for a longer time between river entry and spawning. Spring-run Chinook salmon enter fresh water in March through June to spawn in upstream tributaries in August and September.

Offspring of fall-run spawning may migrate as fry to the ocean soon after yolk absorption (*i.e.*, ocean-type), at 30–45 mm in length (Healey 1991). In the Lower Columbia River system, however, the majority of fall-run Chinook salmon fry migrate either at 60-150 days post-hatching in the late summer or autumn of their first year. Offspring of fall-run spawning may also include a third group of yearling juveniles that remain in fresh water for their entire first year before emigrating. The spring-run Chinook salmon migrates to the sea as yearlings (stream-type) typically in spring. However, the natural timing of LCR spring-run Chinook salmon emigration is obscured by hatchery releases (Myers et al 2006).

Once at sea, the ocean-type LCR Chinook salmon tend to migrate along the coast, while stream-type LCR Chinook salmon appear to move far off the coast into the central North Pacific Ocean (Healey 1991, Myers et al 2006). Adults return to tributaries in the lower Columbia River predominately as three- and four-year-olds for fall-run fish and four- and five-year-olds for spring-run fish.

Status and Trends

NMFS originally listed LCR Chinook salmon as threatened on March 24, 1999 (64 FR 14308), and reaffirmed their threatened status on June 28, 2005 (70 FR 37160). Thirty-one independent Chinook salmon populations – 22 fall- and late fall-runs and 9 spring-runs – are estimated to have existed historically in the Lower Columbia River (Myers et al 2006). The Willamette/Lower Columbia River Technical Review Team (W/LCRTRT) has estimated that 8-10 historic populations have been extirpated, most of them spring-run populations. The fall-run Chinook salmon historically occurred throughout the Lower Columbia River basin, while spring-run Chinook salmon only occurred in the upper portions of Lower Columbia Basins that consist of snowmelt driven flow regimes. The Cowlitz, Kalama, Lewis, White Salmon, and Klickitat Rivers are the major river systems on the Washington side, and the lower Willamette and Sandy Rivers are foremost on the Oregon side.

The basin wide spatial structure has remained generally intact. However, the loss of about 35% of historic habitat has affected distribution within several Columbia River subbasins. Currently, only one population appears to be self sustaining (Good et al 2005). Table 4 identifies populations within the LCR Chinook salmon ESU, their abundances, and hatchery input.

Table 17. **Lower Columbia River Chinook salmon - population structure, abundances, and hatchery contributions (Good et al 2005, Myers et al 2006).**

Run	Populatio	Historical Abundance	Mean Number of Spawners	Hatchery Abundance Contributions
F-R	Grays River (WA)	2,477	99	38%
	Elochoman River (WA)	Unknown	676	68%
	Mill, Abernathy, and German Creeks (WA)	Unknown	734	47%
	Youngs Bay (OR)	Unknown	Unknown	Unknown
	Big Creek (OR)	Unknown	Unknown	Unknown
	Clatskanie River (OR)	Unknown	50	Unknown
	Scappoose Creek (OR)	Unknown	Unknown	Unknown
F-R	Lower Cowlitz River (WA)	53,956	1,562	62%
	Upper Cowlitz River (WA)	Unknown	5,682	Unknown
	Coweeman River (WA)	4,971	274	0%
	Toutle River (WA)	25,392	Unknown	Unknown
	Salmon Creek and Lewis River (WA)	47,591	256	0%
	Washougal River (WA)	7,518	3,254	58%
	Kalama River (WA)	22,455	2,931	67%
	Clackamas River (OR)	Unknown	40	Unknown
	Sandy River (OR)	Unknown	183	Unknown
LF-R	Lewis R-North Fork (WA)	Unknown	7,841	13%
	Sandy River (OR)	Unknown	504	3%
S-R	Upper Cowlitz River (WA)	Unknown	Unknown	Unknown
	Tilton River (WA)	Unknown	Unknown	Unknown

Run	Populatio	Historical Abundance	Mean Number of Spawners	Hatchery Abundance Contributions
	Cispus River (WA)	Unknown	1,787*	Unknown
	Toutle River (WA)	2,901	Unknown	Unknown
	Kalama River (WA)	4,178	98	Unknown
	Lewis River (WA)	Unknown	347	Unknown
	Sandy River (OR)	Unknown	3,085	3%
F-R	Upper Columbia Gorge (WA)	2,363	136	13%
	Big White Salmon R (WA)	Unknown	334	21%
	Lower Columbia Gorge (OR)	Unknown	Unknown	Unknown
	Hood River (OR)	Unknown	18	Unknown
S-R	Big White Salmon R (WA)	Unknown	334	21%
	Hood River (OR)	Unknown	18	Unknown

*Arithmetic mean
Recent 5-year spawner abundance (up to 2001) and historic abundance over more than 20 years is given as a geometric mean, and include hatchery origin Chinook salmon.
F-R is fall run, LF-R is late fall run, and S-R is spring run Chinook salmon.

Historical records of Chinook salmon abundance are sparse. However, cannery records suggest a peak run of 4.6 million fish [43 million lbs see Lichatowich (1999) in 1883. Historically, the number of spring-run Chinook salmon returning to the Lower Columbia River may have almost equaled that of fall-run Chinook salmon (Myers et al 2006). Today, the majority of spring-run LCR Chinook salmon populations are extirpated and total returns are substantially lower than for the fall-run component.

Trend indicators for most populations are negative. The majority of populations for which data are available have a long-term trend of <1; indicating the population is in decline (bennet 2005, Good et al 2005). Only the late-fall run population in Lewis River has an abundance and population trend that may be considered viable (McElhany et al 2007). The Sandy River is the only stream system supporting a natural production of spring-run Chinook salmon of any amount. However, the population is at risk from low abundance and negative to low population growth rates (McElhany et al 2007).

The genetic diversity of all populations (except the late fall-run Chinook salmon) has been eroded by large hatchery influences and periodically by low effective population sizes. The near loss of the spring-run life history type remains an important concern for maintaining diversity within the ESU.

The ESU is at risk from generally low abundances in all but one population, combined with most populations having a negative or stagnant long-term population growth. However, fish from conservation hatcheries do help to sustain several LCR Chinook salmon runs in the short-term though this is unlikely to result in sustainable wild populations in the long-term. Having only one population that may be viable puts the ESU at considerable risk from environmental stochasticity and random catastrophic events. The loss of life history diversity limits the ESU's ability to maintain its fitness in the face of environmental change.

Critical Habitat

NMFS designated critical habitat for LCR Chinook salmon on September 2, 2005 (70 FR 52630). It includes all Columbia River estuarine areas and river reaches proceeding upstream to the confluence with the Hood Rivers as well as specific stream reaches in a number of tributary subbasins.

Of the watersheds (HUC 5s) reviewed in NMFS' assessment of critical habitat for the LCR Chinook salmon ESU, 13 subbasins were rated as having a medium conservation value, four were rated as low, and the remaining subbasins (31), were rated as having a high conservation value to LCR Chinook salmon (Table 18). Additionally, four watersheds were given a "possibly high" rating, *i.e.*, they may be essential to conservation of the species but are currently unoccupied.

Table 18. LCR Chinook salmon HUC 5 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]
Middle-Columbia/Hood	6	(1)	2	(3)	0	
Lower Columbia/Sandy	7	(1, 3)	1	(3, 1)	1	(3)
Lewis	2	(1, 2, 3)	0		0	
Lower Columbia/Clatskanie	2	(3, 1)	3	(3, 2)	1	(2)
Upper Cowlitz River	5	(3)	0		0	
Lower Cowlitz	4	(3, 1)	4	(3, 1)	0	
Lower Columbia	2	(3, 1)	1		0	
Middle Willamette	0		0		1	(2)
Clackamas	1	(1)	0		1	
Lower Willamette	1	(2)	2	(2)	0	
Lower Columbia Corridor	1	(3)	0		0	
Total	31		13		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.

Timber harvest, agriculture, and urbanization have degraded spawning and rearing PCEs by reducing floodplain connectivity and water quality, and by removing natural cover in several rivers. Hydropower development projects have reduced timing and magnitude of water flows, thereby altering the water quantity needed to form and maintain physical habitat conditions and support juvenile growth and mobility. Adult and juvenile migration PCEs are affected by several dams along the migration route.

Upper Columbia River Spring-run Chinook Salmon

The Upper Columbia River (UCR) Spring-run Chinook salmon ESU includes all naturally spawned populations of spring-run Chinook salmon in all Columbia River tributaries upstream of the Rock Island Dam and downstream of Chief Joseph Dam in Washington State. Major tributary subbasins with existing runs are the Wenatchee, Entiat, and Methow Rivers

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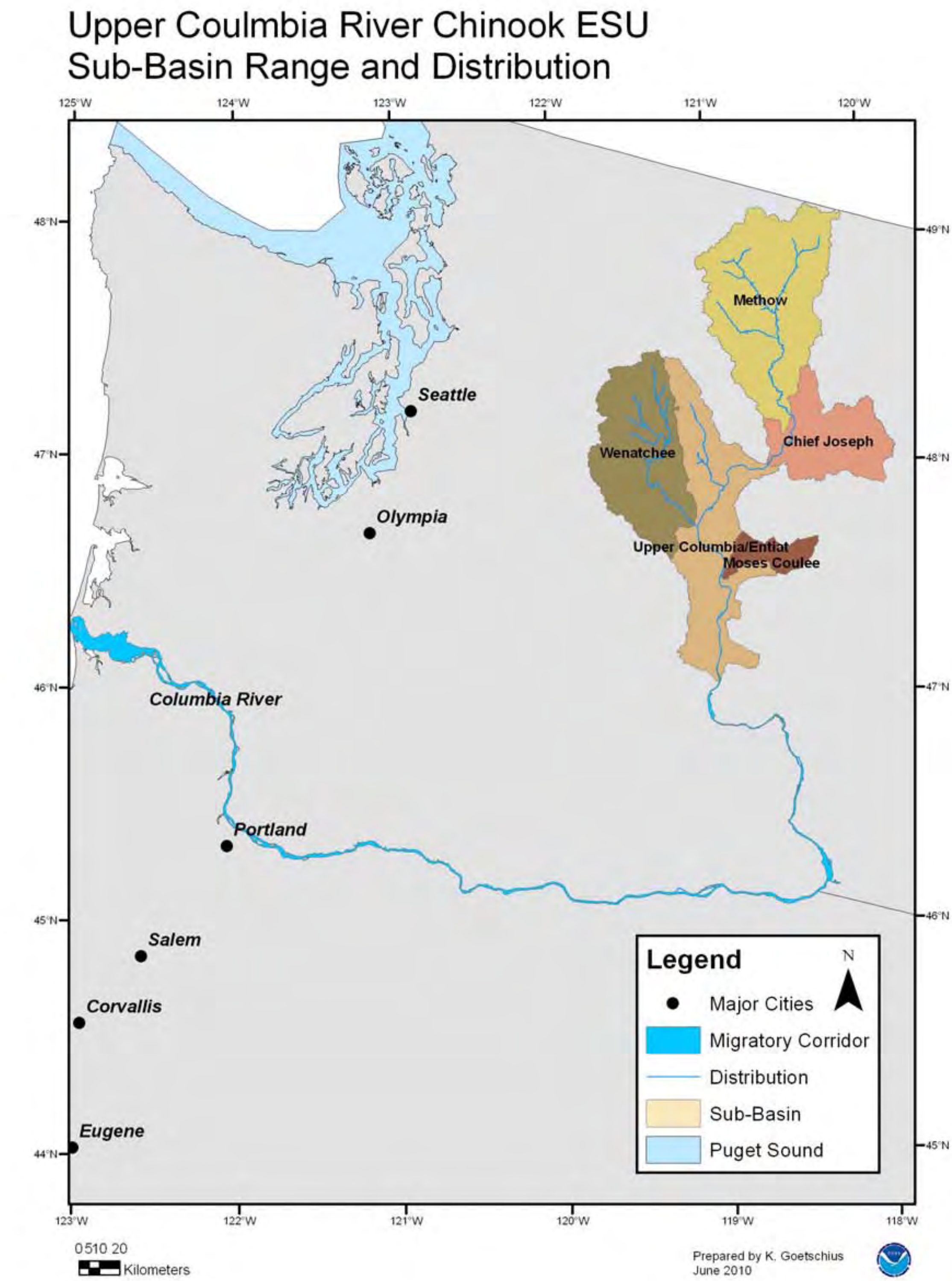


Figure 7). Several hatchery populations are also listed (70 FR 37160). These artificially

propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within this ESU.

Life History

UCR Spring-run Chinook salmon begin returning from the ocean in the early spring. They enter the upper Columbia tributaries from April through July, with the run peaking in mid-May. After migration, UCR Spring-run Chinook salmon hold in freshwater tributaries until spawning occurs in the late summer, peaking in mid- to late August. Juvenile spring-run Chinook salmon spend a year in fresh water before emigrating to salt water in the spring of their second year.

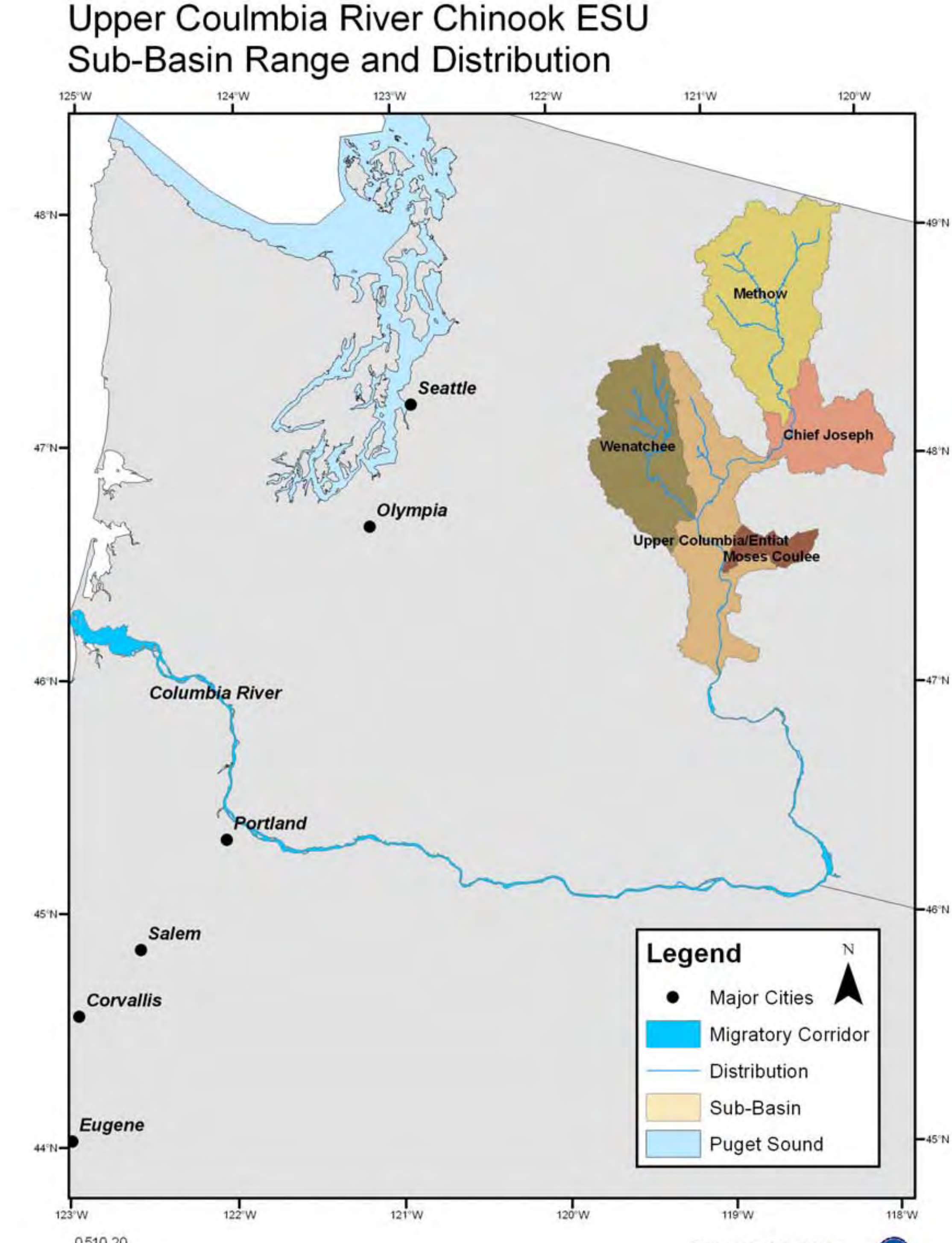


Figure 7. Upper Columbia River Spring-run Chinook salmon distribution

Status and Trends

NMFS listed UCR Spring-run Chinook salmon as endangered on March 24, 1999 (64 FR 14308), and reaffirmed their endangered status on June 28, 2005 (70 FR 37160). The ESU consisted of four populations. Of these, one is now extinct and three are extant. The Interior Columbia Basin Technical Review Team (ICBTRT) characterizes the spatial structure risk to UCR Spring-run Chinook populations as "low" or "moderate." Table 19 identifies populations within the UCR Spring-run Chinook salmon ESU, their abundances, and hatchery input.

Table 19. Upper Columbia River Spring-run Chinook salmon - preliminary population structure, abundances, and hatchery contributions (Good et al 2005)

Population	Historical Abundance	Mean Number of Spawners (Range)[a]	Hatchery Abundance Contributions
Methow River	~2,100	680 (79-9,9-04)	59%
Twisp River	Unknown	58 redds (10-369)	54%
Chewuch River	Unknown	58 redds (6-1,105)	41%
Lost/Early River	Unknown	12 (3-164)	54%
Entiat River	~380	111 (53-444)	42%
Wenatchee River	~2,400	470 (119 - 4,446)	42%
Chiwawa River	Unknown	109 redds (34-1,046)	47%
Nason Creek	Unknown	54 redds (8-374)	39%
Upper Wenatchee River	Unknown	8 redds (0-215)	66%
White River	Unknown	9 redds (1-104)	8%
Little Wenatchee River	Unknown	11 redds (3-74)	21%
Okanogan River	Unkown	Extirpated	NA

[a] 5-year geometric mean number of spawners unless otherwise noted; includes hatchery fish. Range denoted in parenthesis. Means calculated from years 1997 to 2001, except Lost/Early Winter creeks did not include 1998 as no data were available. Data reported in Good et al 2006.

For all populations, average abundance over the recent 10-year period is below the average abundance thresholds that the ICBTRT identifies as a minimum for low risk (ICTRT 2008a, ICTRT 2008b, ICTRT 2008c). The geometric mean spawning escapements from 1997 to 2001 were 273 for the Wenatchee population, 65 for the Entiat population, and 282 for the Methow population. These numbers represent only 8% to

15% of the minimum abundance thresholds. The five-year geometric mean remained low as of 2003. Recently, the 2007 UCR spring Chinook jack counts, an indicator of future adult returns, have increased to their highest level since 1977.

Based on 1980-2004 returns, the lambda for this ESU is estimated at 0.93 (meaning the population is not replacing itself) (Fisher and Hinrichsen 2006). The long-term trend for abundance and lambda for individual populations indicate a decline for all three populations (Good et al 2005). Short-term lambda values indicate an increasing trend for the Methow population, but not for the Wenatchee and Entiat populations (ICTRT 2008a, ICTRT 2008b, ICTRT 2008c).

Finally, the ICBTRT characterizes the diversity risk to all UCR Spring-run Chinook populations as "high". The high risk is a result of reduced genetic diversity from homogenization of populations that occurred under the Grand Coulee Fish Maintenance Project in 1939-1943.

Abundance data showed an increase in spawner returns in 2000 and 2001 (Good et al 2005). However, this increase did not manifest itself in subsequent years. Thus, recent available data on population viability suggest that the ESU continues to be at high risk from small population size; all three UCR Spring-run Chinook salmon populations are affected by low abundances and failing recruitment. Should population growth rates continue at the 1980-2004 levels, UCR Spring-run Chinook salmon populations have a high probability of decline within 50 years. The genetic integrity of all populations has been compromised by periods of low effective population size and low proportion of natural-origin fish.

Critical Habitat

NMFS designated critical habitat for UCR Spring-run Chinook salmon on September 2, 2005 (70 FR 52630). It includes all Columbia River estuarine areas and river reaches proceeding upstream to Chief Joseph Dam and several tributary subbasins.

The UCR Spring-run Chinook salmon ESU has 31 watersheds within its range. Five watersheds received a medium rating and 26 received a high rating of conservation value to the ESU (Table 20). The Columbia River rearing/migration corridor downstream of the spawning range was rated as having a high conservation value.

Table 20. UCR Spring-run Chinook salmon 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]
Chief Joseph	1	(3)	0		0	0
Methow	5	(1, <2, <3)	2	(1, 2)	0	
Upper Columbia/Entiat	3	(3, 2[2], 1[2])	1	(3)	0	
Wenatchee	3	(1, 2, <3)	2	(2, 1)	0	
Moses Coulee	1	(1, =0.8mi)	0		0	
Upper Columbia/Priest Rapids	3	(3)	0		0	
Middle Columbia/Lake Wallula	5	(3)	0		0	
Middle Columbia/Hood	4	(3)	0		0	
Lower Columbia/Sandy	1	(3)	0		0	
Lower Columbia Corridor	all	(3)[3]	0		0	
Total	26		5		0	

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.
2 Only one of the three watersheds, Entiat River, had PCEs 1 and 2.
3 The Lower Columbia Corridor includes 46.5 miles of estuarine PCEs.

Spawning and rearing PCEs are somewhat degraded in quality in tributary systems by urbanization in lower reaches, irrigation and diversion in the major upper drainages, and grazing in the middle reaches. These activities have resulted in excess erosion of fine sediment and silt that smother spawning gravel; reduction in flow quantity necessary for successful incubation, formation of physical rearing conditions, and juvenile mobility; reduction in water quality through contaminated agricultural runoff; and removal of natural cover. Adult and juvenile migration PCEs are heavily degraded by Columbia

River federal dam projects and a number of mid-Columbia River Public Utility District dam projects also obstruct the migration corridor.

Snake River Fall-run Chinook Salmon

The Snake River (SR) Fall-run Chinook salmon ESU includes all naturally spawned populations of fall-run Chinook salmon in the mainstem Snake River below Hells Canyon Dam, and in the Tucannon River, Grande Ronde River, Imnaha River, Salmon River, and Clearwater River subbasins (70 FR 37176,). Four artificial propagation programs are included in the ESU. These artificially propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within this ESU.

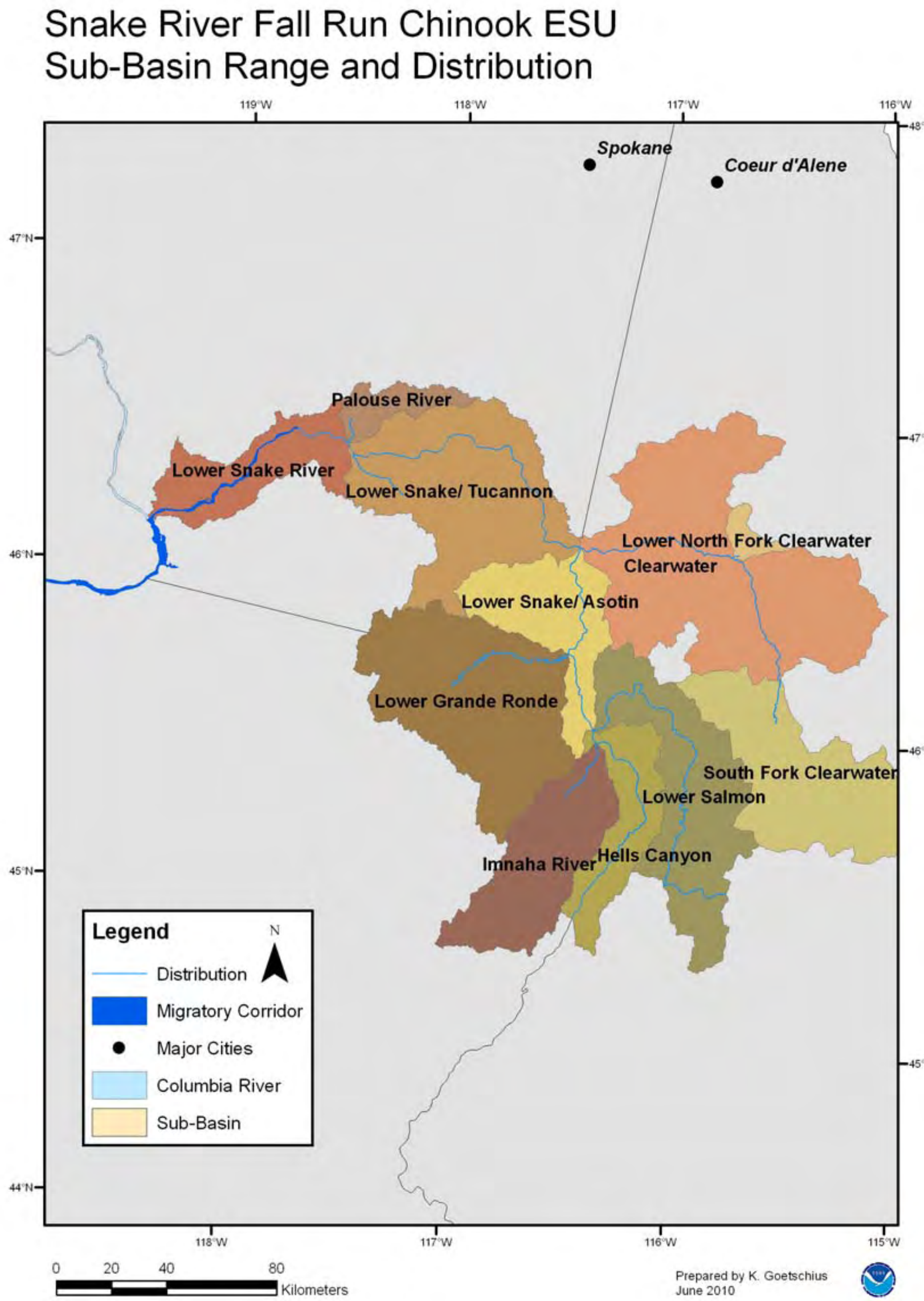


Figure 8. Snake River Fall-run Chinook salmon distribution

Life History

Adult SR Fall-run Chinook salmon enter the Columbia River in July and migrate into the Snake River from August through October. Fall-run Chinook salmon generally spawn from October through November. Fry emerge from redds from April through June (Tiffan et al 2001). Fry rear two months or more in the sandy littoral zone along the river margins above Lower Granite Dam before passing over the dam (Connor et al 2002, Smith et al 2003). Sub-yearling smolts pass over the Lower Granite Dam from June through October with most migration occurring from July through September (Tiffan et al 2001). Sub-yearlings increase their rate of seaward movement as they progress downstream (Smith et al 2003).

Historically, SR Fall-run Chinook salmon exhibited a largely ocean-type life history. However, as a consequence of dam construction, this ESU now resides in water that is cooler than the historic spawning areas, and alteration of the Lower Snake River by hydroelectric dams has created a series of low-velocity pools in the Snake River. Thus, Fall-run Chinook salmon in the Snake River Basin now exhibit one of two life histories: ocean-type and reservoir-type (Connor et al 2005, Tiffan et al 2001). The reservoir-type life history is one where juveniles overwinter in the reservoirs created by the dams, prior to migrating out of the Snake River. SR Fall-run Chinook salmon spend one to four years in the Pacific Ocean before beginning their spawning return migration.

Status and Trends

NMFS originally listed SR Fall-run Chinook salmon as endangered in 1992 (57 FR 14653) but reclassified their status as threatened on June 28, 2005 (70 FR 37160). The SR Fall-run Chinook salmon consists of one extant population that is mostly limited to a core spawning area within a 32-km section of the mainstem Snake River (ICTRT 2003) Two populations have been extirpated.

Estimated annual returns for the period 1938 to 1949 were at 72,000 fish. By the 1950s, numbers had declined to an annual average of 29,000 fish (Bjornn and Hornere 1980). Numbers of SR Fall-run Chinook salmon continued to decline during the 1960s and

1970s as approximately 80% of their historic habitat were eliminated or severely degraded by the construction of the Hells Canyon complex (1958 to 1967) and the lower Snake River dams (1961 to 1975). The abundance of natural-origin spawners in the SR Fall-run Chinook ESU for 2001 (2,652 adults) exceeded 1,000 fish for the first time since counts began at the Lower Granite Dam in 1975. The recent five-year mean abundance of 871 naturally produced spawners at the time of the last status review generated concern that despite recent improvements, the abundance level is very low for an entire ESU. On the other hand, during the years from 1975 to 2000, the ESU fluctuated between 500 to 1,000 natural spawners. This suggests a higher degree of stability in growth rate at low population levels than is seen in other salmonid populations. Further, numbers of natural-origin SR Fall-run Chinook salmon have increased over the last few years, with estimates at Lower Granite Dam of 2,652 fish in 2001, 2,095 fish in 2002, and 3,895 fish in 2003.

Long- and short-term trends in natural returns are positive. Productivity is likely sustained largely by a system of small artificial rearing facilities in the lower Snake River Basin. Depending upon the assumptions made regarding the reproductive contribution of hatchery fish, long- and short-term trends in productivity are at or above replacement.

Low abundances in the 1990s combined with a large proportion of hatchery derived spawners likely have reduced genetic diversity from historic levels. Nevertheless, the SR Fall-run Chinook salmon remains genetically distinct from similar fish in other basins.

As the ESU's single population spawning activities are limited to a relatively short reach of the free flowing mainstem Snake River, it is at considerable risk from environmental variability and stochastic events. The 1997 to 2001 geometric mean natural-origin count over Lower Granite Dam approximate 35% of the proposed delisting abundance criteria of 2,500 natural spawners averaged over eight years. Current observed abundances indicate that the ESU is at moderate risk from low abundances.

Critical Habitat

NMFS designated critical habitat for SR Fall-run Chinook salmon on December 28, 1993 (58 FR 68543). It includes the Columbia River reaches presently or historically accessible to listed fall-run Chinook salmon (except river reaches above impassable natural falls, and Dworshak and Hells Canyon Dams) from the estuary upstream to the confluence of the Snake River; all Snake River reaches from the confluence of the Columbia River upstream to Hells Canyon Dam. It also includes the Palouse River from its confluence with the Snake River upstream to Palouse Falls; the Clearwater River from its confluence with the Snake River upstream to its confluence with Lolo Creek; and the North Fork Clearwater River from its confluence with the Clearwater River upstream to Dworshak Dam. Designated areas consist of the water, waterway bottom, and the adjacent riparian zone (defined as an area 300 feet from the normal high water line on each side of the river channel) (58 FR 68543).

Individual watersheds within the ESU have not been evaluated for their conservation value. However, the lower Columbia River corridor is among the areas of high conservation value to the ESU because it connects every population with the ocean and is used by rearing/migrating juveniles and migrating adults. The Columbia River estuary is a unique and essential area for juveniles and adults making the physiological transition between life in freshwater and marine habitats.

Salmon habitat has been altered throughout the ESU through loss of important spawning and rearing habitat and the loss or degradation of migration corridors. The major degraded PCEs within critical habitat designated for SR Fall-run Chinook salmon include: (1) safe passage for juveniles migration which is reduced by mainstem lower Snake and Columbia River hydropower system; (2) rearing habitat water quality altered by influx of contaminants and changing seasonal temperature regimes caused by water flow management; and (3) spawning/rearing habitat PCE attributes (spawning areas with gravel, water quality, cover/shelter, riparian vegetation, and space to support egg incubation and larval growth and development) that are reduced in quantity (80% loss) and quality due to the mainstem lower Snake River hydropower system.

Water quality impairments in the designated critical habitat of this ESU are common within the range of this ESU. Pollutants such as petroleum products, pesticides, fertilizers, and sediment in the form of turbidity enter the surface waters and riverine sediments from the headwaters of the Snake, Salmon, and Clearwater Rivers to the Columbia River estuary 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 the salmon tissue. This species also requires migration corridors with adequate passage conditions (water quality and quantity available at specific times) to allow access to the various habitats required to complete their life cycle.

Snake River Spring/Summer-Run Chinook Salmon

This ESU includes production areas that are characterized by spring-timed returns, summer-timed returns, and combinations from the two adult timing patterns. The SR Spring/Summer-run Chinook ESU includes all naturally spawned populations of spring/summer-run Chinook salmon in the mainstem Snake River and the Tucannon River, Grande Ronde River, Imnaha River, and Salmon River subbasins (57 FR 23458, Figure 9). Fifteen artificial propagation programs are included in the ESU (70 FR 37176). These artificially propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within this ESU.

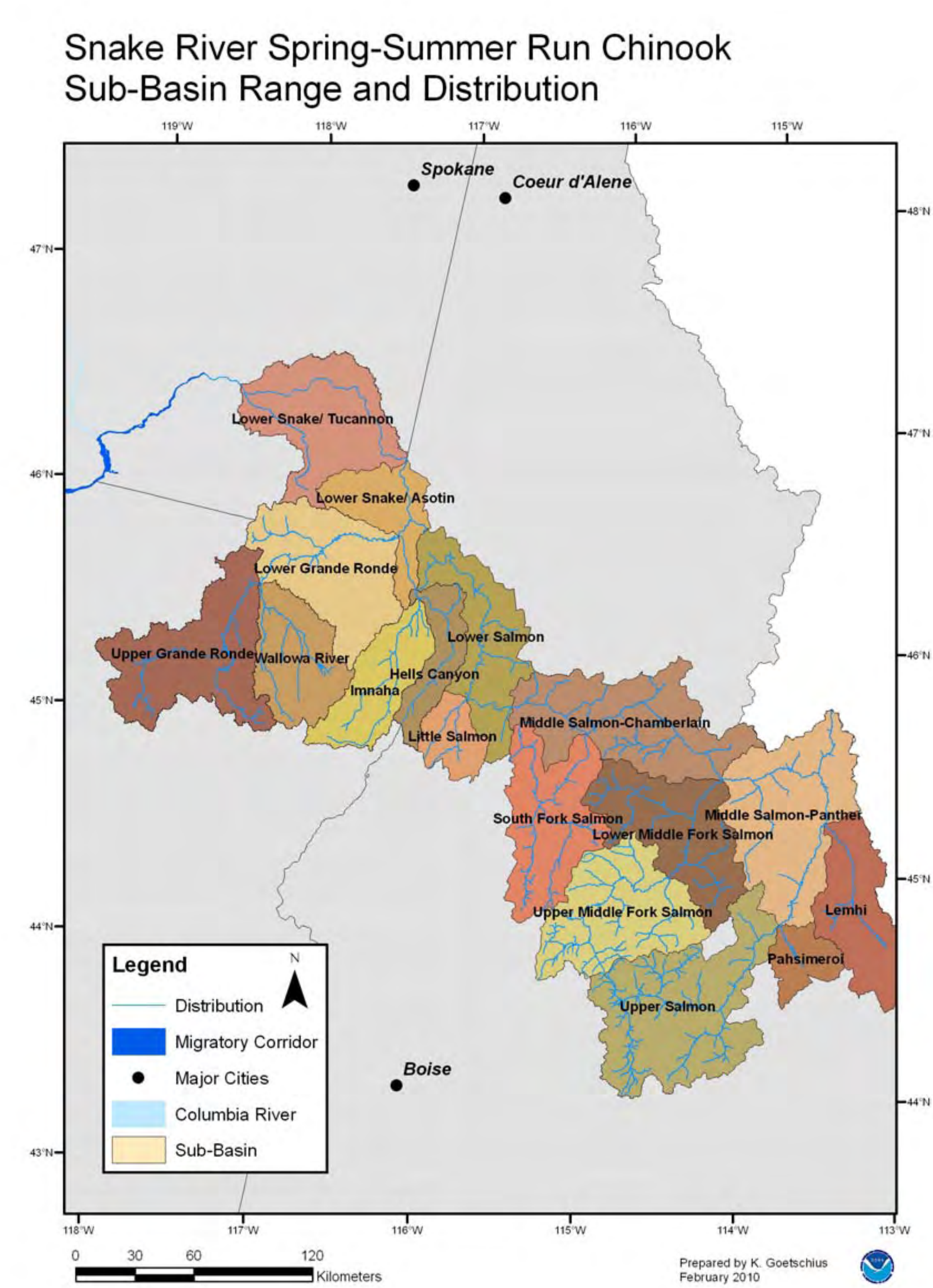


Figure 9. Snake River Spring/Summer-run Chinook salmon distribution.

Life History

Runs classified as spring-run Chinook salmon pass Bonneville Dam beginning in early March to mid-June; runs classified as summer-run Chinook salmon return to the Columbia River from June through August. SR Spring/Summer-run Chinook salmon exhibit a stream-type life history. In general, spring-run type Chinook salmon tend to spawn in higher elevation reaches of major Snake River tributaries while summer-run Chinook salmon tend to spawn lower in the Snake River drainages. However, there is an overlap of summer-run Chinook salmon spawning areas and that of spring-run spawners. Spring-run Chinook salmon spawn in mid- through late August, and summer-run Snake River Chinook salmon spawn approximately one month later than spring-run fish. Eggs incubate over the following winter, and hatch in late winter and early spring of the following year. Juvenile fish mature in fresh water for one year before they migrate to the ocean in the spring of their second year of life. Depending on the tributary and the specific habitat conditions, juveniles may migrate extensively from natal reaches into alternative summer-rearing or overwintering areas. Snake River Spring/Summer-run Chinook salmon return from the ocean to spawn primarily as four and five year-old fish, after two to three years in the ocean.

Status and Trends

NMFS originally listed SR Spring/Summer-run Chinook salmon as threatened on April 22, 1992 (57 FR 14653), and reaffirmed their threatened status on June 28, 2005 (70 FR 37160). The ICBTRT has identified 31 historic populations (Table 21). Historic populations above Hells Canyon Dam are considered extinct (ICTRT 2003). Multiple spawning sites are accessible and natural spawning and rearing are well distributed within the ESU. However, many spawning aggregates have also been extirpated, which has increased the spatial separation of some populations. The South Fork and Middle Fork Salmon Rivers currently support the bulk of natural production in the drainage. Table 21 identifies populations within the Snake River Spring/Summer-run Chinook salmon ESU, their abundances, and hatchery input.

Table 21. Snake River River Spring/Summer-run Chinook salmon populations, abundances, and hatchery contributions (Good et al 2005). Note: rpm denotes redds per mile.

Current Populations	Historical Abundance	Mean Number of Spawners (Range)	Hatchery Abundance Contributions
Tucannon River	Unknown	303 (128-1,012)	76%
Wenaha River	Unknown	225 (67-586)	64%
Wallowa River	Unknown	0.57 redds (0-29)	5%
Lostine River	Unknown	34 redds (9-131)	5%
Minam River	Unknown	180 (96-573)	5%
Catherine Creek	Unknown	50 (13-262)	56%
Upper Grande Ronde River	Unknown	46 (3-336)	58%
Imnaha River	Unknown	564 redds (194-3,041)	62%
Big Sheep Creek	Unknown	0.25 redds (0-1)	97%
Little Salmon	Unknown	Unknown	Unknown
South Fork Salmon River	Unknown	496 redds (277-679)	9%
Secesh River	Unknown	144 redds (38-444)	4%
Johnson Creek	Unknown	131 redds (49-444)	0%
Big Creek spring run	Unknown	53 redds (21-296)	0%
Big Creek summer run	Unknown	5 redds (2-58)	Unknown
Loon Creek	Unknown	27 redds (6-255)	0%
Bear Valley/Elk Creek	Unknown	266 (72-712	0%
Marsh Creek	Unknown	53 (0-164)	0%
North Fork Salmon River	Unknown	5.6 redds (2-19)	Unknown
Lemhi River	Unknown	72 redds (35-216)	0%
Pahsimeroi River	Unknown	161 (72-1,097	Unknown
East Fork Salmon spring run	Unknown	0.27 rpm (0.2 – 1.41)	Unknown
East Fork Salmon summer run	Unknown	1.22 rpm (0.35 – 5.32)	0%
Yankee Fork spring run	Unknown	0	Unknown
Yankee Fork summer run	Unknown	2.9 redds (1-18)	0%
Valley Creek spring run	Unknown	7.4 redds (2-28)	0%
Valley Creek summer run	Unknown	2.14 rpm (0.71 – 9.29)	Unknown
Upper Salmon spring run	Unknown	69 redds (25-357)	Unknown
Upper Salmon summer run	Unknown	0.24 rpm (0.07 – 0.58)	Unknown
Alturas Lake Creek	Unknown	2.7 redds (0-18)	Unknown
Lick Creek	Unknown	1.44 redds (0-29)	59%
ESU Estimate	~1.5 million	~9,700	

According to Matthews and Waples (1991) total annual SR Spring/Summer-run Chinook salmon production may have exceeded 1.5 million adult fish in the late 1800s. Total (natural plus hatchery origin) returns fell to roughly 100,000 spawners by the late 1960s

(Fulton 1968). Between 1981 and 2000, total returns fluctuated between extremes of 1,800 and 44,000 fish. The 2001 and 2002 total returns increased to over 185,000 and 97,184 adults, respectively.

Abundance of summer run Chinook salmon have increased since the low returns in the mid-1990s (lowest run size was 692 fish in 1995). The 1997 to 2008 geometric mean total return for the summer run component at Lower Granite Dam was slightly more than 8,700 fish, compared to the geometric mean of 3,076 fish for the years 1987 to 1996 (Data from the Columbia Basin Fisheries Agencies and Tribes http://www.fpc.org/). However, over 80% of the 2001 return and over 60% of the 2002 return originated from hatcheries (Good et al 2005). Good *et al.* (2005) reported that risks to individual populations within the ESU may be greater than the extinction risk for the entire ESU due to low levels of annual abundance of individual populations. Further, despite the increase in abundance during the last ten years, annual abundance continues to be variable and is most pronounced in natural-origin fish. Thus, although the average abundance in the most recent decade is higher than the previous decade, there is no obvious long-term trend (Good et al 2005) (Data from the Columbia Basin Fisheries Agencies and Tribes http://www.fpc.org/). However, recent trends, buoyed by the last five years, are approaching 1. Additionally, hatchery fish are faring better than wild fish, which comprise roughly 40% of the total returns in the past decade. Overall, most populations are far below their respective interim recovery targets.

There is no evidence of wide-scale genetic introgression by hatchery populations. The high variability in life history traits indicates sufficient genetic variability within the ESU to maintain distinct subpopulations adapted to local environments (Good et al 2005).

Critical Habitat

NMFS designated critical habitat for the SR Spring/Summer-run Chinook salmon on October 25, 1999 (64 FR 57399). This critical habitat encompasses the waters, waterway bottoms, and adjacent riparian zones of river reaches of the Columbia, Snake, and Salmon Rivers, and all tributaries of the Snake and Salmon Rivers, that are or were accessible to listed Snake River salmon (except reaches above impassable natural falls, and Dworshak and Hells Canyon Dams).

NMFS identified spawning, rearing, and migration as PCEs for the SR Spring/Summer-run Chinook salmon. Spawning and juvenile rearing essential features consist of adequate (1) spawning gravel, (2) water quality, (3) water quantity, (4) water temperature, (5) riparian vegetation, (6) food, (7) cover/shelter, and (8) space. Juvenile and adult migration corridor essential features consist of adequate (1) substrate, (2) water quality, (3) water quantity, (4) water temperature, (5) food (juveniles only), (6) riparian vegetation, and (7) access.

Watersheds within the critical habitat designated for the SR Spring/Summer-run Chinook salmon have not been evaluated for their conservation value. However, the lower Columbia River corridor is among the areas of high conservation value to the ESU because it connects every population with the ocean and is used by rearing/migrating juveniles and migrating adults.

Spawning and juvenile rearing PCEs are regionally degraded by changes in flow quantity, water quality, and loss of cover. Juvenile and adult migrations are obstructed by reduced access that has resulted from altered flow regimes from hydroelectric dams. According to the ICBTRT, the Panther Creek population was extirpated because of legacy and modern mining-related pollutants creating a chemical barrier to fish passage (Chapman and Julisu 2005).

Presence of cool water that is relatively free of contaminants is particularly important for the spring/summer run life history as adults hold over the summer and juveniles may rear for a whole year in the river. Water quality impairments are common in the range of the critical habitat designated for this ESU. Pollutants such as petroleum products, pesticides, fertilizers, and sediment in the form of turbidity enter the surface waters and riverine bottom substrate from the headwaters of the Snake, Salmon, and Clearwater Rivers to the Columbia River estuary 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 the salmon tissue. This species also requires migration corridors with adequate passage conditions (water quality and quantity available at specific times) to allow access to the various habitats required to complete their life cycle.

Upper Willamette River Chinook Salmon

The Upper Willamette River (UWR) Chinook salmon ESU includes all naturally spawned populations of spring-run Chinook salmon in the Clackamas River and in the Willamette River, and its tributaries, above Willamette Falls, Oregon (64 FR 14208 Figure 10)**.** Seven artificial propagation programs are included in the ESU (70 FR 37160). These artificially propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within the ESU.

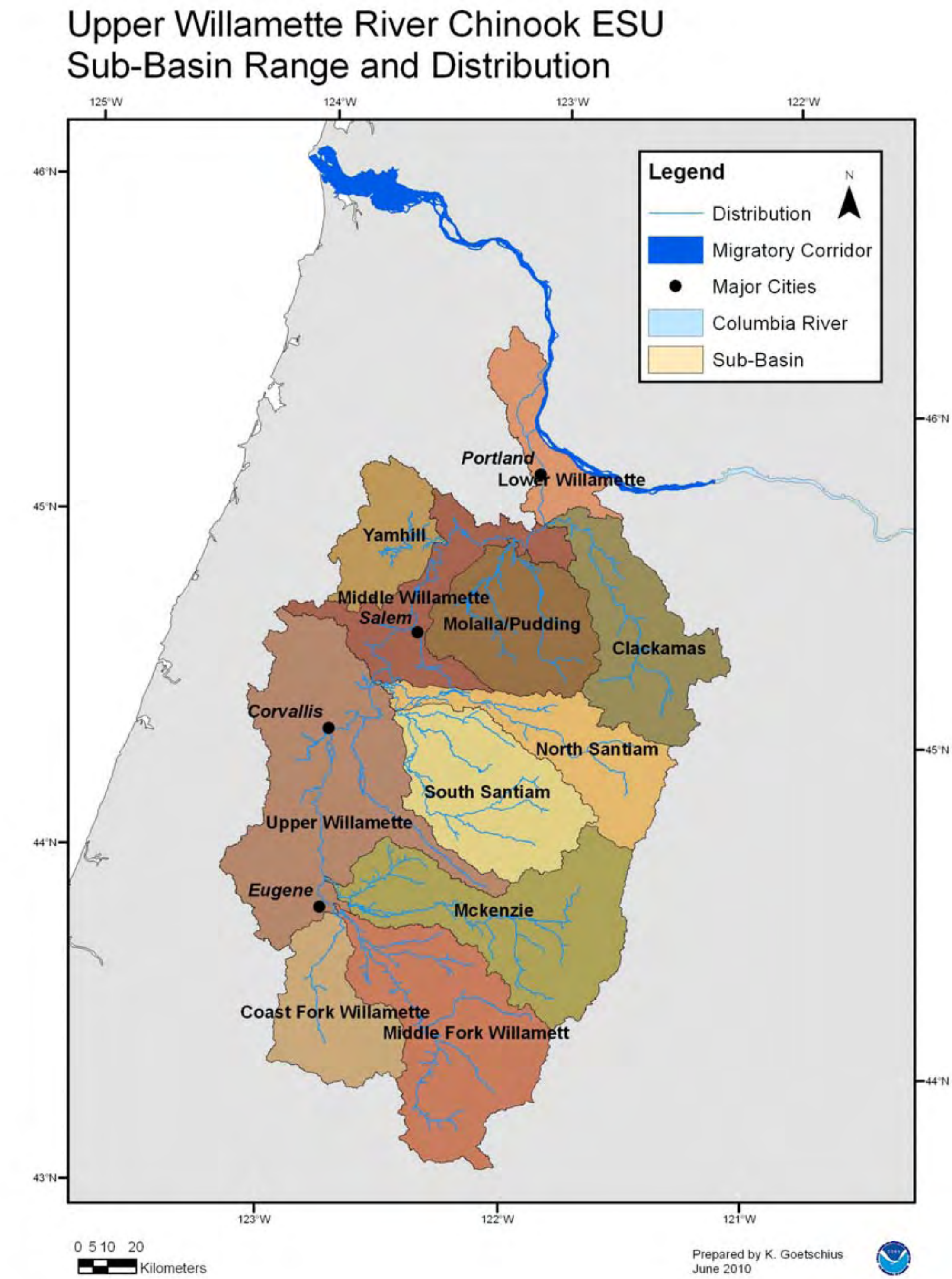


Figure 10. Upper Willamette River Chinook salmon distribution

Life History

UWR Chinook salmon exhibit an earlier time of entry into the Columbia River than other spring-run Chinook salmon ESUs (Myers et al 1998). Adults appear in the lower Willamette River in February, but the majority of the run ascends Willamette Falls in April and May, with a peak in mid- to late May. However, present-day salmon ascend the Willamette Falls via a fish ladder. Consequently, the migration of spring Chinook salmon over Willamette Falls extends into July and August (overlapping with the beginning of the introduced fall-run of Chinook salmon).

The adults hold in deep pools over summer and spawn in late fall or early winter when winter storms augments river flows. Fry may emerge from February to March and sometimes as late as June (Myers et al 2006). Juvenile migration varies with three distinct juvenile emigration “runs”: fry migration in late winter and early spring; sub-yearling (0 yr +) migration in fall to early winter; and yearlings (1 yr +) migrating in late winter to spring. Sub-yearlings and yearlings rear in the mainstem Willamette River where they also use floodplain wetlands in the lower Willamette River during the winter-spring floodplain inundation period.

Status and Trends

NMFS originally listed UWR Chinook salmon as threatened on March 24, 1999 (64 FR 14308), and reaffirmed their threatened status on June 28, 2005 (70 FR 37160). Historically, this ESU included sizable numbers of spawning salmon in the Santiam River, the middle fork of the Willamette River, and the McKenzie River, as well as smaller numbers in the Molalla River, Calapooia River, and Albiqua Creek. Table 22 identifies populations within the UWR Chinook salmon ESU, their abundances, and hatchery input.

The W/LCRTRT identified seven historical independent populations (Myers 2006) (Table 22). Most natural spring Chinook salmon populations of this ESU are likely extirpated or nearly so. The spring Chinook salmon in the McKenzie River is the only remaining naturally reproducing population in this ESU. Current spatial distribution is reduced by the loss of 30 to 40% of the total historic habitat which has restricted spawning to a few areas below dams.

Table 22. Upper Willamette River Chinook salmon independent populations core (C) and genetic legacy (G) populations, and hatchery contributions (Good et al 2005)

Functionally Independent Populations	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Clackamas River	Unknown	2,910	64%
Molalla River	Unknown	52 redds	>93%
North Santiam River	Unknown	~ 7.1 rpm	>95%
South Santiam River	Unknown	982 redds	>84%
Calapooia River	Unknown	16 redds	100%
McKenzie River	Unknown	~2,470	26%
Middle Fork Willamette River	Unknown	235 redds	>39%
Total	>70,000	~9,700	Mostly hatchery

Note: rpm denotes redds per mile

The total abundance of adult spring-run Chinook salmon (hatchery-origin + natural-origin fish) passing Willamette Falls has remained relatively steady over the past 50 years (ranging from approximately 20,000 to 70,000 fish). However, the current abundance is an order of magnitude below the peak abundance levels observed in the 1920s (approximately 300,000 adults). Total number of fish increased during the period from 1996 to 2004 when it peaked at more than 96,000 adult spring-run Chinook salmon passing Willamette Falls. Since then, the run has steadily decreased with only about 14,000 fish counted in 2008, the lowest number since 1960. ESU abundance increased again to about 25,000 adult spring-run Chinook salmon in 2009. Runs consist of a high but uncertain fraction of hatchery-produced fish.

The spring Chinook salmon in the McKenzie River is the only remaining self sustaining naturally reproducing independent population. The other natural-origin populations in this ESU have very low current abundances, and long- and short-term population trends are negative.

Access of fall-run Chinook salmon to the upper Willamette River and the mixing of hatchery stocks within the ESU have threatened the genetic integrity and diversity of the species. Much of the genetic diversity that existed between populations has been homogenized (Myers et al 2006).

Critical Habitat

NMFS designated critical habitat for this species on September 2, 2005 (70 FR 52630). Designated critical habitat includes all Columbia River estuarine areas and river reaches proceeding upstream to the confluence with the Willamette River as well as specific stream reaches in a number of subbasins.

NMFS assessed the conservation value of 59 watersheds within the range of the UWR Chinook salmon (Table 23). Nineteen watersheds received a low rating, 18 received a medium rating, and 22 received a high rating of conservation value to the ESU (NMFS 2005a). The lower Willamette/Columbia River rearing/migration corridor downstream of the spawning range is also considered to have a high conservation value and is the only habitat designated in four of the high value watersheds.

Table 23. UWR Chinook salmon 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]
Middle Fork Willamette	4	(1)	6	(2, 1)	0	
Coastal Fork Willamette	0		0		4	(2, 1)
Upper Willamette	0		3	(2, 1)	3	(2)
McKenzie	5	(1, 2)	2	(2, 1)	0	
North Santiam	2	(1)	1	(2, 1)	0	
South Santiam	3	(1, 2)	3	(2, 1)	0	
Middle Willamette	0		0		4	(2)
Yamhill	0		0		4	(2)
Molalla/Pudding	0		3	(1, 2)	3	(2)
Clackamas	5	(1)[2]	0		1	(1)
Lower Willamette	3	(2)	0		0	
Columbia River Corridor	all	(3)	0		0	
Total	22		18		19	

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.
2 .Lower Clackamas River provides for 13.4 miles of PCE 2

The current condition of PCEs of the UWR Chinook salmon critical habitat indicates that migration and rearing PCEs are not currently functioning or are degraded. These conditions impact their ability to serve their intended role for species conservation. The migration PCE is degraded by dams altering migration timing and water management altering the water quantity necessary for mobility and survival. Migration, rearing, and estuary PCEs are also degraded by loss of riparian vegetation and instream cover. Pollutants such as petroleum products, fertilizers, pesticides, and fine sediment enter the stream through runoff, point source discharge, drift during application, and non-point discharge where agricultural and urban development occurs. Degraded water quality in the lower Willamette River where important floodplain rearing habitat is present affects the ability of this habitat to sustain its role to conserve the species.

California Coastal Chinook Salmon

California Coastal (CC) Chinook salmon includes all naturally-spawned coastal Chinook salmon spawning north from Redwood Creek to, and including, the Russian River to the south as shown in Figure 11. Seven artificial propagation programs are part of this ESU. These artificially propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within this ESU.

Life History

CC Chinook salmon are a fall-run, ocean-type fish. Although a spring-run (river-type) component existed historically, it is now considered extinct (Bjorkstedt et al 2005). The different populations vary in run timing depending on latitude and hydrological differences between watersheds. Entry of CC Chinook salmon into the Russian River depends on increased flow from fall storms, usually in November to January. Juveniles of this ESU migrate downstream from April through June and may reside in the estuary for an extended period before entering the ocean.

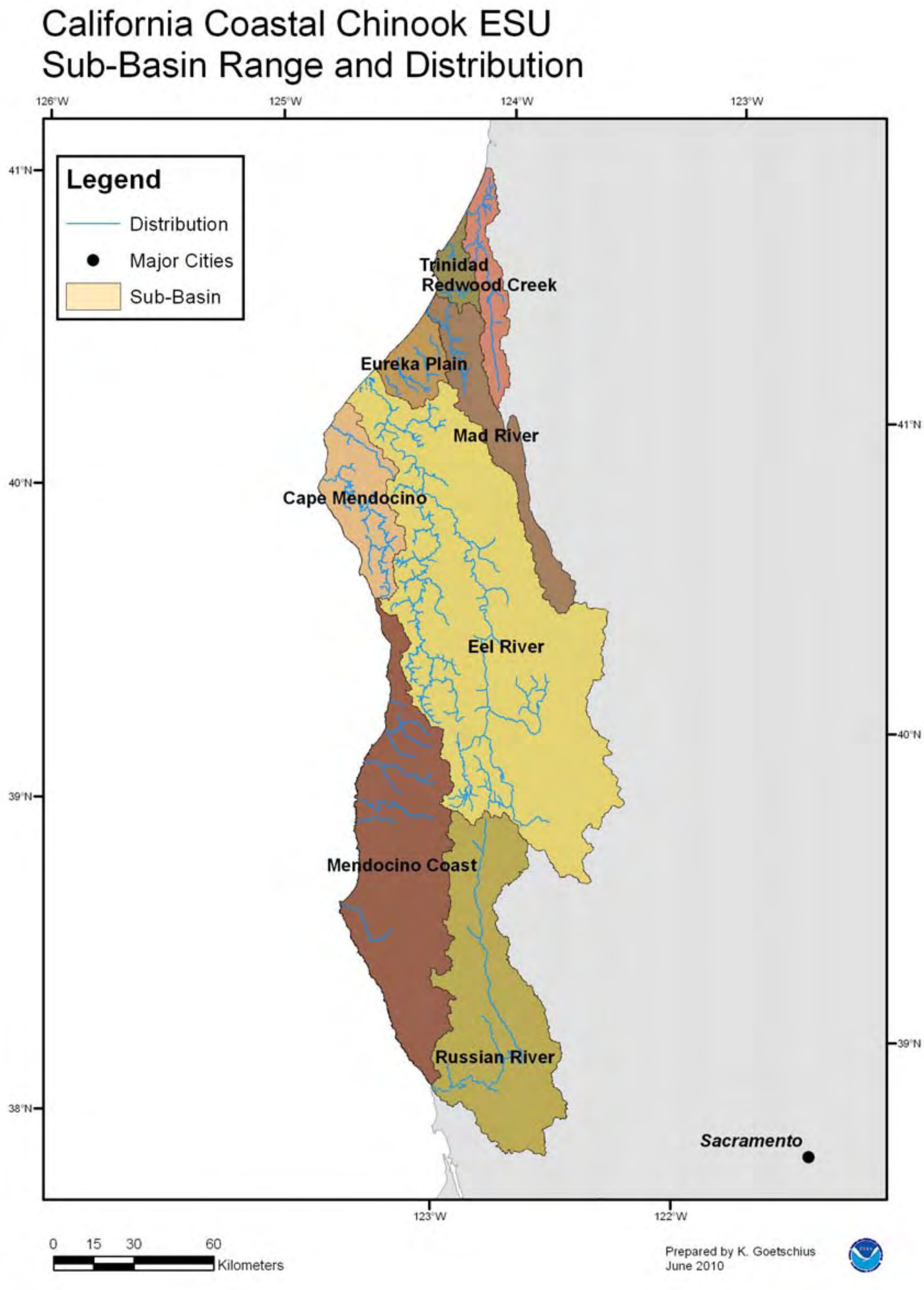


Figure 11. California Coastal Chinook salmon distribution

Table 24. California Coastal Chinook salmon fall-run populations-preliminary population structure, abundances, and hatchery contributions (Good et al 2005)

Population	Historic Spawner Abundance	Mean Number of Spawners	Hatchery Abundance Contributions
Eel River (includes * tributaries below) – 2 populations		156-2,730	~30%
Mainstem Eel River*	13,000	Inc. in Eel River	Unknown
Van Duzen River*	2,500	Inc. in Eel River	Unknown
Middle Fork Eel River*	13,000	Inc. in Eel River	Unknown
South Fork Eel River*	27,000	Inc. in Eel River	Unknown
North Fork Eel River*	Unknown	Inc. in Eel River	Unknown
Upper Eel River*	Unknown	Inc. in Eel River	Unknown
Redwood Creek	1,000-5,000	Unknown	0
Mad River	1,000-5,000	19-103	Unknown
Bear River	100	Unknown	0
Mattole River	1,000-5,000	Unknown	17%
Small Humboldt County rivers	1,500	Unknown	0
Rivers north of Mattole River	600	Unknown	0
Humbolt Bay tributaries	40	120	40 (33%)
Noyo River	50	Unknown	0
Russian River	50-500	>1,383 – >6,103	~0%
Tenmile to Gualala coastal effluents	Unknown	Unknown	0
Total	20,750-72,550	Unknown	

Status and Trends

NMFS listed CC Chinook salmon as threatened on September 16, 1999 (64 FR 50393), and reaffirmed their threatened status on June 28, 2005 (70 FR 37160). The CC Chinook ESU historically consisted of 10 functionally independent populations and 5 potentially independent populations (Bjorkstedt et al 2005). Seventeen basins may have had Chinook salmon runs that relied on immigration from the larger basins. ESU connectivity is substantially reduced by the near extirpation of all historically independent populations between the Russian River in Sonoma County and Mattole River in Humboldt County (NMFS 2008a, Spence et al 2008). The number of extant populations is uncertain.

Historical estimates of escapement suggest abundance was roughly 73,000 in the early 1960s, with the majority of fish spawning in the Eel River, and about 21,000 in the 1980s (Good et al 2005). Table 24 identifies populations within the CC Chinook salmon ESU, their abundances, and hatchery input.

Comparison of historical and current abundance information indicates that independent populations of Chinook salmon are depressed in many basins (Bennet 2005, Good et al 2005, NMFS 2008a). All spring-run populations once occupying the North Mountain Interior are considered extinct or nearly so. Redd counts in Mattole River in the northern portion of the ESU indicate a small but consistent population; the cooler northern climate likely provides for favorable conditions for these populations (Spence et al 2008). The Eel River interior fall-run populations are severely depressed (Spence et al 2008) Two functionally independent populations are believed to have existed along the southern coastal portion of the ESU; of these two, only the Russian River currently has a run of any significance (Bjorkstedt et al 2005). This is also the only population with abundance time series. The 2000 to 2007 median observed (at Mirabel Dam) Russian River Chinook salmon run size is 2,991 with a maximum of 6,103 (2003) and a minimum of 1,125 (2008) adults (Cook 2008, Sonoma County Water Agency 2008) The number of spawners has steadily decreased since its high returns in 2003 with 1,963 fish observed in 2007 and 1,125 observed by December 22, 2008. The time series is too short to estimate lambda.

The CC Chinook ESU is at considerable risk from population fragmentation and reduced spatial diversity. There is little connectivity between the southern and northern portions of their range. At the southern portion of the ESU, only the Russian River population has had a constant run that exceeded 1,000 adult spawning fish over the last 10 years. This places the ESU at risk from random catastrophic events, chronic stressors, and long-term environmental change. Life history diversity has been significantly reduced by loss of the spring-run race and reduction in coastal populations

Critical Habitat

NMFS designated critical habitat for the CC Chinook salmon on September 2, 2005 (70 FR 52488). It includes multiple CALWATER hydrological units north from Redwood Creek and south to Russian River (Table 25). The total area of critical habitat includes 1,500 miles of stream habitat and about 25 square miles of estuarine habitat, mostly within Humboldt Bay. A list and maps of watersheds and streams designated as critical habitat for CC Chinook salmon can be found in the Federal Register (70 FR 52488).

There are 45 occupied CALWATER Hydrologic Subarea (HSA) watersheds within the freshwater and estuarine range of this ESU. Eight watersheds received a low rating, 10 received a medium rating, and 27 received a high rating of conservation value to the ESU (70 FR 52488). Two estuarine habitat areas used for rearing and migration (Humboldt Bay and the Eel River Estuary) also received a high conservation value rating.

Table 25. CC 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]
Redwood Creek	2	(1, 2, 3)	1	(1, 2, 3)	0	
Trindad	1	(1, 2, 3)	0		1	(1, 2, 3)
Mad River	3	(1, 2, 3)	0		0	
Eureka Plain	1	(1, 2, 3)	0		0	
Eel River	12	(1, 2, 3)	4	(1, 2, 3)	3	(1, 2, 3)
Cape Mendocino	2	(1, 2, 3)	0		0	
Mendocino Coast	2	(1, 2, 3)	3	(1, 2, 3)	2	(1, 2, 3)
Russian River	4	(1, 2, 3)	2	(1, 2, 3)	2	(1, 2, 3)
Total	**27**		**10**		**8**	

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.

Critical habitat in this ESU consists of limited quantity and quality summer and winter rearing habitat, as well as marginal spawning habitat. Compared to historical conditions, there are fewer pools, limited cover, and reduced habitat complexity. The current condition of PCEs of the CC 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. CC Chinook salmon spawning PCE in coastal streams is degraded by years of timber harvest that has produced large amounts of sand and silt in spawning gravel and reduced water quality by increased turbidity. Agriculture and urban areas has impacted rearing and migration PCEs in the Russian River by degrading water quality and by disconnecting the river from it floodplains by the construction of levees. Water management from dams within the Russian and Eel River watersheds maintain high flows and warm water during summer which benefits the introduced predatory Sacramento pikeminnow. This has resulted in excessive predation along migration corridors. Breaches of the sandbar at the mouth of the Russian River result in periodic mixing of salt water. This condition degrades the estuary PCE by altering water quality and salinity conditions that support juvenile physiological transitions between fresh- and salt water.

Central Valley Spring-run Chinook Salmon

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 12).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 26 identifies populations within the CV Spring-run Chinook salmon ESU, their abundances, and hatchery input.

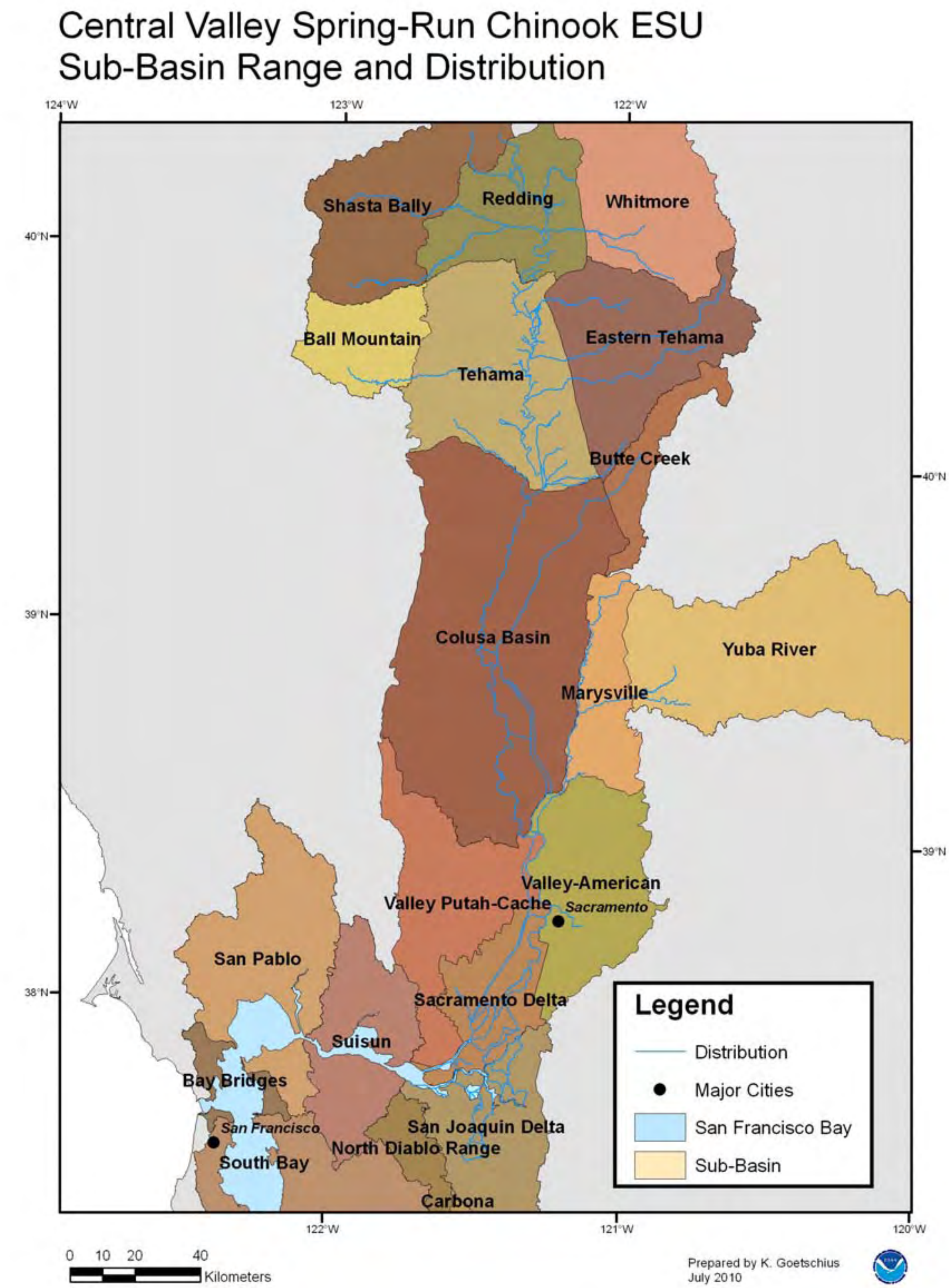


Figure 12. Central Valley Spring-run Chinook salmon distribution

Life History

CV Spring-run Chinook salmon enter the Sacramento River from March to September and spawn from late August through early October, with a peak in September. Chinook salmon require cool fresh water while they mature over the summer. Adult upstream migration may be blocked by temperatures above 21ºC (McCullough 1999). Fry emerge from the gravel November to March. 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.

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). 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. 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 1998).

Table 26. 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 & Reclamation 2007)

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	Unknown
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

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 & 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 & Reclamation 2007).

The Sacramento River trends and lambda show a long- and short- term negative trend and negative population growth (Good et al 2005). Meanwhile, 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 & Reclamation 2007)). Time series data for Mill, Deer, Butte, and Big Chico Creeks spring-run Chinook salmon (updated through 2006) show that all three tributary spring-run Chinook populations have long-and short-term lambdas >1; indicating population growth (Good et al 2005). Although the populations are small, CV spring-run Chinook salmon have some of the highest population growth rates in the Central Valley.

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 27).

There are 38 occupied HSA watersheds within the freshwater and estuarine range of this ESU. Seven watersheds received a low rating, 3 received a medium rating, and 27 received a high rating of conservation value to the ESU (NMFS 2005c). 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 27. 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.

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.

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.

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 the Sacramento River, including all its tributaries, to below Keswick Dam (Figure 13). The ESU now consists of a single spawning population.

Life History

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 delays spawning until May and June. Fry emerge from the gravel in late June to early July and continue through October (Fisher 1994). Young winter-run Chinook salmon start migrating to sea as early as mid July with a peak movement over the Red Bluff Diversion Dam (RBDD) in September. Some offspring move downstream as fry while other rear in the upper Sacramento River and move down as smolt. Normally fry have passed the RBDD by October while smolts may pass over the RBDD until March. Juvenile winter-runs occur in the Delta primarily from November through 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). The winter-run race matures between two and six years of age with the majority returning as three-year olds.

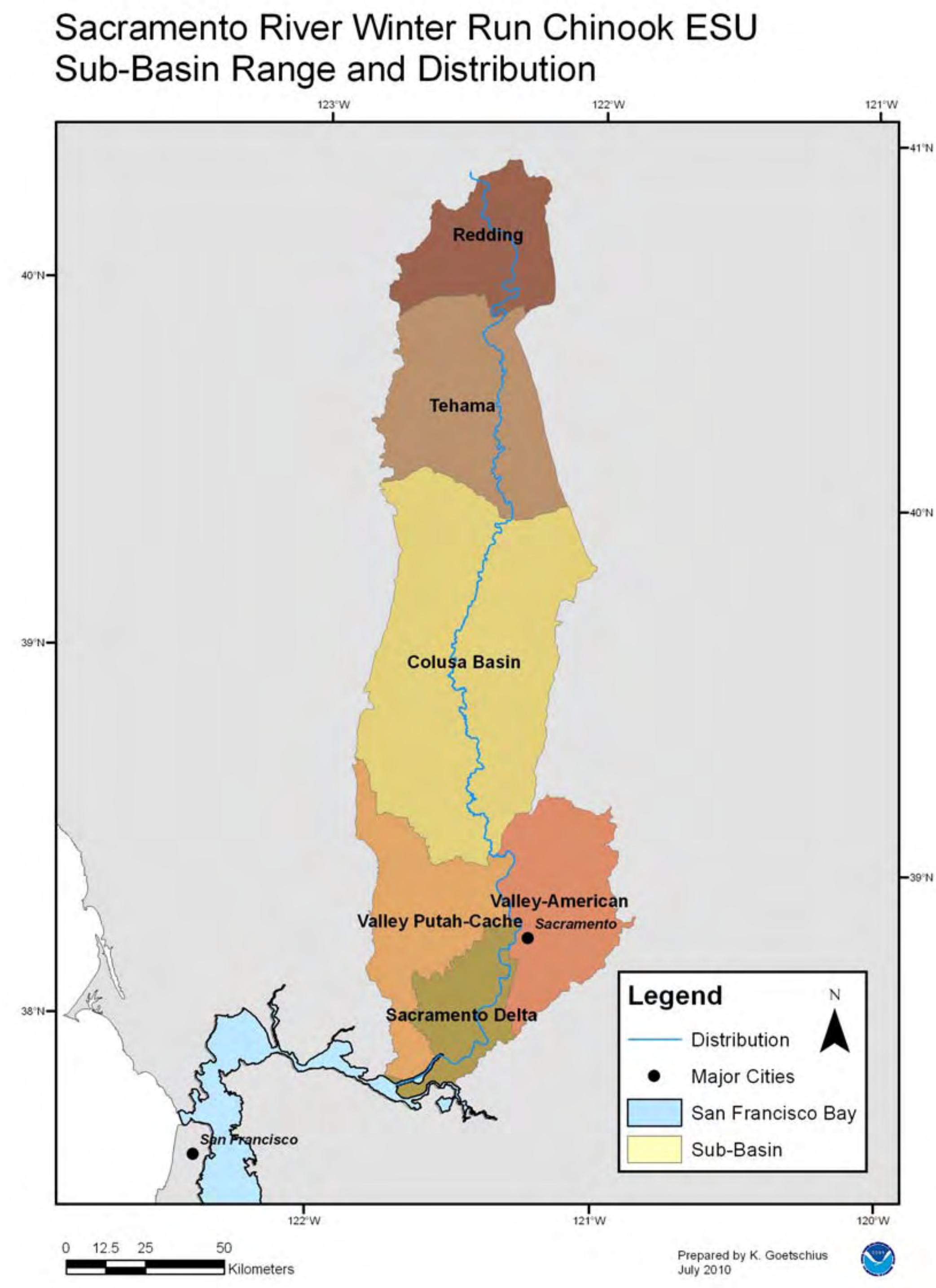


Figure 13. Sacramento River Winter-run Chinook salmon distribution

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 1963, Yoshiyama 1998). Today the Shasta Dam eliminates access to the historic spawning habitat. Cold water releases from the 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. As a result, the Sacramento River Winter-run Chinook salmon has been reduced to a single spawning population confined to a portion of the mainstem Sacramento River.

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 (data from CDFG 2008). Critically low levels were reached during the drought of 1987 to 1992 with an absolute bottom of 191 fish counted. The three-year average run size for the period of 1989 to 1991 was 388 fish.

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 (data from USFWS 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 2,600-2,950 adults for 2008 (USFWS 2008 unpublished, Garwin Yip pers. com).

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.

Critical Habitat

NMFS designated critical habitat for this species on June 16, 1993 (58 FR 33212). It includes: the Sacramento River from Keswick Dam, Shasta County (river mile 302) to Chipps Island (river mile 0) at the westward margin of the Sacramento-San Joaquin Delta, and other specified estuarine waters.

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.

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 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.

Chum Salmon

Description of the Species

Chum salmon have the widest natural geographic and spawning distribution of any Pacific salmonid as their range extend farther along the shores of the Arctic Ocean than other salmonids. Chum salmon have been documented to spawn from Korea and the Japanese island of Honshu, east around the rim of the North Pacific Ocean to Monterey Bay, California. Historically, chum salmon were distributed throughout the coastal regions of western Canada and the U.S. Presently, major spawning populations occur as far south as Tillamook Bay on the northern Oregon coast. We discuss the distribution, life history diversity, status, and critical habitat of the two species of threatened chum salmon separately.

Chum salmon are semelparous, spawn primarily in fresh water, and exhibit obligatory anadromy (there are no recorded landlocked or naturalized freshwater populations). Chum salmon spend two to five years in feeding areas in the northeast Pacific Ocean, which is a greater proportion of their life history than other Pacific salmonids. Chum salmon are distributed throughout the North Pacific Ocean and Bering Sea.

North American chum salmon migrate north along the coast in a narrow coastal band that broadens in southeastern Alaska. However, some data suggest that Puget Sound chum, including Hood Canal Summer-run chum, may not migrate into northern British Columbian and Alaskan waters. Instead, Puget Sound chum salmon travel directly offshore into the North Pacific Ocean.

Chum salmon usually spawn in the lower reaches of rivers. Redds are dug in the mainstem or in side channels of rivers from just above tidal influence to nearly 100 km from the sea. The time to hatching and emergence from the gravel redds are influenced by DO, gravel size, salinity, nutritional conditions, behavior of alevins in the gravel, and incubation temperature (reviewed Bakkala 1970, Salo 1991, Schroder 1977, Schroder et al 1974). For example, fertilized eggs hatch in about 100-150 days at 4°C, but hatch in only 26-40 days at 15°C. Juveniles outmigrate to sea water almost immediately after emerging from the gravel that covers their redds (Salo 1991). The immature salmon distribute themselves widely over the North Pacific Ocean. The maturing adults return to the home streams at various ages, usually at two through five years, and in some cases up to seven years (Bigler 1985). This ocean-type migratory behavior contrasts with the stream-type behavior of some other species in the genus *Oncorhynchus* (*e.g.*, steelhead, coho, and most types of Chinook and sockeye salmon). Stream-type salmonids usually migrate to sea at a larger size, after months or years of freshwater rearing. Thus, survival and growth for juvenile chum salmon depend less on freshwater conditions than on favorable estuarine conditions. Another behavioral difference between chum salmon and other salmonid species is that chum salmon form schools. Presumably, this behavior reduces predation (Pitcher 1986) especially if fish movements are synchronized to swamp predators (Miller and Brannon 1982).

The duration of estuarine residence for chum salmon juveniles are known for only a few estuaries. Observed residence time ranged from 4 to 32 days, with about 24 days as the most common (Johonson et al 1997). Chum salmon juveniles use shallow, low flow habitats for rearing that include inundated mudflats, tidal wetlands and their channels, and sloughs.

Status and Trends

Chum salmon, like the other salmon NMFS has listed, have declined from overharvests, hatcheries, native and non-native exotic species, dams, gravel mining, water diversions, destruction or degradation of riparian habitats, and land use practices (logging, agriculture, and urbanization). Chum salmon are also affected by shifts in climatic conditions that alter patterns and intensity of precipitation.

Hood Canal Summer-run Chum Salmon

The Hood Canal (HC) Summer-run chum salmon ESU includes all naturally spawned populations in Hood Canal and its tributaries as well as populations in Olympic Peninsula rivers between Hood Canal and Dungeness Bay, Washington (Figure 14, 64 FR 14508). Eight artificial propagation programs are included in the ESU: the Quilcene National Fish Hatchery, Hamma Hamma Fish Hatchery, Lilliwaup Creek Fish Hatchery, Union River/Tahuya, Big Beef Creek Fish Hatchery, Salmon Creek Fish Hatchery, Chimacum Creek Fish Hatchery, and the Jimmycomelately Creek Fish Hatchery summer-run chum hatchery programs. These artificially propagated populations are no more divergent relative to the local natural populations(s) than what would be expected between closely related natural populations within the species. Table 28 identifies populations within the HC Summer-run chum salmon ESU, their abundances, and hatchery input.

Table 28. Hood Canal Summer-run Chum salmon populations, abundances, and hatchery contributions (Good et al 2005).

Historically Independent Populations	Stocks (Streams)	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Strait of Juan de Fuca	Chimacum Creek	Unknown	Extinct	N/A
	Dungeness Creek	Unknown	Unknown	Unknown
	Jimmycomelately Creek	Unknown	~60	Unknown
	Salmon/Snow creeks	Unknown	~2,200	0-69%
Hood Canal	Big/Little Quilcene rivers	Unknown	~4,240	5-51%
	Dosewallips River	Unknown	~900	Unknown
	Duckabush River	Unknown	Unknown	Unknown
	Hamma Hamma River	Unknown	~758	Unknown
	Lilliwaup Creek	Unknown	~164	Unknown
	Skokomish River	Unknown	Extinct	N/A
	Big Beef Creek*	Unknown	Extinct	100
	Dewetto Creek*	Unknown	Extinct	Unknown
	Anderson Creek*	Unknown	Extinct	N/A
	Mission Creek*	Unknown	Extinct	N/A
	Tahuya River*	Unknown	Extinct	N/A
	Union River*	Unknown	~690	Unknown

* Streams on the east side of Hood Canal.

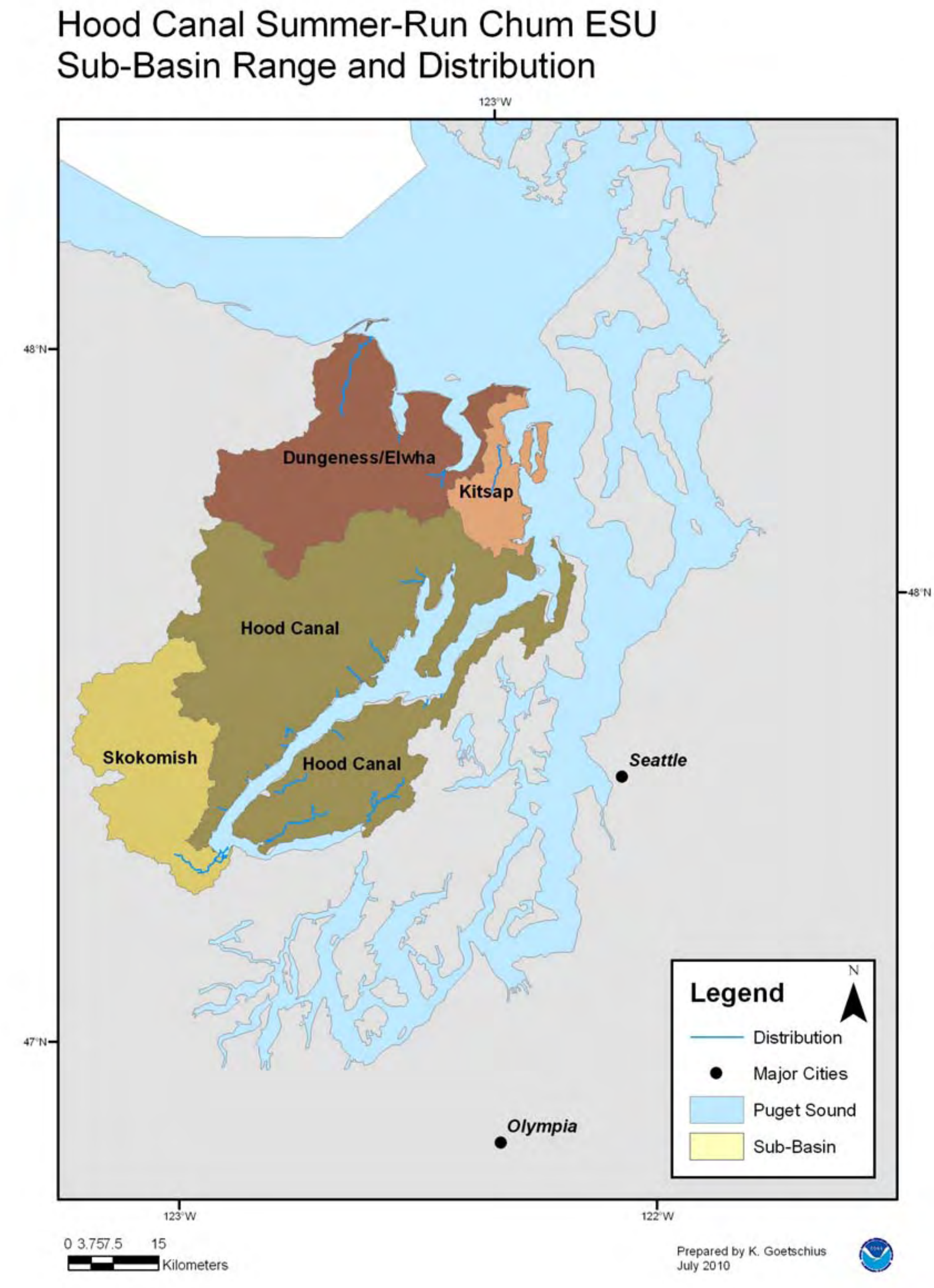


Figure 14. Hood Canal Summer-run Chum salmon distribution.

Life History

Run-timing data from as early as 1913 indicated temporal separation between summer- and fall-run chum salmon in Hood Canal (Johnson et al 1997). The HC Summer-run chum salmon enter natal rivers by late August until October (WDFW 1993). Spawning occurs from mid-September through mid-October. Adults generally spawn in low gradient, lower mainstem reaches of natal streams, typically in center channel areas due to the low flows encountered in the late summer and early fall. Eggs incubate in redds for five to six months and fry emerge between January and May. After hatching, fry move rapidly downstream to subestuarine habitats. HC Summer-run chum salmon seem to have a longer incubation time than fall-run chum salmon in the same streams. Consequently, offspring of summer-run chum salmon have lower average weight and less lipid content than offspring of fall-run chum salmon. Thus, prey availability during their early life history is important for fry survival.

HC Summer-run chum salmon juveniles quickly migrate up the Hood Canal and into the main body of Puget Sound starting in February/March (Johnson et al 1997). The juveniles rear for an average of 23 days in the subestuary deltas which support a diverse array of habitats (tidal channels, mudflats, marshes, and eelgrass meadows). These habitats provide essential rearing and transition environments for this ESU and juveniles rear in these habitats before entering the ocean. Fry in Hood Canal have not been observed to display daily tidal migrations (Bax 1983). Fry movement is associated with prey availability. Juveniles feed primarily on plankton and epibenthic organisms, while subadults feed on similar items as well as larger prey (including fishes and squid).

Fish may emerge from streams over an extended period; some juveniles may remain in Quilcene Bay for several weeks. Most adults return as spawners as three- and four-year old fish.

Status and Trends

NMFS listed HC Summer-run chum salmon as threatened on March 25, 1999 (64 FR 14508), and reaffirmed their threatened status on June 28, 2005 (70 FR 37160). The HC extant summer-run chum ESU consists of two historic independent populations (the Strait of Juan de Fuca and Hood Canal populations) that together were constituted of an estimated 16 historic stocks (Sands et al 2007). Of the 16 historic stocks, seven are considered extirpated. With the extirpation of many local stocks, much of the historical spatial structure has been lost on both the population and the ESU level. Most of the extirpated stocks occurred on the eastern side of Hood Canal, which affects the current spatial structure of the ESU. The widespread loss of estuary and lower floodplain habitat continue to impact the ESU's spatial structure and connectivity.

The Strait of Juan de Fuca population includes three extant stocks that spawn in rivers and streams entering the eastern Strait of Juan de Fuca and Admiralty Inlet. The Hood Canal population consists of six extant stocks within the Hood Canal watershed. HC Summer-run chum salmon are part of an extensive rebuilding program developed and implemented in beginning in 1992 by the state and tribal co-managers. The largest supplemental program occurs at the Big Quilcene River fish hatchery. Reintroduction programs occur in Big Beef (Hood Canal population) and Chimacum (Strait of Juan de Fuca population) creeks. All hatchery fish are marked and can be distinguished from naturally produced fish. There is concern that the Quilcene hatchery stock has high rates of straying, and may represent a risk to historical population structure and diversity.

Adult returns for some of the HC Summer-run chum salmon stocks showed modest improvements in 2000, with upward trends continuing in 2001 and 2002. The recent five-year mean abundance is variable among stocks, ranging from one fish to nearly 4,500 fish. Two stocks (Quilcene and Union River) are above the conservation thresholds established by the rebuilding plan. However, most stocks remain depressed. Estimates of the fraction of naturally spawning hatchery fish exceed 60% for some stocks. This indicates that reintroduction programs are supplementing the numbers of total fish spawning naturally in streams. Both the Strait of Juan de Fuca and the Hood Canal populations have long-term trends above replacement; long-term lambda values range from 0.85 to 1.39 (Good et al 2005). Long-term trends in productivity are above replacement only for the Quilcene and Union River stocks.

Critical Habitat

Critical habitat for this species was designated on September 2, 2005 (70 FR 52630). Of 11 watersheds reviewed in NMFS' assessment of critical habitat for the Hood Canal Summer-run chum salmon ESU, nine watersheds were rated as having a high conservation value while three were rated as having a medium value for conservation (Table 29). Five nearshore marine areas were also given a high conservation value rating. None of the watersheds was considered to be of a low conservation value, primarily because approximately half of the historical populations in this ESU have been extirpated, and the remaining populations are limited to only about 60 stream miles. Many of the watersheds have less than four miles of spawning habitat and none of them have more than 8.5 miles.

Table 29. Hood Canal Summer-run chum salmon 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]
Skokomish	0		1	(1, 3)	0	
Hood Canal	6	(1, 3)	1	(1)[2]	0	
Kitsap	1	(1)	0		0	
Dungeness/Elwha	2	(1)	1	(3, 1)	0	
Total	9		3		0	

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.

Spawning PCE is degraded by excessive fine sediment in the gravel. Rearing PCE is degraded by loss of access to sloughs in the estuary and nearshore areas and excessive predation. Low river flows in several rivers also adversely affect most PCEs. In the estuarine areas, both migration and rearing PCEs of juveniles are impaired by loss of functional floodplain areas necessary for growth and development of juvenile chum salmon. These degraded conditions likely maintain low population abundances across the ESU.

Columbia River Chum Salmon

Columbia River (CR) chum salmon includes all natural-origin chum salmon in the Columbia River and its tributaries in Oregon and Washington. The species consists of two populations: Grays River and Lower Gorge in Washington State (Figure 15). This ESU also includes three artificial hatchery programs. These artificially propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within this ESU.

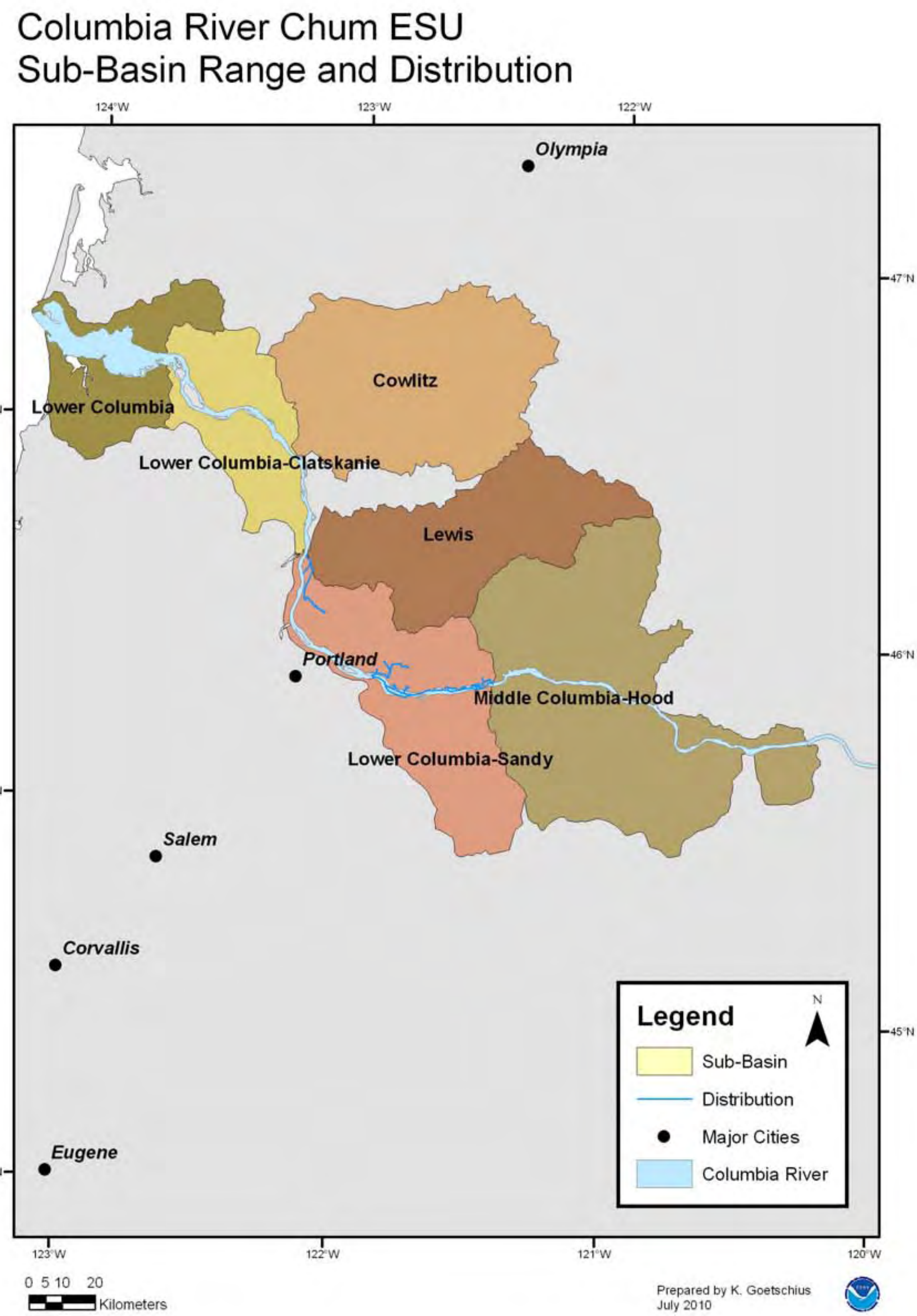


Figure 15**. Columbia River Chum salmon distribution**

Table 30. Populations within the Columbia River chum salmon ESU, their abundances, and hatchery input (Good et al 2005)

Current Populations	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Youngs Bay	Unknown	Not reported	0
Grays River	7,511	3,832 and 2,720*	Unknown
Big Creek	Unknown	Not reported	0
Elochoman River	Unknown	Not reported	0
Clatskanie River	Unknown	Not reported	0
Mill, Abernathy, and German Creeks	Unknown	Not reported	0
Scappoose Creek	Unknown	Not reported	0
Cowlitz River	141,582	Not reported	0
Kalama River	9,953	Not reported	0
Lewis River	89,671	Not reported	0
Salmon Creek	Unknown	Not reported	0
Clackamas River	Unknown	Not reported	0
Sandy River	Unknown	Not reported	0
Washougal River	15,140	Not reported	0
Lower gorge tributaries	>3,141	425	0
Upper gorge tributaries	>8,912	137 and 223*	0

* Salmon Scape Statistics Query 2009: Estimated total number of natural spawners for the years 2007 and 2008.

Life History

Chum salmon return to the Columbia River in late fall (mid-October to December). They primarily spawn in the lower reaches of rivers, digging redds along the edges of the mainstem and in tributaries or side channels. Some spawning sites are located in areas where geothermally-warmed groundwater or mainstem flow upwells through the gravel.

Chum salmon fry emigrate from March through May shortly after emergence. Juvenile chum salmon reside and feed in estuaries before beginning their long distance oceanic migration. Chum salmon may choose either the upper or lower estuaries depending on the relative productivity of each. The timing of entry of juvenile chum salmon into sea water is correlated with the warming of the nearshore waters and the accompanying plankton blooms (Burgner 1991). The movement offshore generally coincides with the decline of inshore prey resources and when fish have grown to a size that allows them to feed upon neritic organisms and avoid predators (Burgner 1991). The period of estuarine residence is a critical life history phase and plays a major role in determining the size of the subsequent adult run back to fresh water.

Status and Trends

NMFS listed CR chum salmon as threatened on March 25, 1999, and reaffirmed their threatened status on June 28, 2005 (71 FR 37160). Regarding spatial structure, historically this ESU was highly prolific; CR chum salmon were reported in almost every river in the Lower Columbia River basin. However, few CR chum salmon have been observed in tributaries between the Dalles and Bonneville dams in recent years. Chum salmon were not observed in any of the upper gorge tributaries, including the White Salmon River, during the 2003 and 2004 spawning ground surveys. Surveys of the White Salmon River in 2002 found only one male and one female carcass; the female had not spawned (Ehlke and Keller 2003). However, in the Cascades, chum salmon sampled from each tributary recently appeared as remnants of genetically distinct populations (Greco et al 2007).

Historically, the ESU was composed of 17 populations in Oregon and Washington between the mouth of the Columbia River and the Cascade crest (Myers et al 2006) (Table 30). Only two populations with any significant spawning remain today, both on the Washington side (Good et al 2005). They are the Grays River and the Lower Gorge (which include Hardy and Hamilton Creeks) populations (Good et al 2005). In addition, during the first years after 2000, new (or newly discovered) spawning was observed in the Washougal River mainstem and in the Washington side of the Columbia River mainstem below the mouth of Washougal River (Good et al 2005). It is unclear whether this spawning has been maintained. An extensive 2000 survey in Oregon streams supports that chum salmon are extirpated from the Oregon portion of this ESU (Good et al 2005).

The CR chum salmon runs have declined substantially from historic levels concurrently with the drastic reduction of spawning populations. In the early 1900s, the ESU numbered in the hundreds of thousands to a million returning adults that supported a large commercial fishery in the first half of this century. However, by the 1950s, most runs had disappeared and fisheries landings in later years rarely exceeded 2,000 chum salmon per year (Fulton 1970, Marr 1943, Rich 1942). During the 1980s and 1990s, the estimated combined abundance of natural spawners for the Lower Gorge, Washougal, and Grays River populations was below 4,000 adults. However, in 2002, the abundance of natural spawners increased to an estimate of total natural spawners exceeding 20,000 adults. The cause of this dramatic increase in abundance is unknown and was not maintained in the following years.

Current ESU abundance is mostly driven by the Lower Gorge and Grays River populations. The estimated size of the Lower Gorge population is at 400-500 individuals, down from a historical level of greater than 8,900 (Good et al 2005). A significant increase in spawner abundance occurred in 2001 and 2002 to around 10,000 adults (Good et al 2005). However, spawner surveys indicate that the abundance again decreased to low levels during 2003 through 2008 though the spawner surveys may underestimate abundance since the proportion of tributary and mainstem spawning differ between years and the surveys do not include spawners in the Columbia River mainstem (Good et al 2005, WDFW 2009). In the 1980s, estimates of the Grays River population ranged from 331 to 812 individuals. However, the population increased in 2002 to as many as 10,000 individuals (Good et al 2005). Based on data for number of spawners per river mile, this increase continued through 2003 and 2004. However, fish abundance fell again to less than 5,000 fish during the years 2005 through 2008 (WDFW 2009)

Estimates of abundance and trends are available only for the Grays River and Lower Gorge populations. The lambda values indicate a long-term downward trend at 0.954 and 0.984, respectively (Good et al 2005). The 10-year trend (up to 2001) was negative for the Grays River population and just over 1.0 for the Lower Gorge. Long- and short-term productivity trends for populations are at or below replacement.

Critical Habitat

Critical habitat was originally designated for the CR chum salmon on February 16, 2000 (65 FR 7764) and was re-designated on September 2, 2005 (70 FR 52630). Sixteen of the 19 subbasins reviewed in NMFS' assessment of critical habitat for the CR chum salmon ESU were rated as having a high conservation value (Table 31). The remaining three subbasins were given a medium conservation value. Washington's federal lands were rated as having high conservation value to the species.

Table 31. CR chum salmon 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]
Middle Columbia/Hood	3	(3)	0		0	
Lower Columbia/Sandy	3	(3, 1)	0		0	
Lewis	2	(3)	0		0	
Lower Columbia/Clatskanie	3	(3, 2, 1)	0		0	
Cowlitz	3	(3)	3	(3)	0	
Lower Columbia	2	(3, 2, 1)	0		0	
Lower Columbia Corridor	all	(3, 1)	0		0	
Total	16		3		0	

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.

Limited information exists on the quality of essential habitat characteristics for CR chum salmon. However, migration PCE has been significantly impacted by dams obstructing adult migration and access to historic spawning locations. Water quality and cover for estuary and rearing PCEs have decreased in quality to the extent that the PCEs are not likely to maintain their intended function to conserve the species.

Coho Salmon

Description of the Species

Coho salmon occur naturally in most major river basins around the North Pacific Ocean from central California to northern Japan (Laufle et al 1986). In this section, we discuss the distribution, life history diversity, status, and critical habitat of the four endangered and threatened coho species separately.

As with other salmon, the coho salmon life cycle consists of a juvenile freshwater phase and a growth phase in the ocean before fish return to rivers to spawn. Along the

Oregon/California coast, coho salmon primarily return to rivers to spawn as three-year olds, having spent approximately 18 months rearing in fresh water and 18 months in salt water. In some streams, a smaller proportion of males may return as two-year olds. The presence of two-year old males can allow for substantial genetic exchange between brood years. The relatively fixed three-year life cycle exhibited by female coho salmon limits demographic interactions between brood years. This makes coho salmon more vulnerable to environmental perturbations than other salmonids that exhibit overlapping generations, *i.e.*, the loss of a coho salmon brood year in a stream is less likely than for other Pacific salmon to be reestablished by females from other brood years.

Most coho salmon enter rivers between September and February. In many systems, coho salmon will have to wait to enter until fall rainstorms have provided the river with sufficiently strong flows and depth. Coho salmon spawn from November to January, and occasionally into February and March. Spawning occurs in a few third-order streams. Most spawning activity occurs in fourth- and fifth-order streams. Spawning generally occurs in tributaries with gradients of 3% or less.

Depending on temperature, egg incubation ranges from 35 to 50 days (Sandercock 1991). Hatchlings remain in the gravel as alevins for several weeks while absorbing the yolk sac before emerging from the gravel. In Oregon coastal streams, total average time from egg deposition to emergence is 110 days (Sandercock 1991). Following emergence, fry move to areas with weak water currents such as backwaters and shallow areas near the stream banks. As the fry grow, they disperse upstream and downstream to establish and defend territories. Territorial behavior limits summer density in streams and subordinate individuals may congregate in pools (Sandercock 1991).

Juvenile coho salmon commonly rear in small streams less than five ft. wide and occasionally in larger ponds and lakes (Pollock et al 2004). Juvenile rearing rarely occurs in tributaries exceeding gradients of 3% although they may move to streams with gradients of 4 to 5%. Preferred water quality consists of water with low turbidity, DO levels of 4 to 9 mg/l, and water temperatures ranging from 10° to 15°C (Bell 1973, McMahon 1983). Growth is slowed down considerably at 18°C and ceases at 20°C (Bell 1973, Stein et al 1972). The likelihood of juvenile coho salmon occupying habitat that exceed 16.3°C maximum weekly average temperature declines significantly (Welsh et al 2001).

During spring and summer, the emphasis is on growth and sustained invertebrate forage production and renewal are necessary. During the growth period, coho salmon fry show low risk averseness and position themselves in open water when sufficient food is available (Bugert et al 1991, Giannico 2000, Reinhardt 1999). The main prey are primarily drifting aquatic invertebrates produced in interstices of the gravel substrate and in the leaf litter within pools, and drifting terrestrial insects produced in the riparian canopy (Sandercock 1991). Important food organisms include aquatic insects such as chironomid larvae, mayfly, caddisflies, and stonefly. Coho salmon juveniles also feed opportunistically on non-insects, such as small fish and salmon eggs, and terrestrial insects.

Studies of stream habitat use show that there are a velocity threshold for rearing fry and juveniles. Juveniles prefer focal positions that have water velocity less than 20 cm/s (with a preference of 3 – 6 cm/s) with faster flowing adjacent areas with high food renewal through drift (Beecher et al 2002, Fausch 1984, Fausch 1993, Rosenfield et al 2000, Shirvell 1990). High food abundance (*i.e.*, drift) may increase the potential for net energy gain at higher velocities, allowing fish to move into faster waters where fish experience higher growth rate despite the greater swimming costs (Giannico and Healey 1999, Rosenfeld et al 2005). High prey availability also reduces territory size and may increase a stream's rearing capacity (Dill and Fraser 1984, Dill et al 1981, Mason 1976). Reduction in food availability reduces growth by subdominants and less for dominant fish (Rosenfeld et al 2005).

Coho salmon juveniles seek river margins, backwater, and pools during fall and winter; they are rarely found in mid-stream locations of the stream channel during November and February (Bilby and Bisson 1987, Bilby and Bisson 2001, Fausch and Northcote 1992, Tschaplinkski and Hartman 1983). High densities of juvenile coho salmon also occur in log jams (Brown 1985, Tschaplinski and Hartmann 1983). In early fall with the onset of the first seasonal freshets, a large portion of the juvenile population may also migrate to overwinter in off-channel habitat such as larger pools, beaver ponds, off-stream side channels and alcoves, ephemeral swamps, and inundated floodplains (Brown 1985, Bustard and Narver 1975, Nickelson et al 1992b, Nickelson et al 1992c, Peterson 1982, Tschaplinski and Hartman 1983).

During the winter period, juveniles typically reduce feeding activity and growth rates slow down or stop. In spring, juvenile activity increases. By March of their second spring, the juveniles feed heavily on insects and crustaceans and grow rapidly before smoltification and outmigration (Olegario 2006). Juveniles that overwinter in off-channel habitat, ephemeral streams, and floodplains often experience higher survival and growth than juveniles that overwinter in mainstream channels (Brown 1985, Dolloff 1987, Olegario 2006, Quinn and Peterson 1996, Swales et al 1988.)

Availability of suitable overwintering habitat has been suggested to determine smolt production in streams (Bustard and Narver 1975, Nickelson et al 1992). Adult return or smolt production is related to the area of wetlands, lakes, and ponds within watersheds (Beechie et al 1994, Pess et al 2002, Sharma and Hilborn 2001).

Coho salmon juveniles usually migrate to the ocean as smolts in their second spring. Relative to species such as chum salmon, Chinook salmon, and steelhead, coho salmon smolts usually spend a short time in the estuary with little feeding (Magnusson and Hilborn 2003, Thorpe 1994). Estuarine residence times can average one to three days (Miller and Sadro 2003). However, some coho salmon fry may migrate to and rear in the tidally influenced portions of the stream. In one Oregon stream, a portion of the coho salmon fry were observed remaining in the upper estuary to rear after moving into the estuary during their first spring (Miller and Sadro 2003).

After entering the ocean, immature coho salmon initially remain in nearshore waters close to the parent stream. North American coho salmon will migrate north along the coast in a narrow coastal band that broadens in southeastern Alaska. During this migration, juvenile coho salmon tend to occur in both coastal and offshore waters. During spring and summer, coho salmon will forage in waters between 46ºN, the Gulf of Alaska, and along Alaska's Aleutian Islands.

Status and Trends

Coho salmon depend on the quantity and quality of the freshwater aquatic systems where they spawn and the juveniles rear, and on the ocean conditions where they grow to maturity. Coho salmon have declined from overharvests, hatcheries, native and non-native species, dams, gravel mining, water diversions, the destruction or degradation of riparian habitat, and land use practices (logging, agriculture, and urbanization).

Lower Columbia River Coho Salmon

The LCR coho salmon include all naturally spawned populations of coho salmon in the Columbia River and its tributaries in Oregon and Washington, from the mouth of the Columbia up to and including the Big White Salmon and Hood Rivers, and includes the Willamette River to Willamtte Falls, Oregon (Figure 16). This ESU also includes 25 artificial propogation programs (70FR 37160).

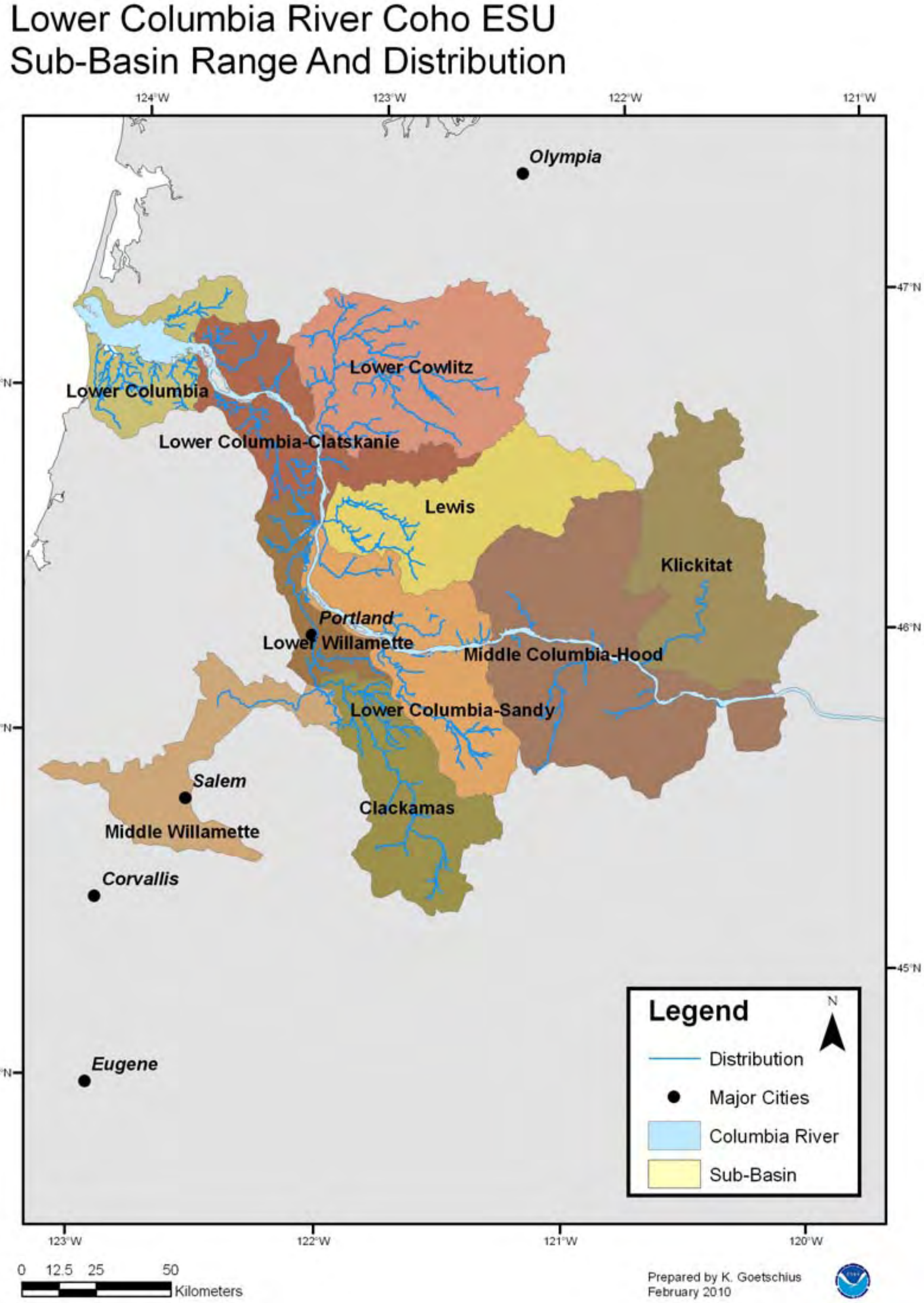


Figure 16. LCR coho salmon distribution.

Life History

The majority of the LCR coho salmon are of hatchery origin. Hatchery runs are currently managed for two distinct runs: early returning (Type S) and late returning (Type N) (Johnson et al 1991). Type S coho salmon return to fresh water in mid-August and to the spawning tributaries in early September. Spawning peaks from mid-October to early November. Type N coho salmon return to the Columbia River from late September through December and enter the tributaries from October through January. Most Type N spawning occurs from November through January.

Analysis of run timing of coho salmon suggests that the Clackamas River population is composed of one later returning population and one early returning population. The late returning population is believed to be descended from the native Clackamas River population. The early returning population is believed to descend from hatchery fish introduced from Columbia River populations outside the Clackamas River basin (Good et al 2005). The naturally produced coho salmon return to spawn between December and March (Johnson et al 1991).

Fry emerge from the redds during a three-week period between early March and late July. The juveniles rear in fresh water for a year and smolt outmigration occurs from April through June with a peak in May. Smolts migrate through the Columbia River estuary during dusk and dawn. During movement they are found in mid-river areas of the estuary. However, during mid-morning to late afternoon they reside near the shores of the estuary (Johnson et al 1991).

Status and Trends

NMFS listed the LCR coho salmon as threatened on June 28, 2005 (70 FR 37160). The LCR coho salmon ESU historically consisted of 25 independent populations. The vast majority (over 90%) of these are either extirpated or nearly so (Table 32). Today, only 2 of the 25 populations have any significant natural production in the Sandy and Clackamas Rivers. In addition, wild coho salmon have re-appeared in two additional basins

(Scappoose and Clatskanie) after a 10-year period during the 1980s and 1990s when they were largely absent (McElhany et al 2007).

Table 32. Lower Columbia River coho salmon populations, estimated natural spawner abundances, and hatchery contributions (Good et al 2005, McElhany et al 2007)

River/Region	Historical Abundance	2002-2004 Spawner Abundance[1]: Max/Geometric mean	Hatchery Abundance Contributions
Youngs Bay and Big Creek	Unknown	~4,470/200	91%
Grays River	Unknown	Unknown	Unknown
Elochoman River	Unknown	Unknown	Unknown
Clatskanie River	Unknown	~550/286	0-80%
Mill, Germany, and Abernathy creeks	Unknown	Unknown	Unknown
Scappoose Rivers	Unknown	~850/470	0%
Cispus River	Unknown	Unknown	Unknown
Tilton River	Unknown	Unknown	Unknown
Upper Cowlitz River	Unknown	Unknown	Unknown
Lower Cowlitz River	Unknown	Unknown	Unknown
North Fork Toutle River	Unknown	Unknown	Unknown
South Fork Toutle River	Unknown	Unknown	Unknown
Coweeman River	Unknown	Unknown	Unknown
Kalama River	Unknown	Unknown	Unknown
North Fork Lewis River	Unknown	Unknown	Unknown
East Fork Lewis River	Unknown	Unknown	Unknown
Upper Clackamas River	Unknown	~1,770/1,264	12%
Lower Clackamas River	Unknown	~1,180/843	78%
Salmon Creek	Unknown	Unknown	Unknown
Upper Sandy River	Unknown	~1,170/720	0%
Lower Sandy River	Unknown	271/?	97%
Washougal River	Unknown	Unknown	Unknown
Lower Columbia River gorge tributaries	Unknown	Unknown	Unknown
Big White Salmon river	Unknown	Unknown	Unknown
Upper Columbia River gorge tributaries	Unknown	1,317/?	>65%
Hood River	Unknown	~600/~230	Unknown

Prior to 1900, the Columbia River had an estimated annual run of more than 600,000 adults with about 400,000 spawning in the lower Columbia River (Johnson et al 1991). By the 1950s, the estimated number of coho salmon returning to the Columbia River had decreased to 25,000 adults or about 5% of historic levels. Massive hatchery releases since 1960 have increased the Columbia River run size. Between 1980 and 1989, the run varied from 138,000 adults to a historic high of 1,553,000 adults. However, only a small portion of these spawned naturally, and available information indicates that the naturally produced portion has continuously declined since the 1950s. The current number of naturally spawning fish during October and late November ranges from 3,000 to 5,500 fish. The majority of these are of hatchery origin. The 1996 to 1999 geometric mean for the late run in the Clackamas River, the only-run which is considered consisting mainly of native coho salmon, was 35 fish.

Both the long- and short-term trend, and lambda for the natural origin (late-run) portion of the Clackamas River coho salmon are negative but with large confidence intervals (Good et al 2005). The short-term trend for the Sandy River population is close to 1, indicating a relatively stable population during the years 1990 to 2002 (Good et al 2005). The long-term trend (1977 to 2002) for this same population shows that the population has been decreasing (trend=0.54); there is a 43% probability that the median population growth rate (lambda) was less than one.

Hatchery-origin spawners dominate the majority of populations. However, both the upper Clackamas River and the upper Sandy River spawner populations range from zero to very few hatchery origin spawneres. Recent reviews by the W/LCRTRT placed most populations in the high to moderate risk category from eroded diversity (McElhany et al 2004, McElhany et al 2006).

Critical Habitat

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

Oregon Coast Coho Salmon

The Oregon Coast (OC) coho salmon ESU includes all naturally spawned populations of coho salmon in Oregon coastal streams south of the Columbia River and north of Cape Blanco (63 FR 42587; August 10, 1998, Figure 17). One hatchery stock, the Cow Creek (ODFW stock # 37) hatchery coho, is included in the 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.

Life History

The OC coho salmon exhibit the general three year life cycle as described above. Two-year old males commonly occur in some streams and on average make up 20% of spawning males. However, the proportion of two-year old males is highly variable between years and river systems.

There is some variation in run timing between Oregon watersheds but adults generally start to migrate into rivers at the first fall freshet, usually in late October or early November. A delay in rain can delay river entry considerably. Once in the stream, some coho may spend up to two months in fresh water before spawning. Spawning usually occurs from November through January and may continue into February. Juveniles emerge from the gravel in spring and typically spend a summer and winter in fresh water before migrating to the ocean as smolts, usually in April or May, in their second spring. However, the timing varies between years, among river systems, and based on small-scale habitat variability (Lawson et al 2007). Coastal coho salmon spend little time in estuarine environments during outmigration. Once in coastal waters, the OC coho salmon eventually move northward. By late summer, juveniles are observed distributed off the mouth of Columbia River and the Washington Coast. In fall and winter juvenile coho salmon continue to move northward and have been caught off the coast of Alaska (Lawson et al 2007). Southward movement starts in winter or early spring with adults starting to home to natal streams by August.

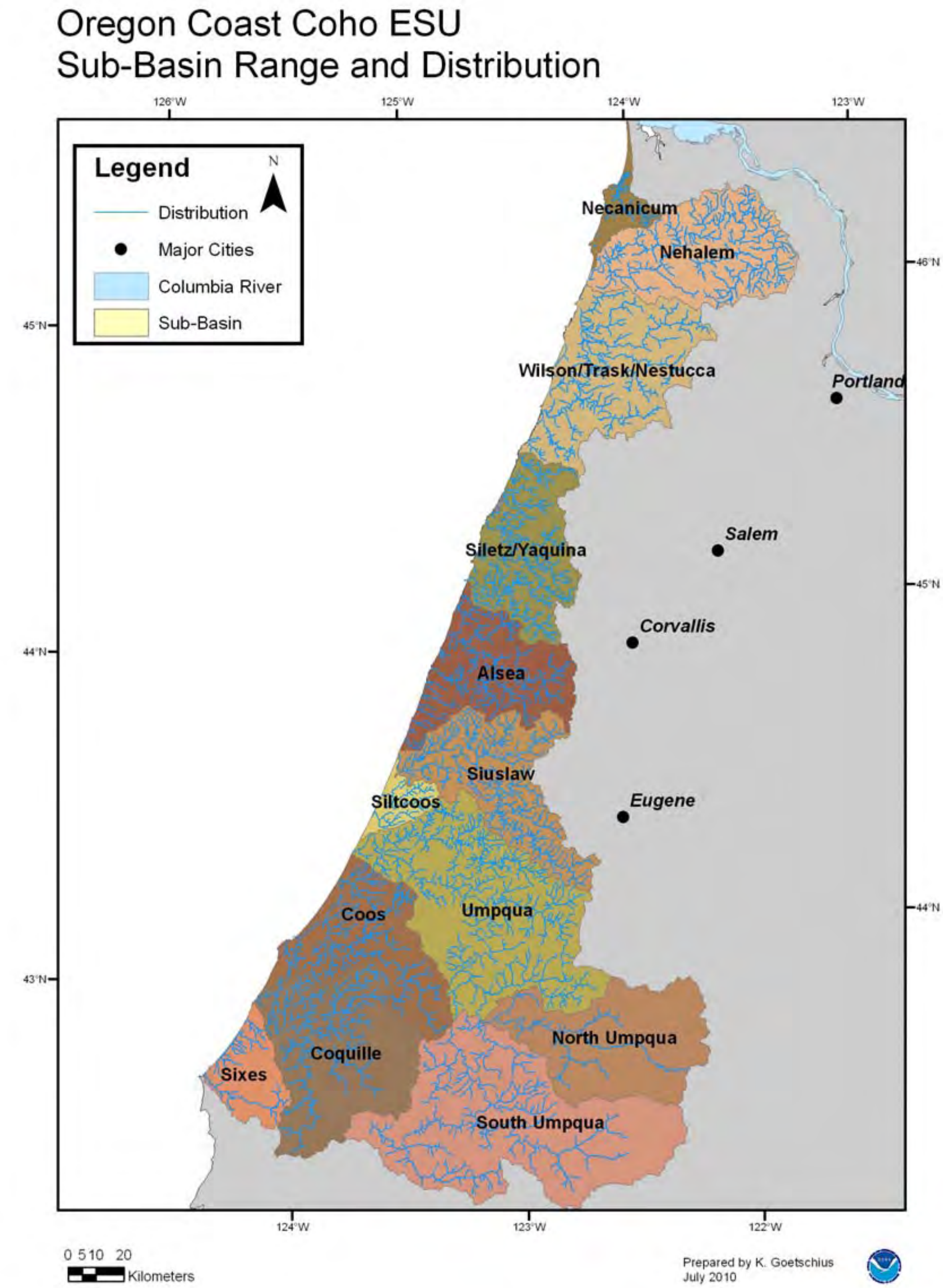


Figure 17. Oregon Coast Coho salmon distribution.

Status and Trends

NMFS listed the OC coho salmon as a threatened species on February 11, 2008 (73 FR 7816). Lawson *et al.* (2007) considered the ESU to have historically consisted of 13 functionally independent populations and 8 potentially dependent populations. Current coho salmon coastal distribution has not changed markedly compared to historical distribution (Lawson et al 2007). However, river alterations and habitat destruction have significantly modified use and distribution within several river basins.

The OC coho salmon historical escapement in the 10 larger basins has been estimated to about 2.4 to 2.9 million spawners (from Table C-1 in Lawson et al 2007). Recent ESU abundances have decreased drastically since then. The estimated median spawning population during the years 1990 to 1999 was 43,183 (min. 21,279, max. 74,021) coho salmon spawners in the ESU (ODFW 2009). After 1999, total ESU abundance increased. A median of 165,324 native OC coho salmon spawners was estimated for the period 2000 through 2008 with a range from a low of 66,169 to a high of 260,000 naturally produced spawners. Table 33 identifies independent populations within the OC coho salmon ESU, historic and recent abundances, and hatchery input.

Table 33. Oregon Coast Coho salmon potential historic and estimated recent spawner abundances, and hatchery contributions (Lawson et al 2007, Good et al 2005).

Basin	Population historic status	Historic Abundance	Recent Spawner Abundance	Hatchery Abundance Contributions
Necanicum	P-I	68,500	1,889	35-40%
Nehalem	F-I	333,000	18,741	40-75%
Tillamook	F-I	329,000	3,949	30-35%
Nestucca	F-I	104,000	3,846	~5%
Siletz	F-I	122,000	2,295	~50%
Yaquina	F-I	122,000	3,665	~25%
Alsea	F-I	163,000	3,621	~40%
Siuslaw	F-I	267,000	16,213	~40%
Umpqua	F-I*	820,000	24,351	<10%
Siltcoos and Tahhenitch	P-I	100,000	15,967**	0%
Tenmile	P-I	53,000	3,251**	0%
Coos	F-I	206,000	20,136	<5%
Coquille	F-I	417,000	8,847	<5%
Total		924,000	107,553	

*The Umpqua Rive basin is believed to have supported four functionally independent populations.
** Abundance in 2002, ODFW data http://oregonstate.edu/dept/ODFW/spawn/data.htm
F-I = Functionally Independent, P-I = Potentially Independent.

The abundance and productivity of OC coho salmon since the 1997 status review represented some of the best and worst years on record (Good et al 2005). Yearly adult returns for this ESU were in excess of 160,000 natural spawners in 2001 and 2002. However, these encouraging increases in spawner abundance in 2000–2002 were preceded by three consecutive brood years (the 1994–1996 brood years returning in 1997–1999, respectively) exhibiting recruitment failure. Recruitment failure is when a given year class of natural spawners fails to replace itself when its offspring return to the spawning grounds three years later. At the time of the 2005 status report, these three years of recruitment failure were the only such instances observed thus far in the entire 55-year abundance time series for OC coho salmon (Good et al 2005). The encouraging 2000–2002 increases in natural spawner abundance were primarily observed in populations in the northern portion of the ESU (Good et al 2005). Although encouraged by the increase in spawner abundance in 2000–2002, the long-term trends in ESU productivity remained negative due to the low abundances observed during the 1990s (Good et al 2005).

Recent data indicate that the total abundance of natural spawners in the OC coho salmon ESU again steadily decreased until 2007 with an estimated spawner abundance of 66,169 fish or approximately 25% of the 2002 peak abundance (260,555 spawners) (ODFW 2009). Thus, recruitment failed during the five years from 2002 through 2007 but abundance increased again in 2008 to 165,324 spawners. There is no apparent weak brood year for the ESU.

Critical Habitat

NMFS designated critical habitat for Oregon Coast coho salmon on February 11, 2008 (73 FR 7816). The designation include 72 of 80 watersheds and total about 6,600 stream miles including all or portions of the Nehalem, Nestucca/Trask, Yaquina, Alsea, Umpqua, and Coquille basins.

There are 80 watersheds within the range of this ESU. Eight watersheds received a low conservation value rating, 27 received a medium rating, and 45 received a high rating to the ESU (Table 34).

Table 34. OC coho salmon 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]
Necanicum	0		1	(1, 2)	0	
Nehalem	5	(1, 2)	0		1	(2, 1)
Wilson/Trask/Nestucca	7	(1, 2)	2	(1, 2)	0	
Siletz/Yaquina	3	(1, 2)	5	(1, 2)	0	
Alsea	4	(1, 2)	3	(1, 2)	1	(1, 2=1.5mi)
Siuslaw	6	(1, 2, <3)	2	(1, 2)	0	
Siltcoos	1	(2, 1)	0		0	
North Umpqua	1	(1, <2)	3	(1, 3, <2)	3	(1)
South Umpqua	3	(1, <2, <<3)	8	(1, 2, 3)	1	(1)
Umpqua	6	(1, 3, 2)	1	(1, 3)	1	(1, 2, 3)
Coos	4	(1, 2, <3)	0		0	
Coquille	4	(1, 2, 3))	1	(1, 2)	1	(1, 2)
Sixes	1	(1, 20	1	(1, 2)		
Total	45		27		8	

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.

Spawning PCE has been impacted in many watersheds by inclusion into spawning gravel of fine sediment from timber harvest and forestry related activities, agriculture, and grazing. These activities have also diminished the channels' rearing and overwintering capacity by reducing the amount of large woody debris in stream channels, removing riparian vegetation, disconnecting floodplains from stream channels, and changing the quantity and dynamics of stream flows. Rearing PCE has been degraded by elevated water temperatures in 29 of the 80 HUC 5 watersheds; rearing PCE within the Nehalem, North Umpqua, and the inland watersheds of the Umpqua subbasins have elevated stream temperatures. Water quality is impacted by contaminants from agriculture and urban areas in low laying areas in the Umpqua subbasins, and in coastal watersheds within the Siletz/Yaquina, Siltcoos, and Coos subbasins; A reduction in water quality have been observed in 12 watersheds due to contaminants and excessive nutrition. Migration PCE is impacted throughout the ESU by culverts and road crossings that restrict passage. Thus, quality of PCEs varies widely throughout the critical habitat area designated for OC coho salmon. As described above, many watersheds are heavily impacted with low quality of PCEs while habitat in other coho salmon bearing watersheds are of sufficient quality for supporting the conservation purpose of designated critical habitat.

Southern Oregon/Northern California Coast Coho Salmon

The Southern Oregon/Northern California Coast (SONCC) coho salmon ESU consists of all naturally spawning populations of coho salmon that reside below long-term, naturally impassible barriers in streams between Punta Gorda, California and Cape Blanco, Oregon (Figure 18). This ESU also includes three artificial propagation programs. These artificially propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within this ESU.

Life History

In Oregon, the SONCC coho salmon enter rivers in September or October. River entry is later south of the Klamath River Basin, occurring in November and December, in basins south of the Klamath River to the Mattole River, California. River entry occurs from mid-December to mid-February in rivers farther south. Because coho salmon enter rivers late and spawn late south of the Mattole River, they spend much less time in the river prior to spawning compared to populations farther north. Juveniles emerge from the gravel in spring, and typically spend a summer and winter in fresh water before migrating to the ocean as smolts in their second spring. Coho salmon adults spawn at age three, spending about a year and a half in the ocean.

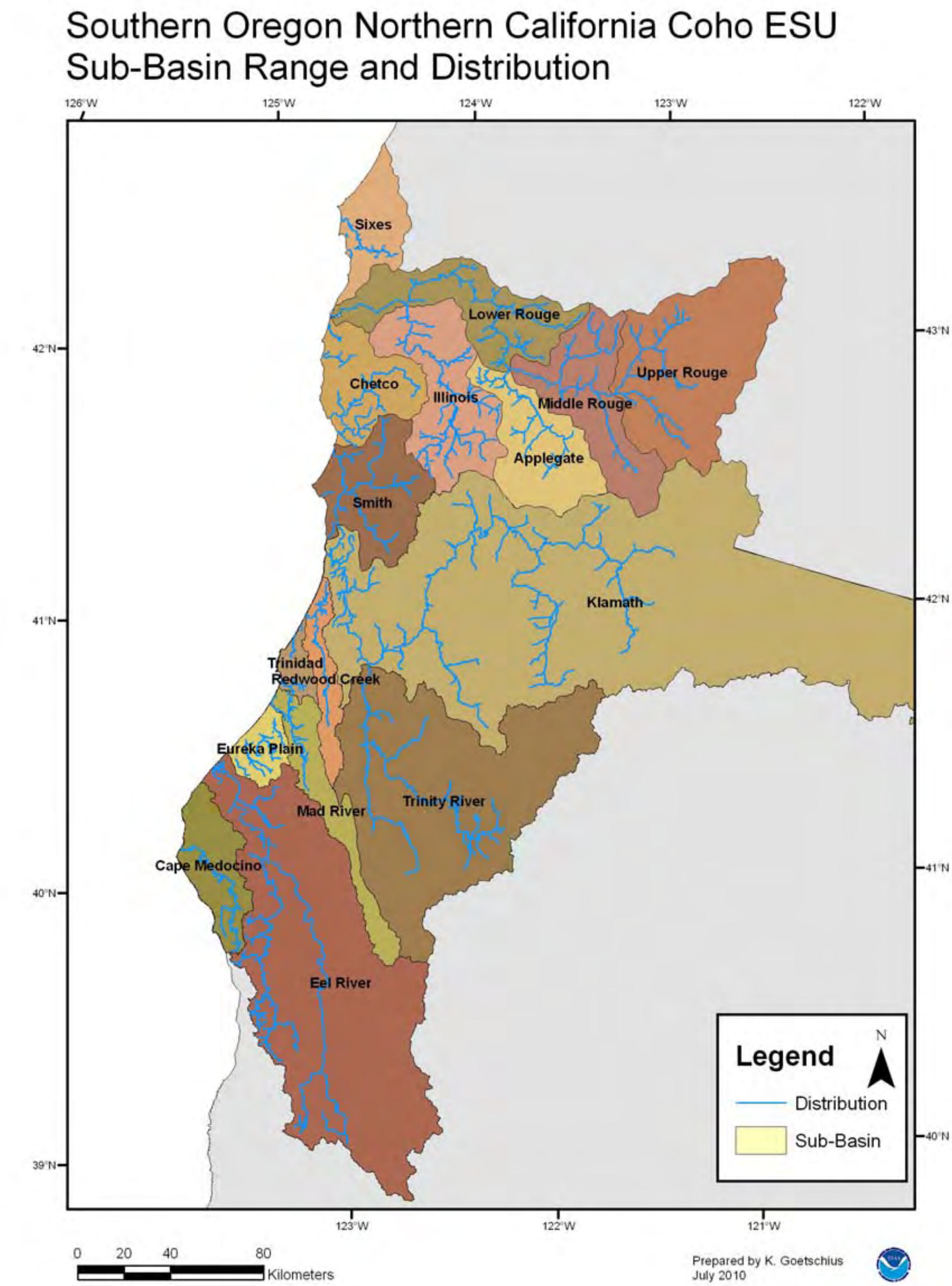


Figure 18. SONCC coho salmon distribution.

Status and Trends

NMFS listed SONCC coho salmon as threatened on May 7, 1997 (62 FR 24588), and reaffirmed their threatened status on June 28, 2005 (70 FR 37160). The ESU consists of three major basins: the Rough (OR), Klamath (OR/CA), and the Eel (CA) Rivers. Three historically independent interior populations have been identified for the Rough River basin, eight for the Klamath River basin, and six in the Eel River basin (Williams et al 2006b). In addition, eight coastal basins within the ESU likely supported functionally independent populations under historical conditions, six basins likely supported potentially independent populations, and 13 supported dependent populations. Presence-absence data indicate a disproportionate loss of southern populations compared to the northern portion of the ESU.

Data on population abundance and trends are limited for this ESU. Historical point estimates of coho salmon abundance for the early 1960s and mid-1980s suggest that California statewide coho spawning escapement in the 1940s ranged between 200,000 and 500,000 fish. Numbers declined to about 100,000 fish by the mid-1960s with about 43% originating from this ESU. Brown *et al.* (1994) estimated that about 7,000 wild and naturalized coho salmon were produced in the California portion of this ESU. Further, presence-absence surveys indicate that the SONCC coho salmon have declined in California compared to past abundances (Good et al 2005). Data from surveys in Oregon contrast the California portion of the ESU in that fish presence has been steadily increasing from 1998 through 2007 (Bennet 2005, Good et al 2005, Jepsen and Leader 2008.)

There is no consistent monitoring of any SONCC coho salmon populations. Trend and median population growth for single populations have therefore not been calculated. Information on abundance and production from California streams is limited. However, presence-absence data show that distributions within watersheds have remained suppressed compared to the historic distribution. Some hatchery releases has occurred but there is not enough information to evaluate the impacts of hatchery on fish diversity.

Critical Habitat

NMFS designated critical habitat for the SONCC coho salmon on May 5, 1999 (64 FR 24049). Species critical habitat encompasses all accessible river reaches between Cape Blanco, Oregon, and Punta Gorda, California and consists of the water, substrate, and river reaches (including off-channel habitats) in specified areas. Accessible reaches are those within the historical range of the ESU that can still be occupied by any life stage of coho salmon. Watersheds within the ESU have not been evaluated for their conservation value.

Critical habitat designated for the SONCC coho salmon is generally of good quality in northern coastal streams. Spawning PCE has been degraded throughout the ESU by logging activities that has increased fines in spawning gravel. Rearing PCE has been considerably degraded in many inland watersheds by the loss of riparian vegetation that has resulted in unsuitable high water temperatures. Rearing PCE and juvenile migration PCE has been reduced by disconnecting floodplains and off-channel habitat in low gradient reaches of streams, thereby reducing winter rearing capacity.

Central California Coast Coho Salmon

The Central California Coast (CCC) coho salmon ESU includes all naturally spawned populations of coho salmon from Punta Gorda in northern California south to and including the San Lorenzo River in central California, as well as populations in tributaries to San Francisco Bay, excluding the Sacramento-San Joaquin River system (Figure 19) The ESU also includes four artificial propagation programs. These artificially propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within this ESU.

Life History

In general, coho salmon within California exhibit a three-year life cycle. However, two-year old males commonly occur in some streams. Both run and spawn timing of coho salmon in this region are late (both peaking in January) relative to northern populations, with little time spent in fresh water between river entry and spawning.

Spawning runs coincide with the brief peaks of river flow during the fall and winter. Most CCC coho salmon juveniles undergo smoltification and start their seaward migration one year after emergence from the redd. Juveniles spending two winters in fresh water have, however, been observed in at least one coastal stream within the range of the ESU (Bjorkstedt et al 2005). Smolt outmigration generally peaks in April and May (Shapovalov and Taft 1954, Weitkamp et al 1995).

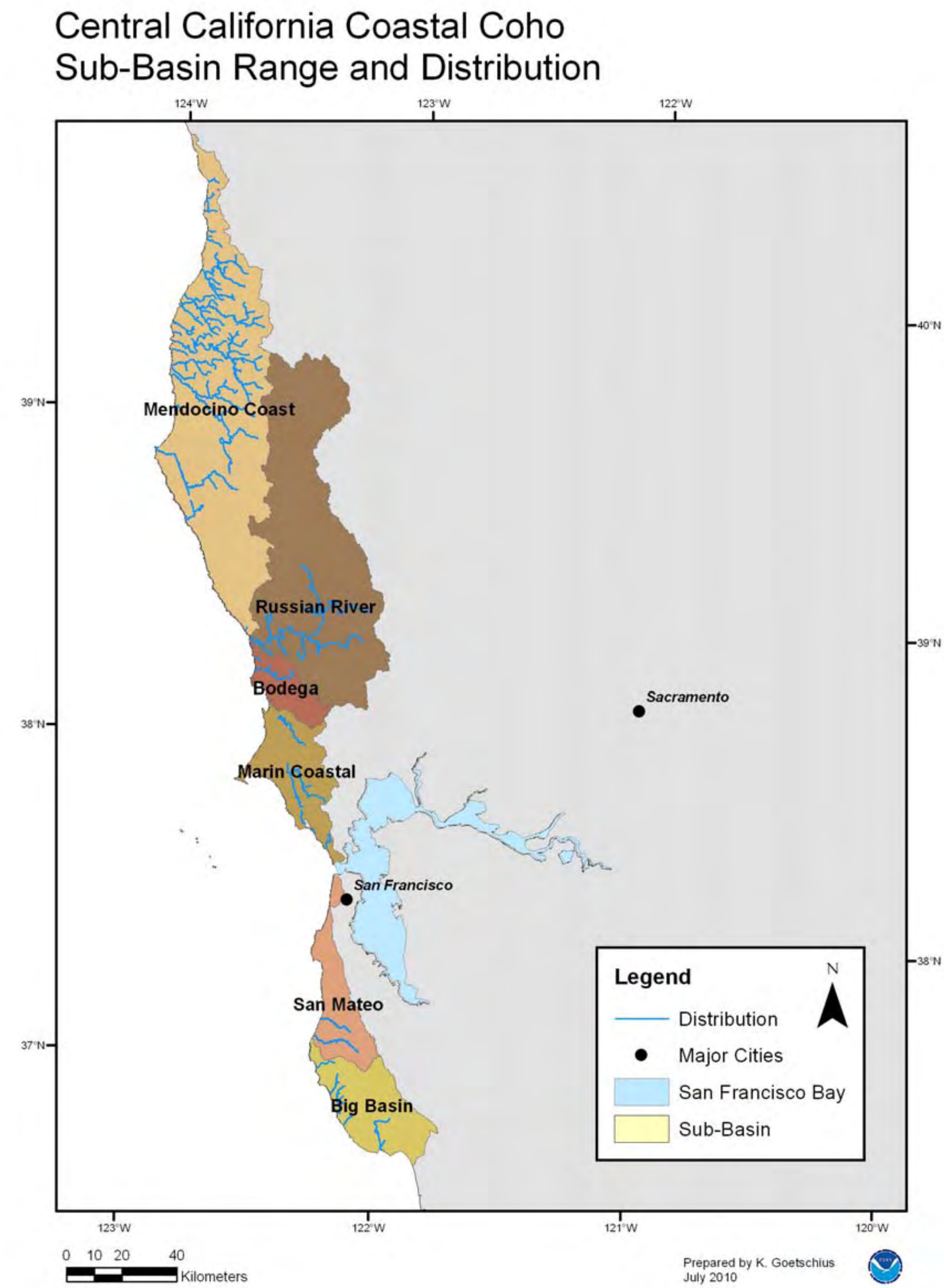


Figure 19. CCC Coho salmon distribution.

Status and Trends

NMFS originally listed the CCC coho salmon as threatened on October 31, 1996 (61 FR 56138), and reclassified their status to endangered on June 28, 2005 (70 FR 37160). The ESU consisted historically of 11 functionally independent populations and a larger number of dependent populations (Spence et al 2008). ESU spatial structure has been substantially modified due to lack of viable source populations and loss of dependent populations. One of the two historically independent populations in the Santa Cruz mountains (*i.e.*, South of the Golden Gate Bridge) is extirpated (Good et al 2005, Spence et al 2008). Coho salmon are considered effectively extirpated from the San Francisco Bay (NMFS 2001, Spence et al 2008). The Russian River population, once the largest and most dominant source population in the ESU, is now at high risk of extinction because of low abundance and failed productivity (Spence et al 2008). The Lost Coast to Navarro Point to the north contains the majority of coho salmon remaining in the ESU.

Limited information exists on abundance of coho salmon within the CCC coho salmon ESU. About 200,000 to 500,000 coho salmon were produced statewide in the 1940s (Good et al 2005). This escapement declined to about 99,000 by the 1960s with approximately 56,000 (56%) originating from streams within the CCC coho salmon ESU. The estimated number of coho salmon produced within the ESU in the late 1980s had further declined to 6,160 (46% of the estimated statewide production) (Good et al 2005).

Information on the abundance and productivity trends for the naturally spawning component in individual rivers of the CCC coho salmon ESU is extremely limited (Good et al 2005, Spence et al 2008). There are no long-term time series of spawner abundance for individual river systems. Returns increased in 2001 in streams within the northern portion of the ESU (Good et al 2005). However, recent CCC coho salmon returns (2006/07 and 2007/08) have been discouragingly low (McFarlane et al 2008). About 500 fish have returned in 2010 across the entire range. This is the third straight year of abysmal returns for CCC coho salmon. This year’s low return suggests that all three year classes are faring poorly across the species’ range.

Table 35. Central California Coast Coho salmon populations, abundances, and releases of hatchery raised smolt (Good et al 2005, Bjorkstedt et al 2005)

River/Region	Historical Escapement (1963)	1987-1991 Escapement Abundance	Hatchery Abundance Contributions*
Ten Mile River	6,000	160	892 – 796,561
Noyo River	6,000	3,740	940,970 – 242,808
Big River	6,000	280	9,988 – 191,310
Navarro River	7,000	300	20,020 – 143,812
Garcia River	2,000	500 (1984-1985)	183,153
Other Mendacino County rivers	10,000	470	Unknown
Gualala River	4,000	200	10,005 – 135,050
Russian River	5,000	255	7,998 – 415,730
Other Sonoma County rivers	1,000	180	Unknown
Marin County	5,000	435	5,760 – 305,421**
San Mateo County	1,000	Unknown	Unknown
San Francisco Bay	Unknown	Extirpated	NA
Santa Cruz County	1,500	50 (1984-1985)	Unknown
San Lorenzo River	1,600	Unknown	17,160 – 145,960
Total	200,000-500,000	6,570 (min)	

*Most coho salmon hatchery contributions have been infrequent and the numbers indicate the range of documented releases. All hatchery data are from Bjorkstedt *et al.* 2005.
**Lagunitas and Walker Creeks

The best data available for the CCC coho salmon are presence-absence surveys and they are used as a proxy for abundance changes. At the time of the 1996 listing, coho salmon occurred in about 47% of the streams (62) and were considered extirpated from 53% (71) of the streams that historically harbored coho salmon within the ESU (Brown et al 1994) Later reviews have concluded that the number of occupied streams relative to historic has not changed and may actually have declined (Good et al 2005, NMFS 2001).

Hatchery raised smolt have been released infrequently but occasionally in large numbers in rivers throughout the ESU (Bjorkstedt et al 2005). Releases have included transfer of stocks within California and between California and other Pacific states as well as smolt raised from eggs collected from native stocks. However, genetic studies show little homogenization of populations, *i.e.*, transfer of stocks between basins have had little effect on the geographic genetic structure of CCC coho salmon (SCWA 2002). The CCC coho salmon likely has considerable diversity in local adaptations given that the ESU spans a large latitudinal diversity in geology and ecoregions, and include both coastal and inland river basins.

Critical Habitat

Critical habitat for the CCC coho salmon ESU was designated on May 5, 1999 (64 FR 24049). It encompasses accessible reaches of all rivers (including estuarine areas and tributaries) between Punta Gorda and the San Lorenzo River (inclusive) in California. Critical habitat for this species also includes two streams entering San Francisco Bay: Arroyo Corte Madera Del Presidio and Corte Madera Creek. Individual watersheds within the ESU have not been evaluated for their conservation value.

NMFS (2008a) evaluated the condition of each habitat attribute in terms of its current condition relative to its role and function in the conservation of the species. The assessment of habitat for this species showed a distinct trend of increasing degradation in quality and quantity of all PCEs as the habitat progresses south through the species range, with the area from the Lost Coast to the Navarro Point supporting most of the more favorable habitats and the Santa Cruz Mountains supporting the least. However, all populations are generally degraded regarding spawning and incubation substrate, and juvenile rearing habitat. Elevated water temperatures occur in many streams across the entire ESU.

Sockeye Salmon

Description of the Species

Sockeye salmon occur in the North Pacific and Arctic oceans and associated freshwater systems. This species ranges south as far as the Klamath River in California and northern Hokkaido in Japan, to as far north as Bathurst Inlet in the Canadian Arctic and the Anadyr River in Siberia. We discuss the distribution, life history diversity, status, and critical habitat of the two endangered and threatened sockeye species separately.

Spawning generally occurs in late summer and autumn, but the precise time can vary greatly among populations. Males often arrive earlier than females on the spawning grounds, and will persist longer during the spawning period. Average fecundity ranges from about 2,000 eggs per female to 5,000 eggs, depending upon the population and age of the female.

The vast majority of sockeye salmon spawn in outlet streams of lakes or in the lakes themselves. In lakes, the species commonly spawn along “beaches” where underground seepage creates upwelling that provides eggs and alevins with fresh oxygenated water. Incubation is a function of water temperature, but generally lasts between 100 and roughly 200 days (Burgner 1991). Sockeye salmon fry primarily use lakes as rearing areas with river emerged fry migrating into lakes to rear. Fry emerging in streams emptying into lakes usually move rapidly with the water flow downstream into lakes. Fry emerging from lake outlet spawning areas migrate upstream into lakes. In these cases, fry hold for a period in the stream and may feed actively before moving upstream into the lake. During upstream migration, they move along the low velocity stream margin. Fry emerging from lakeshore or island spawning grounds distribute along the shoreline of the lake or move offshore into deep water (Burgner 1991). The juvenile sockeye salmon rear in lakes from one to three years after emergence.

Some sockeye spawn in rivers without lake habitat for juvenile rearing. Offspring of these riverine spawners use the lower velocity sections of rivers as juvenile rearing environment for one to two years. Alternatively, juveniles may also migrate to sea in their first year.

Certain populations of *O. nerka* become resident in the lake environment and are called kokanee or little redfish (Burgner 1991). Kokanee and sockeye often co-occur in many interior lakes, where access to the sea is possible but energetically costly. On the other hand, coastal lakes, where the migration to sea is relatively short and energetic costs are minimal, rarely support kokanee populations.

During freshwater rearing, sockeye salmon feeding behavior change as the juvenile transit through stages from emergence to the time of smoltification. As the alevins emerge from gravel, they feed little and depend mostly on the yolk sack, if it is still present, for growth (Burgner 1991). It is therefore critical for the small fry to start feeding as the yolk sack reserves are being depleted; a high mortality is observed when fishes are starved for more than two weeks after yolk absorption (Bilton and Robins 1973). In the earlier fry stage from spring to early summer, juveniles forage exclusively in the warmer littoral (*i.e.*, shoreline) zone where they depend mostly on dipteran insects (mostly chironomidae larvae and pupae) and on cyclopoid copepods and cladocerans. In summer, underyearling sockeye salmon transit from the littoral habitat to a pelagic existence where they feed on larger zooplankton. However, diptera, especially chironomids, can contribute substantially in caloric value. Older and larger fish may also prey on fish larvae. Distribution in lakes and prey preference is, however, a dynamic process that changes diurnally and annually, with water temperature, with the presence and abundance of particular prey species, presence of predators and competitors, and the size of the sockeye salmon juveniles.

Upon smoltification, anadromous sockeye migrate to the ocean. Peak emigration to the ocean occurs in mid-April to early May in southern sockeye populations (<52ºN latitude) and as late as early July in northern populations (62ºN latitude) (Burgner 1991). River-type sockeye populations make little use of estuaries during their emigration to the marine environment. Upon entering marine waters, sockeye may reside in the nearshore or coastal environment for several months but are typically distributed offshore by fall (Burgner 1991). Adult sockeye salmon return to their natal lakes to spawn after spending one to four years at sea.

Status and Trends

Sockeye salmon depend on the quantity and quality of aquatic systems. Sockeye salmon, like the other salmon NMFS has listed, have declined from overharvests, hatcheries, native and non-native exotic species; dams, gravel mining, water diversions, destruction or degradation of riparian habitat, and land use practices (logging, agriculture, and urbanization).

Ozette Lake Sockeye Salmon

Distribution

This ESU includes sockeye salmon that migrate into and rear in the Ozette Lake near the northwest tip of the Olympic Peninsula in Olympic National Park, Washington (Figure 20). The Ozette Lake sockeye salmon ESU includes all naturally spawned anadromous populations of sockeye salmon in Ozette Lake, Ozette River, Coal Creek, and other tributaries flowing into Ozette Lake. Composed of only one population, the Ozette Lake sockeye salmon ESU consists of five spawning aggregations or subpopulations which are grouped according to their spawning locations. The five spawning locations are Umbrella and Crooked creeks, Big Rive, and Olsen's and Allen's beaches (Rawson et al 2009). Two artificial populations are also considered part of this ESU. These artificially propagated populations are no more divergent relative to the local natural population than would be expected between closely related natural populations (70 FR 37160).

Sockeye salmon stock reared at the Makah Tribe's Umbrella Creek Hatchery were included in the ESU, but were not considered essential for recovery of the ESU. However, once the hatchery fish return and spawn in the wild, their progeny are considered as listed under the ESA.

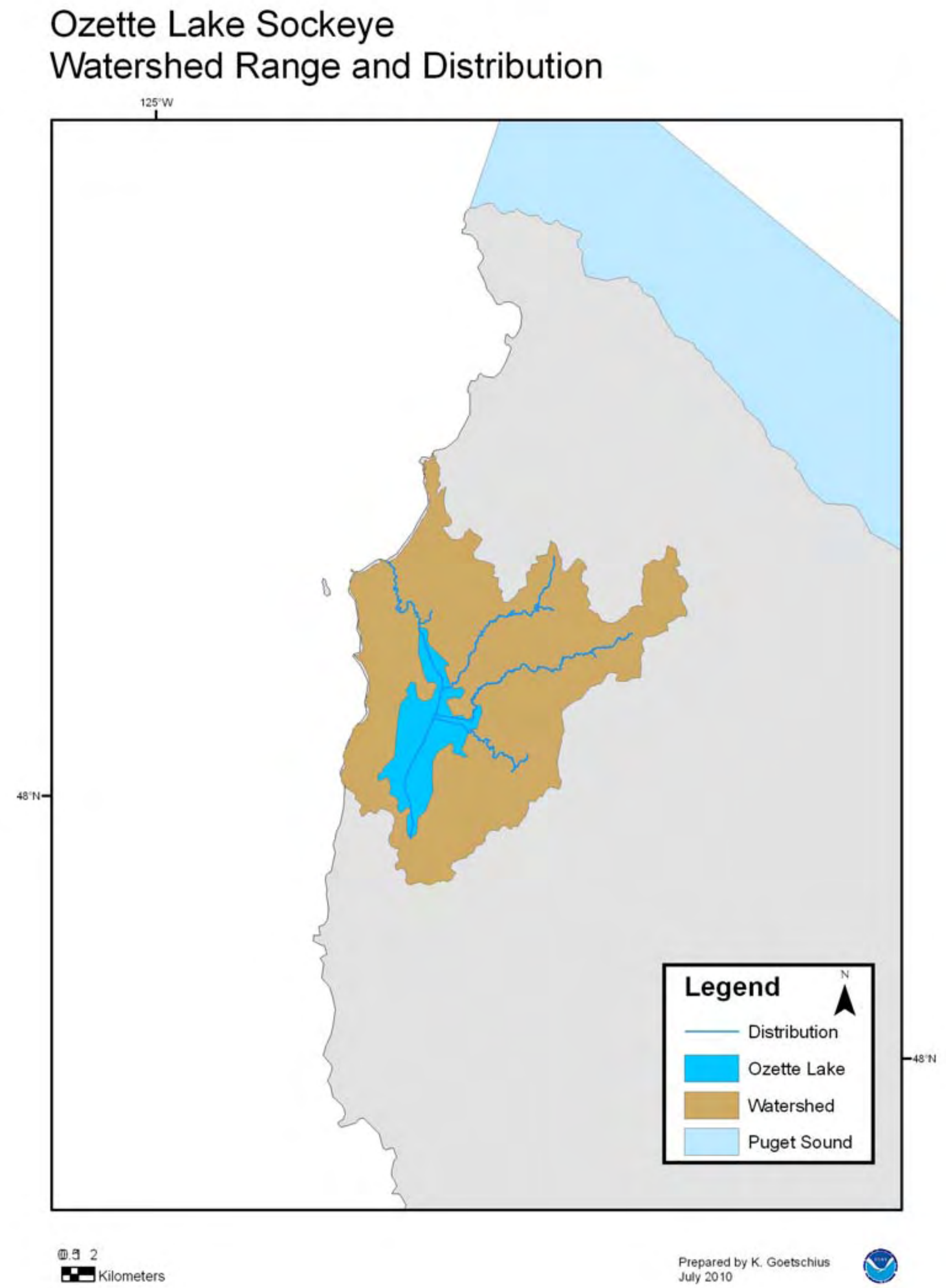


Figure 20**. Ozette Lake Sockeye salmon distribution.**

Life History

Adult Ozette Lake sockeye salmon enter Ozette Lake through the Ozette River from April to early August. Of these, about 99% are four-year old adults. Adults remain in the lake for an extended period before spawning from late October through February. Sockeye salmon spawn primarily in lakeshore upwelling areas in Ozette Lake. Minor spawning may occur below Ozette Lake in the Ozette River or in Coal Creek, a tributary of the Ozette River. Native sockeye salmon do not presently spawn in tributary streams to Ozette Lake but they may have spawned there historically. However, a hatchery program has initiated tributary-spawning by hatchery fish in Umbrella Creek and Big River (Good et al 2005).

Egg incubation occurs from October through May. Emergence and dispersal in the lake occurs from late-February through May. Fry disperse to the limnetic zone in Ozette Lake, where the fish rear. Tributary fry also migrate to the lake soon after emergence. In their second spring after one year of rearing, Ozette Lake sockeye salmon emigrate seaward as age 1+ smolts. The lake is highly productive and water fleas dominate the diet. Sockeye salmon smolts produced in Ozette Lake are documented as the third largest, averaging 4 ½ to 5 inches in length, among west coast sockeye populations examined for average smolt size. The majority of Ozette Lake sockeye salmon return to spawn after two years in the ocean (NMFS 2008b). Ozette Lake also supports a population of kokanee which is not listed under the ESA. There is a large genetic difference between the anadromous and the resident *O. nerka* populations (Crewson et al 2001).

Status and Trends

NMFS originally listed the Ozette Lake sockeye salmon as a threatened species in 1999 (64 FR 14528), and reaffirmed their threatened status on June 28, 2005 (70 FR 37160).

The Ozette Lake sockeye salmon ESU is composed of one historical population, with substantial substructuring of individuals into multiple spawning aggregations.

Historically at least four beaches in the lake were used for spawning but only two beach spawning locations – Allen's and Olsen's beaches – remain today.

The historical abundance of Ozette Lake sockeye salmon is poorly documented, but may have been as high as 50,000 individuals (Blum 1988). Kemmerich (1945) reported a decline in the run size since the 1920s weir counts and Makah Fisheries Management (Makah Fisheries Management 2000) concluded a substantial decline in the Tribal catch of Ozette Lake sockeye salmon occurred at the beginning of the 1950s. Whether decrease in abundance compared to historic estimates is a result of fewer spawning aggregations, lower abundances at each aggregation, or both, is unknown (Good et al 2005).

The most recent (1996-2006) escapement estimates (run size minus broodstock take) range from a low of 1,404 in 1997 to a high of 6,461 in 2004, with a median of approximately 3,800 sockeye per year (geometric mean: 3,353) (Rawson et al 2009). No statistical estimation of trends is reported. However, comparing four year averages (to include four brood years in the average since the species primarily spawn as four-year olds) shows an increase during the period 2000 to 2006: For return years 1996 to 1999 the run size averaged 2,460 sockeye salmon, for the years 2000 to 2003 the run size averaged just over 4,420 fish, and for the years 2004 to 2006, the three-year average abundance estimate was 4,167 sockeye (Data from appendix A in (Rawson et al 2009)). It is estimated that between 35,500 and 121,000 spawners could be normally carried after full recovery (Hard et al 1992b).

The supplemental hatchery program began with out-of-basin stocks and make up an average of 10% of the run. The proportion of beach spawners originating from the hatchery is unknown but it is likely that straying is low. Hatchery originated fish is therefore not believed to have had a major effect on the genetics of the naturally spawned population. However, Ozette Lake sockeye has a relatively low allelic diversity at microsatellite DNA loci compared to other *O. nerka* populations examined in Washington State (Crewson et al 2001). Genetic differences occur between age cohorts.

As different age groups do not spawn with each other, the population may be more vulnerable to significant reductions in population structure due to catastrophic events or unfavorable conditions affecting one year class. Based on this, the Puget Sound TRT's diversity viability criterion is one or more persistent spawning aggregation(s) with each major genetic and life history group being present within the aggregation (Rawson et al 2009). Currently this is not the case; both spawning aggregations are at risk from losing year classes.

Critical Habitat

NMFS designated critical habitat for Ozette Lake sockeye salmon on September 2, 2005 (70 FR 52630). It encompasses areas within the Hoh/Quillayute subbasin, Ozette Lake, and the Ozette Lake watershed. The entire occupied habitat for this ESU is within the single watershed for Ozette Lake. This watershed was given a high conservation value rating. Spawning and rearing PCEs are found in the lake and in portions of three lake tributaries. Ozette River also provides rearing and migration PCEs. The river mouth provides estuarine habitat.

Spawning habitat has been affected by loss of tributary spawning areas and exposure of much of the available beach spawning habitat due to low water levels in summer. Further, native and non-native vegetation as well as sediment have reduced the quantity and suitability of beaches for spawning. The rearing PCE is degraded by excessive predation and competition with introduced non-native species, and by loss of tributary rearing habitat. Migration habitat may be adversely affected by high water temperatures and low water flows in summer which causes a thermal block to migration (La Riviere 1991).

Snake River Sockeye Salmon

The Snake River (SR) sockeye salmon ESU includes all anadromous and residual sockeye from the Snake River basin, Idaho, as well as artificially propagated sockeye salmon from the Redfish Lake Captive Broodstock Program (June 28, 2005, 70 FR

37160). The Redfish Lake is located in the Salmon River basin, a subbasin within the larger Snake River basin (Figure 21).

Life History

SR sockeye salmon are unique compared to other sockeye salmon populations. Sockeye salmon returning to Redfish Lake in Idaho's Stanley Basin travel a greater distance from the sea (approximately 900 miles) to a higher elevation (6,500 ft) than any other sockeye salmon population and are the southern-most population of sockeye salmon in the world (Bjornn et al 1968). Stanley Basin sockeye salmon are separated by 700 or more river miles from two other extant upper Columbia River populations in the Wenatchee River and Okanogan River drainages. These latter populations return to lakes at substantially lower elevations (Wenatchee at 1,870 ft, Okanagon at 912 ft) and occupy different ecoregions.

A resident form of *O. nerka* (kokanee), also occur in the Redfish Lake. The residuals are non-anadromous; they complete their entire life cycle in fresh water. However, studies have shown that some ocean migrating juveniles are progeny of resident females (Rieman et al 1994). The residents also spawn at the same time and in the same location as anadromous sockeye salmon.

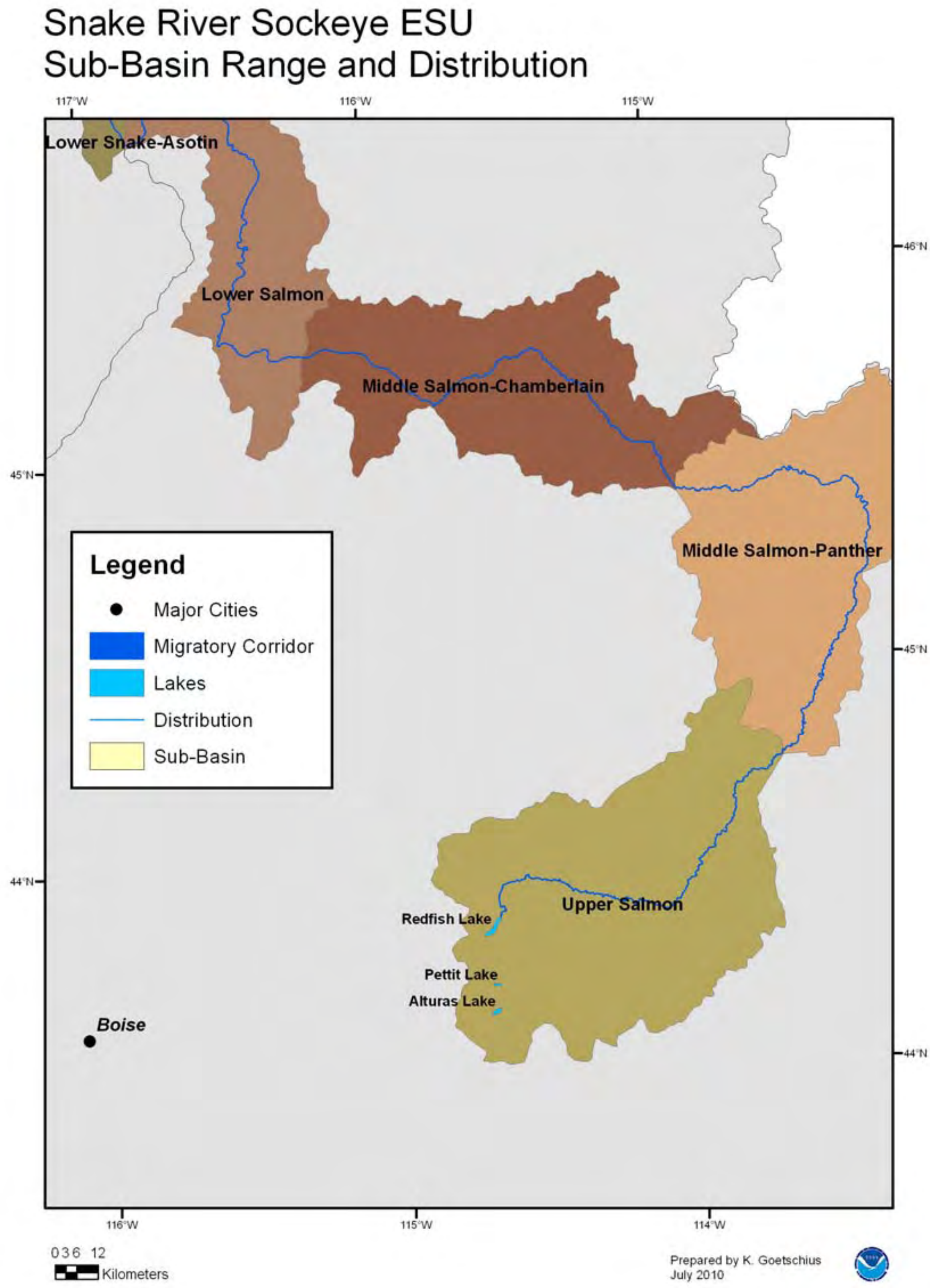


Figure 21. SR Sockeye Salmon distribution

Historically, sockeye salmon entered the Columbia River system in June and July, and arrived at Redfish Lake between August and September (FCRPS 2008). Spawning occurred in lakeshore gravel and generally peaked in October. Fry emerged in the spring (generally April and May) then migrated to open waters of the lake to feed. Juvenile sockeye remained in the lake for one to three years before migrating through the Snake and Columbia Rivers to the ocean. While pre-dam reports indicate that sockeye salmon smolts migrate in May and June, PIT tagged sockeye smolts from Redfish Lake pass Lower Granite Dam from mid-May to mid-July. Adult anadromous sockeye spent two or three years in the open ocean before returning to Redfish Lake to spawn.

Status and Trends

NMFS originally listed SR sockeye salmon as endangered in 1991, and reaffirmed their endangered status on June 28, 2005 (70 FR 37160). Subsequent to the 1991 listing, the residual form of sockeye residing in Redfish Lake was identified. In 1993, NMFS determined that residual sockeye salmon in Redfish Lake was part of the SR sockeye salmon ESU.

The only extant sockeye salmon population in the Snake River basin at the time of listing occurred in Redfish Lake, in the Stanley Basin (upper Salmon River drainage) of Idaho. Other lakes in the Salmon River basin that historically supported sockeye salmon include Alturas Lake above Redfish Lake which was extirpated in the early 1900s as a result of irrigation diversions, although residual sockeye may still exist in the lake (Chapman and Witty 1993). From 1955 to 1965, the Idaho Department of Fish and Game eradicated sockeye salmon from Pettit, Stanley, and Yellowbelly lakes, and built permanent structures on each of the lake outlets that prevented re-entry of anadromous sockeye salmon (Chapman and Witty 1993). Other historic sockeye salmon populations within the Snake River basin include Wallowa Lake (Grande Ronde River drainage, Oregon), Payette Lake (Payette River drainage, Idaho), and Warm Lake (South Fork Salmon River drainage, Idaho) (Gustafson et al 1997a). These populations are now considered extinct.

Recent annual abundances of natural origin sockeye salmon in the Stanley Basin have been extremely low. No natural origin anadromous adults have returned since 1998 and the abundance of residual sockeye salmon in Redfish Lake is unknown. This species is currently entirely supported by adults produced through the captive propagation program.

Adult returns to Redfish Lake during the period 1954 through 1966 ranged from 11 to 4,361 fish (Bjornn et al 1968). In 1985, 1986, and 1987, 11, 29, and 16 sockeye, respectively, were counted at the Redfish Lake weir (Good et al 2005). Only 18 natural origin sockeye salmon have returned to the Stanley Basin since 1987. The first adult returns from the captive brood stock program returned to the Stanley Basin in 1999. From 1999 through 2005, a total of 345 captive brood adults that had migrated to the ocean returned to the Stanley Basin. Recent years have seen an increase in returns to over 600 in 2008 and more than 700 returning adults in 2009. Current smolt-to-adult survival of sockeye originating from the Stanley Basin lakes is rarely greater than 0.3% (Hebdon et al 2004).

Critical Habitat

NMFS designated critical habitat for SR sockeye salmon on December 28, 1993 (58 FR 68543). Designated habitat encompass the waters, waterway bottoms, and adjacent riparian zones of specified lakes and river reaches in the Columbia River that are or were accessible to listed Snake River salmon (except reaches above impassable natural falls, and Dworshak and Hells Canyon Dams). SR sockeye critical habitat areas include the Columbia River from a straight line connecting the west end of the Clatsop jetty (Oregon side) and the west end of the Peacock jetty (Washington side), all river reaches from the estuary upstream to the confluence of the Snake River, and all Snake River reaches upstream to the confluence of the Salmon River; all Salmon River reaches to Alturas Lake Creek; Stanley, Redfish, Yellow Belly, Pettit, and Alturas Lakes (including their inlet and outlet creeks); Alturas Lake Creek and that portion of Valley Creek between Stanley Lake Creek; and the Salmon River.

Conservation values of individual watersheds have not been reported (58 FR 68543). However, all areas occupied and used for migration by the SR sockeye salmon should be considered of high conservation value as the species' distribution is limited to a single lake within the Snake River basin.

The quality and quantity of rearing and juvenile migration PCEs have been reduced by factors such as tilling, water withdrawals, timber harvest, grazing, mining, and alteration of floodplains and riparian vegetation. These activities disrupt access to foraging areas, increase the amount of fines in the steam substrate that support production of aquatic insects, and reduce instream cover. Adult and juvenile migration PCE is affected by four dams in the Snake River basin that obstructs migration and increases mortality of downstream migrating juveniles.

Water quality impairments in the designated critical habitat of the SR sockeye salmon include inputs from fertilizers, insecticides, fungicides, herbicides, surfactants, heavy metals, acids, petroleum products, animal and human sewage, dust suppressants (*e.g.*, magnesium chloride), radionuclides, sediment in the form of turbidity, and other anthropogenic pollutants. Pollutants enter the surface waters and riverine sediments from the headwaters of the Salmon River to the Columbia River estuary 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 the salmon tissue. Sockeye salmon require migration corridors with adequate passage conditions (water quality and quantity available at specific times) to allow access to the various habitats required to complete their life cycle. Multiple exposures to contaminants occur to all life stages throughout the entire range of the SR sockeye salmon.

Steelhead

Description of the Species

Steelhead are native to Pacific Coast streams extending from Alaska south to northwestern Mexico. We discuss the distribution, life history, status, and critical habitat of the 11 endangered and threatened steelhead species separately.

Steelhead have a protracted run time relative to Pacific salmon and do not tend to travel in large schools. Nevertheless, steelhead can be divided into two basic run-types: the stream-maturing type, or summer steelhead, and the ocean-maturing type, or winter steelhead. The summer steelhead enters fresh water in a sexually immature condition between May and October (Busby et al 1996, Nickelson et al 1992a). They then hold in cool, deep holding pools during summer and fall before moving to spawning sites as mature adults in January and February (Barnhart 1986, Nickelson et al 1992). Summer steelhead most commonly occur in streams where snowmelt contributes substantially to the annual hydrograph. The winter steelhead enters fresh water between November and April with well-developed gonads and spawns shortly after river entry (Busby et al 1996, Nickelson et al 1992a). Variations in migration timing exist between populations. Some adults enter coastal streams in the spring, just before spawning (Meehan and Bjornn 1991).

Steelhead typically spawn in small tributaries rather than large, mainstem rivers; spawning distribution often overlap with coho salmon. However, steelhead tend to prefer higher gradients (generally 2-7%, sometimes up to 12% or more) and their distribution tend to extend farther upstream than for coho salmon. Summer steelhead commonly spawn higher in a watershed than do winter steelhead, sometimes even using ephemeral streams from which juveniles are forced to emigrate as flows diminish.

Unlike Pacific salmon, steelhead are iteroparous, or capable of spawning more than once before death (Busby et al 1996). Mostly females spawn more than once but rarely more than twice before dying (Nickelson et al 1992a). Iteroparity is more common among southern steelhead populations than northern populations (Busby et al 1996).

Juveniles rear in fresh water from one to four years, then smolt and migrate to the ocean in March and April (Barnhart 1986). After two to three weeks, in late spring, and following yolk sac absorption, alevins emerge from the gravel and begin actively feeding. The fry usually inhabit shallow water along banks and stream margins of streams (Nickelson et al 1992). As they grow, steelhead juveniles commonly occupy faster flowing water such as riffles. Older and larger juveniles are more risk averse; they stay in deeper water and keep close to cover (Bisson et al 1992, Bisson et al 1988). Some older juveniles move downstream to rear in larger tributaries and mainstem rivers (Nickelson et al 1992a)

Steelhead juveniles are highly territorial, dominance is based on initial size, and high densities result in increased migration. Juvenile steelhead that have established territories migrate little during their first summer (Bisson et al 1988). Steelhead fry and parr hold close to the substratum where flows are lower and sometimes counter to the main stream. Here, steelhead foray up into surface currents for drifting food or prey at invertebrates on the stream bottom (Bisson et al 1988, Kalleberg 1958). Older steelhead commonly uses deeper pools (Bisson et al 1982, Bisson et al 1988).

Juvenile steelhead are opportunistic and feed on a wide variety of aquatic and terrestrial insects (Chapman and Bjornn 1969). Prey species varies with season and availability; they utilize higher prey diversity than sympatric coho salmon (Pert 1987). Prey includes common aquatic stream insects such as caddisflies, mayflies, and stoneflies but also other insects (especially chironomid pupae), zooplankton, and benthic organisms (Merz 2002, Pert 1987). Older juveniles sometimes prey on emerging fry, other fish larvae, crayfish, and even small mammals but these are not a major food source (Merz 2002).

All listed salmonids use shallow, low flow habitats at some point in their life cycle. However, steelhead juveniles use such habitat less than coho salmon and prefer faster flowing stream sections. During winter and spring, juveniles often seek protection under rocks and boulders to escape high flows. Contrary to coho salmon, steelhead seem to avoid overwintering in channels that have organic matter or "muck" as bottom substrate. They may move into inundated floodplains to forage during the high flow season.

In Oregon and California, steelhead may enter estuaries where sand bars close off the estuary, thereby creating low salinity lagoons. The migration of juvenile steelhead to lagoons occurs throughout the year, but is concentrated in the late spring/early summer and in the late fall/early winter period (Shaovalov and Taft 1954, Zedonis 1992). In southern California, two discrete groups of juvenile steelhead use different habitat provided by lagoons: steelhead juveniles that use the upper and fresher areas of coastal lagoons for freshwater rearing throughout the year, and smolts that drop down from the watershed and use the lagoon primarily in the spring prior to seawater entry (Cannata 1998, Zedonis 1992).

Immature steelhead migrate directly offshore during their first summer from whatever point they enter the ocean rather than along the coastal belt as salmon do. During the fall and winter, juveniles move southward and eastward (Hartt and Dell 1986, Nickelson 1992). Steelhead typically reside in marine waters for two or three years prior to returning to their natal stream to spawn as four or five-year olds.

Status and Trends

Steelhead survival depends on the quantity and quality of those aquatic systems they occupy. Steelhead have declined from overharvests, hatcheries, native and non-native exotic species, dams, gravel mining, water diversions, destruction or degradation of riparian habitat, and land use practices (logging, agriculture, and urbanization).

Puget Sound Steelhead DPS

This DPS includes all naturally spawned anadromous winter-run and summer-run steelhead in streams in the river basins of the Strait of Juan de Fuca, Puget Sound, and Hood Canal, Washington, bounded to the west by the Elwha River (inclusive) and to the north by the Nooksack River and Dakota Creek (inclusive), as well as the Green River natural and Hamma Hamma winter-run steelhead hatchery stocks (Figure 22). The remaining hatchery programs are not considered part of the DPS because they are more than moderately diverged from the local native populations.

Life History

The Puget Sound steelhead DPS contains both winter-run and summer-run steelhead. Adult winter-run steelhead generally return to Puget Sound tributaries from December to April (NMFS 2005). Spawning occurs from January to mid-June, with peak spawning occurring from mid-April through May. Prior to spawning, maturing adults hold in pools or in side channels to avoid high winter flows. Less information exists for summer-run steelhead as their smaller run size and higher altitude headwater holding areas have not been conducive for monitoring. Based on information from four streams, adult run time occur from mid-April to October with a higher concentration from July through September (NMFS 2005g).

The majority of juveniles reside in the river system for two years with a minority migrating to the ocean as one or three-year olds. Smoltification and seaward migration occur from April to mid-May. The ocean growth period for Puget Sound steelhead ranges from one to three years in the ocean (Busby et al 1996). Juveniles or adults may spend considerable time in the protected marine environment of the fjord-like Puget Sound during migration to the high seas.

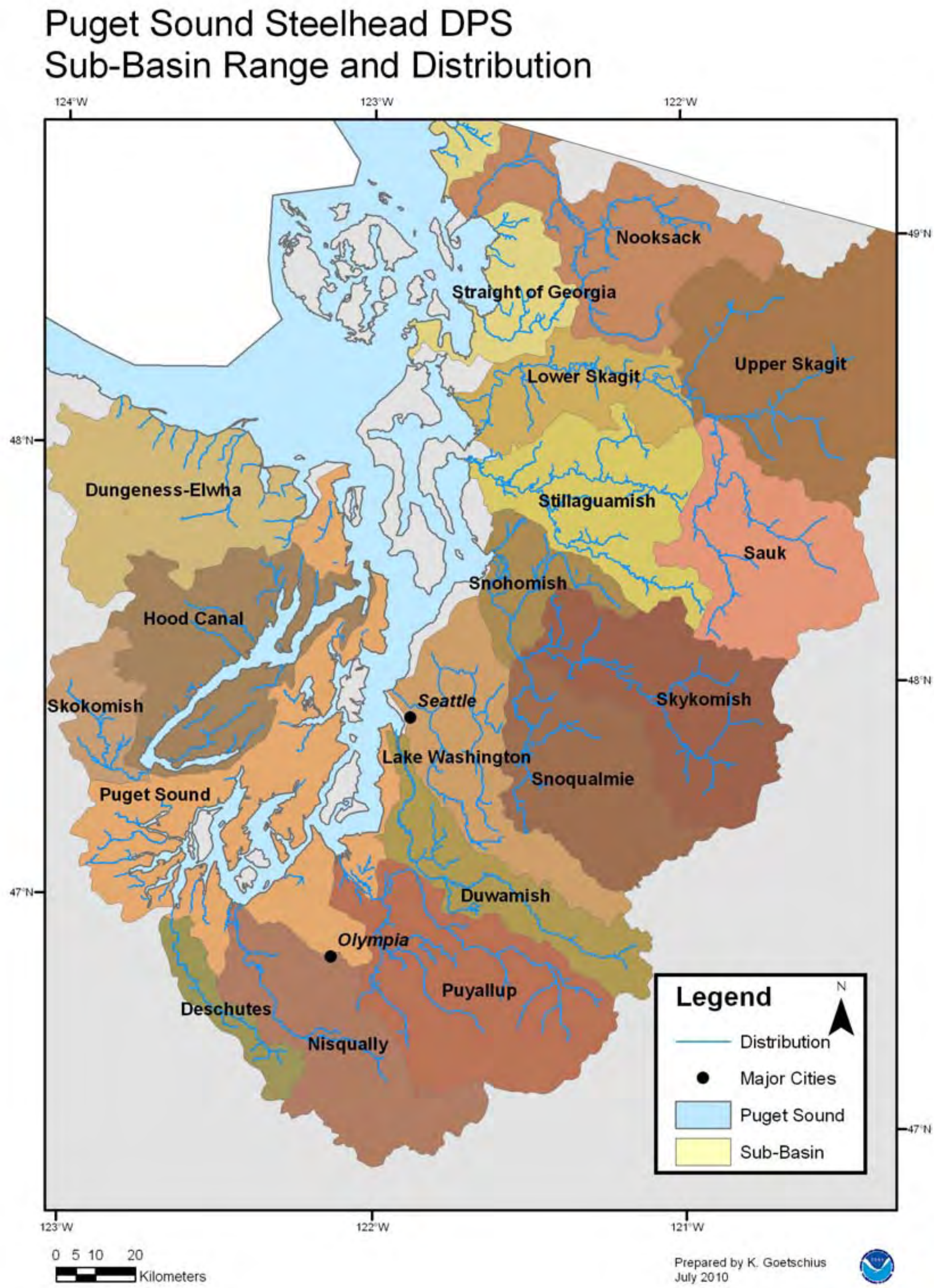


Figure 22. Puget Sound steelhead distribution.

Status and Trends

NMFS listed Puget Sound steelhead as threatened on May 11, 2007 (72 FR 26722). Fifty-three populations of steelhead have been identified in this DPS, of which 37 are winter-run. Summer-run populations are distributed throughout the DPS but are concentrated in northern Puget Sound and Hood Canal; only the Elwha River and Canyon Creek support summer-run steelhead in the rest of the DPS. The Elwha River run, however, is descended from introduced Skamania Hatchery summer-run steelhead. Historical summer-run steelhead in the Green River and Elwha River were likely extirpated in the early 1900s.

In the early 1980s, run size for this DPS was calculated at about 100,000 winter-run fish and 20,000 summer-run fish. By the 1990s, the total run size for four major stocks exceeded 45,000, roughly half of which were natural escapement. The Washington Department of Fish and Wildlife (WDFW) concluded that DPS escapement (excluding the Hamma Hamma population, see below) further declined by 23% during the years from 1999 through 2004 relative to the period from 1994 through 1998 (WDFW 2008). Of the 53 known stocks of Puget Sound steelhead, the WDFW 2002 stock assessment categorized five stocks as healthy, 19 as depressed, one as critical, and 27 of unknown status. The WDFW (2002) data show escapement of natural spawners for the period 1980 to 2004 and the period 2000 to 2004.

Table 36. Geometric mean estimates of escapement of natural spawners for Puget Sound steelhead

Population	Run type	Long Term	5-Year
Canyon	SSH	N/A	Table 36N/A
Skagit	SSH	N/A	N/A
Snohomish	SSH	N/A	N/A
Stillaguamish	SSH	N/A	N/A
Canyon	WSH	N/A	N/A
Dakota	WSH	N/A	N/A
Nooksack	WSH	N/A	N/A
Samish	WSH	501	852
Skagit	WSH	6,994	5,419
Snohomish	WSH	5,283	3,230

Population	Run type	Long Term	5-Year
Stillaguamish	WSH	1,028	550
Tolt	SSH	129	119
Green	SSH	N/A	N/A
Cedar	WSH	138	37
Green	WSH	1,802	1,620
Lk. Washington	WSH	308	37
Nisqually	WSH	1,116	392
Puyallup	WSH	1,714	907
Dewatto	WSH	24	25
Dosewallips	WSH	71	77
Duckabush	WSH	17	18
Hamma Hamma	WSH	30	52
Quilcene	WSH	17	18
Skokomish	WSH	439	203
Tahuya	WSH	114	117
Union	WSH	55	55
Elwha	SSH	N/A	N/A
Dungeness	WSH	311	174
Elwha	WSH	N/A	N/A
McDonald	WSH	150	96
Morse	WSH	106	103

For each population, estimates are provided for both long term (all yr, ca. 1980-2004 for most populations) and for a recent five year period (5 yr, 2000-2004). SSH, summer steelhead; WSH, winter steelhead. (NMFS 2005 status review updated for Puget Sound steelhead, http://www.nwr.noaa.gov/ESA-Salmon-Listings/Salmon-Populations/Steelhead/STPUG.cfm)

In the 1996 and 2005 status reviews, the Skagit and Snohomish Rivers (North Puget Sound) winter-run steelhead were found to produce the largest escapements (Busby et al 1996, NMFS 2005g). The two rivers still produce the largest wild escapement with a recent (2005 to 2008) four-year geometric mean of 5,468 for the Skagit River and an average 2,944 steelhead in Snohomish River for the two years 2005 and 2006 (WDFW 2009). Lake Washington has the lowest abundances of winter-run steelhead with an escapement of less than 50 fish in each year from 2000 through 2004 (WDFW 2008). The stock is now virtually extirpated with only eight and four returning fish in 2007 and 2008, respectively (WDFW 2009). No abundance estimates exist for most of the summer-run populations; all appear to be small, most averaging less than 200 spawners annually.

Long-term trends (1980 to 2004) for the Puget Sound steelhead natural escapement have declined significantly for most populations, especially in southern Puget Sound, and in some populations in northern Puget Sound (Stillaguamish winter-run), Canal (Skokomish winter-run), and along the Strait of Juan de Fuca (Dungeness winter-run) (NMFS 2005g). Positive trends were observed in the Samish winter-run (northern Puget Sound) and the Hamma Hamma winter-run (Hood Canal) populations. The increasing trend on the Hamma Hamma River may be due to a captive rearing program rather than to natural escapement (NMFS 2005g).

The negative trends in escapement of naturally produced fish resulted from peaks in natural escapement in the early 1980s. Still, the period 1995 through 2004 (short-term) showed strong negative trends for several populations. This is especially evident in southern Puget Sound (Green, Lake Washington, Nisqually, and Puyallup winter-run), Hood Canal (Skokomish winter-run), and the Strait of Juan de Fuca (Dungeness winter-run) (NMFS 2005g). As with the long-term trends, positive trends were evident in short-term natural escapement for the Samish and Hamma Hamma winter-run populations, and also in the Snohomish winter-run populations.

Median population growth rates (λ) using 4-year running sums is less than 1, indicating declining population growth, for nearly all populations in the DPS (NMFS 2005g). However, some of the populations with declining recent population growth show only slight declines, (*e.g.*, Samish and Skagit winter-run in northern Puget Sound, and Quilcene and Tahuya winter-run in Hood Canal).

Only two hatchery stocks genetically represent native local populations (Hamma Hamma and Green River natural winter-run). The remaining programs, which account for the vast preponderance of production, are either out-of-DPS derived stocks or were within-DPS stocks that have diverged substantially from local populations. The WDFW (WDFW 1993) estimated that 31 of the 53 stocks were of native origin and predominantly natural production.

Intentional and inadvertent hatchery selection on life history in Chambers Creek winter-run steelhead has resulted in a domesticated strain with a highly modified average run and spawn timing. If interbreeding occurs, such changes can have a detrimental effect on fitness in the wild. However, genetic analyses by Phelps *et al*. (1997) indicated reproductive isolation of and/or poor spawning success by hatchery-origin fish. There is, however, some evidence for introgression by hatchery releases into winter-run steelhead populations in tributaries to the Strait of Juan de Fuca. However, this may have been due to the small size of the naturally-spawning populations relative to the hatchery introductions.

Critical Habitat

NMFS has not designated critical habitat for the Puget Sound steelhead.

Lower Columbia River Steelhead

The LCR steelhead DPS includes all naturally spawned steelhead populations below natural and manmade impassable barriers in streams and tributaries to the Columbia River between the Cowlitz and Wind Rivers, Washington (inclusive), and the Willamette and Hood Rivers, Oregon (inclusive) (Figure 23). Two hatchery populations are included in this species, the Cowlitz Trout Hatchery winter-run population and the Clackamas River population but neither was listed as threatened.

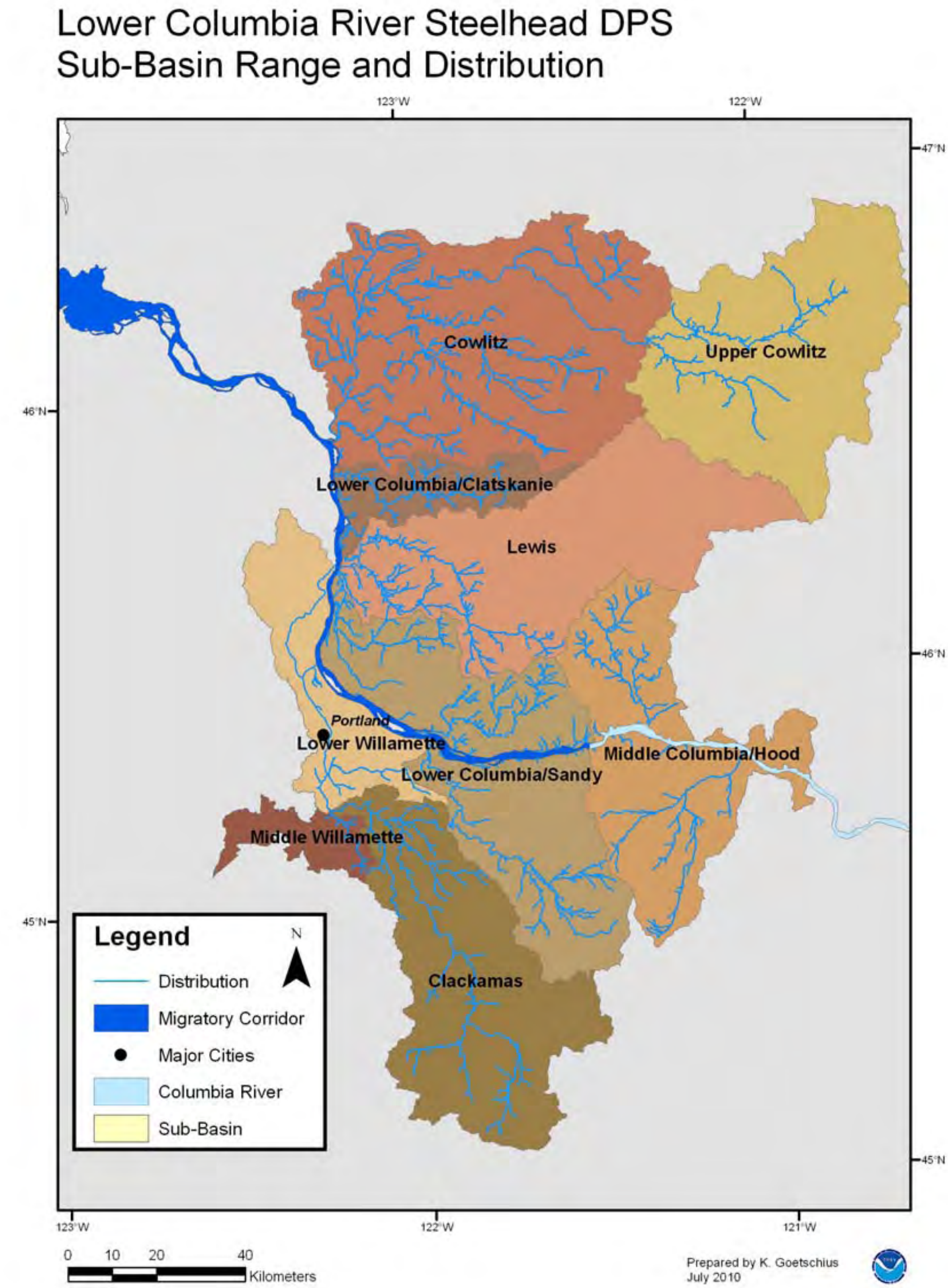


Figure 23. Lower Columbia River steelhead distribution.

Life History

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

The majority of juvenile LCR steelhead remain for two years in freshwater environments before ocean entry in spring. Both winter- and summer-run adults normally return after two years in the marine environment.

Status and Trends

NMFS listed LCR steelhead as threatened on March 19, 1998 (63 FR 13347), and reaffirmed their threatened status on January 5, 2006 (71 FR 834). The LCR steelhead had 17 historically independent winter steelhead populations and 6 independent summer steelhead populations (McElhany *et al.* 2003; Myer *et al*. 2006). All historic LCR steelhead populations are considered extant. However, spatial structure within the historically independent populations, especially on the Washington side, has been substantially reduced by the loss of access to the upper portions of some basins due to tributary hydropower development.

Table 37. LCR Steelhead salmon populations, historic abundances (Good et al. 2005), 1998 – 2002 and 2004 to 2005 geometric mean abundance (Good et al. 2005)(Salmon Scape Query 2009), and hatchery contributions (McElhany et al. 2003; Good et al. 2005).

Population	Run	Historical Abundance	Recent Geometric Mean Total Abundances	Hatchery Abundance Contributions
Cispus River	Winter	Unknown	Unknown	Unknown
Tilton River		Unknown	2,787/--	~73%
Upper Cowlitz River		Unknown	Unknown	Unknown
Lower Cowlitz River		1,672	Unknown	Unknown
Coweeman River		2,243	466/488	~50%

Population	Run	Historical Abundance	Recent Geometric Mean Total Abundances	Hatchery Abundance Contributions
SF Toutle River		2,627	504/616	~2%
NF Toutle River		3,770	196/169	0%
Kalama River		3,165	726/1440	0%
NF Lewis River		713	Unknown	Unknown
EF Lewis River		3,131	Unknown/514	Unknown
Salmon Creek		Unknown	Unknown	Unknown
Washougal River		2,497	323/528	0%
Clackamas River		Unknown	560/--	41%
Sandy River		Unknown	977/--	42%
Lower tributaries		793	Unknown	Unknown
Upper tributaries		243	Unknown	Unknown
Hood River		Unknown	756/--	~52%
Kalama River	Summer	Unknown	--/384	
NF Lewis River	Summer	Unknown	Unknown	Unknown
EF Lewis River	Summer	Unknown	--/474	
Washougal River	Summer	Unknown	--/668	
Hood River	Summer	Unknown	931/--	~83%
Wind River	Summer	2,288	--/627	~5%

All LCR steelhead populations declined from 1980 to 2000, with sharp declines beginning in 1995. Historical counts in some of the larger tributaries (Cowlitz, Kalama, and Sandy Rivers) suggest the population probably exceeded 20,000 fish. During the 1990s, fish abundance dropped to 1,000 to 2,000 fish. Recent abundance estimates of natural-origin spawners range from completely extirpated for some populations above impassable barriers to over 700 fishes for the Kalama and Sandy winter-run populations. A number of the populations have a substantial fraction of hatchery-origin spawners in spawning areas. Many of the long-and short-term trends in abundance of individual populations are negative.

There is a difference in population stability between winter- and summer-run LCR steelhead. The winter-run steelhead in the Cascade region has the highest likelihood of being sustained as it includes a few populations with moderate abundance and positive short-term population growth rates (McElhany et al 2007, Good et al 2005). The Gorge summer-run steelhead is at the highest risk over the long-term as the Hood River population is at high risk of being lost (McElhany et al 2007).

Critical habitat

Critical habitat was designated for the LCR steelhead on September 2, 2005 (70 FR 52488). Of 41 subbasins listed as critical habitat for the LCR steelhead, 28 subbasins were rated as having a high conservation value. Eleven subbasins were rated as having a medium value and two were rated as having a low value to the conservation of the DPS (Table 38).

Table 38. LCR steelhead 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]
Middle-Columbia/Hood	4	(1, 3, <2)	1	(3, 1)	1	(3, 1)
Lower Columbia/Sandy	4	(1, 3)	5	(3, 1)	0	
Lewis	2	(3, 1, 2)	0		0	
Lower Columbia/Clatskanie	1	(3, 1)	0		0	
Upper Cowlitz River	5	(3)	0		0	
Cowlitz	3	(3, 1)	5	(3, 1, 2)	0	
Middle Willamette	0		0		1	(1, 2)
Clackamas	6	(1, <2)	0		0	
Lower Willamette	3	(2, 1, 3)	0		0	
Lower Columbia Corridor	all	(3, 2)	0		0	
Total	28		11		2	

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

Critical habitat is affected by reduced quality of rearing and juvenile PCBs within the lower portion and alluvial valleys of many watersheds; contaminants from agriculture affect both water quality and food production in these reaches of tributaries and in the mainstem Columbia River. Several dams affect adult migration PCE by obstructing the migration corridor. Watersheds which consist of a large proportion of federal lands such as is the case with the Sandy River watershed, have relatively healthy riparian corridors that support attributes of the rearing PCE such as cover, forage, and suitable water quality.

Upper Willamette River Steelhead

The UWR steelhead DPS includes all naturally spawned winter-run steelhead populations below natural and manmade impassable barriers in the Willamette River, Oregon, and its tributaries upstream from Willamette Falls to the Calapooia River (inclusive) (Figure 24)**.** No artificially propagated populations that reside within the historical geographic range of this DPS are included in this listing. Hatchery summer-run steelhead occur in the Willamette Basin but are an out-of-basin population that is not included in this DPS.

Life History

Native steelhead in the Upper Willamette are a late-migrating winter group that enters fresh water in January and February (Howell et al 1985). UWR steelhead do not ascend to their spawning areas until late March or April, which is late compared to other West Coast winter steelhead. Spawning occurs form April to June 1. The unusual run timing may be an adaptation for ascending the Willamette Falls, which may have facilitated reproductive isolation of the stock. The smolt migration past Willamette Falls also begins in early April and proceeds into early June, peaking in early- to mid-May (Howell et al 1985). Smolts generally migrate through the Columbia via Multnomah Channel rather than the mouth of the Willamette River. As with other coastal steelhead, the majority of juveniles smolt and outmigrate after two years; adults return to their natal rivers to spawn after spending two years in the ocean. Repeat spawners are predominantly female and generally account for less than 10% of the total run size (Busby et al 1996).

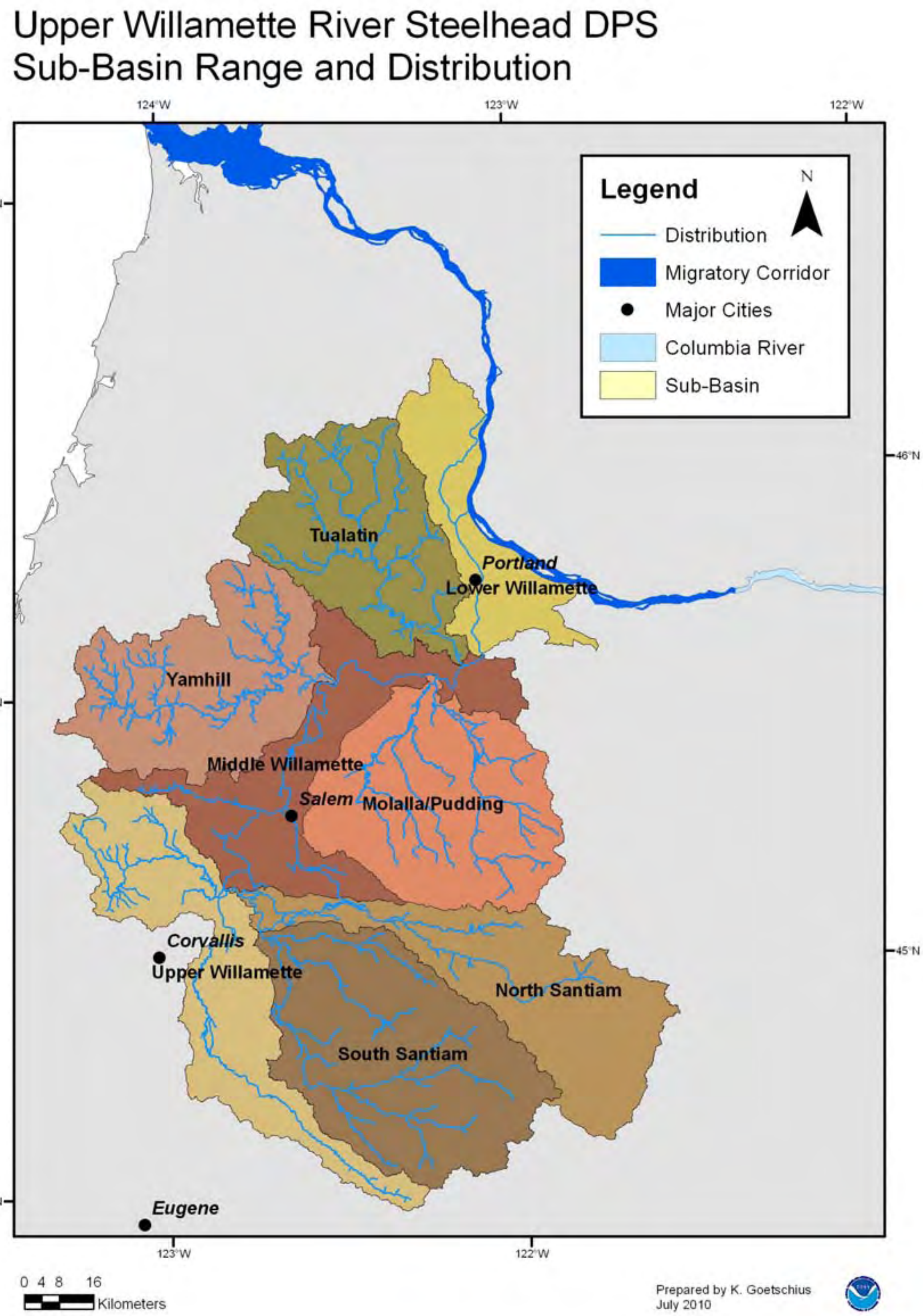


Figure 24. UWR Steelhead distribution.

Status and Trends

NMFS originally listed UWR steelhead as threatened on March 25, 1999 (64 FR 14517), and reaffirmed their threatened status on January 5, 2006 (71 FR 834). Four basins on the east side of the Willamette River historically supported independent populations for the UWR steelhead. All of which remain extant. Data reported in McElhaney et al. (2007) indicate that currently the two largest populations within the DPS are the Santiam River populations. Meaner spawner abundance in both the North and South Santiam River is about 2,100 native winter-run steelhead. However, about 30% of all habitat has been lost due to human activities (McElhany et al 2007). The North Santiam population has been substantially affected by the loss of access to the upper North Santiam basin. The South Santiam subbasin has lost habitat behind non-passable dams in the Quartzville Creek watershed. Notwithstanding the lost spawning habitat, the DPS continues to be spatially well distributed, occupying each of the four major subbasins.

Table 39. Upper Willamette River steelhead salmon populations, core (C) and genetic legacy (G) populations, abundances, and hatchery contributions (McElhany et al 2003, Good et al 2005).

Historic Independent Populations	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Mollala Rivers	Unknown	0.972 rpm	Unknown
North Santiam River	Unknown	0.963 rpm	Unknown
South Santiam River	Unknown	0.917 rpm	Unknown
Calapooia River	Unknown	1.053 rpm	Unknown
Total	Unknown	5,819	

Note: rpm denotes redds per mile.

UWR steelhead are moderately depressed from historical levels (McElhany et al 2007). Average number of late-fall steelhead passing Willamette Falls decreased during the 1990s to less than 5,000 fish. The number again increased to over 10,000 fish in 2001 and 2002. The geometric and arithmetic mean number of late-run steelhead passing Willamette Falls for the period 1998 to 2001 were 5,819 and 6,795, respectively.

Population information for individual basins exist as redds per (river) mile. These redd counts show a declining long-term trend for all populations (Good et al 2005). One population, the Calapooia, had a positive short-term trend during the years from 1990 to 2001. McElhany *et al.* (2007) however, found that the populations had a low risk of extinction. Two of the populations were considered at moderate risk from failed abundances and recruitment levels and two (North and South Santiam Rivers) were considered at low risk given current abundances and recruitment.

Hatchery raised winter-run steelhead were released in the Upper Willamette River up to 1999. These fish were out of basin stocks and had an earlier return timing than the native steelhead. The impact of these releases on the genetic diversity and life history of the native population is unknown. Nevertheless, remains of the early run still exist and the release of hatchery fish has been discontinued.

Critical Habitat

NMFS designated critical habitat for this species on September 2, 2005 (70 FR 52488). It includes all Columbia River estuarine areas and river reaches proceeding upstream to the confluence with the Willamette River and specific stream reaches in the following subbasins: Upper Willamette, North Santiam, South Santiam, Middle Willamette, Molalla/Pudding, Yamhill, Tualatin, and Lower Willamette (NMFS 2005a).

Of the subbasins reviewed in NMFS' assessment of critical habitat for the UWR steelhead, 14 subbasins were rated as having a high conservation value, six were rated as having a medium value, and 17 were rated as having a low conservation value (Table 40).

Table 40. UWR steelhead 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]
Upper Willamette	1	(1, 2)	2	(2, 1)	0	
North Santiam	3	(1, 2)	0		0	
South Santiam	6	(1, 2)	0		0	
Middle Willamette	0		0		4	(2, 1)
Yamhill	0		1	(1, 2)	6	(2, 1)
Molalla/Pudding	1	(1)	2	(2, 1)	3	(2, 1)
Tualatin	0		1	(1, 2)	4	(1, 2, 3)
Lower Willamette	3	(2)	0		0	
Columbia River Corridor	all	(3)	0		0	
Total	14		6		17	

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 critical habitat designated for the UWR steelhead is degraded, and provides a reduced the conservation value necessary for species recovery. Critical habitat is affected by reduced quality of rearing and juvenile PCEs within many watersheds; contaminants from agriculture affect both water quality and food production in several watersheds and in the mainstem Columbia River. Several dams affect adult migration PCE by obstructing the migration corridor.

Middle Columbia River Steelhead

The MCR steelhead DPS includes all naturally spawned steelhead populations below natural and manmade impassable barriers in streams from above the Wind River, Washington, and the Hood River, Oregon (exclusive), upstream to, and including, the Yakima River, Washington, excluding *O. mykiss* from the Snake River Basin. Steelhead from the Snake River basin (described later in this section) are excluded from this DPS. Seven artificial propagation programs are part of this DPS. They include: the Touchet River Endemic, Yakima River Kelt Reconditioning Program (in Satus Creek, Toppenish Creek, Naches River, and Upper Yakima River), Umatilla River, and the Deschutes River steelhead hatchery programs (Figure 25). These artificially propagated populations are considered no more divergent relative to the local natural populations than would be expected between closely related natural populations within the DPS.

According to the ICBTRT (2003), this DPS is composed of 16 populations in four major population groups (Cascade Eastern Slopes Tributaries, John Day River, Walla Walla and Umatilla Rivers, and Yakima River), and one unaffiliated population (Rock Creek).

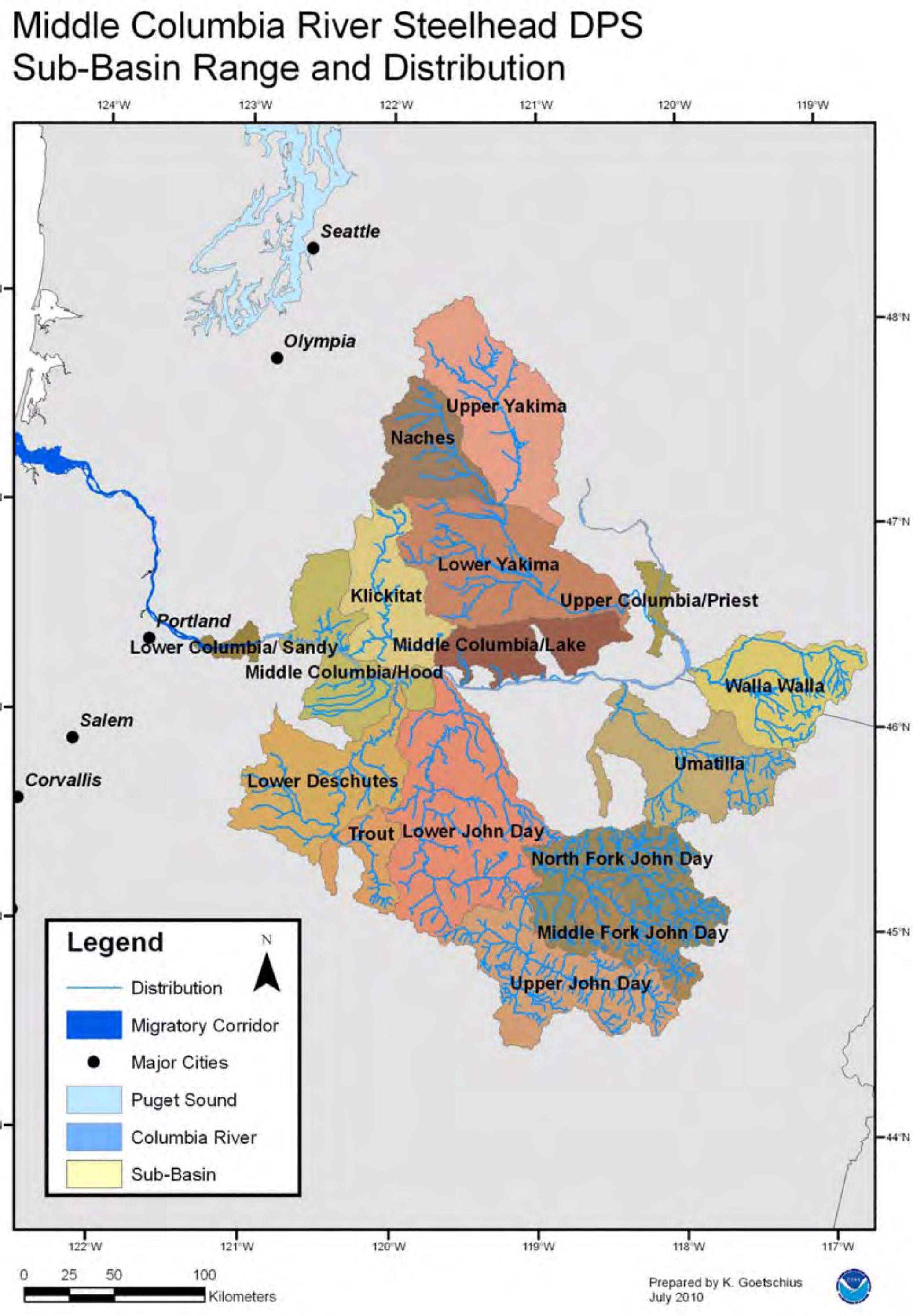


Figure 25. MCR Steelhead distribution.

Life History

MCR steelhead populations are mostly of the summer-run type. Adult steelhead enter fresh water from June through August. The only exceptions are populations of inland winter-run steelhead which occur in the Klickitat River and Fifteenmile Creek (Busby et al 1996).

The majority of juveniles smolt and outmigrate as two-year olds. Most of the rivers in this region produce about equal or higher numbers of adults having spent one year in the ocean as adults having spent two years. However, summer-run steelhead in Klickitat River have a life cycle more like LCR steelhead whereby the majority of returning adults have spent two years in the ocean (Busby et al 1996). Adults may hold in the river up to a year before spawning.

Status and Trends

NMFS listed MCR steelhead as threatened on March 25, 1999 (64 FR 14517), and reaffirmed their threatened status on January 5, 2006 (71 FR 834). The ICBTRT identified 16 extant populations in four major population groups (Cascades Eastern Slopes Tributaries, John Day River, Walla Walla and Umatilla Rivers, and Yakima River) and one unaffiliated independent population (Rock Creek) (ICTRT 2003). There are two extinct populations in the Cascades Eastern Slope major population group: the White Salmon River and the Deschutes Crooked River above the Pelton/Round Butte Dam complex. Present population structure is delineated largely on geographical proximity, topography, distance, ecological similarities or differences.

Table 41. Middle Columbia River steelhead independent populations, abundances, and hatchery contributions (ICTRT 2003; Good et al. 2005)

Major Basins	Population	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Cascade Eastern Slope Tributaries	Klickitat River	Unknown	97-261 reds	Unknown
	White Salmon River	*Unknown*	*Extirpated*	*N/A*
	Fifteenmile Creek	Unknown	2.87 rpm	100%
	East and West Deschutes River*	Unknown	10,026-21,457	38%
	Crooked River	*Unknown*	*Extirpated*	N/A
John Day	John Day upper main	Unknown	926-4,168	96%
	John Day lower main	Unknown	1.4 rpm	0%
	John Day NF			
	upper NF	Unknown	2.57 rpm	0%
	lower NF	Unknown	.52 rpm	0%
	John Day MF	Unknown	3.7 rpm	0%
	John Day SF	Unknown	2.52 rpm	0%
Walla Walla and Umatilla	Umatilla River	Unknown	1,480-5,157	60%
	Walla Walla River	Unknown	Unknown	Unknown
	Touchet River	Unknown	273-527	Unknown
	Willow Creek	*Unknown*	*Extirpated*	*N/A*
Yakima	Yakima River Basin	Unknown	1,058-4,061	97%
	Satus Creek	Unknown	Unknown	Unknown
	Toppenish Creek	Unknown	Unknown	Unknown
	Naches River	Unknown	Unknown	Unknown
	Upper Yakima	Unknown	Unknown	Unknown

*Deschutes River is divided into two historically independent populations: the Eastside and Westside Tributaries

Historic run estimates for the Yakima River imply that annual species abundance may have exceeded 300,000 returning adults (Busby et al 1996). The five-year average (geometric mean) return of natural MCR steelhead for 1997 to 2001 was up from previous years' basin estimates. Returns to the Yakima River, the Deschutes River, and sections of the John Day River system were substantially higher compared to 1992 to 1997 (Good et al 2005). The five-year average for these basins is 298 and 1,492 fish, respectively (Good et al 2005).

Good *et al.* (2005) calculated that the median estimate of long-term trend over 12 indicator data sets was –2.1% per year (–6.9 to 2.9), with 11 of the 12 being negative. Long-term annual population growth rates (λ) were also negative. The median long-term

λ was 0.98, assuming that hatchery spawners do not contribute to production, and .97 assuming that both hatchery- and natural-origin spawners contribute equally.

The median short-term (1990–2001) annual population growth rate assuming no hatchery contribution is estimated to 1.045 (Good et al 2005). Of the 12 datasets, 8 indicator trends have a positive growth rate. Assuming that potential hatchery spawners contributed at the same rate as natural-origin spawners resulted in lower estimates of population growth rates. The median short-term λ under the assumption of equal hatchery- and natural-origin spawner effectiveness was 0.967, with 6 of the 12 indicator trends exhibiting positive growth rates.

The Yakima River populations are at a risk from overall depressed abundances and the majority of spawning occurring in only one tributary (Good et al 2005). The Cascade populations are at risk by the only population with large runs being dominated by out-of-basin strays (Good et al 2005). Returns to sections of the John Day River system increased in the late 1990s and these populations are the only ones with returns consisting mainly of natural spawners (Good et al 2005). However, degraded habitat conditions in the John Day River basin (NMFS 1999) may affect the populations' ability to maintain a positive recruitment during less productive ocean conditions (Good et al 2005).

Status reviews in the 1990s noted considerable reduction in abundances in several basins, loss and degraded freshwater habitat, and stray steelhead in Deschutes River. The population experienced a substantial increase in abundance in some basins since these reviews (Good et al 2005).

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52630).

The CHART assessment for this DPS addressed 15 (HUC4) subbasins containing 106 occupied watersheds (HUC5), as well as the Columbia River rearing/migration corridor (NMFS 2005a). Of all the watersheds, 73 were rated as having a high conservation value, 24 as medium value, and 9 as low value (Table 42). The lower Columbia River rearing/migration corridor downstream of the spawning range is also considered to have a high conservation value.

Table 42. MCR steelhead 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]
Upper Yakima	3	(1, 3, 2)	1	(2, 1)	0	
Naches	3	(1, 3)	0		0	
Lower Yakima	3	(1, 3)	3	(3[1], 2)	0	
Middle Columbia/Lake Wallula	2	(3, <1)	3	(3)	0	
Walla Walla	5	(1, 3, 2)	3	(3, 1, 2)	1	(3)
Umatilla	6	(1, 2)	1	(1, 2)	3	(1, 2)
Middle Columbia/Hood	3	(1, 3)	4	(3, <2)	1	(1)
Klickitat	4	(3, 1)	0		0	
Upper John Day	12	(1, 2, 3)	1	(1, 2)	0	
North Fork John Day	9	(1, 2, 3)	1	(1, 2)	0	
Middle Fork John Day	4	(1, 3)	0		1	(2, 1)
Lower John Day	7	(1, 3)	6	(1, 3, 2)	1	(3, <2)
Lower Deschutes	8[3]	(1, 2)	0		1	(1, =1.9mi)
Trout	2	(1)	1	(1)	1	(1,=1.5mi)
Lower Columbia/Sandy	1	(3)	0		0	
Upper Columbia/Priest Rapids	1	(3)	0		0	
Lower Columbia Corridor	all	(3)[2]				
Total	73		24		9	

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 critical habitat designated for the MCR steelhead is moderately degraded. Critical habitat is affected by reduced quality of rearing and juvenile PCEs within many watersheds; contaminants from agriculture affect both water quality and food production in several watersheds and in the mainstem Columbia River. Loss of riparian vegetation to grazing has resulted in high water temperatures in the John Day basin. Reduced quality of the rearing PCEs has diminished its contribution to the conservation value necessary for the recovery of the species. Several dams affect adult migration PCE by obstructing the migration corridor.

Upper Columbia River Steelhead

The UCR steelhead DPS includes all naturally spawned steelhead populations below natural and man-made impassable barriers in streams in the Columbia River basin upstream from the Yakima River, Washington, to the U.S. - Canada border (Figure 26). The UCR steelhead DPS also includes six artificial propagation programs: the Wenatchee River, Wells Hatchery (in the Methow and Okanogan Rivers), Winthrop NFH, Omak Creek, and the Ringold steelhead hatchery programs. These artificially propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within this DPS.

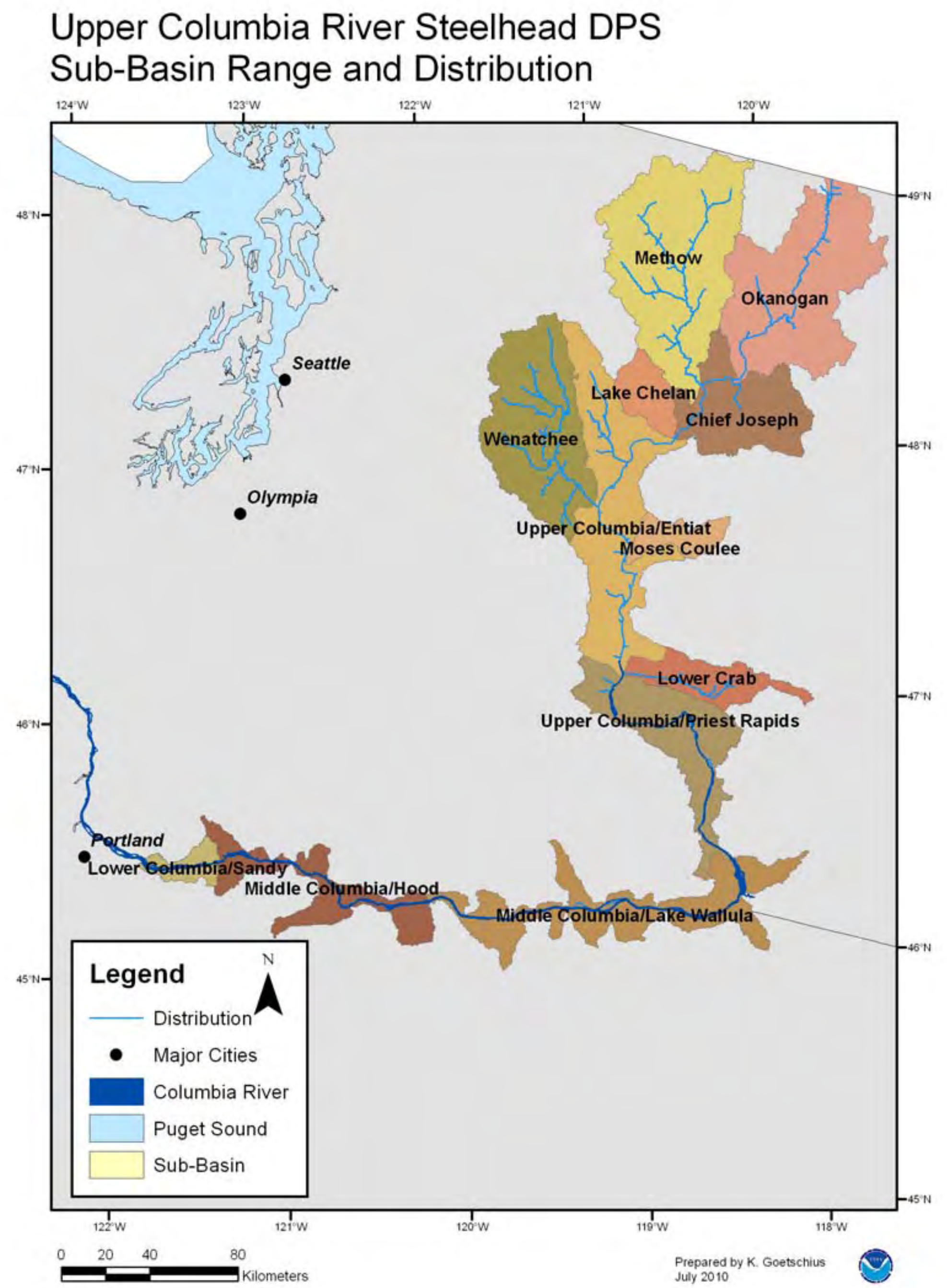


Figure 26. UCR Steelhead distribution.

Life History

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

Status and Trends

NMFS originally listed UCR steelhead as endangered on August 19, 1997 (62 FR 43937). On June 18, 2009, the species' status was changed to threatened (74 FR 42605). The UCR steelhead consisted of four historical independent populations: the Wenatchee, Entiat, Methow, and Okanogan. All populations are extant. The UCR steelhead must navigate over several dams to access spawning areas. The construction of Grand Coulee Dam in 1939 blocked access to over 50% of the river miles formerly available to UCR steelhead (ICTRT 2003).

Returns of both hatchery and naturally produced steelhead to the upper Columbia River have increased in recent years. The average 1997 to 2001 return counted through the Priest Rapids fish ladder was approximately 12,900 fish. The average for the previous five years (1992 to 1996) was 7,800 fish. Abundance estimates of returning naturally produced UCR steelhead were based on extrapolations from mainstem dam counts and associated sampling information (Good et al 2005). The natural component of the annual steelhead run over Priest Rapids Dam increased from an average of 1,040 (1992-1996), representing about 10% of the total adult count, to 2,200 (1997-2001), representing about 17% of the adult count during this period of time (ICTRT 2003).

Table 43. Upper Columbia River Steelhead salmon populations, abundances, and hatchery contributions (Good et al 2005).

Population	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Wenatchee/Entiat rivers	Unknown	1,899-8,036	71%
Methow/Okanogan rivers	Unknown	1,879-12,801	91%
Total	Unknown	3,778-20,837	

Recent population abundances for the Wenatchee and Entiat aggregate population and the Methow population remain well below the minimum abundance thresholds developed for these populations (ICTRT 2003). A five-year geometric mean (1997 to 2001) of approximately 900 naturally produced steelhead returned to the Wenatchee and Entiat rivers (combined). The abundance is well below the minimum abundance thresholds but it represents an improvement over the past (an increasing trend of 3.4% per year).

Regarding the population growth rate of natural production, on average, over the last 20 full brood year returns (1980/81 through 1999/2000 brood years), including adult returns through 2004-2005, UCR steelhead populations have not replaced themselves. Overall adult returns are dominated by hatchery fish, and detailed information is lacking on the productivity of the natural population.

All UCR steelhead populations have reduced genetic diversity from homogenization of populations that occurred during the Grand Coulee Fish Maintenance project from 1939-1943, from 1960, and 1981 (Chapman et al 1994).

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52630).

The CHART assessment for this ESU addressed 10 (HUC4) subbasins containing 41 occupied watersheds (HUC5), as well as the Columbia River rearing/migration corridor. Thirty-one of the watersheds were rated as having a high conservation value, seven as medium value, and three as low value (Table 44). The lower Columbia River rearing/migration corridor downstream of the spawning range is of high conservation value.

Table 44. UCR Steelhead 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]
Chief Joseph	1	(3, 2)	0		2	(2)
Okanogan	2	(3, 1)	3	(3)	0	
Similkameen	1	(3)	0		0	
Methow	7	(1, 3)	0		0	
Lake Chelan	0		1	(1, 3)	0	
Upper Columbia/Entiat	3	(3, 1)	1	(3)	0	
Wenatchee	4	(1, 2, 3)	1	(3, 1)	0	
Moses Coulee	0		0		1	(2)
Lower Crab	0		1	(3)	0	
Upper Columbia/Priest Rapids	3	(3)	0		0	
Middle Columbia/Lake Wallula	5	(3)	0		0	
Middle Columbia/Hood	4	(3)	0		0	
Lower Columbia/Sandy	1	(3)	0		0	
Lower Columbia Corridor	all	(3)	0		0	
Total	31		7		3	

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 critical habitat designated for the UCR steelhead is moderately degraded. Habitat quality in tributary streams varies from excellent in wilderness and roadless areas to poor in areas subject to heavy agricultural and urban development Critical habitat is affected by reduced quality of rearing and juvenile PCEs within many watersheds; contaminants from agriculture affect both water quality and food production in several watersheds and in the mainstem Columbia River. Several dams affect adult migration PCE by obstructing the migration corridor.

Snake River Steelhead

The Snake River (SR) basin steelhead DPS includes all naturally spawned steelhead populations below natural and man-made impassable barriers in streams in the Columbia

River Basin upstream from the Yakima River, Washington, to the U.S. - Canada border (Figure 27). Six artificial propagation programs are also included in the DPS: the Tucannon River, Dworshak National Fish Hatchery, Lolo Creek, North Fork Clearwater, East Fork Salmon River, and the Little Sheep Creek/Imnaha river hatchery programs. These artificially propagated populations are no more divergent relative to the local natural populations than what would be expected between closely related natural populations within the DPS.

Life History

SR basin steelhead are generally classified as summer-run fish. They enter the Columbia River from late June to October. After remaining in the river through the winter, SR basin steelhead spawn the following spring (March to May). Managers recognize two life history patterns within this DPS primarily based on ocean age and adult size upon return: A-run or B-run. A-run steelhead are typically smaller, have a shorter freshwater and ocean residence (generally one year in the ocean), and begin their up-river migration earlier in the year. B-run steelhead are larger, spend more time in fresh water and the ocean (generally two years in ocean), and appear to start their upstream migration later in the year. SR basin steelhead usually smolt after two or three years.

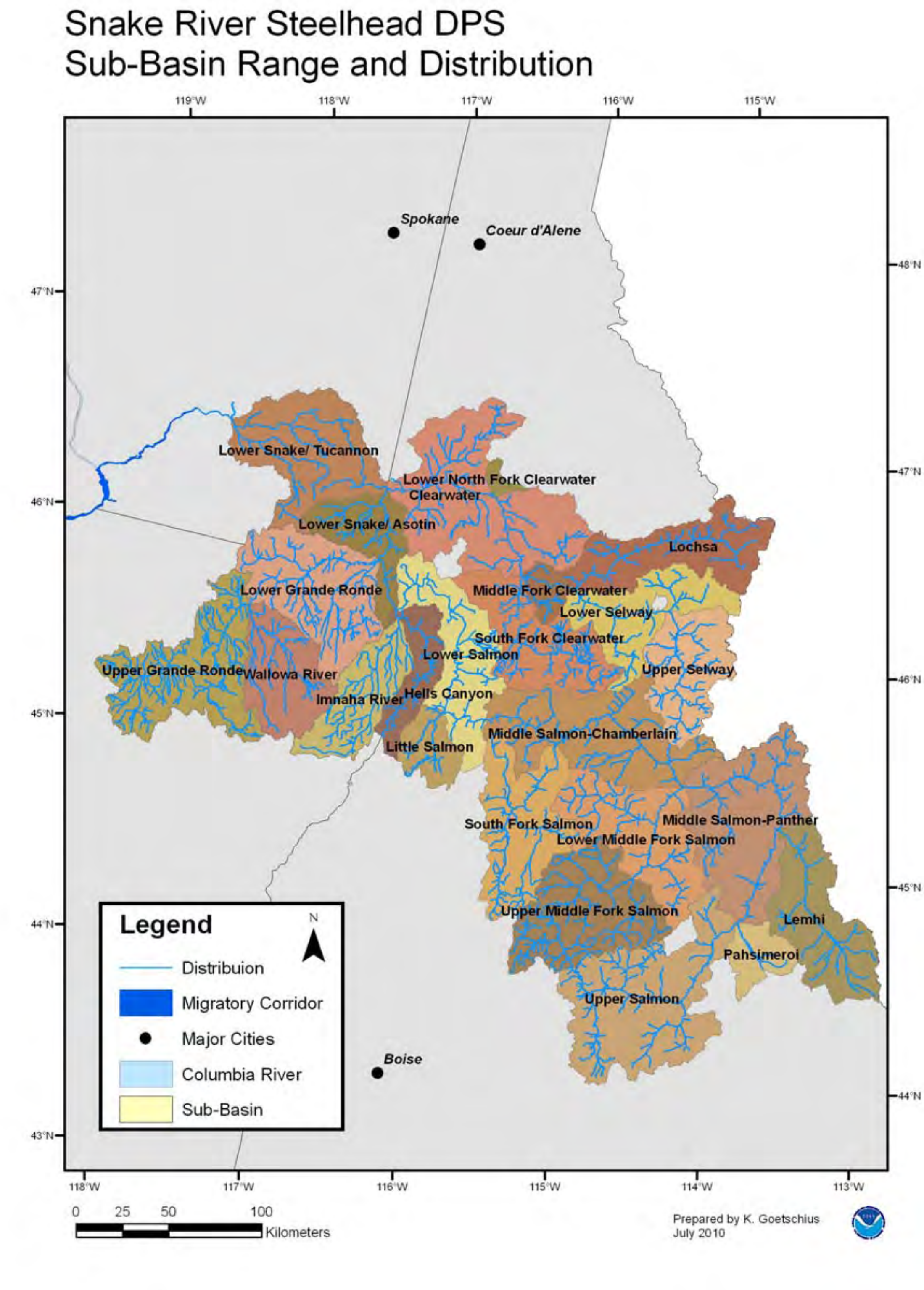


Figure 27. **SR Basin Steelhead distribution.**

Status and Trends

NMFS listed SR basin steelhead as threatened on August 18, 1997 (62 FR 43937), and reaffirmed their threatened status on January 5, 2006 (71 FR 834). The ICBTRT (2003a) identified 23 populations. SR basin steelhead remain spatially well distributed in each of the six major geographic areas in the Snake River basin (Good et al 2005). The SR basin steelhead B- run populations remain particularly depressed.

Table 45. SR Basin Steelhead salmon populations, abundances, and hatchery contributions (Good et al 2005)

River	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Tucannon River	3,000	257-628	26%
Lower Granite run	Unknown	70,721-259,145	86%
Snake A-run	Unknown	50,974-25,950	85%
Snake B-run	Unknown	9,736-33,195	89%
Asotin Creek	Unknown	0-543 redds	Unknown
Upper Grande Ronde River	15,000	1.54 rpm	23%
Joseph Creek	Unknown	1,077-2,385	0%
Imnaha River	4,000	3.7 rpm	20%
Camp Creek	Unknown	55-307	0%
Total	22,000 (min)	?	

Note: rpm denotes redds per mile.

A quantitative assessment for viability of SR steelhead is difficult given limited data on adult spawning escapement for specific tributary production areas. Annual return estimates are limited to counts of the aggregate return over Lower Granite Dam, and spawner estimates for the Tucannon, Asotin, Grande Ronde, and Imnaha Rivers (Table 45). The 2001 return over Lower Granite Dam was substantially higher relative to the low levels seen in the 1990s; the recent geometric five-year mean abundance (14,768 natural returns) was approximately 28% of the interim recovery target level (52,00 natural spawners). The 10-year average for natural-origin steelhead passing Lower Granite Dam between 1996 and 2005 is 28,303 adults. Parr densities in natural production areas, which are another indicator of population status, have been substantially below estimated capacity for several decades. The Snake River supports approximately 63% of the total natural-origin production of steelhead in the Columbia

River Basin. The current condition of Snake River Basin steelhead (Good et al 2005) is summarized below:

There is uncertainty for wild populations given limited data for adult spawners in individual populations. Regarding population growth rate, there are mixed long- and short-term trends in abundance and productivity. Regarding spatial structure, the SR basin steelhead are well distributed with populations remaining in six major areas. However, the core area for B-run steelhead, once located in the North Fork of the Clearwater River, is now inaccessible to steelhead. Finally, genetic diversity is affected by the displacement of natural fish by hatchery fish (declining proportion of natural-origin spawners).

Overall, the abundances remain well below interim recovery criteria. The high proportion of hatchery produced fish in the runs remains a major concern.

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52630). Of the watersheds assessed, 229 were rated as having a high conservation value, 42 as medium value, and 12 as low value (Table 46). The Columbia River migration corridor was also given a high conservation value rating (NMFS 2005a).

Table 46. SR steelhead 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]
Hells Canyon	3	(1, 2, 3)	0		0	
Imnaha River	5	(1)	0		0	
Lower Snake/Asotin	3	(1, 2, 3)	0		0	
Upper Grande Ronde	9	(1, 2)	2	(2, 1)	0	
Wallowa River	5	(1)	1	(1)	0	
Lower Grande Ronde	7	(1)	0		0	
Lower Snake/Tucannon	2	(1, 3)	2	(3, 1)	4	(1, 3)
Palouse River	0		1	(3, 1)	0	
Upper Salmon	20	(1)	6	(1)	1	(1)

HUC 4 Subbasin	HUC 5 Watershed conservation Value (CV)					
	High CV	PCE(s)[1]	Medium CV	PCE(s)[1]	Low CV	PCE(s)[1]
Pahsimeroi	1	(1)	2	(1)	0	
Middle Salmon-Panther	16	(1, <3)	6	(1)	1	(1)
Lemhi	11	(1)[4]	1	(1)	0	
Upper Middle Fork Salmon	13	(1)	0		0	
Lower Middle Fork Salmon	17	(1, <2)	0		0	
Middle Salmon-Chamberlain	14	(1, <3)	3	(3, 1)	1	(1)
South Fork Salmon	15	(1)	0		0	
Lower Salmon	12	(1, 3)	5	(1, 3)	0	
Upper Selway	9	(1, 3)	0		0	
Lower Selway	13	(1, 2)	0		0	
Lochsa	14	(1)	0		0	
Middle Fork Clearwater	2	(1)	0		0	
South Fork Clearwater	8	(1, 3)	3	(1)	2	(1, <3)
Clearwater	16	(1)	10	(1, 2, 3)	3	(1)
Lower Snake River	3	(3)	0		0	
Upper Columbia/Priest Rapids	1	(2)	0		0	
Middle Columbia/Lake Wallula	5	(2)	0		0	
Middle Columbia/Hood	4	(2)	0		0	
Lower Columbia/Sandy	1	(2)	0		0	
Lower Columbia Corridor	all	(3)	0		0	
Total	229		42		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 critical habitat designated for SR basin steelhead is moderately degraded. Critical habitat is affected by reduced quality of rearing and juvenile PCEs within many watersheds; contaminants from agriculture affect both water quality and food production in several watersheds and in the mainstem Columbia River. Loss of riparian vegetation to grazing has resulted in high water temperatures in the John Day basin. These factors have substantially reduced the rearing PCEs contribution to the conservation value necessary for species recovery. Several dams affect adult migration PCE by obstructing the migration corridor.

Northern California Steelhead

The NC steelhead DPS includes all naturally spawned steelhead populations below natural and manmade impassable barriers in California coastal river basins from Redwood Creek southward to, but not including, the Russian River, as well as two artificial propagation programs: the Yeager Creek Hatchery, and North Fork Gualala River Hatchery (Gualala River Steelhead Project) steelhead hatchery programs (Figure 28).

Life History

This DPS includes both winter- and summer –run steelhead. In the Mad and Eel Rivers, immature steelhead may return to fresh water as "half-pounders" after spending only two to four months in the ocean. Generally, a half-pounder will overwinter in fresh water and return to the ocean in the following spring.

Juvenile out-migration appears more closely associated with size than age but generally, throughout their range in California, juveniles spend two years in fresh water (Busby et al 1996). Smolts range from 14-21 cm in length. Juvenile steelhead may migrate to rear in lagoons throughout the year with a peak in the late spring/early summer and in the late fall/early winter period (Shapovalov and Taft 1954, Zedonis 1992).

Steelhead spend anywhere from one to five years in salt water, however, two to three years are most common (Busy et al 1996). Ocean distribution is not well known but coded wire tag recoveries indicate that most NC steelhead migrate north and south along the continental shelf (Barnhart 1986).

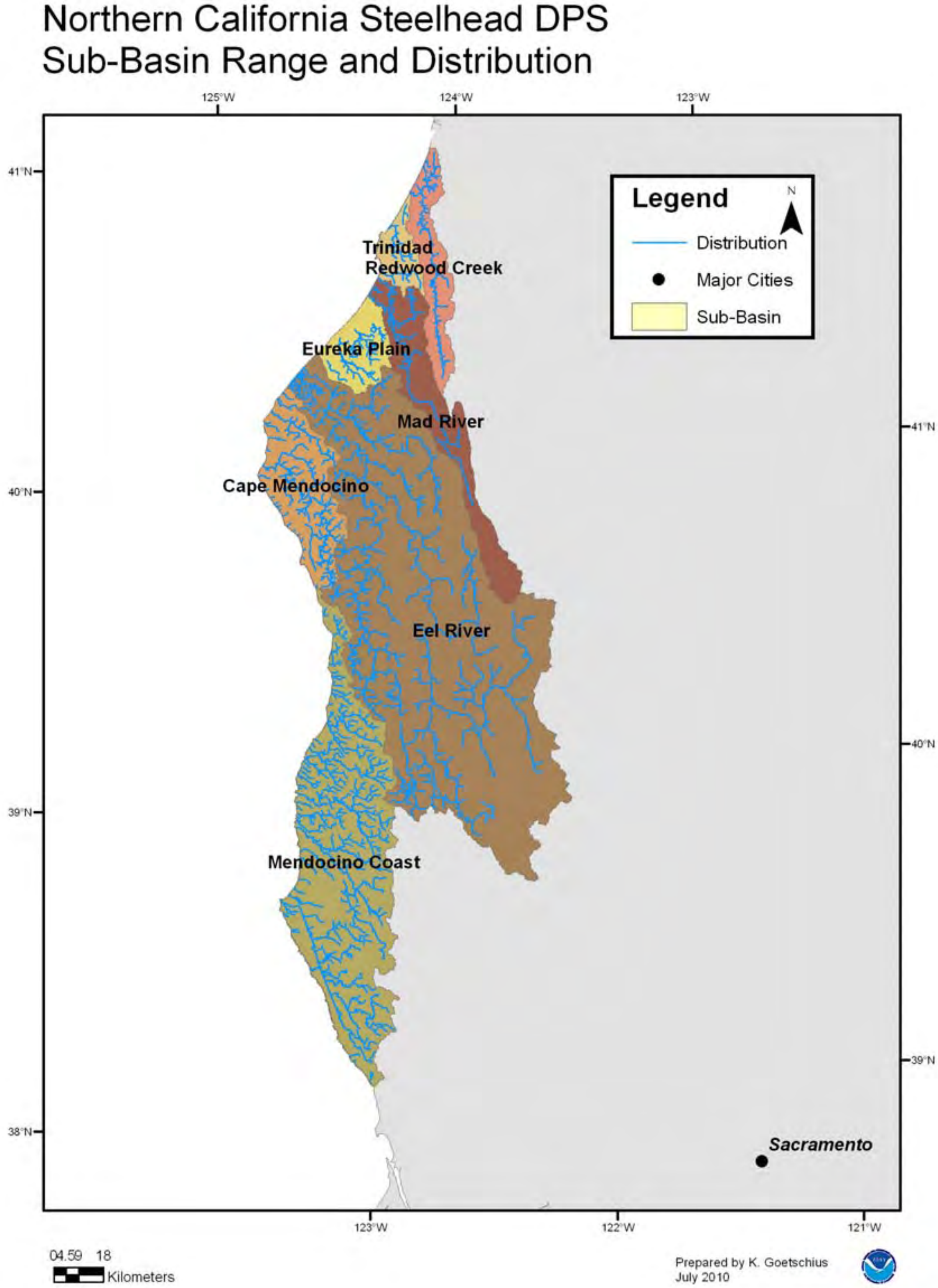


Figure 28. Northern California Steelhead distribution.

Status and Trends

NMFS listed NC steelhead as threatened on June 7, 2000 (65 FR 36074), and reaffirmed their threatened status on January 5, 2006 (71 FR 834). The DPS encompass 15 historic functionally independent populations (and 22 potentially independent populations) of winter steelhead and 10 historic independent populations of summer steelhead (Bjorkstedt et al 2005). Although the DPS spatial structure is relatively intact, the spatial structure and distribution within most watersheds have been adversely affected by barriers and high water temperatures. One of the basins, the Upper Mainstem Eel, has lost too much of its habitat to sustain an independent population today (Spence et al 2008). Production in the Mad River has been substantially reduced by the loss of 36% of its potential steelhead habitat. Large portions of the interior Russian River have been lost to the Coyote Valley Dam on the Russian River and the Warm Springs Hydroelectric Facility on Dry Creek, a major tributary to the Russian River. Spatial distribution in several smaller coastal watersheds has been impacted by constructed barriers blocking access to tributaries and headwaters.

Long-term data sets are limited for the NC steelhead. Before 1960, estimates of abundance specific to this DPS were available from dam counts in the upper Eel River (Cape Horn Dam–annual avg. no. adults was 4,400 in the 1930s), the South Fork Eel River (Benbow Dam–annual avg. no. adults was 19,000 in the 1940s), and the Mad River (Sweasey Dam– annual avg. no. adults was 3,800 in the 1940s). Estimates of steelhead spawning populations for many rivers in this DPS totaled 198,000 by the mid-1960s (Table 47).

Table 47. NC Steelhead salmon historic functionally independent populations and their abundances and hatchery contributions (Good et al 2005)

Population	Historical Abundance	Recent Spawner Abundance	Hatchery Abundance Contributions
Mad River (S)	6,000	162-384	2%
MF Eel River (S)	Unknown	384-1,246	0%
NF Eel River (S)	Uknown	Extirpated	N/A
Mattole River (S)	Unknown	9-30*	Unknown
Redwood Creek (S)	Unknown	6*	Unknown
Van Duzen (W)	10,000	Unknown	Unknown
Mad River (W)	6,000	Unknown	Unknown

Population	Historical Abundance	Recent Spawner Abundance	Hatchery Abundance Contributions
SF Eel River (W)	34,000	2743-20,657	Unknown
Mattole River (W)	12,000	Unknown	Unknown
Redwood Creek (W)	10,000	Unknown	Unknown
Humboldt Bay (W)	3,000	Unknown	Unknown
Freshwater Creek (W)		25-32	
Ten Mile River (W)	9,000	Unknown	Unknown
Noyo River (W)	8,000	186-364*	Unknown
Big River (W)	12,000	Unknown	Unknown
Navarro River (W)	16,000	Unknown	Unknown
Garcia River (W)	4,000	Unknown	Unknown
Gualala River (W)	16,000	Unknown	Unknown
Total	198,000	Unknown	

*From Spence et al. (2008). Redwood Creek abundance is mean count over four generations. Mattole River abundances from surveys conducted between 1996 and 2005. Noyo River abundances from surveys conducted since 2000.
Summer –run steelhead is noted with a (S) and winter-run steelhead with a (W)

During the first status review on this DPS, adult escapement trends were computed from seven populations. Five of the seven populations exhibited declines while two exhibited increases with a range of almost a 6% annual decline to a 3.5% increase. At that time, little information existed for the actual contribution of hatchery fish to natural spawning, and on present total run sizes for the DPS (Busby et al 1996).

More recent time series data are from snorkel counts conducted on adult summer-run steelhead in the Middle Fork Eel River. Good *et al.* (2005) estimated lambda at 0.98 with a 95% confidence interval of 0.93 and 1.04. The result is an overall downward trend in both the long- and short- term. Juvenile data were also recently examined. Both upward and downward trends were apparent (Good et al 2005).

Reduction of summer-run steelhead populations has significantly reduced current DPS diversity compared to historic conditions. Of the 10 summer-run steelhead populations, only four are extant. Of these, only the Middle Fork Eel River population is at moderate risk of extinction, the remaining three are at high risk (Spence et al 2008). Hatchery influence has likely been limited.

Critical Habitat

NMFS designated critical habitat for NC steelhead on September 2, 2005 (70 FR 52488). Specific geographic areas designated include the following CALWATER hydrological units: Redwood Creek, Trinidad, Mad River, Eureka Plain, Eel River, Cape Mendocino, and the Mendocino Coast. The total area of critical habitat includes about 3,000 miles of stream habitat and about 25 square miles of estuarine habitat, mostly within Humboldt Bay.

There are 50 occupied CALWATER Hydologic Subareas (HSA) watersheds within the freshwater and estuarine range of this ESU. Nine watersheds received a low rating, 14 received a medium rating, and 27 received a high rating of conservation value to the ESU NMFS 2005c (Table 48). Two estuarine habitat areas used for rearing and migration (Humboldt Bay and the Eel River Estuary) also received a high conservation value rating.

Table 48. NC steelhead 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]
Redwood Creek	2	(1, 2, 3)	1	(1, 2, 3)	0	
Trindad	1	(1, 2, 3)	0		1	(1, 2, 3)
Mad River	3	(1, 2, 3)	0		1	(1, 2, 3)
Eureka Plain	1	(1, 2, 3)	0		0	
Eel River	10	(1, 2, 3)	9	(1, 2, 3)	0	
Cape Mendocino	1	(1, 2, 3)	0		2	(1, 2, 3)
Mendocino Coast	9	(1, 2, 3)	4	(1, 2, 3)	5	(1, 2, 3)
Total	27		14		9	

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 critical habitat designated for the NC steelhead is moderately degraded. Nevertheless, it does provide some conservation value necessary for species recovery. Within portions of its range, especially the interior Eel River, rearing PCE quality is affected by elevated temperatures by removal of riparian vegetation. Spawning PCE attributes such as the quality of substrate supporting spawning, incubation, and larval development have been generally degraded throughout designated critical habitat by silt and sediment fines in the spawning gravel. Bridges and culverts further restrict access to tributaries in many watersheds, especially in watersheds with forest road construction, thereby reducing the function of adult migration PCE.

Central California Coast Steelhead

The CCC steelhead DPS includes all naturally spawned steelhead populations below natural and manmade impassable barriers in California streams from the Russian River (inclusive) to Aptos Creek (inclusive), and the drainages of San Francisco, San Pablo, and Suisun Bays eastward to Chipps Island at the confluence of the Sacramento and San Joaquin Rivers (Figure 29).

Life History

The DPS is entirely composed of winter-run fish, as are those DPSs to the south. Adults return to the Russian River and migrate upstream from December – April, and smolts emigrate between March – May). (Shapovalov and Taft 1954, Hayes 2008Most spawning takes place from January through April. While age at smoltification typically ranges for one to four years, recent studies indicate that growth rates in Soquel Creek likely prevent juveniles from undergoing smoltification until age two (Sogard et al 2009). Survival in fresh water reaches tends to be higher in summer and lower from winter through spring for year classes 0 and 1 (Sogard et al 2009). Larger individuals also survive more readily than do smaller fish within year classes (Sogard et al 2009). Greater movement of juveniles in fresh water has been observed in winter and spring versus summer and fall time periods. Smaller individuals are more likely to be observed to exceed 0.3 mm per day, and are highest in winter through spring, potentially due to higher water flow rates and greater food availability (Boughton et al 2007, Sogard et al 2009).

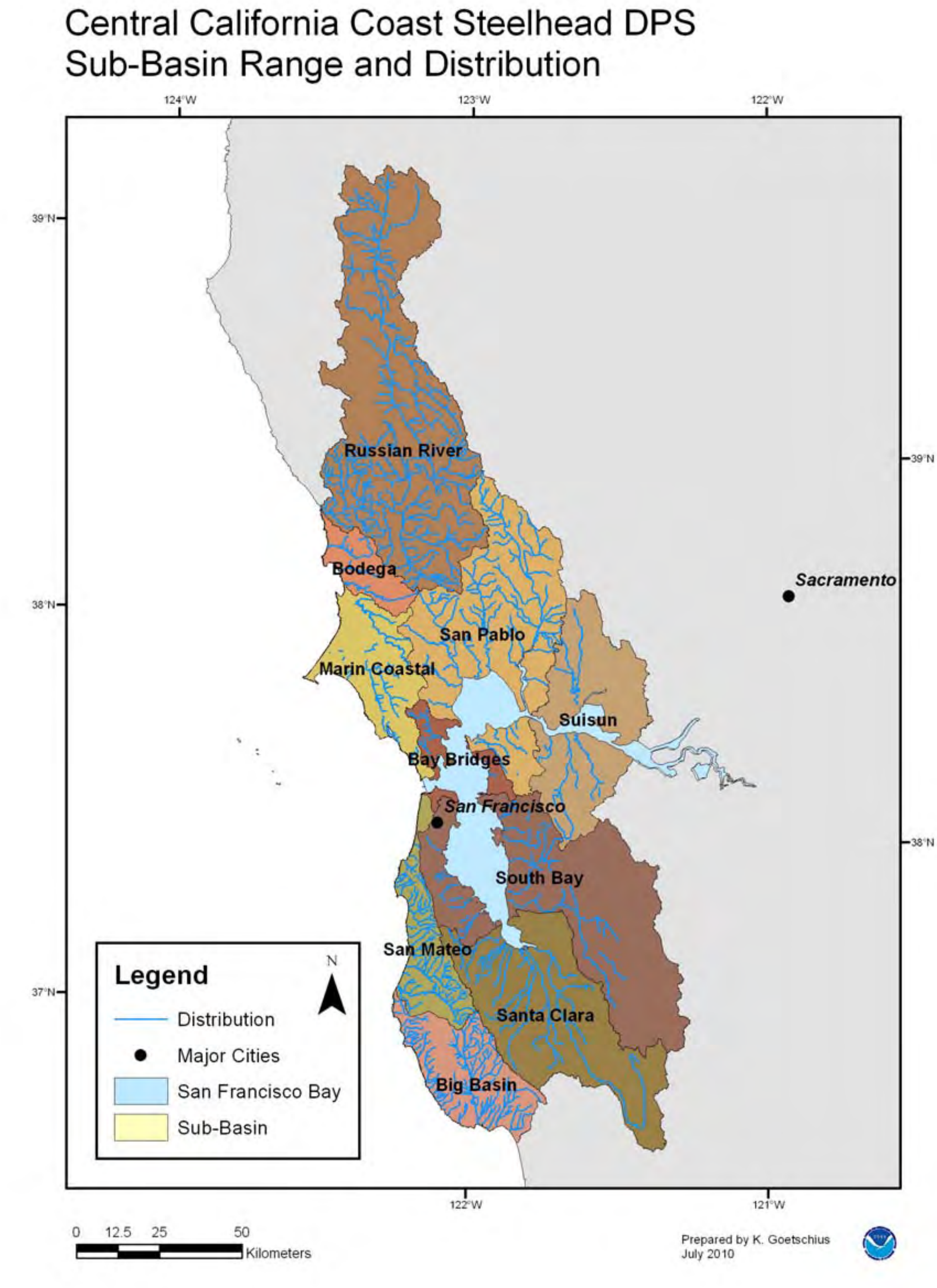


Figure 29. CCC steelhead. Land Cover Class Legend in Error! Reference source not found.**.**

Status and Trends

NMFS listed CCC steelhead as threatened on August 18, 1997 (62 FR 43937), and reaffirmed their threatened status on January 5, 2006 (71 FR 834). The CCC steelhead consisted of nine historic functionally independent populations and 23 potentially independent populations {Bjorkstedt, 2005 #584}. Of the historic functionally independent populations, at least two are extirpated while most of the remaining are nearly extirpated. Current runs in the basins that originally contained the two largest steelhead populations for CCC steelhead, the San Lorenzo and the Russian Rivers, both have been estimated at less than 15% of their abundances just 30 years earlier (Good et al. 2005). Steelhead access to significant portions of the upper Russian River has also been blocked {Busby, 1996 #588;NMFS, 2008 #961}.

Table 49. CCC Steelhead populations, historic population type, abundances, and hatchery contributions {Good, 2005 #601;NMFS, 2008 #961}

Basin	Pop. Type	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Upper Russian River	FI	65,000 (1970)	1,750-7,000 (1994)	Unknown
Lagunitas Creek	PI	Unknown	400-500 (1990s)	Unknown
Stemple Creek	PI	Unknown	Extirpated	N/A
Americano Creek	PI	Unknown	Extirpated	N/A
San Gregorio	FI	1,000 (1973)	Unknown	Unknown
Waddell Creek	PI	481	150 (1994)	Unknown
Scott Creek	D	Unknown	<100 (1991)	Unknown
San Vicente Creek	D	150 (1982)	50 (1994)	Unknown
San Lorenzo River	FI	20,000	<150 (1994)	Unknown
Soquel Creek	PI	500-800 (1982)	<100 (1991)	Unknown
Aptos Creek	PI	200 (1982)	50-75 (1994)	Unknown
Guadalupe River	FI	Unknown	Unknown	Unknown
Napa River	FI	Unknown	Unknown	Unknown
San Leandro River	FI	Unknown	Extirpated*	N/A
San Lorenzo River	FI	20,000 pre-1965	<150 (1994)	N/A
Alameda Creek	FI	Unknown	Extirpated	N/A
Total		94,000	2,400-8,125	

*A remnant stray run may still exist {Leidy, 2005 #1032}
Population type: FI, historic functionally independent; PI, historic potentially independent.

Historically, the entire CCC steelhead DPS may have consisted of an average runs size of 94,000 adults in the early 1960s {Good, 2005 #601}. Information on current CCC steelhead populations consists of anecdotal, sporadic surveys that are limited to only smaller portions of watersheds. Presence-absence data indicated that most (82%) sampled streams (a subset of all historical steelhead streams) had extant populations of juvenile *O. mykiss* (Adams 2000; Good *et al.* 2005). Table YY identifies populations within the CCC steelhead salmon ESU, their abundances, and hatchery input.

Though the information for individual populations is limited, available information strongly suggests that no population is viable. Long-term population sustainability is extremely low for the southern populations in the Santa Cruz mountains and in the San Francisco Bay {NMFS, 2008 #961}. Declines in juvenile southern populations are consistent with the more general estimates of declining abundance in the region {Good, 2005 #601}. The interior Russian River winter-run steelhead has the largest runs with an estimate of an average of over 1,000 spawners; it may be able to be sustained over the long-term but hatchery management has eroded the population’s genetic diversity {Bjorkstedt, 2005 #584;NMFS, 2008 #961}.

Data on abundance trends do not exist for the DPS as a whole or for individual watersheds. Thus, it is not possible to calculate long-term trends or lambda.

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52630). It includes the Russian River watershed, coastal watersheds in Marin County, streams within the San Francisco Bay, and coastal watersheds in the Santa Cruz Mountains down to Apos Creek.

There are 47 occupied HSA watersheds within the freshwater and estuarine range of this ESU. Fourteen watersheds are considered of low conservation value, 13 as having a medium conservation value, and 19 as having a high conservation value to the ESU {NMFS, 2005 #1031} (Table 50). Five 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 50. CCC steelhead 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]
Russian River	7	(1, 2, 3)	2	(1, 2, 3)	1	(1, 2, 3)
Bodega Bay	0		1	(1, 2, 3)	1	(1, 2, 3)
Coastal Marin County	1	(1, 2, 3)	1	(1, 2, 3)	2	(1, 2, 3)
San Mateo	2	(1, 2, 3)	2	(1, 2, 3)	1	(1, 2, 3)
Bay Bridges	1	(estuarine PCEs)	1	(1, 2, 3)	1	(1, 2, 3)
South Bay	1	(estuarine PCEs)	1	(1, 2, 3)	1	(1 mi of 2 and 3)
Santa Clara	1	(estuarine PCEs)	2	(1, 2, 3)	2	(1, 2, 3)
San Pablo	3	(1, 2, 3)	1	(1, 2, 3)	2	(1, 2, 3)
Suisun	0		1	(1, 2, 3)	4	(1, 2, 3)
Big Basin	3	(1, 2, 3)	1	(1, 2, 3)	0	
Total	19		13		15	

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.

Streams throughout the critical habitat have reduced quality of spawning PCEs; sediment fines in spawning gravel have reduced the ability of the substrate attribute to provide well oxygenated and clean water to eggs and alevins. High proportions of fines in bottom substrate also reduce forage by limiting the production of aquatic stream insects adapted to running water. Elevated water temperatures and impaired water quality have further reduced the quality, quantity and function of the rearing PCE within most streams. These impacts have diminished the ability of designated critical habitat to conserve the CCC steelhead.

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 30).

Life History

CCV steelhead are considered winter steelhead and have the longest freshwater migration of any population of winter steelhead. CCV steelhead generally leave the ocean from August through April {Busby, 1996 #588}, and spawn from December through April, with peaks from January though March, in small streams and tributaries where cool, well oxygenated water is available year-round {Hallock, 1961 #604;McEwan, 1996 #618}. 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.

Newly emerged fry move to the shallow, protected areas associated with the stream margin {McEwan, 1996 #618}. 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 also may be used for juvenile rearing. Migratory corridors are downstream of the spawning areas and include the lower mainstems of the Sacramento and San Joaquin rivers and the Delta.

Hallock *et al*. {, 1961 #604} 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 the spring, 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.

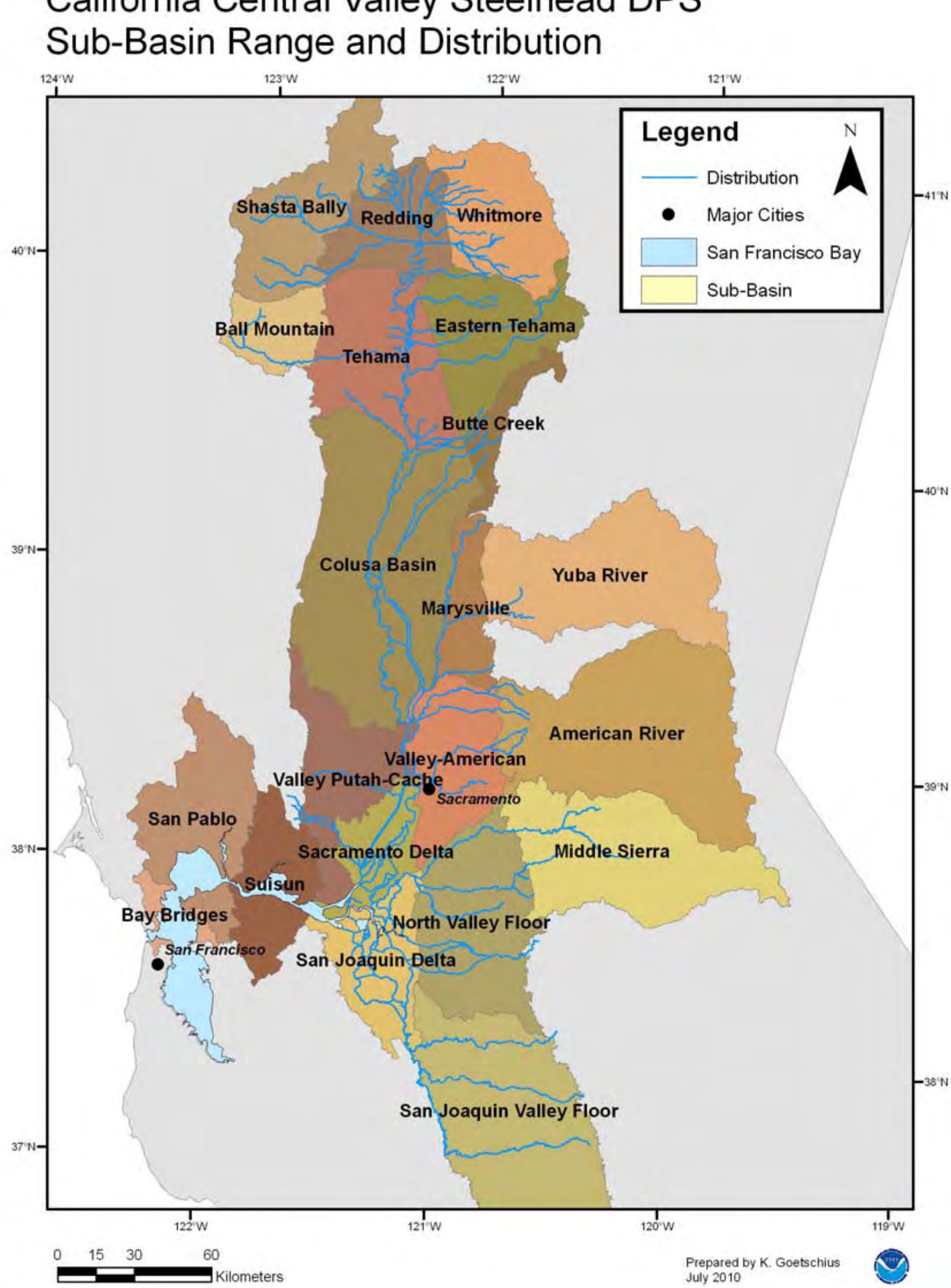


Figure 30. CCV steelhead distribution.

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, 2006 #1931}. Spatial structure and patchiness strongly influenced suitable habitats being isolated due largely to high summer temperatures on the valley floor.

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, 2006 #1931}. 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, 2005 #601}. Steelhead have also been observed in Clear Creek and Stanislaus River {Good, 2005 #601;Demko, 2000 #596}. 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, 2005 #601}. 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, 2001 #619}. By the early 1960s, the steelhead run size had declined to about 40,000 adults {McEwan, 2001 #619}. Over the past 30 years, the naturally spawned steelhead populations in the upper Sacramento River have declined substantially. Hallock *et al.* {, 1961 #604} estimated an average of 20,540 adult steelhead in the Sacramento River, upstream of the Feather River, through the 1960s. 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, 1996 #618;McEwan, 2001 #619}. 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, 2005 #601}.

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, 2009 #1930}.

The CCV steelhead distribution ranged over a wide variety of environmental conditions and likely contained biologically significant amounts of spatially structured genetic diversity {Lindley, 2006 #1931}. 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.

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. 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 #1031}. 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 51. 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
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. 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 (*i.e.*, 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.

South-Central California Coast Steelhead

South-Central California Coast (S-CCC) steelhead include all naturally spawned steelhead populations below natural and manmade impassable barriers in streams from the Pajaro River (inclusive) to, but not including the Santa Maria River, California. No artificially propagated steelhead populations that reside within the historical geographic range of this DPS are included in this designation. The two largest basins overlapping within the range of this DPS include the inland basins of the Pajaro River and the Salinas River (Figure 31).

Life History

Only winter steelhead are found in this DPS. Migration and spawn timing are similar to adjacent steelhead populations. There is limited life history information for steelhead in this DPS.

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52488). There are 29 occupied HSA watersheds within the freshwater and estuarine range of this ESU. The conservation value of 6 watersheds is low, 11 are of medium conservation value, and 12 are of a high conservation value to the ESU {NMFS, 2005 #1031}. One of these occupied watershed units is Morro Bay, which is used as rearing and migratory habitat for steelhead populations that spawn and rear in tributaries to the Bay.

Table 52. Number of South-Central California Coast steelhead 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]
Pajaro River	2	(2, 3, 1)	3	(2, 3, 1)	0	
Carmel River	1	(1, 2, 3)	0		0	
Santa Lucia	1	(1, 2, 3)	0		0	
Salinas	2	(2, 3, 1)	1	(1, 2)	4	(2, 3, <1)
Estero Bay	6	(2, 1, 3)	7	(1, 2, 3)	2	(1, 2, 3)
Total	12		11		6	

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.

Migration and rearing PCEs are degraded throughout critical habitat by elevated stream temperatures and contaminants from urban and agricultural areas. Estuarine PCE is impacted by most estuaries being breached, removal of structures, and contaminants.

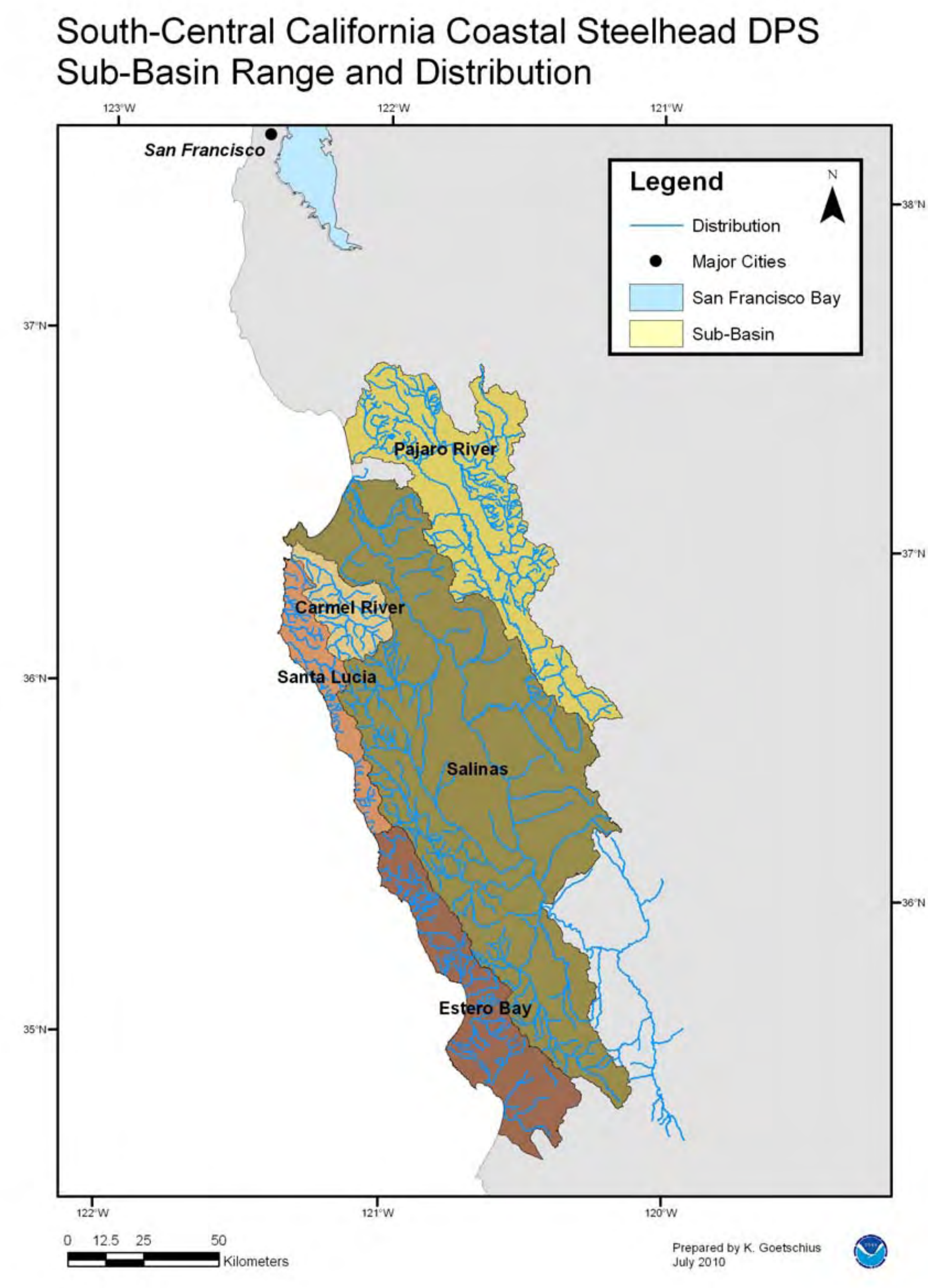


Figure 31**. S-CCC steelhead distribution.**

Southern California Steelhead

The Southern California (SC) steelhead DPS includes all naturally spawned steelhead populations below natural and manmade impassable barriers in streams from the Santa Maria River, San Luis Obispo County, California, (inclusive) to the U.S. - Mexico Border (Figure 32**).** Artificially propagated steelhead that reside within the historical geographic range of this DPS are not included in the listing.

Life History

There is limited life history information for SC steelhead. In general, migration and life history patterns of SC steelhead populations are dependent on rainfall and stream flow {Moore, 1980 #681}. Steelhead within this DPS can withstand higher temperatures compared to populations to the north. The relatively warm and productive waters of the Ventura River have resulted in more rapid growth of juvenile steelhead compared to the more northerly populations {Moore, 1980 #681}.

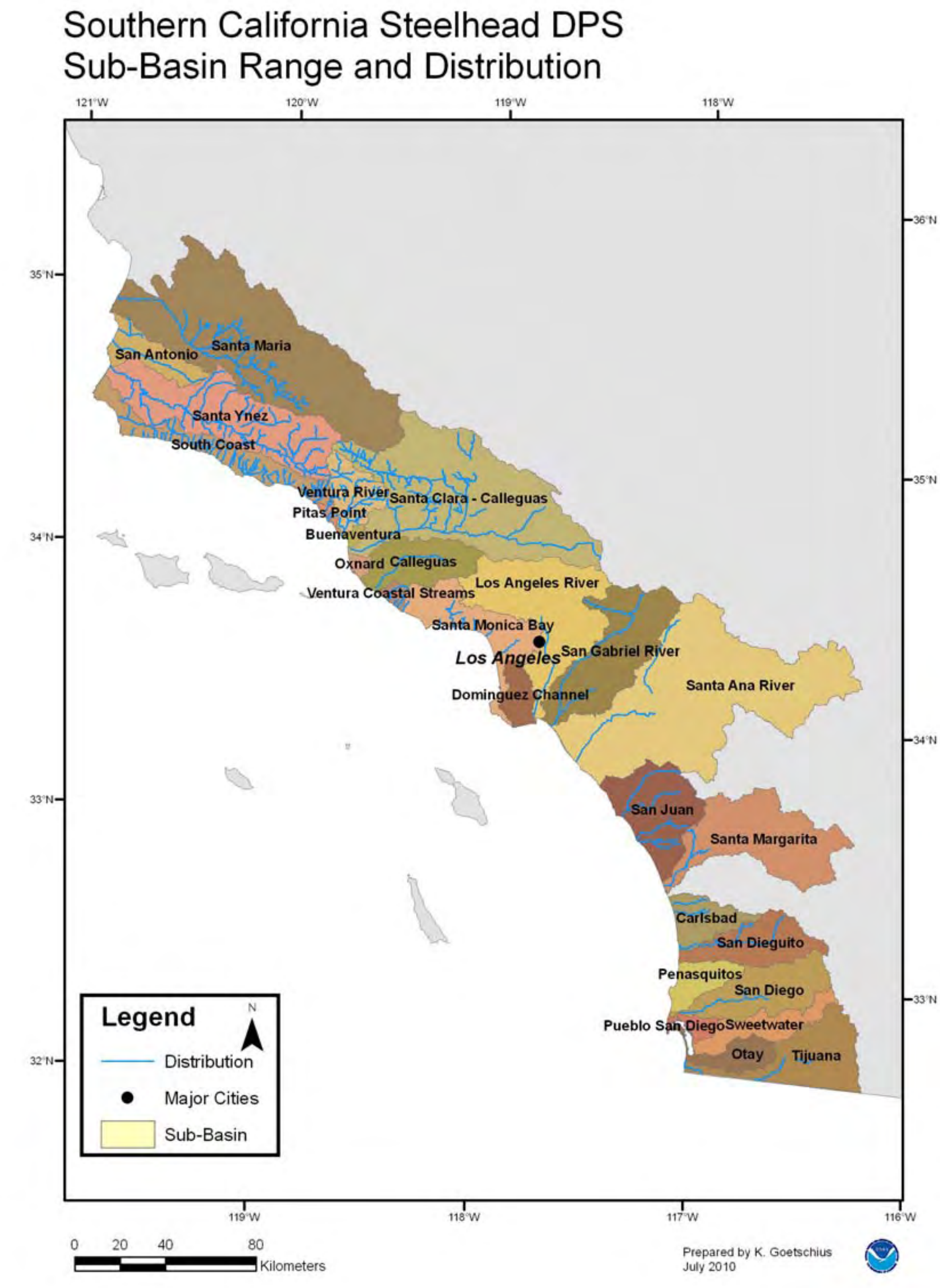


Figure 32. Southern California steelhead distribution.

Status and Trends

NMFS listed the SC steelhead as endangered on August 18, 1997 (62 FR 43937), and reaffirmed their endangered status on January 5, 2006 (71 FR 834). Historic population structure and evaluation of potential stratification of the DPS have not been conducted for this DPS.

Table 53. Southern California Steelhead salmon populations, abundances, and hatchery contributions {Good, 2005 #601}

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

Construction of dams and increasing water temperatures have excluded steelhead distribution in many watersheds throughout southern California. Streams in southern California with steelhead present have declined over the last decade with a southward increase in the proportional loss of populations. Consequently, the SC steelhead have experienced a contraction of its southern range limit {Boughton, 2005 #1029}. This contraction affects the SC steelhead's ability to maintain genetic and life history diversity for adaptation to environmental change

Limited information exists on SC steelhead runs. Based on combined estimates for the Santa Ynez, Ventura, and Santa Clara rivers, and Malibu Creek, an estimated 32,000 to 46,000 adult steelhead occupied this DPS historically. In contrast, less than 500 adults are estimated to occupy the same four waterways presently. The last estimated run size for steelhead in the Ventura River, which has its headwaters in Los Padres National Forest, is 200 adults {Busby, 1996 #588}. Table 53 identifies populations within the SC Steelhead salmon ESU, their abundances, and hatchery input.

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52630). There are 29 HSA watersheds within the freshwater and estuarine range of this ESU designated as critical habitat (Table 54). Three watersheds received a low, five received a medium, and 21 received a high conservation value rating for the ESU {NMFS, 2005 #1031}.

Table 54. Southern California steelhead 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]
Santa Maria	1	(1, 2, 3)	0		1	(1, 2, 3)
Santa Ynez	2	(2, 3, 1)	2	(1, 2, 3)	1	(2, 3, 1)
South Coast	5	(2, 3, 1)	0		0	
Ventura River	2	(2, 3, 1)	2	(1, 2, 3)	0	
Santa Clara-Calleguas	5	(2, 3, 1)	1	(2, 3)	0	
Santa Monica Bay	3	(2, 1, 3)	0		0	
Calleguas	0		0		1	(2, 3)
San Juan	3	(2, 3, 1)	0		0	
Total	21		5		3	

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.

All PCEs have been affected by degraded water quality by pollutants from densely populated areas and agriculture within the DPS. Elevated water temperatures impact rearing and juvenile migration PCEs in all river basins and estuaries. Rearing and spawning PCEs have also been affected throughout the DPS by management or reduction in water quantity. The spawning PCE has also been affected by the combination of erosive geology and land management activities that have resulted in an excessive amount of fines in the spawning gravel of most rivers.

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 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 geographically focused on the aquatic ecosystems in the states of California, Idaho, Oregon, and Washington. Accordingly, the environmental baseline for this consultation focuses on the general status and trends of the aquatic ecosystems in these four states and the consequences of that status for listed resources under NMFS' jurisdiction. We describe the overall principal natural phenomena affecting all listed Pacific salmonids under NMFS jurisdiction in the action area.

We further describe anthropogenic factors through the predominant land and water uses within a region, as land use patterns vary by 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.

Natural Mortality Factors

Available data indicate high natural mortality rates for salmonids, especially in the open ocean/marine environment. According to Bradford {, 1997 #706}, 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, 2004 #533;Bradford, 1997 #706}. In freshwater rearing habitats, the natural mortality rate averages about 70% for all salmonid species {Bradford, 1997 #706}. 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, 1997 #706;Marshall, 1990 #691}. 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.

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, 1993 #707;Kier Associates, 1991 #706;Foott, 2003 #708}. However, disease outbreaks may occur when water quality is diminished and fish are stressed from crowding and diminished flows {Spence, 1996 #523;Guillen, 2003 #692}. Young coho salmon or other salmonid species may become stressed and lose their resistance in higher temperatures {Spence, 1996 #523}. 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 #539}. 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.

Sea lice also cause deadly infestations of wild and farm-grown salmon. *Henneguya salminicola*, a protozoan parasite, is commonly found in the flesh of salmonids. The fish responds by walling off the parasitic infection into a number of cysts that contain milky fluid. This fluid is an accumulation of a large number of parasites. Fish with the longest freshwater residence time as juveniles have the most noticeable infection. The order of prevalence for infection is coho followed by sockeye, Chinook, chum, and pink salmon.

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 #692;CDFG, 2003 #637}.

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.

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, 2005 #696;Hard, 1992 #487;Ford, 2006 #694}. Generally, harbor seals do not feed on salmonids as frequently as California sea lions {Pearcy, 1997 #547}. 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, 1997 #426}. In the Columbia River, salmonids may contribute substantially to sea lion diet at specific times and locations {Pearcy, 1997 #547}. Spring Chinook salmon and steelhead are subject to pinniped predation when they return to the estuary as adults {NMFS, 2006 #986}. 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 {FCRPS, 2008 #658}.

NOAA Fisheries has granted permits to the states of Idaho, Oregon, and Washington for the lethal removal of individual California sea lions that prey on adult spring-run Chinook salmon in the tailrace of Bonneville Dam under section 120 of the Marine Mammal Protection Act {NMFS, 2006 #986}. This action may increase the survival of adult Chinook salmon and steelhead. The Humane Society of the U.S. unsuccessfully challenged NOAA Fisheries' issuance of these permits in the U.S. District Court of Oregon (Humane Society of the U.S. v. Gutierrez, 625 F. Supp. 2d 1052 (D. Or. 2008)). The Ninth Circuit denied the Humane Society's request for a stay of the action pending appeal (Humane Society of the U.S. v. Gutierrez, 5558 F.3d 896 (9th Cir. 2009)). The appeal is currently pending in the Ninth Circuit Court of Appeals.

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 #547}. 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, 2005 #178}. Recent 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, 2006 #715}. 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, 2006 #697;Collis, 2007 #1437}.

Antolos *et al.* {, 2005 #722} 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, 2008 #988}. 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 {FCRPS, 2008 #658}.

Fish Predation

Pikeminnows (*Ptychocheilus oregonensis*) are significant predators of yearling juvenile migrants {Friesen, 1999 #690}. Chinook salmon were 29% of the prey of northern pikeminnows in lower Columbia reservoirs, 49% in the lower Snake River, and 64% downstream of Bonneville Dam. Sockeye smolts comprise a very small fraction of the overall number of migrating smolts {Ferguson, 2006 #714} in any given year. The significance of fish predation on juvenile chum is unknown. There is little direct evidence that piscivorous fish in the Columbia River consume juvenile sockeye salmon. The ongoing Northern Pikeminnow Management Program (NPMP) has reduced predation-related juvenile salmonid mortality since 1990. Benefits of recent northern pikeminnow management activities to chum salmon are unknown. However, it may be comparable to those for other salmon species with a sub-yearling juvenile life history {Friesen, 1999 #690}.

The primary fish predators in estuaries are probably adult salmonids or juvenile salmonids which emigrate at older and larger sizes than others. They include cutthroat trout (*O. clarki*) or steelhead smolts preying on chum or pink salmon 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 #541;Beamish, 1995 #686;Beamish, 1992 #685}.

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 #674;Buchwalter, 2004 #106}. Nevertheless, fire is also one of the dominant habitat-forming processes in mountain streams {Bisson, 2003 #678}. 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 #674;Greswell, 1999 #675}. 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 #673}. 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 #675}.

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, 2000 #677}. 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, 2003 #107}, {Minshall, 2001 #672;Buchwalter, 2004 #106}. 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, 2003 #107}.

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, 1993 #1942;Beamish, 2009 #1943;Finney, 2002 #1944}. Sediment cores reconstructed for 2,200-year records have shown that Northeastern Pacific fish stocks have historically been regulated by these climate regimes {Finney, 2002 #1944}. 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, 2009 #1943}. A trend of an increasing Aleutian Low pressure indicates high pink and chum salmon production and low production of coho and Chinook salmon {Beamish, 2009 #1943}. The abundance and distribution of salmon and zooplankton also relate to shifts in North Pacific atmosphere and ocean climate {Francis, 1994 #1945}.

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, 2000 #1946;Mantua, 1997 #736}. The reversals in 1947 and 1977 correspond to dramatic shifts in salmon production regimes in the North Pacific Ocean {Mantua, 1997 #736}. 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 #1947}. During the post-1977 regime when ocean productivity was generally lower (residuals were negative), the PDO was negative {Levin, 2003 #1947}.

A smaller, less pervasive regime shift occurred in 1989 {Hare, 2000 #1946}. Beamish *et al.*{, 2000 #1949} 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, 2000 #1949}.

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 #1950}. 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 #709}. 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, 1992 #487}.

Carbon dioxide emissions are also predicted to have major environmental impacts along the west coast of North America during the 21st century and beyond {CIG, 2004 #594;IPCC, 2001 #709}. 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 #1950}. The Intergovernmental Panel on Climate Change (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 #1950}. 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 #1950}.

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, 2009 #1951}. Food availability and water temperature may affect proper maturation and smoltification and feeding behavior {Mangel, 1994 #1952}. 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, 2005 #1953}. 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, 2004 #1954}. If climate change leads to an overall decrease in the availability of food, then returning fish will likely be smaller {Mangel, 1994 #1952}. Finally, future climatic warming could lead to alterations of river temperature regimes, which could further reduce available fish habitat {Yates, 2008 #1955}.

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 {CIG, 2004 #594;IPCC, 2001 #709}.

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.

Anthropogenic Mortality Factors

In this section we address anthropogenic threats in the geographic regions across the action area. Among the threats discussed are the “four Hs”: hatcheries, harvest, hydropower, and habitat. Prior to discussion of each geographic region, three major issues are highlighted: pesticide contamination, elevated water temperature, and loss of habitat/habitat connectivity. These three factors are the most relevant to the current analysis. We provide information on pesticide detections in the aquatic environment and highlight their background levels from past and ongoing anthropogenic activities. This information is pertinent to EPA’s proposed registration of azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, methamidophos, methidathion, methyl parathion, naled, phosmet, and phorate in the U.S. and its territories. As water temperature plays such a strong role in salmonid distribution, we also provide a general discussion of anthropogenic temperature impacts. Finally, we discuss the health of riparian systems and floodplain connectivity, as this habitat is vital to salmonid survival.

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, 2006 #563}. 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.* {, 2006 #563}, 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 also present more recent unpublished data on the chemicals and degradates addressed in this Opinion from the NAWQA program and state databases maintained by California and Washington. As far as NMFS was able to ascertain, neither Oregon nor Idaho maintain publicly available state-wide water quality databases. The California and Washington databases include some data from the NAWQA, and the data are from more localized studies. The NAWQA database included measurements for all parent pesticides considered in this Opinion except bensulide, methamidophos, and naled. Both the California Department of Pesticide Regulations (CDPR) and the Department of Washington Environmental

Information Management (EIM) databases included monitoring data for all parent pesticides. Measurements of toxicologically important degradates varied more between databases but at least one of the databases included monitoring data for dichlorvos, azinphos methyl oxon, methidathion oxon, methyl paraxon, phorate oxon, phosmet oxon, disulfoton sulfoxide, disulfoton sulfone, fenamiphos sulfoxide, and/or fenamiphos sulfone. Generally the degradates were detected in <1% of samples taken, with the exception of disulfoton sulfone, which was detected in 3 – 5% of samples in the Washington EIM database and the NAWQA database. Disulfoton sulfone is more persistent in the environment than the parent disulfoton.

Overall, data from those databases are relatively consistent in regard to pesticides addressed in this Opinion, with azinphos methyl, ethoprop, and dimethoate generally being the most frequently quantifiable parent compounds. Azinphos methyl was measured in concentrations ranging from 0.002 – 7.35 μg/L, and ethoprop was measured in concentrations ranging from 0.001 – 5.75 μg/L in the NAWQA database. Dimethoate appeared most frequently in the CDPR database, and was measured in concentrations ranging from 0.030 – 11.31 μg/L. Azinphos methyl was also the most frequently detected in the Washington EIM database, with concentrations ranging from 0.0003 – 0.740 μg/L. Disulfoton sulfone was measured in concentrations ranging from 0.004 – 0.28 μg/L.

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, 2006 #563}.

About 40 pesticide compounds accounted for most detections in water, fish, or bed sediment. Twenty-four pesticides and one degradate were each detected in more than 10% of streams in agricultural, urban, or mixed land use settings. These 25 pesticide compounds include 11 herbicides used most heavily in agriculture during the study period (plus the atrazine degradate, deethylatrazine); 7 herbicides used extensively for non-agricultural purposes; and 6 insecticides used in both agricultural and urban settings, but most intensively in urban settings. Five of these 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 {, 2008 #989} determined that current use of chlorpyrifos, diazinon, and malathion is likely to jeopardize the continued existence of 27 listed salmonid ESUs/DPSs. NMFS (2009) further determined that current use of carbaryl and carbofuran is likely to jeopardize the continued existence of 22 ESUs/DPSs; and the current use of methomyl is likely to jeopardize the continued existence of 18 ESUs/DPSs of listed salmonids.

During the 1992 – 2001 study period, thirteen organochlorine pesticide compounds, including historically used parent pesticides and their degradates and by-products, were each found in more than 10% of fish or bed-sediment samples from streams draining watersheds with either agricultural, urban, or mixed land use {Gilliom, 2006 #563}.

Additionally, more frequent detections and higher concentrations of insecticides occur in sampled urban streams {Gilliom, 2006 #563}. Diazinon, chlorpyrifos, carbaryl, and malathion nationally ranked 2nd, 4th, 8th, and 15th among pesticides in frequencies of outdoor applications for home- and garden use in 1992 {Whitmore, 1992 #724}. These same insecticides accounted for the most insecticide detections in urban streams. Diazinon and carbaryl were the most frequently detected and were found at frequencies and levels comparable to those for the common herbicides. Insecticides applied prior to this study were also found most frequently in fish and bed sediment from urban streams. The highest detection frequencies were for chlordane compounds, dichloro-diphenyl-trichloroethane (DDT) compounds, and dieldrin. Urban streams also had the highest concentrations of total chlordane and dieldrin in both sediment and fish tissue. Chlordane and aldrin were widely used for termite control until the mid-to-late 1980s. Their agricultural uses were restricted during the 1970s.

Gilliom *et al.*{Gilliom, 2006 #563} also presented pesticide detection data on three OPs that were previously assessed by NMFS (2008) and are part of the baseline habitat conditions. Specifically, they are chlorpyrifos and diazinon. Both insecticides were commonly used in agricultural and urban areas from 1992 - 2001 and prior to the sampling period. About 13 million lbs of chlorpyrifos and about 1 million lbs of diazinon were applied for agricultural use during this period. Non-agricultural uses of chlorpyrifos and diazinon totaled about 5 million and 4 million lbs per year in 2001, respectively {Gilliom, 2006 #563}. For both insecticides, concentrations in most urban streams were higher than in most agricultural streams, and were similar to those found in agricultural areas with the greatest intensities of use. Diazinon and chlorpyrifos were detected about 75% and 30% of the time in urban streams, respectively {Gilliom, 2006 #563}.

Another dimension of pesticides and their degradates in the aquatic environment is their simultaneous occurrence as mixtures {Gilliom, 2006 #563}. 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, 2006 #563}. Fish experiencing coincident exposure to multiple pesticides 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. Azinphos methyl, dimethoate, disulfoton, ethoprop, fenamiphos, methamidophos, methidathion, methyl parathion, naled, phosmet, and phorate are all AChE inhibitors. In California, there are 61 pesticides that inhibit AChE approved for use {CDPR, 2007 #990}. According to CDPR, the amount of these chemicals used has decreased {CDPR, 2007 #990}. However, some AChE a.i.s – such as bensulide and naled – are increasing in use {CDPR, 2007 #990}. While the trend indicates decreased reliance on these products, we note that their current use remains significant.

Table 55. Use figures for AChE inhibiting pesticides in California {CDPR, 2007 #990}

	1996	2006
lbs a.i. applied	15,473,843	6,857,530
Acres treated (agriculture use only)	11,720,058	5,729,958

Mixtures of organochlorine pesticide compounds were also common in fish-tissue samples from most streams. About 90% of fish samples collected from urban steams contained two or more pesticide compounds and 33% contained 10 or more pesticides. Similarly, 75% of fish samples from streams draining watersheds with agricultural and mixed land use contained 2 or more pesticide compounds and 10% had 10 or more compounds {Gilliom, 2006 #563}.

NAWQA analysis of all detections indicates that more than 6,000 unique mixtures of 5 pesticides were detected in agricultural streams {Gilliom, 2006 #563}. The number of unique mixtures varied with land use. Mixtures of the most often detected individual pesticides include the herbicides atrazine (and its degradate deethylatrazine), metolachlor, simazine, and prometon. Each herbicide occurred in more than 30% of all mixtures found in agricultural and urban uses in streams. Also present in more than 30% of the mixtures were cyanazine, alachlor, metribuzin, and trifluralin in agricultural streams. Dacthal and the insecticides diazinon, chlorpyrifos, carbaryl, and malathion were also present in urban streams. Carbaryl occurred in at least 50% of urban streams. In 15% of urban streams carbaryl concentration was over 0.1 μg/L {Gilliom, 2006 #563}. Insecticides are typical constituents in environmental mixtures and are commonly found in both agricultural and urban streams.

The numbers of unique mixtures of organochlorine pesticide compounds found in whole-fish tissue samples were greater in urban streams than in streams from agricultural or mixed land use watersheds. About 1,400 unique 5-compound mixtures were found in fish from urban steams compared to fewer than 800 unique 5-compound mixtures detected in fish from agricultural and mixed land use steams. The relative contributions of most organochlorine compounds to mixtures in fish were about the same for urban and agricultural streams.

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, 2006 #563}. 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.

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 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, quantify, and are not regulated. Examples of nonpoint source pollution include, but are not limited to, urban runoff from impervious surfaces, areas of fertilizer and pesticide application, and manure.

According to EPA's database of NPDES permits, about 243 NPDES permits are co-located with listed Pacific salmonids in California. Collectively, the total number of EPA-recorded NPDES permits in Idaho, Oregon, and Washington, that are co-located with listed Pacific salmonids is 1,978. See ESU/DPS Figures x – x for NPDES permits co-located within listed salmonid ESUs/DPSs within the states of California, Idaho, Oregon, and Washington in the *Status of Listed Resources* chapter (pages xxx-xxx).

On November 27, 2006, EPA issued a final rule which exempted pesticides from the NPDES permit process, provided that application was approved under FIFRA. The NPDES permits, then, do not include any point source application of pesticides to waterways in accordance with FIFRA labels. 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. States are expected to review their current NPDES permitting requirements for aquatic pesticide use in light of the Sixth Circuit Decision.

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. These are water quality limited estuaries, lakes, and streams that do not meet state surface water quality standards, and are not expected to improve within the next two years. This process is in accordance with section 303(d) of the CWA. Water bodies listed under 303(d) are those that are considered impaired or threatened by pollution.

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.

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 within two years of the 303(d) listing process.

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 tolerance varies between species and life stages. Optimal temperatures for rearing salmonids range from 10ºC and 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, 1996 #523;McCullough, 1999 #539}. 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, 1997 #688}.

Sublethal temperatures (above 24ºC) could be detrimental to salmon by increasing susceptibility to disease {Colgrove, 1966 #536} or elevating metabolic demand {Brett, 1995 #535}. Substantial research demonstrates that many fish diseases become more virulent at temperatures over 15.6ºC {McCullough, 1999 #539}. 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 (Table 56).

Table 56. Washington State water temperature thresholds for salmonid habitat. These temperatures are representative of limits set by California, Idaho, and Oregon {WSDE, 2006 #781}.

Category	Highest 7-DADMax
Salmon and Trout Spawning	13°C (55.4°F)
Core Summer Salmonid Habitat	16°C (60.8°F)
Salmonid Spawning, Rearing, and Migration	17.5°C (63.5°F)
Salmonid Rearing and Migration Only	17.5°C (63.5°F)

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 57). Because the 303(d) list is limited to the subset of rivers tested, the chart values should be regarded as lower-end estimates.

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 in wastewater discharge, decreased water flow, minimal shading by riparian areas, and climatic variation.

Table 57. Number of kilometers of river, stream and estuaries included in state 303(d) lists due to temperature that are located within each salmonid ESU/DPS. Data was taken from the most recent GIS layers available from state water quality assessments reports*

Species	ESU	California	Oregon	Washington	Idaho	Total
Chinook Salmon	California Coastal	39.3	–	–	–	39.3
	Central Valley Spring - Run	0.0	–	–	–	0.0
	Lower Columbia River	–	56.6	229.8	–	286.4
	Upper Columbia River Spring - Run	–	–	254.6	–	254.6
	Puget Sound	–	–	705.0	–	705.0
	Sacramento River Winter - Run	0.0	–	–	–	0.0
	Snake River Fall - Run	–	610.1	246.6	400.2	1,256.9
	Snake River Spring / Summer - Run	–	809.3	243.2	543.8	1,596.3
	Upper Willamette River	–	2,468.0	–	–	2,468.0
Chum Salmon	Columbia River	–	56.6	225.0	–	281.6
	Hood Canal Summer - Run	–	–	90.1	–	90.1
Coho Salmon	Central California Coast	39.3	–	–	–	39.3
	Lower Columbia River	–	291.9	233.5	–	525.4
	Southern Oregon and Northern California Coast	1,416.2	1,833.0	–	–	3,249.2
	Oregon Coast	–	3,715.8	–	–	3,715.8
Sockeye Salmon	Ozette Lake	–	–	4.8	–	4.8
	Snake River	–	–	–	0.0	0.0
Steelhead	Central California Coast	0.0	–	–	–	0.0
	California Central Valley	0.0	–	–	–	0.0
	Lower Columbia River	–	201.2	169.3	–	370.5
	Middle Columbia River	–	3,518.5	386.2	–	3,904.7
	Northern California	39.3	–	–	–	39.3
	Puget Sound	–	–	704.9	–	704.9
	Snake River	–	990.7	246.6	737.6	1,974.9
	South-Central California Coast	0.0	–	–	–	0.0
	Southern California	0.0	–	–	–	0.0
	Upper Columbia River	–	–	282.3	–	282.3
	Upper Willamette River	–	1,668.0	–	–	1,668.0

*CA 2006, Oregon 2004/2006, Washington 2004, and Idaho 1998. (California EPA TMDL Program 2007b, Oregon Department of Environmental Quality 2007, Washington State Department of Ecology 2005, Idaho Department of Environmental Quality 2001).

Baseline Habitat Condition

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, 2001 #973}. 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, 2001 #858}. While not necessary for pool formation, LWD is associated with around 80% of pools in northern California, Washington, and the Idaho pan-handle {Bilby, 2001 #972}.

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. 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 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 #981}.

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, 2001 #971}. 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. Chum salmon use adjacent floodplain areas for spawning. Soon after their emergence, chum salmon use the riverine system to rapidly reach the estuary where they mature, rear, and migrate to the ocean. Coho salmon use the floodplain landscape extensively for rearing. Estuarine floodplains can provide value to juveniles of all species once they reach the salt water interface.

Once floodplain areas have been disturbed, it can take decades for their recovery {Smith, 2005 #981}. Consequently, most land use practices cause some degree of impairment. Development leads to construction of levees and dikes, which isolate the mainstem 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 #981}.

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.

Geographic Regions

For a more fine scale analysis, we divided the action area into geographic regions: the

Southwest Coast Region (California and the southern parts of the State of Oregon) and the Pacific Northwest Region (Idaho, Oregon, and Washington). The Pacific Northwest Region was further subdivided according to ecoregions or other natural features important to NMFS trust resources. Use of these geographic regions is consistent with previous NMFS consultations conducted at the national level NMFS 2007a. We summarize the principal anthropogenic factors occurring in the environment that influence the current status of listed species within each region. Table 4 provides a breakdown of these regions and includes the USGS subregions and accounting units for each region. It also provides a list of ESUs/DPSs found in each accounting unit, as indicated by Federal Register listing notices.

Table 58. USGS Subregions and accounting units within the Northwest and Southwest Regions, along with ESUs/DPSs present within the area {Seaber, 1987 #978}.

Region	USGS Subregion	Accounting Unit	State	HUC no.	ESU/DPS
Pacific Northwest: Columbia River Basin	Upper Columbia River Basin	—	WA	170200	Upper Columbia Spring-run Chinook; Upper Columbia Steelhead; Middle Columbia Steelhead
	Yakima River Basin	—	WA	170300	Middle Columbia Steelhead
	Lower Snake River Basin	Lower Snake River Basin	ID, OR, WA	170601	Snake River Steelhead; Snake River Spring/Summer-run Chinook; Snake River Fall-run Chinook; Snake River Sockeye
		Salmon River Basin	ID	170602	Snake River Steelhead; Snake River Spring/Summer - Run Chinook; Snake River Fall - Run Chinook; Snake River Sockeye
		Clearwater River Basin	ID, WA	170603	Snake River Steelhead; Snake River Fall - Run Chinook
	Middle Columbia River Basin	Middle Columbia River Basin	OR, WA	170701	Middle Columbia Steelhead; Lower Columbia Chinook; Columbia Chum; Lower Columbia Coho
		John Day River Basin	OR	170702	Middle Columbia Steelhead
		Deschutes River Basin	OR	170703	Middle Columbia Steelhead

Region	USGS Subregion	Accounting Unit	State	HUC no.	ESU/DPS
	Lower Columbia River Basin	—	OR, WA	170800	Lower Columbia Chinook; Columbia Chum; Lower Columbia Steelhead; Lower Columbia Coho
	Willamette River Basin	—	OR	170900	Upper Willamette Chinook; Upper Willamette Steelhead; Lower Columbia Chinook; Lower Columbia Steelhead; Lower Columbia Coho
Pacific Northwest: Coastal Drainages	Oregon-Washington Coastal Basin	Washington Coastal	WA	171001	Ozette Lake Sockeye
		Northern Oregon Coastal	OR	171002	Oregon Coast Coho
		Southern Oregon Coastal	OR	171003	Oregon Coast Coho; Southern Oregon and Northern California Coast Coho
Pacific Northwest: Puget Sound	Puget Sound	—	WA	171100	Puget Sound Chinook; Hood Canal Summer -Run Chum; Puget Sound Steelhead
Southwest Coast	Klamath-Northern California Coastal	Northern California Coastal	CA	180101	Southern Oregon and Northern California Coast Coho; California Coastal Chinook; Northern California Steelhead; Central California Coast Steelhead; Central California Coast Coho
		Klamath River Basin	CA, OR	180102	Southern Oregon and Northern California Coast Coho
	Sacramento River Basin	Lower Sacramento River Basin	CA	180201	Central Valley Spring-run Chinook; California Central Valley Steelhead; Sacramento River Winter-run Chinook
	San Joaquin River Basin	—	CA	180400	California Central Valley Steelhead
	San Francisco Bay	—	CA	180500	Central California Coast Steelhead; Southern Oregon and Northern California Coast Coho; Central California Coast Coho; Sacramento River Winter-run Chinook

Region	USGS Subregion	Accounting Unit	State	HUC no.	ESU/DPS
	Central California Coastal	—	CA	180600	Central California Coast Steelhead; Southern Oregon and Northern California Coast Coho; South-Central California Coast Steelhead; Southern California Steelhead; Central California Coast Coho; Sacramento River Winter-run Chinook
	Southern California Coastal	Ventura-San Gabriel Coastal	CA	180701	Southern California Steelhead
		Laguna-San Diego Coastal	CA	180703	Southern California Steelhead

Southwest Coast Region

The basins in this section occur in the States of California and the southern parts of Oregon. Ten of the 28 species addressed in the Opinion occur in the Southwest Coast Region. They are the California Coastal Chinook (CC) salmon, Central Valley (CV) Spring-run Chinook salmon, Sacramento River winter-run Chinook salmon, Southern Oregon/Northern California Coast (SONCC) coho salmon, Central California Coast (CCC) coho salmon, Northern California (NC) steelhead, Central California Coast (CCC) steelhead, California Central Valley (CCV) steelhead, South-Central California Coast (S-CCC) steelhead, and Southern California (SC) steelhead (Table 4). Table 5 and Table 6 show land area in km² for each ESU/DPS located in the Southwest Coast Region.

Table 59. Area of land use categories within the range Chinook and Coho Salmon ESUs in km² where bolded numbers are totals for each category. Land cover image data were taken from Multi-Resolution Land Characteristics (MRLC) Consortium, a consortium of nine federal agencies (USGS, EPA, USFS, NOAA, NASA, BLM, NPS, NRCS, and USFWS) (National Land Cover Data 2001). Land cover class definitions are available at: http://www.mrlc.gov/nlcd_definitions.php

Land Cover		Chinook Salmon			Coho Salmon	
sub category	code	CA Coastal	Central Valley	Sacramento River	So. Oregon and No. CA	Central CA Coast
Water		**128**	**346**	**12**	**208**	**157**
Open Water	11	128	346	0	197	157
Perennial Snow/Ice	12	0	0	12	11	0
Developed Land		**1,138**	**2,588**	**681**	**1,985**	**991**
Open Space	21	826	1,150	16	1,384	629
Low Intensity	22	137	578	313	225	171
Medium Intensity	23	95	567	0	92	138
High Intensity	24	10	135	313	23	30
Barren Land	31	70	158	40	261	23
Undeveloped Land		**19,079**	**15,169**	**87**	**43,314**	**9,185**
Deciduous Forest	41	850	664	7	1,057	208
Evergreen Forest	42	10,700	3,761	1	28,080	4,752
Mixed Forest	43	1,554	479	51	2,426	922
Shrub/Scrub	52	3,801	3,203	0	8,864	1,620
Herbaceous	71	2,114	6,317	12	2,708	1,646
Woody Wetlands	90	42	191	0	130	25
Emergent Wetlands	95	18	553	18	50	13
Agriculture		**395**	**5,878**	**11**	**1,189**	**239**
Hay/Pasture	81	183	769	11	736	6
Cultivated Crops	82	212	5,110	0	454	233
TOTAL (inc. open water)		20,740	23,982	792	46,697	10,572
TOTAL (w/o open water)		20,612	23,636	792	46,499	10,415

Table 60. Area of Land Use Categories within the Range of Steelhead Trout DPSs (km²). Land cover image data were taken from Multi-Resolution Land Characteristics (MRLC) Consortium, a consortium of nine federal agencies (USGS, EPA, USFS, NOAA, NASA, BLM, NPS, NRCS, and USFWS) (National Land Cover Data 2001). Land cover class definitions are available at: http://www.mrlc.gov/nlcd_definitions.php

Land Cover		Steelhead				
sub category	code	Northern CA	Central CA Coast	CA Central Valley	South-Central CA coast	Southern CA
Water		**106**	**1,406**	**409**	**127**	**86**
Open Water	11	106	1,406	409	127	86
Perennial Snow/Ice	12	0	0	0	0	0
Developed Land		**757**	**3,677**	**3,252**	**1,759**	**1,385**
Open Space	21	610	1,224	1,431	1,019	685
Low Intensity	22	50	876	693	247	364
Medium Intensity	23	32	1,223	744	168	262
High Intensity	24	3	327	181	23	12
Barren Land	31	63	26	202	303	62
Undeveloped Land		**16,117**	**11,041**	**19,216**	**14,959**	**7,689**
Deciduous Forest	41	763	179	751	1	0
Evergreen Forest	42	9,790	2,506	3,990	1,721	835
Mixed Forest	43	1,159	2,086	598	1,925	897
Shrub/Scrub	52	2,878	2,253	3,745	4,952	4,370
Herbaceous	71	1,478	3,588	9,435	6,194	1,516
Woody Wetlands	90	32	36	248	93	35
Emergent Wetlands	95	17	392	450	73	35
Agriculture		**193**	**522**	**10,724**	**1,500**	**794**
Hay/Pasture	81	179	36	1,671	203	141
Cultivated Crops	82	14	486	9,054	1,297	653
TOTAL (inc. open water)		17,173	16,645	33,601	18,345	9,954
TOTAL (w/o open water)		17,067	15,240	33,193	18,218	9,868

Select watersheds described herein characterize the past, present, and future human activities and their impacts on the area. The Southwest Coast region encompasses all Pacific Coast rivers south of Cape Blanco, Oregon through southern California. NMFS has identified the Cape Blanco area as an ESU biogeographic boundary for Chinook and coho salmon, and steelhead based on strong genetic, life history, ecological and habitat differences north and south of this landmark. Major rivers contained in this grouping of watersheds are the Sacramento, San Joaquin, Salinas, Klamath, Russian, Santa Ana, and Santa Margarita Rivers (Table 61).

Table 61. Select rivers in the southwest coast region {Carter, 2005 #457}.

Watershed	Approx Length (mi)	Basin Size (mi^2)	Physiographic Provinces*	Mean Annual Precipitation (in)	Mean Discharge (cfs)	No. Fish Species (native)	No. Endangered Species
Rogue River	211	5,154	CS, PB	38	10,065	23 (14)	11
Klamath River	287	15,679	PB, B/R, CS	33	17,693	48 (30)	41
Eel River	200	3,651	PB	52	7,416	25 (15)	12
Russian River	110	1,439	PB	41	2,331	41 (20)	43
Sacramento River	400	27,850	PB, CS, B/R	35	23,202	69 (29)	>50 T & E spp.
San Joaquin River	348	83,409	PB, CS	49	4,662	63	>50 T & E spp.
Salinas River	179	4,241	PB	14	448	36 (16)	42 T & E spp.
Santa Ana River	110	2,438	PB	13	60	45 (9)	54
Santa Margarita River	27	1,896	LC, PB	49.5	42	17 (6)	52

* Physiographic Provinces: PB = Pacific Border, CS = Cascades-Sierra Nevada Range, B/R = Basin & Range.

Land Use

Forest and vacant land are the dominant land uses in the northern basins of the Southwest Coast Region. Grass, shrubland, and urban uses are the dominant land uses in the southern basins (Table 8). Overall, the most developed watersheds are the Santa Ana, Russian, and Santa Margarita rivers. The Santa Ana watershed encompasses portions of San Bernardino, Los Angeles, Riverside, and Orange counties. About 50% of the coastal subbasin in the Santa Ana watershed is dominated by urban land uses and the population density is about 1,500 people per square mile. When steep and undevelopable lands are excluded from this area, the population density in the watershed is about 3,000 people per square mile. However, the most densely populated portion of the basin is near the City of Santa Ana. Here, the population density reaches 20,000 people per square mile {Belitz, 2004 #456;Burton, 1998 #455}. The basin is home to nearly 5 million people. However, this population is projected to increase two-fold in the next 50 years {Belitz, 2004 #456;Burton, 1998 #455}.

Table 62. Land uses and population density in several southwest coast watersheds {Carter, 2005 #457}.

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

Rogue River	6	83	<1	9 grass & shrub	32
Klamath River	6	66	<1	24 grass, shrub, wetland	5
Eel River	2	65	<1	31 grass & shrub	9
Russian River	14	50	3	31 (23 grassland)	162
Sacramento River	15	49	2	30 grass & shrub	61
San Joaquin River	30	27	2	36 grass & shrub	76
Salinas River	13	17	1	65 (49 grassland)	26
Santa Ana River	11	57	32	---	865
Santa Margarita River	12	11	3	71 grass & shrub	135

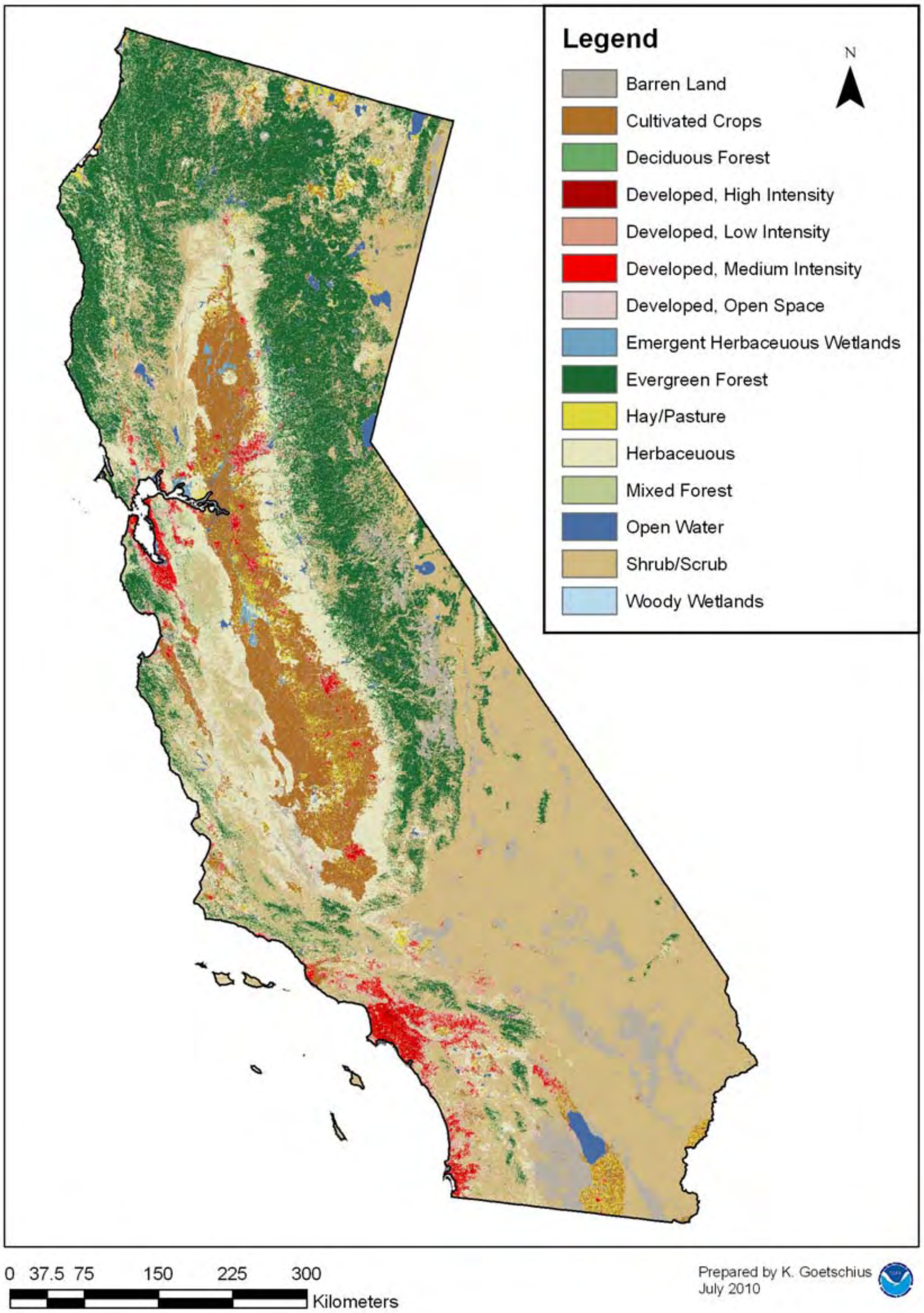


Figure 33. Landuse in Southwest Region

As a watershed becomes urbanized, population increases and changes occur in stream habitat, water chemistry, and the biota (plants and animals) that live there. The most obvious effect of urbanization is the loss of natural vegetation which results in an increase in impervious cover and dramatic changes to the natural hydrology of urban streams. Urbanization generally results in land clearing, soil compaction, modification and/or loss of riparian buffers, and modifications to natural drainage features {Richter, 2002 #514}. The increased impervious cover in urban areas leads to increased volumes of runoff, increased peak flows and flow duration, and greater stream velocity during storm events.

Runoff from urban areas also contains all the chemical pollutants from automobile traffic and roads as well as those from industrial sources and residential use. Urban runoff is also typically warmer than receiving waters and can significantly increase temperatures in small urban streams. Warm stream water is detrimental to native aquatic life resident fish and the rearing and spawning needs of anadromous fish. Wastewater treatment plants (WWTP) replace septic systems, resulting in point discharges of nutrients and other contaminants not removed in the processing. Additionally, some cities have combined sewer/stormwater overflows and older systems may discharge untreated sewage following heavy rainstorms. WWTP outfalls often discharge directly into the rivers containing salmonids. These urban nonpoint and point source discharges affect the water quality and quantity in basin surface waters.

In many basins, agriculture is the major water user and the major source of water pollution to surface waters. During general agricultural operations, pesticides are applied on a variety of crops for pest control. These pesticides may contaminate surface water via runoff especially after rain events following application. Usage of some of the 12 a.i.s in California are reported below. Usage information from California may or may not apply for states in the Pacific Northwest Region. Pesticide use data for the southern parts of Oregon are unreported as Oregon lacks a database with comparable information as California. Pesticide detection data for these same a.i.s are reported in the Targeted Monitoring subsection of the *Effects of the Proposed Action* chapter.

Agricultural Pesticide Usage for some of the 12 a.i.s in the Southwest Coast Region

The State of California tracks pesticide usage closely, unlike Idaho, Oregon, and Washington. EPA used California's pesticide data in its 2008 and 2009 Red-legged frog (*Rana aurora draytonii*) evaluations and these data were from 2002 - 2005. Other cited sources of pesticide usage data are from EPA's BEs and Science Chapter documents for these 12 a.i.s.

<u>Azinphos methyl</u>. Azinphos methyl is used in crop operations in the Central Valley of California. Crops with highest usage of azinphos methyl during the mid-1990s were apples, cotton, almonds, sugarcane, and alfalfa. However, use of azinphos methyl in California has declined from nearly 475,000 lbs a.i. on 325,000 treated acres in 1993 to about 160,000 lbs a.i. on 117,484 treated acres in 2001 {EPA, 2003 #1676}. About 55% of the amount of azinphos methyl applied in 2001 was to almonds, with 17% applied to apples, 11% to pears, and 7% to walnuts. Between 2002 and 2005, azinphos methyl was reportedly used in 37 counties in California. The principal use was on orchard and vineyard crops. Non-orchard uses also occurred although most of these non-orchard/vineyard applications were limited to three or fewer counties. Additional non-agricultural applications were reported as landscape maintenance, greenhouse flowers, and structural pest control. The greatest average usage (average of lbs applied per commodity across all four years) was to almonds in Kern county at 24,784 lbs. Recent data show that the greatest use of azinphos methyl in California is on almonds at an annual average of 48,000 lbs, followed by pistachios at 29,000 lbs, apples at 18,000 lbs, pears at 11,000 lbs, and walnuts at 7,000 lbs. All remaining crops had less than 1,000 lbs applied annually, and one use (nursery stock) had reported application of 2 lbs in Santa Clara county {EPA, 2007 #1688}.

<u>Bensulide.</u> According to EPA {EPA, 2007 #1689} up to 6 lbs of bensulide/acre are

applied on a variety of crops for a maximum of 12 lbs a.i./acre/year. Crops include arugula, artichoke, beet, bok choy, broccoli, Brussels sprouts, cabbage, canola, cantaloupe, cardoon, cauliflower, celery, chicory, Chinese cabbage, Chinese greens, cilantro, collard, corn, cotton, cucumber, dandelion green, eggplant, endive, fennel, gai choy, gai lon, grape herbs and spices, kale, kohlrabi, landscape maintenance, lettuce melon, mizuna, mustard, ornamentals onion, parsley, peas, fruiting and spice peppers, pumpkin, radish, rapini, rights-of-way, spinach, squash, summer squash, swiss chard, turf/sod, watermelon, winter squash, and zucchini squash. The highest application rate is ground application of 32 lbs a.i./acre on golf course turf. Of the 188,854 lbs reported to have been used in California in 2001, 66,339 lbs (35%) was within the range of one or more salmon and steelhead ESUs {EPA, 2002 #1677}. A large portion of salmon habitat occurs in coastal counties south of San Francisco. Approximately half of the 1999 - 2001 bensulide lbs used in California was in Imperial County. The inland southern counties from Fresno county south accounted for 144,804 lbs in 2000 and 122,515 lbs in 2001, or 67% and 65%, respectively of the total California use. Most of the bensulide use in the remainder of California was in coastal counties, with Monterey county having the highest amount, 42,106 lbs in 2000 and 37,402 lbs in 2001. Ventura, Santa Barbara, and San Benito counties all had more than 1,000 lbs used in 2000 and 2001, while San Luis Obispo and Santa Clara counties had more than 1,000 lbs used in 2001. Stanislaus County was the only Central Valley county with more than 1,000 lbs of use, and this occurred in 2000 and 2001.

<u>Dimethoate.</u> According to EPA {EPA, 2008 #1690}, dimethoate is nationally registered for over 40 uses in agriculture and ornamental production. Use data from 2001 - 2005 for California indicate that dimethoate is applied throughout the year, with the majority of applications occurring during the summer months (June – August). From 2001 - 2005, the percentage of total dimethoate use in California was highest on alfalfa (19.7% of total use), tomato (13.5%), beans (11.3%), broccoli (10.6%), corn (9.3%), citrus (8.4%), lettuce (7.5%), and cotton (7.1%) {CDPR, 2007 #990}. Dimethoate use on non-cropland areas adjacent to vineyards is permitted under a SLN and is applied only to Napa, Sonoma, Mendocino, and Lake counties in northern California.

Disulfoton. Disulfoton is used on a variety of terrestrial food and nonfood crops and terrestrial feed crops. The National Pesticide Use Database (National Center for Agriculture Policy (NCFAP) 2001) indicates that for major crops in California and the Pacific Northwest, total application of disulfoton in 1992 (census report) was 821,337 lbs a.i./year. In the 1997 data, 560,367 lbs a.i. of disulfoton (IRED QUA; attachment 2) were applied to the same crops. The greatest decline was observed in total wheat application, which fell from 498,288 to 188,498 lbs a.i./year.

Ethoprop. Ethoprop may be applied to a variety of crops. There are no registered homeowner uses {EPA, 2006 #1703}. The National Quantitative Use Database (NCFAP) indicates total ethoprop use in California and the Pacific Northwest on the major crops of potatoes, corn, and sweet potatoes was 569,203 lbs a.i. in 1992 and 470,831 lbs a.i. in 1997. For most commodities, ethoprop use increased during 1992 - 1997. However, use on potatoes and sweet potatoes declined significantly in some states. The CDPR reports a steady decline in lbs of ethoprop use in California from 1993 (62,143 lbs a.i./acre) to 2002 (16,531 lbs a.i./acre) {EPA, 2003 #1680}.

Fenamiphos. Fenamiphos was used on a variety of crops and nonagricultural uses such as golf courses and turf farms {EPA, 1999 #1921}. Based on data from 1996 to 1999, California was the major user of fenamiphos for agricultural uses (30% of the total national use of fenamiphos) and it was the fourth ranking state for the total use of fenamiphos in turf and ornamentals (2% of the total national use) {EPA, 2003 #1681}. According to CDPR trend data, use of fenamiphos decreased steadily in California from 232,510 lbs in 1993 to 70,939 lbs in 2002. The cumulative acres treated with fenamiphos also decreased from 142,069 acres in 1993 to 38,297 acres in 2002 {EPA, 2003 #1681}. Current data on fenamiphos use in California are not readily available and therefore unreported.

Methamidophos. Methamidophos is used on cotton, potatoes, tomatoes, and alfalfa grown for seed. EPA estimates that 640,000 lbs of a.i. are applied nationally on an annual basis {EPA, 2004 #1682}. Crops with a high percentage of acreage treated are fresh tomatoes (46%) and potatoes (21%). The trend shows increasing cotton acreage treated by methamidophos from a current treated acreage of 1% (BEAD usage data up to 1996) to a projected usage of 10% (registrant-provided information 1997). Specific use information for methamidophos in California is limited, not readily available, and therefore unreported.

Methidathion. Methidathion use in California occurs on citrus, all other orchard crops, and cotton {EPA, 2004 #1683}. Reported use of methidathion in California declined from 1993 (451,890 lbs a.i.) to 2002 (67,455 lbs a.i.) {EPA, 2004 #1683}. Based on national use data compiled by EPA from 2001 - 2006, on average, about 110,000 lbs of methidathion are applied annually to agricultural crops, 95% of it in California. These data show that usage is highest on almonds and oranges, with annual average applications to each of 20,000 lbs a.i. The highest average percent crop treated with methidathion is artichokes (60%). EPA's summary of CDPR data from 1999 – 2006, report that an average of 82,301 lbs of methidathion were applied in California to an average of 52,823 acres per year. Use was at a maximum of 177,105 lbs in 1999 and then dropped by nearly half the following year to 98,129 lbs. The total amount of methidathion used in California in 2002 was 67,833 lbs on 37,644 acres {EPA, 2004 #1683}. Methidathion use remained relatively stable between 2003 and 2006 with average applications ranging from around 50,000 lbs/year to 60,000 lbs/year. From 1999 - 2006, methidathion was used in a total of 34 counties involving 41 different uses. Four counties accounted for 70% of the total lbs applied on average per county [Kern (25%), Tulare (20%), Monterey (14%), Fresno (11%)]. Fruit orchards accounted for about 30% of the total lbs applied per year in California on average. Other major crops include almonds (23%), oranges (17%), and artichokes (14%) {EPA, 2009 #1694}.

Methyl parathion. Methyl parathion use in California occur on walnut orchards, agricultural crops, and grass used as hay, pasture, or forage. There are no residential uses {EPA, 2008 #1695}. Based on data reported in the CDPR database, use of methyl parathion in California has increased during 2002 - 2005. During this time frame total use of 292,000 lbs of methyl parathion was reported. Use of about 56,000 lbs a.i. was reported in 2002. Total lbs applied ranged from about 75,000 – 83,000 lbs a.i. in 2003 – 2005. In all years, the dominant use was on walnuts (94% of all applied from 2002-2005). The only other uses accounted for greater than or equal to 1% were reported for corn (5%) and onions (1%). Of the total applied, 75% was used in only four counties (Tulare, San Joaquin, Stanislaus, and Kings) with seven other counties accounting for an additional 24% of reported use {EPA, 2008 #1695}.

Naled. The dominant naled uses in California are on cotton, broccoli, flying insect control, strawberry, and sugar beet. Use of naled in California peaked in 1995 and 1997 and has been relatively decreasing since then. In 2002 about 200,000 lbs a.i. were used {EPA, 2004 #1685}. The highest reported uses of naled in California from 2002 - 2005 were cotton (representing 38% of the total applied), broccoli (about 12%), public health (about 11%), strawberry (10%), and sugar beet (6%). All other uses individually comprised less than 5% of total naled applied. During this same three-year period, at least 40 counties in California reported naled use {EPA, 2008 #1696}. In October 2003, CDPR's list of the top 100 pesticides used in agriculture in California ranked naled at 73 based on the use of 201,504 lbs a.i. applied to 154,963 acres statewide. According to 2002 - 2005 California Pesticide Use and Reporting System (PURS) data, about 743,280 lbs of naled were applied in California during that period. Fresno and King counties had the highest naled use for this same period (255,250 lbs and 106,305 lbs, respectively), followed by Monterey county (88,629 lbs), and Sutter county (43,010). Lake, El Dorado, Tehama, and San Francisco counties reported the lowest amounts used, with the latter two counties reporting zero lbs used from 2002 - 2005.

Phorate. Phorate use on agricultural crops in California include wheat, sugar beets, sorghum, potato, peanut, cotton, corn, sweet corn, and beans, and for non-agricultural use on ornamentals. Fresno county has the highest average annual lbs of phorate applied (11,000 lbs) for all uses, followed by Riverside (about 10,000 lbs), San Joaquin, and Tulare counties. Alameda and Orange counties show negligible use. No use was reported for Mendocino county. However, if average annual application rates by county for all uses are compared, Del Norte county has the highest rate at 7 lb a.i./acre followed by Santa Barbara (2.4 lb a.i./acre) and San Luis Obispo (2.3 lb a.i./acre). Uses with the highest average annual amounts applied in California are cotton (27,264 lbs), potato (12,431 lbs), corn (11,290 lbs), sweet corn (5,746 lbs), sugar beet (6,613 lbs), and ornamentals (3,070 lbs). All other uses combined total 535 lbs per year on average {EPA, 2008 #1697}.

Phosmet. Phosmet is primarily applied by commercial applicators {EPA, 2008 #1698} onto grapes, pears, and apples along the California coast, to grapes, alfalfa, pears, apple stone fruit, and nuts in the California Central Valley, and to grapes and alfalfa in the California desert. According to EPA's 1999 Quantitative Use Assessment (QUA) for phosmet, an average of 1 million lbs of a.i. was applied to about 402,000 acres of crop annually from 1988 through 1997. Most use occurred in California. USGS data from the mid-1990s reported that the highest use of phosmet was applied to apples (506,000 lbs a.i.), pears (96,000 lbs a.i.), alfalfa hay (87,000 lbs a.i.), and peaches (81,000 lbs a.i.). In 2001, about 20% of phosmet was applied to almonds, and 10% or more was applied to peaches, apples, walnuts, and nectarines {EPA, 2003 #1687}. CDPR data from 2002 - 2005 reported that the highest average annual usage of phosmet in California include almond (11,612 lbs) and pistachio (8,893 lbs). These use sites are followed by nectarine (2,939 lbs), peach (2,654), and plum (1,800 lbs).

As part of the baseline pesticide conditions in the Southwest Coast region, pesticide reduction programs also exist in California to minimize levels of the above a.i.s into the aquatic environment. They are described below:

Pesticide Reduction Programs in the Southwest Coast Region

Monitoring of water resources is handled by the California Environmental Protection Agency's Regional Water Boards. Each Regional Board makes water quality decisions for its region including setting standards and determining waste discharge requirements. The Central Valley Regional Water Quality Control Board (CVRWQCB) addresses issues in the Sacramento and San Joaquin River Basins. These river basins are characterized by crop land, specifically orchards, which historically rely heavily on organophosphates for pest control.

In 2003, the CVRWQCB adopted the Irrigated Lands Waiver Program (ILWP). Participation was required for all growers with irrigated lands that discharge waste which may degrade water quality. However, the ILWP allowed growers to select one of three methods for regulatory coverage {Markle, 2005 #1501}. These options included: 1) join a Coalition Group approved by the CVRWQCB, 2) file for an Individual Discharger Conditional Waiver, and 3) comply with zero discharge regulation {Markle, 2005 #792}. Many growers opted to join a Coalition as the other options were more costly. Coalition Groups were charged with completing two reports – a Watershed Evaluation Report and a Monitoring and Reporting Plan. The Watershed Evaluation Report included information on crop patterns and pesticide/nutrient use, as well as mitigation measures that would prevent orchard runoff from impairing water quality. Similar programs are in development in other agricultural areas of California.

As a part of the Waiver program, the Central Valley Coalitions undertook monitoring of "agriculture dominated waterways". Some of the monitored waterways are small agricultural streams and sloughs that carry farm drainage to larger waterways. The coalition was also required to develop a management plan to address exceedance of State water quality standards. Currently, the Coalitions monitor toxicity to test organisms, stream parameters (*e.g.*, flow, temperature, etc.), nutrient levels, and pesticides used in the region, including diazinon and chlorpyrifos. Sampling diazinon exceedances within the Sacramento and Feather Rivers resulted in the development of a TMDL. The Coalitions were charged with developing and implementing management and monitoring plans to address the TMDL and reduce diazinon runoff.

The Coalition for Urban/Rural Environmental Stewardship (CURES) is a non-profit organization that was founded in 1997 to support educational efforts for agricultural and urban communities focusing on the proper and judicious use of pest control products. CURES educates growers on methods to decrease diazinon surface water contamination in the Sacramento River Basin. The organization has developed best-practice literature for pesticide use in both urban and agricultural settings (www.curesworks.org). CURES also works with California’s Watershed Coalitions to standardize their Watershed Evaluation Reports and to keep the Coalitions informed. The organization has worked with local organizations, such as the California Dried Plum Board and the Almond Board of California, to address concerns about diazinon, pyrethroids, and sulfur. The CURES site discusses alternatives to organophosphate dormant spray applications. It lists pyrethroids and carbaryl as alternatives, but cautions that these compounds may impact non-target organisms. The CURES literature does not specifically address the 12 a.i.s discussed in this Opinion.

California also has PURS legislation whereby all agricultural uses of registered pesticides must be reported. In this case “agricultural” use includes applications to parks, golf courses, and most livestock uses.

In 2006, CDPR put limitations on dormant spay application of most insecticides in orchards, in part to adequately protect aquatic life in the Central Valley region. While the legislation was prompted by diazinon and chlorpyrifos exceedences, these limitations also apply to other organophosphates, pyrethroids, and carbamates.

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 that apply to azinphos methyl, disulfoton, ethoprop, methamidophos, methidathion, methyl parathion, naled, phorate, and phosmet use in salmonid habitat are:

- Do not use in currently occupied habitat.
- Provide a 20 ft minimum strip of vegetation (on which pesticides should not be applied) along rivers, creeks, streams, wetlands, vernal pools and stock ponds, or on the downhill side of fields where runoff could occur. Prepare land around fields to contain runoff by proper leveling, etc. Contain as much water "on-site" as possible. The planting of legumes, or other cover crops for

several rows adjacent to off-target water sites is recommended. Mix pesticides in areas not prone to runoff such as concrete mixing/loading pads, disked soil in flat terrain or graveled mix pads, or use a suitable method to contain spills and/or rinsate. Properly empty and triple-rinse pesticide containers at time of use. This measure does not apply to methamidophos.

- Conduct irrigations efficiently to prevent excessive loss of irrigation waters through runoff. Schedule irrigations and pesticide applications to maximize the interval of time between the pesticide application and the first subsequent irrigation. Allow at least 24 hours between applications of pesticides listed in this bulletin and any irrigation that results in surface runoff into natural waters. Time applications to allow sprays to dry prior to rain or sprinkler irrigations. Do not make aerial applications while irrigation water is on the field unless surface runoff is contained for 72 hours following the application. This measure applies to bensulide but does not apply to methamidophos.

- 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 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. This measure also applies to bensulide.

Dimethoate and fenamiphos are not listed in the bulletins. In addition to the two limitations noted above, the following apply to bensulide use:

- Do not use in currently occupied habitat except: (1) as specified in Habitat Descriptors, (2) in organized habitat recovery programs, or (3) for selective control of invasive exotic plants.
- Do not apply within 30 yards upslope of habitat unless a suitable method is used to contain or divert runoff waters.

In addition to pesticide usage for agriculture, this land use further impacts salmonid aquatic habitats through water diversions or withdrawals from rivers and tributaries. Associated impacts from water diversion in the Southwest Coast region are described below.

Water Diversions for Agriculture in the Southwest Coast Region

In 1990, nearly 95% of the water diverted from the San Joaquin River was diverted for agriculture. Additionally, 1.5% of the water was diverted for livestock {Carter, 2005 #457}. The amount and extent of water withdrawals or diversions for agriculture impact streams and their inhabitants via reduced water flow/velocity and dissolved oxygen levels. For example, adequate water flow is required for migrating salmon along freshwater, estuarine, and marine environments in order to complete their life cycle. Low flow events may delay salmonid migration or lengthen fish presence in a particular water body until favorable flow conditions permit fish migration along the migratory corridor or into the open ocean.

Water diversions may also increase nutrient load, sediments (from bank erosion), and temperature. Flow management and climate changes have decreased the delivery of suspended particulate matter and fine sediment to the estuary. The conditions of the habitat (shade, woody debris, overhanging vegetation) whereby salmonids are constrained by low flows also may make them more or less vulnerable to predation, elevated temperatures, crowding, and disease. Water flow effects on salmonids may seriously impact adult migration and water quality conditions for spawning and rearing salmonids. High temperature may also result from the loss of vegetation along streams that used to shade the water and from new land uses (buildings and pavement) whereby rainfall picks up heat before it enters into an adjacent stream. Runoff inputs from multiple land use may further pollute receiving waters inhabited by fish or along fish migratory corridors.

Currently, California has over 500 water bodies on its 303(d) list {Wu, 2000 #546}. The 2006 list includes 779 stream segments, rivers, lakes, and estuaries and 12 pollutant categories {CEPA, 2007 #780}. Pollutants represented on the list include pesticides, metals, sediments, nutrients or low dissolved oxygen, temperature, bacteria and pathogens, and trash or debris. There are 2,237 water body/pollutant listings; a water body is listed separately for each pollutant detected {CEPA, 2007 #780}. The 2006 303(d) list identifies water bodies listed due to the presence of specific pollutants, including carbofuran and elevated temperature (Table 63). See species ESU/DPS maps for NPDES permits and 303(d) waters co-located within listed salmonid ESUs/DPSs in California.

Table 63. California's 2006 Section 303(d) List of Water Quality Limited Segments: segments listed for exceeding temperature and azinphos methyl and methyl parathion limits {CEPA, 2007 #779}.

Pollutant	Estuary Acres Affected	River / Stream Miles Affected	# Water Bodies
Temperature	-	16,907.2	41
Azinphos methyl	-	61	3
Methyl parathion	-	49	1

Estuary systems of the region are consistently exposed to anthropogenic pressures stemming from high human density sources. For example, the largest west coast estuary is the San Francisco Estuary. This water body provides drinking water to 23 million people, irrigates 4.5 million acres of farmland, and drains roughly 40% of California’s land area. As a result of high use, many environmental measures of the San Francisco Estuary are poor. Water quality suffers from high phosphorus and nitrogen loads, primarily from agricultural, sewage, and storm water runoff. Water clarity is also compromised. Sediments from urban runoff and historical activities contain high levels of contaminants. They include pesticides, polychlorinated biphenyls (PCBs), nickel, selenium, cadmium, mercury, copper, and silver. Specific pesticides include pyrethroids, malathion, carbaryl, and diazinon. Other pollutants include DDT and polynuclear aromatic hydrocarbons (PAHs).

Other wastes are also discharged into San Francisco Bay. Approximately 150 industries discharge wastewater into the bay. Discharge of hot water from power plants and industrial sources may elevate temperatures and negatively affect aquatic life. Additionally, about 60 sewage treatment plants discharge treated effluent into the bay and elevate nutrient loads. However, since 1993, many of the point sources of pollution have been greatly reduced. Pollution from oil spills also occur due to refineries in the bay area. Gold mining has also reduced estuary depths in much of the region, causing drastic changes to habitat. As these stressors persist in the marine environment, the estuary system will likely carry loads for future years, even with strict regulation.

Large urban centers are foci for contaminants. Contaminant levels in surface waters near San Francisco, Oakland, and San Jose are highest. These areas are also where water clarity is at its worst. Some of the most persistent contaminants (PCBs, dioxins, DDT, etc.) are bioaccumulated by aquatic biota and can biomagnify in the food chain. Fish tissues contain high levels of PCB and mercury. Concentrations of PCB were 10 times above human health guidelines for consumption. Birds, some of which are endangered (clapper rail and least tern), have also concentrated these toxins.

As mentioned earlier in this chapter, the distribution of the most prevalent pesticides in streams and ground water correlate with land use patterns and associated past or present pesticide use. The USGS conducted NAWQA analyses for three basins within the Southwest Coast Region. The NAWQA data reported some of the 12 a.i.s evaluated in this Opinion including the OPs and carbamates that NMFS previously evaluated in its 2008 and 2009 Opinions. NAWQA data for these basins are summarized below:

Santa Ana Basin: NAWQA Analysis

The Santa Ana watershed is the most heavily populated study site out of more than 50 assessment sites studied across the nation by the NAWQA Program. According to Belitz *et al.* {, 2004 #399}, treated wastewater effluent is the primary source of baseflow to the Santa Ana River. Secondary sources that influence peak river flows include stormwater runoff from urban, agricultural, and undeveloped lands {Belitz, 2004 #456}. Stormwater and agricultural runoff frequently contain pesticides, fertilizers, sediments, nutrients, pathogenic bacteria, and other chemical pollutants to waterways and degrade water quality. The above inputs have resulted in elevated concentrations of nitrates and pesticides in surface waters of the basin. Nitrates and pesticides were more frequently detected here than in other national NAWQA sites {Belitz, 2004 #456}.

Additionally, Belitz *et al.* {, 2004 #399} found that pesticides and volatile organic compounds (VOCs) were frequently detected in surface and ground water in the Santa Ana Basin. Of the 103 pesticides and degradates routinely analyzed for in surface and ground water, 58 were detected. Pesticides included diuron, diazinon, carbaryl, chlorpyrifos, lindane, malathion, and chlorothalonil. Carbaryl was detected in 42% of urban samples, though it generally did not exceed the standard for protection of aquatic life (Belitz *et al.* 2004). Carbofuran was also detected, but did not exceed any water quality standards. Azinphos methyl, disulfoton, ethoprop, methomyl, and phorate were tested for but not detected. Of the 85 VOCs routinely analyzed for, 49 were detected. VOCs included methyl *tert*-butyl ether (MTBE), chloroform, and trichloroethylene (TCE). Organochlorine compounds were also detected in bed sediment and fish tissue. Organochlorine concentrations were also higher at urban sites than at undeveloped sites in the Santa Ana Basin. Organochlorine compounds include DDT and its breakdown product diphenyl dicloroethylene (DDE), and chlordane. Other contaminants detected at high levels included trace elements such as lead, zinc, and arsenic. According to Belitz *et al.* {, 2004 #399}, the biological community in the basin is heavily altered as a result from these pollutants.

San Joaquin-Tulare Basin: NAWQA Analysis

A study was conducted by the USGS in the mid-1990s on water quality within the San Joaquin-Tulare basins. Concentrations of dissolved pesticides in this study unit were among the highest of all NAWQA sites nationwide. The USGS detected 49 of the 83 pesticides it tested for in the mainstem and three subbasins. Pesticides were detected in all but one of the 143 samples. The most common detections were of the herbicides simazine, dacthal, metolachlor, and EPTC (Eptam), and the insecticides diazinon and chlorpyrifos. Twenty-two pesticides were detected in 20% of the samples {Dubrovsky, 1998 #459}. Carbaryl and methomyl were detected in all three subbasins, despite land use differences. Carbaryl was detected in roughly 20% of samples from each subbasin, while methomyl detections ranged from 5% to 25%. Further, most samples contained mixtures of between 7 and 22 pesticides.

Many pesticides had the maximum concentration of all 20 study units and all sites exceeded the aquatic life criteria for at least 1 pesticide at least 17% of the time. Azinphos methyl, disulfoton, ethoprop, and methyl parathion were detected. Although phorate was tested for, it was not detected. Criteria for the protection of aquatic life were exceeded in 37% of samples of streams {Dubrovsky, 1998 #459}. Only seven pesticides exceeded this criteria: diuron, trifluralin, azinphos methyl, carbaryl, chlorpyrifos, diazinon, and malathion. Forty percent of these exceedances were attributed solely to diazinon. However, criteria do not exist yet for over half of the detected compounds (Dubrovsky et al.1998).

Azinphos methyl is used on a number of orchard crops in this NAWQA unit. Based on 1992 National Agricultural Statistics Service (NASS) data, this unit had the highest azinphos methyl usage among the 20 NAWQA units initiated in 1991 and the second among all 60 NAWQA study units. Of 40 different sites sampled in the San Joaquin-Tulare Basin, nine had at least one detect or 22.5%. The maximum level of azinphos methyl detected in any sample from 1993 - 1997 was from a site in the San-Joaquin-Tulare Basin.

Organochlorine insecticides in bed sediment and tissues of fish or clams were also detected. They include DDT and toxaphene. Levels at some sites were among the highest in the nation. Concentrations of trace elements in bed sediment generally were higher than concentrations found in other NAWQA study units {Dubrovsky, 1998 #459}.

Sacramento River Basin: NAWQA Analysis

Another study conducted by the USGS from 1996 - 1998 within the Sacramento River Basin detected up to 24 out of 47 pesticides in surface waters {Domagalski, 2000 #515}. Pesticides included thiobencarb, carbofuran, molinate, simazine, metolachlor, dacthal, chlorpyrifos, carbaryl, and diazinon. Land use differences between sites are reflected in pesticide detections. Carbofuran was detected in 100% of samples from the agricultural site, but only 6.7% of urban samples (Domagalski 2000). Carbaryl, however, was detected in 100% of urban samples and 42.9% of agricultural samples. Some pesticides were detected at concentrations higher than criteria for the protection of aquatic life in the smaller streams, but were diluted to safer levels in the mainstem river. Intensive agricultural activities also impact water chemistry. In the Salinas River and in areas with intense agriculture use, water hardness, alkalinity, nutrients, and conductivity are also high.

Other Land Uses in the Southwest Coast Region

Habitat Modification

The Central Valley area, including San Francisco Bay and the Sacramento and San Joaquin River Basins, has been drastically changed by development. Salmonid habitat has been reduced to 300 miles from historic estimates of 6,000 miles {CDFG, 1993 #974}. In the San Joaquin Basin alone, the historic floodplain covered 1.5 million acres with 2 million acres of riparian vegetation {CDFG, 1993 #974}. Roughly 5% of the Sacramento River Basin's riparian forests remain. Impacts of development include loss of LWD, increased bank erosion and bed scour, changes in sediment loadings, elevated stream temperature, and decreased base flow. Thus, lower quantity and quality of LWD and modified hydrology reduce and degrade salmonid rearing habitat.

The Klamath Basin in Northern California has been heavily modified as well. Water diversions have reduced spring flows to 10% of historical rates in the Shasta River, and dams block access to 22% of historical salmonid habitat. The Scott and Trinity Rivers have similar histories. Agricultural development has reduced riparian cover and diverted water for irrigation {NRC, 2003 #976}. Riparian habitat has decreased due to extensive logging and grazing. Dams and water diversions are also common. These physical changes resulted in water temperatures too high to sustain salmonid populations. The Salmon River, however, is comparatively pristine; some reaches are designated as Wild and Scenic Rivers. The main cause of riparian loss in the Salmon River basin is likely wild fires – the effects of which have been exacerbated by salvage logging {NRC, 2003 #976}.

Mining

Famous for the gold rush of the mid-1800s, California has a long history of mining. Extraction methods such as suction dredging, hydraulic mining, and strip mining may cause water pollution problems. In 2004, California ranked top in the nation for non-fuel mineral production with 8.23% of total production {NMA, 2007 #464}. Today, gold, silver, and iron ore comprise only 1% of the production value. Primary minerals include construction sand, gravel, cement, boron, and crushed stone. California is the only state to produce boron, rare-earth metals, and asbestos {NMA, 2007 #464}.

California contains approximately 1,500 abandoned mines. Roughly 1% of these mines are suspected of discharging metal-rich waters into the basins. The Iron Metal Mine in the Sacramento Basin releases more than 1,100 lbs of copper and more than 770 lbs of zinc to the Keswick Reservoir below Shasta Dam. The Iron Metal Mine also released elevated levels of lead {Cain et al. 2000 in \Carter, 2005 #457}. Metal contamination reduces the biological productivity within a basin. Metal contamination can result in fish kills at high levels or sublethal effects at low levels. Sublethal effects include a reduction in feeding, overall activity levels, and growth. The Sacramento Basin and the San Francisco Bay watershed are two of the most heavily impacted basins within the state from mining activities. The basin drains some of the most productive mineral deposits in the region. Methyl mercury contamination within San Francisco Bay, the result of 19th century mining practices using mercury to amalgamate gold in the Sierra Nevada Mountains, remains a persistent problem today. Based on sediment cores, pre-mining concentrations were about five times lower than concentrations detected within San Francisco Bay today {Conaway, 2003 #466}.

Hydromodification Projects

Several of the rivers within California have been modified by dams, water diversions, drainage systems for agriculture and drinking water, and some of the most drastic channelization projects in the nation (see species distribution maps). In all, there are about 1,400 dams within the State of California, more than 5,000 miles of levees, and more than 140 aqueducts {Mount, 1995 #463}. In general, the southern basins have a warmer and drier climate and the more northern, coastal-influenced basins are cooler and wetter. About 75% of the runoff occurs in basins in the northern half of California, while 80% of the water demand is in the southern half. Two water diversion projects meet these demands—the federal Central Valley Project (CVP) and the California State Water Project (CSWP). The CVP is one of the world's largest water storage and transport systems. The CVP has more than 20 reservoirs and delivers about 7 million acre-ft per year to southern California. The CSWP has 20 major reservoirs and holds nearly 6 million acre-ft of water. The CSWP delivers about 3 million acre-ft of water for human use. Together, both diversions irrigate about 4 million acres of farmland and deliver drinking water to roughly 22 million residents.

Both the Sacramento and San Joaquin rivers are heavily modified, each with hundreds of dams. The Rogue, Russian, and Santa Ana rivers each have more than 50 dams, and the Eel, Salinas, and the Klamath Rivers have between 14 and 24 dams each. The Santa Margarita is considered one of the last free flowing rivers in coastal southern California with nine dams occurring in its watershed. All major tributaries of the San Joaquin River are impounded at least once and most have multiple dams or diversions. The Stanislaus River, a tributary of the San Joaquin River, has over 40 dams. As a result, the hydrograph of the San Joaquin River is seriously altered from its natural state. Alteration of the temperature and sediment transport regimes had profound influences on the biological community within the basin. These modifications generally result in a reduction of suitable habitat for native species and frequent increases in suitable habitat for non-native species. The Friant Dam on the San Joaquin River is attributed with the extirpation of spring-run Chinook salmon within the basin. A run of the spring-run Chinook salmon once produced about 300,000 to 500,000 fish {Carter, 2005 #457}.

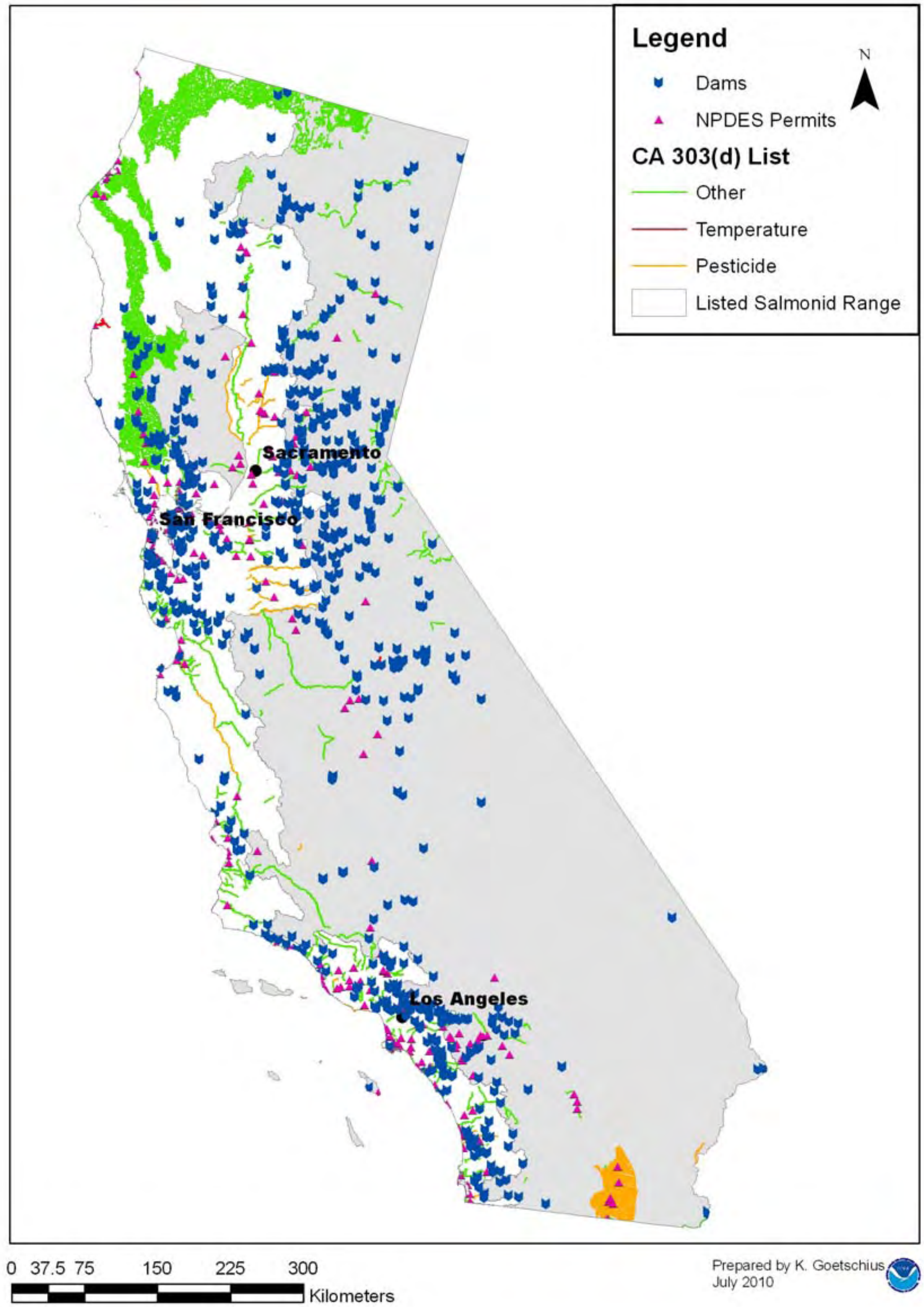


Figure 34 Southest Coast 303(d) waters, dams, and NPDEs permit sites

Artificial Propagation

Anadromous fish hatcheries have existed in California since establishment of the McCloud River hatchery in 1872. There are nine state hatcheries: the Iron Gate (Klamath River), Mad River, Trinity (Trinity River), Feather (Feather River), Warm Springs (Russian River), Nimbus (American River), Mokelumne (Mokelumne River), and Merced (Merced River). The California Department of Fish and Game (CDFG) also manages artificial production programs on the Noyo and Eel rivers. The Coleman National Fish Hatchery, located on Battle Creek in the upper Sacramento River, is a federal hatchery operated by the USFWS. The USFWS also operates an artificial propagation program for Sacramento River winter run Chinook salmon.

Of these, the Feather River, Nimbus, Mokelumne, and Merced River facilities comprise the Central Valley Hatcheries. Over the last ten years, the Central Valley Hatcheries have released over 30 million young salmon. State and the federal (Coleman) hatcheries work together to meet overall goals. State hatcheries are expected to release 18.6 million smolts in 2008 and Coleman is aiming for more than 12 million. There has been no significant change in hatchery practices over the year that would adversely affect the current year class of fish. A new program marking 25% of the 32 million Sacramento River Fall-run Chinook smolts may provide data on hatchery fish contributions to the fisheries in the near future.

Commercial and Recreational Fishing

The region is home to many commercial fisheries. The largest in terms of total California landings in 2006 were northern anchovy, Pacific sardine, Chinook salmon, sablefish, Dover sole, Pacific whiting, squid, red sea urchin, and Dungeness crab {CDFG, 2007 #468}. Red abalone is also harvested.

Despite regulated fishing programs for salmonids, listed salmonids are also caught as bycatch. There are several approaches under the ESA to address tribal and state take of ESA-listed species that may occur as a result of harvest activities. Section 10 of the ESA provides for permits to operate fishery harvest programs. ESA section 4(d) rules provide exemptions from take for resource, harvest, and hatchery management plans.

Management of salmon fisheries in the Southwest Coast Region is a cooperative process involving federal, state, and tribal representatives. The Pacific Fishery Management Council sets annual fisheries in federal waters from three to 200 miles off the coasts of Washington, Oregon, and California. Inland fisheries are those within state boundaries, including those extending out three miles from state coastlines. The states of Oregon, Idaho, California, and Washington issue salmon fishing licenses for inland fisheries. The California Fish and Game Commission (CFGC) sets the salmon seasons and issues permits for all California waters and the Oregon Department of Fish and Game sets the salmon seasons and issues permits for all Oregon waters.

In 2008, there was an unprecedented collapse of the Sacramento River fall-run Chinook salmon that led to complete closure of the commercial and sport Chinook fisheries in California and in Oregon south of Cape Falcon. U.S. Department of Commerce Secretary Gary Locke released a 2008 West Coast salmon disaster declaration for California and Oregon in response to poor salmon returns to the Sacramento River, which led to federal management reducing commercial salmon fishing off southern Oregon and California to near zero. Secretary Locke also released $53.1 million in disaster funds to aid affected fishing communities.

In 2009, federal fishery managers severely limited commercial salmon fishing in California and Oregon for the second year in a row due to low Sacramento River fall-rn Chinook salmon returns. California State sport and commercial ocean salmon seasons were closed by the CFGC through August 28, 2009. There was a 10-day ocean sport fishery in the Klamath Management Zone (Horse Mountain to the California-Oregon border) from August 29 through September 7, 2009. A limited in-river salmon season was considered by the CFGC at its May meeting. The CFGC decided to leave open the Sacramento River between the Highway 113 bridge near Knight's Landing and just below the Lower Red Bluff (Sycamore) Boat Ramp from November 16 through December 31,

2009. The Klamath-Trinity River Basin had a salmon sport fishing season for Klamath River fall Chinook salmon that began August 15, 2009.

Non-native Species

Plants and animals that are introduced into habitats where they do not naturally occur are called non-native species. They are also known as non-indigenous, exotic, introduced, or invasive species, and have been known to affect ecosystems. Non-native species are introduced through infested stock for aquaculture and fishery enhancement, through ballast water discharge and from the pet and recreational fishing industries (http://biology.usgs.gov/s+t/noframe/x191.htm.). The Aquatic Nuisance Species Task Force suggests that it is inevitable that cultured species will eventually escape confinement and enter U.S. waterways. Non-native species were cited as a contributing cause in the extinction of 27 species and 13 subspecies of North American fishes over the past 100 years {Miller, 1989 #516}. Wilcove, Rothstein *et al.* {, 1998 #788} note that 25% of ESA-listed fish are threatened by non-native species. By competing with native species for food and habitat as well as preying on them, non-native species can reduce or eliminate populations of native species.

Surveys performed by CDFG state that at least 607 non-native species are found in California coastal waterways (Foss *et al.* 2007). The majority of these species are representatives of four phyla: annelids (33%), arthropods (22%), chordates (13%), and mollusks (10%). Non-native chordate species are primarily fish and tunicates which inhabit fresh and brackish water habitats such as the Sacramento-San Joaquin Delta {Foss, 2007 #787}. The California Aquatic Invasive Species Management Plan includes goals and strategies for reducing the introduction rate of new invasive species as well as removing those with established populations.

Atmospheric Deposition in the Southwest Coast Region

In developing the California Red-legged frog (CRLF) assessments, EPA considered ambient air monitoring and atmospheric deposition processes for some of the 12 a.i.s evaluated in this Opinion. This information is discussed in the *Targeted Monitoring Studies* subsection within the *Effects of the Proposed Action* Chapter.

Pacific Northwest Region

This region encompasses Idaho, Oregon, and Washington and includes parts of Nevada, Montana, Wyoming, and British Columbia. In this section we discuss three major areas that support salmonid populations within the action area. They include the Columbia River Basin and its tributaries, the Puget Sound Region, and the coastal drainages north of the Columbia River.

Eighteen of the 28 ESUs/DPSs addressed in the Opinion occur within the Pacific Northwest Region. They are the Puget Sound Chinook salmon, Lower Columbia River (LCR) Chinook salmon, Upper Columbia River (UCR) Spring-run Chinook salmon, Snake River (SR) Fall-run Chinook salmon, SR Spring/Summer-run Chinook salmon, Upper Willamette River (UWR) Chinook salmon, Hood Canal (HC) Summer-run chum, Columbia River (CR) chum, LCR coho, Oregon Coast (OC) coho, Ozette Lake sockeye, SR sockeye, Puget Sound steelhead, LCR steelhead, UWR steelhead, Middle Columbia River (MCR) steelhead, UCR steelhead, and the SR steelhead (Table 4).

Table 10, Table 11, and Table 12 show the types and areas of land use within each salmonid ESU/DPS.

Table 64. Area of land use categories within Chinook Salmon ESUs in km² where bolded numbers are totals of each category. Land cover image data were taken from Multi-Resolution Land Characteristics (MRLC) Consortium, a consortium of nine federal agencies (USGS, EPA, USFS, NOAA, NASA, BLM, NPS, NRCS, and USFWS) {NLCD, 2001 #785}. Land cover class definitions are available at: http://www.mrlc.gov/nlcd_definitions.php

Landcover Type		Chinook Salmon					
sub category	code	Puget Sound	Lower Columbia River	Upper Columbia River Spring Run	Snake River Fall Run	Snake River Spring/ Summer Run	Upper Willamette River
Water		**6,485**	**653**	**203**	**346**	**293**	**130**
Open Water	11	6,172	641	188	346	253	124
Perennial Snow/Ice	12	313	12	16	0	40	7
Developed Land		**5,271**	**1,861**	**847**	**2,588**	**974**	**2,008**
Open Space	21	1,601	649	203	1,150	328	632
Low Intensity	22	1,694	517	218	578	113	722
Medium Intensity	23	668	290	55	567	30	322
High Intensity	24	266	118	11	135	2	112
Barren Land	31	1,042	287	360	158	500	220
Undeveloped Land		**22,481**	**10,692**	**16,155**	**15,168**	**52,573**	**14,159**
Deciduous Forest	41	999	551	21	664	10	248
Evergreen Forest	42	14,443	6,497	8,138	3,761	27,701	9,531
Mixed Forest	43	2,526	927	7	479	4	1,130
Shrub/Scrub	52	2,415	1,598	6,100	3,203	13,618	1,940
Herbaceous	71	957	520	1,737	6,317	11,053	801
Woody Wetlands	90	648	377	92	191	96	431
Emergent Wetlands	95	492	223	59	553	92	78
Agriculture		**1,447**	**825**	**964**	**5,879**	**4,316**	**5,972**
Hay/Pasture	81	1,188	547	327	769	456	3,617
Cultivated Crops	82	258	278	636	5,110	3,860	2,355
TOTAL (inc. open water)		35,683	14,031	18,168	23,982	58,157	22,269
TOTAL (w/o open water)		29,511	13,390	17,981	23,636	57,904	22,146

Table 65. Area of land use categories within chum and coho ESUs in km². Land cover image data were taken from Multi-Resolution Land Characteristics (MRLC) Consortium, a consortium of nine federal agencies (USGS, EPA, USFS, NOAA, NASA, BLM, NPS, NRCS, and USFWS) {NLCD, 2001 #785}. Land cover class definitions are available at: http://www.mrlc.gov/nlcd_definitions.php

Landcover Type		**Chum Salmon**		**Coho Salmon**		**Sockeye Salmon**	
sub category	code	Hood Canal Summer Run	Columbia River	Lower Columbia River	Oregon Coast	Ozette Lake	Snake River
Water		**755**	**656**	**687**	**200**	**30**	**36**
Open Water	11	704	655	675	200	30	19
Perennial Snow/Ice	12	51	1	12	0	0	18
Developed Land		**403**	**1,684**	**1,990**	**1,807**	**3**	**15**
Open Space	21	134	605	708	1,107	1	3
Low Intensity	22	77	463	563	163	0	2
Medium Intensity	23	20	258	305	49	0	0
High Intensity	24	6	110	124	20	0	0
Barren Land	31	166	247	290	467	2	9
Undeveloped Land		**3,324**	**8,198**	**13,254**	**24,589**	**195**	**1,259**
Deciduous Forest	41	97	548	575	418	3	0
Evergreen Forest	42	2,477	4,294	8,487	14,943	158	755
Mixed Forest	43	200	892	999	4,126	3	0
Shrub/Scrub	52	299	1,353	1,982	3,134	14	185
Herbaceous	71	61	363	386	263	8	269
Woody Wetlands	90	56	222	225	226	8	16
Emergent Wetlands	95	133	526	600	1,478	1	34
Agriculture		**66**	**746**	**1,028**	**925**	**0**	**13**
Hay/Pasture	81	64	533	680	860	0	12
Cultivated Crops	82	2	213	348	64	0	1
TOTAL (inc. open water)		4,548	11,284	16,959	27,520	228	1,323
TOTAL (w/o open water)		3,843	10,628	16,284	27,320	199	1,304

Table 66. Area of land use categories within sockeye ESUs and steelhead DPSs in km² where bolded numbers are totals for each category. Land cover image data were taken from Multi-Resolution Land Characteristics (MRLC) Consortium, a consortium of nine federal agencies (USGS, EPA, USFS, NOAA, NASA, BLM, NPS, NRCS, and USFWS) {NLCD, 2001 #785}. Land cover class definitions are available at: http://www.mrlc.gov/nlcd_definitions.php

Landcover		Steelhead					
sub category	Code	Puget Sound	Lower Columbia River	Upper Willamette River	Middle Columbia River	Upper Columbia River	Snake River
Water		**6,485**	**262**	**62**	**588**	**375**	**327**
Open Water	11	6,172	250	62	575	359	285
Perennial Snow/Ice	12	313	12	0	13	16	42
Developed Land		**5,271**	**1,601**	**1,278**	**2,304**	**1,092**	**1,205**
Open Space	21	1,601	518	382	1,276	343	515
Low Intensity	22	1,694	506	513	627	294	144
Medium Intensity	23	668	287	231	192	80	40
High Intensity	24	266	116	75	25	13	3
Barren Land	31	1,042	174	77	183	361	504
Undeveloped Land		**22,481**	**10,339**	**6,942**	**53,790**	**19,621**	**67,839**
Deciduous Forest	41	999	382	171	54	25	35
Evergreen Forest	42	14,443	7,023	4,133	18,347	8,223	39,556
Mixed Forest	43	2,526	611	791	41	7	17
Shrub/Scrub	52	2,415	1,589	994	32,089	9,351	15,644
Herbaceous	71	957	398	519	2,752	1,823	12,361
Woody Wetlands	90	648	244	292	217	109	116
Emergent Wetlands	95	492	93	43	291	81	111
Agriculture		**1,447**	**927**	**4,373**	**12,771**	**3,684**	**6,690**
Hay/Pasture	81	1,188	605	2,529	863	448	463
Cultivated Crops	82	258	322	1,844	11,908	3,236	6,227
TOTAL (inc. open water)		35,683	13,128	12,655	69,453	24,771	76,061
TOTAL (w/o open water)		29,511	12,878	12,593	68,878	24,411	75,777

Columbia River Basin

The most notable basin within the Pacific Northwest region is the Columbia River. The Columbia River is the largest river in the Pacific Northwest and the fourth largest river in terms of average discharge in the U.S. The Columbia River drains over 258,000 square miles, and is the sixth largest in terms of drainage area. Major tributaries include the

Snake, Willamette, Salmon, Flathead, and Yakima rivers. Smaller rivers include the Owyhee, Grande Ronde, Clearwater, Spokane, Methow, Cowlitz, and the John Day Rivers (see Table 67 for a description of select Columbia River tributaries). The Snake River is the largest tributary at more than 1,000 miles long. The headwaters of the Snake River originate in Yellowstone National Park, Wyoming. The second largest tributary is the Willamette River in Oregon {Hinck, 2004 #472;Kammerer, 1990 #471}. The Willamette River is also the 19th largest river in the nation in terms of average annual discharge {Kammerer, 1990 #471}. The basins drain portions of the Rocky Mountains, Bitteroot Range, and the Cascade Range.

Table 67. Select tributaries of the Columbia River {Carter, 2005 #457}

Watershed	Approx Length (mi)	Basin Size (mi^2)	Physiographic Provinces*	Mean Annual Precipitation (in)	Mean Discharge (cfs)	No. Fish Species (native)	No. Endangered Species
Snake/Salmon rivers	870	108,495	CU, NR, MR, B/R	14	55,267	39 (19)	5 fish (4 T, 1 E), 6 (1 T, 5 E) snails, 1 plant (T)
Yakima River	214	6,139	CS, CU	7	3,602	50	2 fish (T)
Willamette River	143	11,478	CS, PB	60	32,384	61 (~31)	5 fish (4 T, 1 E),

* Physiographic Provinces: CU = Columbia-Snake River Plateaus, NR = Northern Rocky Mountains, MR = Middle Rocky Mountains, B/R = Basin & Range, CS = Cascade-Sierra Mountains, PB = Pacific Border

The Columbia River and estuary were once home to more than 200 distinct runs of Pacific salmon and steelhead with unique adaptations to local environments within a tributary {Stanford, 2005 #473}. Salmonids within the basin include Chinook salmon, chum salmon, coho salmon, sockeye salmon, steelhead, redband trout, bull trout, and cutthroat trout.

Land Use in the Columbia River Basin

More than 50% of the U.S. portion of the Columbia River Basin is in federal ownership (most of which occurs in high desert and mountain areas). Approximately 39% is in private land ownership (most of which occurs in river valleys and plateaus). The remaining 11% is divided among the tribes, state, and local governments {Hinck, 2004 #472}. See Table for a summary of land uses and population densities in several subbasins within the Columbia River watershed [data from {Stanford, 2005 #502}].

Table 14. Land use and population density in select tributaries of the Columbia River {Stanford, 2005 #502}.

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

The interior Columbia Basin has been altered substantially by humans causing dramatic changes and declines in native fish populations. In general, the basin supports a variety of mixed uses. Predominant human uses include logging, agriculture, ranching, hydroelectric power generation, mining, fishing, a variety of recreational activities, and urban uses. The decline of salmon runs in the Columbia River is attributed to loss of habitat, blocked migratory corridors, altered river flows, pollution, overharvest, and competition from hatchery fish. In the Yakima River, 72 stream and river segments are listed as impaired by the Washington State Department of Ecology (DOE) and 83% exceed temperature standards. In the Yakima River, non-native grasses and other plants are commonly found along the lower reaches of the river {Stanford, 2005 #473}. In the Willamette River, riparian vegetation was greatly reduced by land conversion. By 1990, only 37% of the riparian area within 120 m was forested, 30% was agricultural fields, and 16% was urban or suburban lands.

Ranching and Agriculture

Ranching, agriculture, and related services in the Pacific Northwest employ more than nine times the national average [19% of the households within the basin {NRC, 2004 #475}]. Ranching practices have led to increased soil erosion and sediment loads within adjacent tributaries. The worst of these effects may have occurred in the late 1800s and early 1900s from deliberate burning to increase grass production {NRC, 2004 #475}. Several measures are currently in place to reduce the impacts of grazing. Measures include restricted grazing in degraded areas, reduced grazing allotments, and lowered stocking rates. Today, the agricultural industry impacts water quality within the basin. Agriculture is second only to the large-scale influences of hydromodification projects regarding power generation and irrigation. Water quality impacts from agricultural activities include alteration of the natural temperature regime, insecticide and herbicide contamination, and increased suspended sediments. During general agricultural operations, pesticides are applied on a variety of crops for pest control. These pesticides may contaminate surface water via runoff especially after rain events following application. Available data on usage of the 12 a.i.s in the Pacific Northwest are reported below. Pesticide detection data for these same a.i.s are reported in the *Targeted Monitoring* subsection of the *Effects of the Proposed Action* chapter.

Agricultural Usage for some of the 12 a.i.s in the Pacific Northwest Region

<u>Azinphos methyl.</u> Azinphos methyl usage information for Washington, Oregon, and Idaho was obtained from the U.S. Department of Agriculture's NASS Agricultural Chemical Usage report cited in {EPA, 2003 #1676}. This compound is used for crops in central Washington. They include apples, pears, sweet cherries, plums, and grapes. Most uses of azinphos methyl are on apples (85%) (EPA BE 2003). In 1995, the maximum use was on apples between 1990 and 2001 (474,400 lbs a.i./acres). In 2001, about 241,400 lbs azinphos methyl/acres were applied, indicating a declining use trend. About 3% of the land area in Washington was in apple orchards and 55% of the total apple acreage was within 400 m of flowing water. An additional 5% of the total apple acreage was within 400 m of a static water body. In Oregon, azinphos methyl is used mostly on pears and apples, with some use on sweet cherries, potatoes, and caneberries. In Idaho, azinphos methyl is used on potatoes, where an average of 6% of the crop was treated during this period {EPA, 2003 #1676}.

<u>Bensulide.</u> Actual use of bensulide on crops for this region is unclear. For the Pacific Northwest, the QUA highlights sugar beets in Oregon with an average annual use of 5,000 lbs, "other crops" in Oregon with an average annual use of 1,000 lbs, and onions in Idaho, Oregon, and Texas with an average annual use at 99,000 lbs. More recent usage data beyond 1996 are unavailable for Idaho, Oregon, and Washington {EPA, 2002 #1677}. However, it appears most use in areas with listed salmonid and steelhead is on onions {EPA, 2007 #1689}.

Disulfoton. Disulfoton usage on crops in the Pacific Northwest and California include asparagus, broccoli, peppers, barley, potatoes, and wheat {EPA, 2002 #1677}. Asparagus is grown in six counties in eastern Washington. They include Franklin (10,900 acres), Yakima (7,300 acres), Walla Walla (1,600 acres), Grant (1,000 acres), Benton (500 acres), and Adams (500 acres) (Washington State Department of Agriculture/Endangered Species Program 2003). The National Pesticide Use Database (NCFAP 2001) indicates that for major crops in the Pacific Northwest and California, total application of disulfoton in 1992 (census report) was 821,337 lbs a.i./year. In the 1997 data, 560,367 lbs a.i. of disulfoton (IRED QUA; attachment 2) were applied to the same crops. The greatest decline was observed in total wheat application, which fell from 498,288 to 188,498 lbs a.i./year {EPA, 2002 #1677}. Most disulfoton usage on barley occurs in the eastern portion of Idaho.

Dimethoate. Dimethoate usage information for Idaho, Oregon, and Washington are limited. This compound is not a recognized product for forestry use in the Pacific Northwest. Although dimethoate is mainly applied to cottonwood, birch, oak, douglas fir, fraser fir, cypress, and cedar trees, specific usage information for this compound is unavailable {EPA, 2004 #1678}.

Ethoprop. Ethoprop is used on a variety of crops. There are no registered homeowner uses {EPA, 2006 #1703}. The National Quantitative Use Database (NCFAP) indicates total ethoprop use in the Pacific Northwest and California on potatoes, corn, and sweet potatoes was 569,203 lbs a.i. in 1992 and 470,831 lbs a.i. in 1997 {EPA, 2003 #1680}. The Central Basin of Washington State is a large potato growing region of the country where ethoprop is commonly used to control nematodes and wireworms.

For most commodities, ethoprop use increased during 1992 - 1997. Ethoprop use in Washington increased from 4,228 lbs a.i. to 9,784 lbs a.i. for green beans; from 13,121 lbs a.i. to 41,315 lbs a.i. for corn; and from 90,288 lbs a.i. to 113,499 lbs a.i. for potatoes. Similar increased use of ethoprop occurred in Idaho for green beans (from 0 lbs a.i. in 1993 to 3,409 lbs a.i. in 1997); and in Oregon for corn (from 9,642 lbs a.i. in 1992 to

25,843 lbs a.i. in 1997; and potatoes (from 44,136 lbs a.i. in 1992 to 59,262 in 1997). 1997 ethoprop use data for Idaho indicate 161,151 lbs a.i. compared to 313,135 lbs a.i. used in 1992 for potatoes. 1997 data for Oregon indicate 15,285 lbs a.i. used on green beans compared to 18,430 used in 1992 {EPA, 2003 #1680}. However, use on potatoes and sweet potatoes declined significantly in Idaho and Oregon {EPA, 2003 #1680}. Washington state counties that grow green beans, corn, and potatoes include: Adams, Grant, Franklin, Skagit, Whatcom, and Yakima.

Fenamiphos. Idaho, Oregon, and Washington are not listed as states with significant use of fenamiphos. However fenamiphos has been applied to raspberries, apples, and pears in the Pacific Northwest. According to EPA's 2000 QUA (based on data from 1990 to 1998), Washington and Oregon are the states with the greatest amount of fenamiphos use on raspberries. However, only a weighted average of 9% of the crop was treated, with an estimated maximum of 21% of the crop being treated in both states (EPA BE 2003). About 77% of all raspberries grown in Washington occur in Whatcom county. The remaining raspberry acreage include Skagit, Clark, Cowlitz, and Pierce counties (Washington State Department of Agriculture/Endangered Species Program 2003).

Methidathion. Current usage information for methidathion is limited for the Pacific Northwest. In Washington State, methidathion is applied to apples, apricots, cherries, peaches, nectarines, plums, pears, artichokes, alfalfa, and sunflowers. Counties that grow some or most of the above crops include Yakima, Douglas, Benton, Grant, and Okanogan (Washington State Department of Agriculture/Endangered Species Program 2004). In Idaho and Oregon, methidathion is applied on alfalfa hay, cherries, plums, all pome fruits, and English walnuts (Oregon only).

Methamidophos. Recent usage data for methamidophos in the Pacific Northwest are not readily available and are therefore unreported. Known applications of methamidophos are for potatoes grown in Washington and Oregon. However, the extent of this compound's use in both states is not readily available. Potato growing counties in Washington include Benton, Franklin, Grant, Okanogan, Walla Walla, and Yakima.

Potato growing counties in Oregon include Morrow, Multnomah, Umatilla, and Union {EPA, 2004 #1682}.

Methidathion. Recent usage data for methidathion in the Pacific Northwest are not readily available and are therefore unreported. Known applications of methidathion in this region are for alfalfa hay and alfalfa grown for seed, almonds, apples, plums, cherries, nectarines, and all pome fruits. Counties in Washington that produce the above crops include Benton, Franklin, Grant, Kittitas, Klickitat, Okanogan, Walla Walla, Wallowa, Whitman, and Yakima. Counties in Oregon that grow these same crops include Gilliam, Morrow, Multnomah, Umatilla, Union, Wallowa, and Wasco. Similarly, counties in Idaho that grow the above crops are Idaho, Latah, Lemhi, Lewis, and Nez Perce {EPA, 2004 #1683} .

Methyl parathion. Recent usage data for methyl parathion in the Pacific Northwest are not readily available and are therefore unreported. Known applications of methyl parathion in this region are for hay, alfalfa hay, barley, corn, lentils, potations, sugar beets, wheat, peas, oats, potatoes, and all beans (except lima). Counties in Washington that produce the above crops include Adams, Benton, Franklin, Grant, Klickitat, Walla Walla, Whatcom, Whitman, and Yakima. Counties in Oregon that grow the above crops include Gilliam, Klamath, Morrow, Multnomah, Umatilla, and Union. Similarly, counties in Idaho that grow these same crops are Idaho, Latah, Lemhi, Lewis, and Nez Perce {EPA, 2004 #1684}.

Naled. Recent usage data for naled in the Pacific Northwest are not readily available and are therefore unreported. Known applications of naled in Washington include peas, beans, hops, alfalfa for seed, grapes, and sugar beets. The predominant counties in Washington that produce these crops are Adams, Benton, Franklin, Grant, King, Skagit, Walla Walla, Whitman, and Yakima (Washington State Department of Agriculture/Endangered Species Program 2004). Known applications of naled in Idaho and Oregon are for alfalfa grown for seed {EPA, 2004 #1685}. The predominant counties in Idaho that produce the above crops are Clackamas, Idaho, Latah, Lemhi, Marion, Nez Perce, Washington, and Union {EPA, 2004 #1685}. Although naled is used for mosquito control, it is not an important agent in controlling adult mosquitoes in the Pacific Northwest {EPA, 2004 #1685}.

Phorate. Recent usage information on phorate in the Pacific Northwest is not readily available and is therefore unreported. Known applications of phorate in this region are for potatoes, field and sweet corn, sugar beets, and beans. Counties in Washington that produce the above crops include Adams, Benton, Franklin, Grant, Walla Walla, and Yakima. Counties in Oregon that produce these same crops include Benton, Linn, Marion, Morrow, Multnomah, Umatilla, Wallowa, and Washington {EPA, 2003 #1686}. Small amounts of beets and corn are grown in Latah county, ID (www.quickstats.nass.usda.gov, accessed 8/24/10).

Phosmet. Recent usage data on phosmet in the Pacific Northwest are not readily available and are therefore unreported. Known applications of phosmet in this region are for alfalfa, apples, cherries, nectarines, peas, pears, peaches, and potatoes. In Washington, phosmet was applied to apples, pears, potatoes, peaches, and nectarines. Counties in Washington that produce these crops include Adams, Benton, Chelan, Douglas, Franklin, Grant, Klickitat, Okanogan, Walla Walla, and Yakima. In 1997, approximately 317,520 lbs and 14,000 lbs of phosmet were applied to apples and pears in Washington, respectively (Washington State Department of Agriculture 2003). In 2000, approximately 32,200 lbs and 9,400 lbs of phosmet were applied to pears and apples in Oregon, respectively (as cited in {EPA, 2003 #1687}). No information is provided for phosmet use in Idaho.

As part of the baseline pesticide conditions in the Pacific Northwest region, pesticide reduction programs exist in Idaho, Oregon, and Washington to minimize levels of the above a.i.s into the aquatic environment. They are described below.

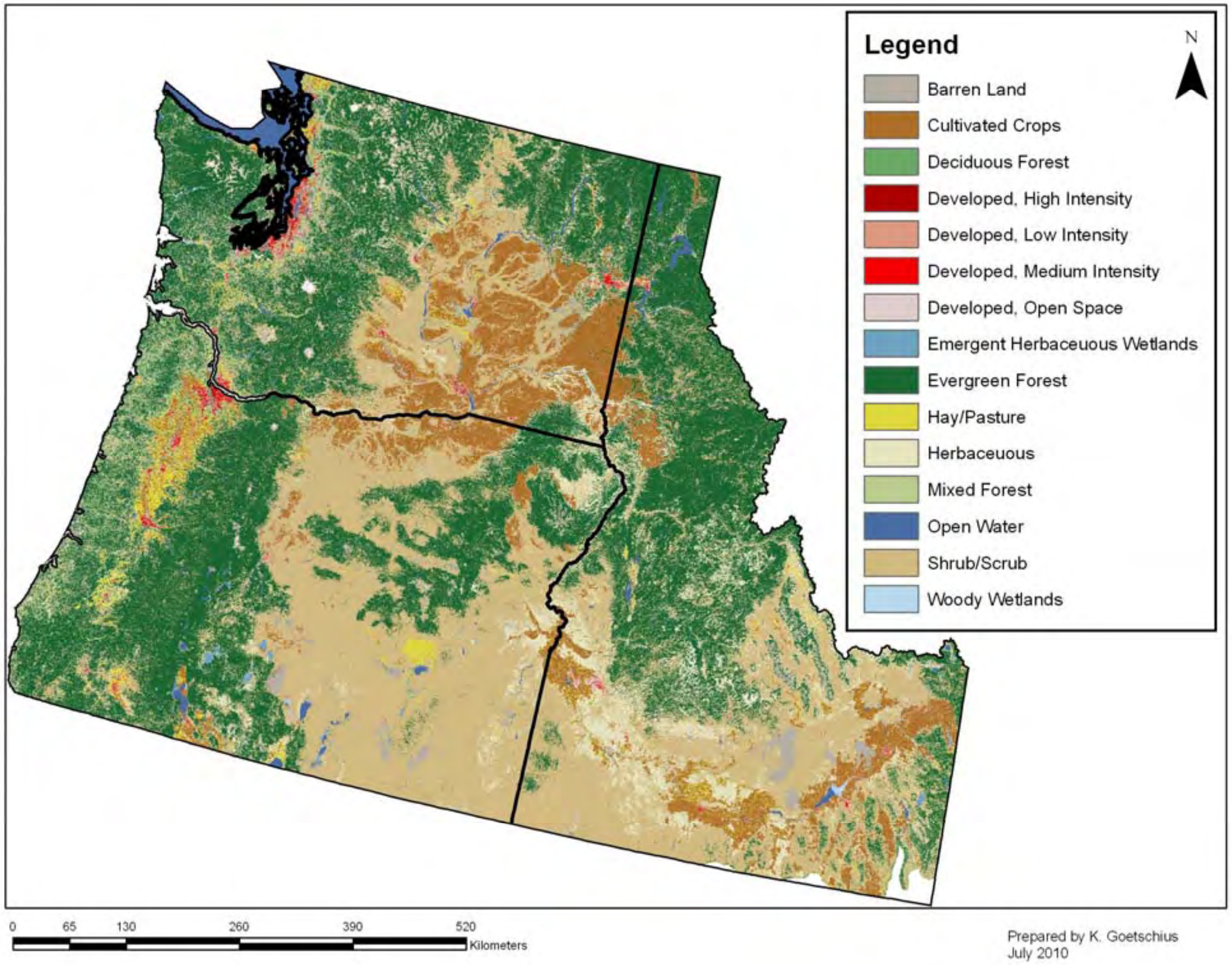


Figure 35 Pacific Nortwest Landsue

Pesticide Reduction Programs in the Pacific Northwest Region

When using all 12 a.i.s, growers must adhere to the court-ordered injunctive relief, requiring buffers of 20 yards for ground application and 100 yards for any aerial application. These measures are mandatory in all four states, pending completion of consultation.

The Idaho State Department of Agriculture has published a BMP guide for pesticide use. The BMPs include eight "core" voluntary measures that will prevent pesticides from leaching into soil and groundwater. These measures include applying pest-specific controls, being aware of the depth to ground water, and developing an Irrigation Water Management Plan.

Oregon has PURS legislation that requires all agricultural uses of registered pesticides be reported. In this case "agricultural" use includes applications to parks, golf courses, and most livestock uses. Oregon requires reporting if application is part of a business, for a government agency, or in a public place. However, the Governor of Oregon has suspended the PURS program until January 2013 due to budget shortages.

Oregon has also implemented a voluntary program. The Pesticide Stewardship Partnerships (PSP) program began in 1999 through the Oregon Department of Environmental Quality. The PSP's goal is to involve growers and other stakeholders in water quality management at a local level. Effectiveness monitoring is used to provide feedback on the success of mitigation measures. As of 2006, there were six pilot PSPs planned or in place. Early results from the first PSPs in the Columbia Gorge Hood River and in Mill Creek demonstrate reductions in chlorpyrifos and diazinon levels and detection frequencies. DEQ's pilot programs suggest that PSPs can help reduce contamination of surface waters.

Oregon is in the process of developing a Pesticide Management Plan for Water Quality

Protection, as required under FIFRA. This plan describes how government agencies and stakeholders will collaboratively reduce pesticides in Oregon water supplies. The PSP program is a component of this plan, and will provide information on the effectiveness of mitigation measures.

Washington State has a Surface Water Monitoring Program that looks at pesticide concentrations in some salmonid bearing streams and rivers. The program was initiated in 2003 and now monitors four areas. Three of these were chosen due to high overlap with agriculture: the Skagit-Samish watershed, the Lower Yakima Watershed, and the Wenatchee and Entiat watersheds. The final area, in the Cedar-Sammamish watershed, is an urban location, intended to look at runoff in a non-agriculture setting. It was chosen due to detection of pesticides coincident with pre-spawning mortality in coho salmon. The Surface Water Monitoring program is relatively new and will continue to add watersheds and testing for additional pesticides over time.

Washington State also has a voluntary program that assists growers in addressing water rights issues within a watershed. Several watersheds have elected to participate, forming Comprehensive Irrigation District Management Plans (CIDMPs). The CIDMP is a collaborative process between government and landowners and growers; the parties determine how they will ensure growers get the necessary volume of water while also guarding water quality. This structure allows for greater flexibility in implementing mitigation measures to comply with both the CWA and the ESA.

The Columbia Gorge Fruit Growers Association is a non-profit organization dedicated to the needs of growers in the mid-Columbia area. The association brings together over 440 growers and 20 shippers of fruit from Oregon and Washington. It has issued a BMP handbook for OPs, including information on alternative methods of pest control. The mid-Columbia area is of particular concern, as many orchards are in close proximity to streams.

Stewardship Partners is a non-profit organization in Washington State that works to build partnerships between landowners, government, and non-profit organizations. In large part, its work focuses on helping landowners to restore fish and wildlife habitat while maintaining the economic viability of their farmland. Projects include restoring riparian areas, reestablishing floodplain connectivity, and removing blocks to fish passage. Another current project is to promote rain gardens as a method of reducing surface water runoff from developed areas. Rain gardens mimic natural hydrology, allowing water to collect and infiltrate the soil.

Stewardship Partners also collaborates with the Oregon-based Salmon-Safe certification program. Salmon-Safe is an independent eco-label recognizing organizations who have adopted conservation practices that help restore native salmon habitat in Pacific Northwest rivers and streams. These practices protect water quality, fish and wildlife habitat, and overall watershed health. While the program began with a focus on agriculture, it has since expanded to include industrial and urban sites as well. The certification process includes pesticide restrictions. Salmon-Safe has produced a list of "high risk" pesticides which, if used, would prevent a site from becoming certified. If a grower wants an exception, they must provide written documentation that demonstrates a clear need for use of the pesticide, that no safer alternatives exist, and that the method of application (such as timing, location, and amount used) represents a negligible risk to water quality and fish habitat. Bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, naled, and phosmet are all on the high risk list. Over 250 farms and businesses currently have the Salmon-Safe certification.

In addition to pesticide usage for agriculture, this land use further affects available salmonid aquatic habitat. The amount and extent of water withdrawals or diversions for agriculture impact streams and their inhabitants via reduced water flow/velocity and dissolved oxygen levels. These impacts are described below:

Water Diversions for Agriculture in the Pacific Northwest Region

Agriculture and ranching increased steadily within the Columbia River basin from the mid- to late-1800s. By the early 1900s, agricultural opportunities began increasing at a much more rapid pace with the creation of more irrigation canals and the passage of the Reclamation Act of 1902 {NRC, 2004 #475}. Today, agriculture represents the largest water user within the basin (>90%).

Roughly 6% of the annual flow from the Columbia River is diverted for the irrigation of 7.3 million acres of croplands within the basin. The vast majority of these agricultural lands are located along the lower Columbia River, the Willamette, Yakima, Hood, and Snake rivers, and the Columbia Plateau {Hinck, 2004 #472}.

The impacts of these water diversions include an increase nutrient load, sediments (from bank erosion), and temperature. Flow management and climate changes have further decreased the delivery of suspended particulate matter and fine sediment to the estuary. The conditions of the habitat (shade, woody debris, overhanging vegetation) whereby salmonids are constrained by low flows also may make fish more or less vulnerable to predation, elevated temperatures, crowding, and disease. Water flow effects on salmonids may seriously impact adult migration and water quality conditions for spawning and rearing salmonids. High temperature may also result from the loss of vegetation along streams that used to shade the water and from new land uses (buildings and pavement) whereby rainfall picks up heat before it enters into an adjacent stream. Runoff inputs from multiple land use may further pollute receiving waters inhabited by fish or along fish migratory corridors.

As mentioned earlier in this chapter, the distribution of the most prevalent pesticides in streams and ground water correlate with land use patterns and associated past or present pesticide use. The USGS conducted NAWQA analyses for five basins within the Pacific Northwest Region. The NAWQA data reported some of the 12 a.i.s evaluated in this Opinion as well as the OPs and carbamates that NMFS previously evaluated in its two pesticide Opinions. NAWQA data for these basins are summarized below:

The USGS has a number of fixed water quality sampling sites throughout various tributaries of the Columbia River. Many of the water quality sampling sites have been in place for decades. Water volumes, crop rotation patterns, crop type, and basin location are some of the variables that influence the distribution and frequency of pesticides within a tributary. Detection frequencies for a particular pesticide can vary widely. One study conducted by the USGS between May 1999 and January 2000 in the surface waters of Yakima Basin detected 25 pesticide compounds {Ebbert, 2002 #547}. Atrazine was the most widely detected herbicide and azinphos methyl was the most widely detected insecticide. Other detected compounds include simazine, terbacil, trifluralin; deethylatrazine, carbaryl, diazinon, malathion, and DDE. In addition to current use-chemicals legacy, chemicals continue to pose a serious problem to water quality and fish communities despite their ban in the 1970s and 1980s {Hinck, 2004 #472}.

Fish and macroinvertebrate communities exhibit an almost linear decline in condition as the level of agriculture intensity increases within a basin {Cuffney, 1997 #332;Fuhrer, 2004 #474}. A study conducted in the late 1990s examined 11 species of fish, including anadromous and resident fish collected throughout the basin, for a suite of 132 contaminants. They included 51 semi-volatile chemicals, 26 pesticides, 18 metals, 7 PCBs, 20 dioxins, and 10 furans. Sampled fish tissues revealed PCBs, metals, chlorinated dioxins and furans (products of wood pulp bleaching operations), and other contaminants.

Yakima River Basin: NAWQA Analysis

The Yakima River Basin is one of the most agriculturally productive areas in the U.S. {Fuhrer, 2004 #474}. Croplands within the Yakima Basin account for about 16% of the total basin area of which 77% is irrigated. The extensive irrigation-water delivery and drainage system in the Yakima River Basin greatly controls water quality conditions and aquatic health in agricultural streams, drains, and the Yakima River {Fuhrer, 2004 #474}. From 1999 to 2000, the USGS conducted a NAWQA study in the Yakima River Basin. Fuhrer *et al.* {, 2004 #399} reported that nitrate and orthophosphate were the dominant forms of nitrogen and phosphorus found in the Yakima River and its agricultural tributaries. Arsenic, a known human carcinogen, was also detected in agricultural drains at elevated concentrations during the nonirrigation season when ground water is the primary source of stream flow.

The USGS also detected 76 pesticide compounds in the Yakima River Basin. They include 38 herbicides (including metribuzin), 17 insecticides (such as carbaryl, diazinon, and malathion), 15 breakdown products, and 6 others. Insecticides were detected more frequently in the Yakima Basin than the national average. In agricultural drainages, insecticides were detected in 80% of samples compared to the national average of 37%. Insecticides were also detected more frequently in mixed land-use streams – 71% of samples rather than 53%. Fuhrer *et al.* 2004 attributes this difference to the heavy use of insecticides in fruit orchards (2004). The most frequently detected pesticides were 2,4-D, terbacil, azinphos methyl, atrazine, carbaryl, and deethylatrazine. Generally, compounds were detected in tributaries more often than in the Yakima River itself. Azinphos methyl was detected in 47% of tributary samples, but not in any samples from the mainstem river. Further, azinphos methyl was regularly detected at levels which exceed EPA freshwater chronic-toxicity criterion for the protection of aquatic life. While azinphos methyl was only detected in tributaries during the irrigation season, it was detected at or above the nanogram/L criterion in all but one instance.

NAWQA data from 1999 - 2000 also reported that azinphos methyl was the most widely detected insecticide. Sites with the highest (greater than 70%) azinphos methyl detection rates were associated with drainage basins whereby azinphos methyl was applied only to apples. The maximum detected concentration of azinphos methyl was 0.523 μg/L. Disulfoton concentration exceeded a drinking water human-health advisory limit in one sample. In addition to azinphos methyl, ethoprop and disulfoton were also detected but not reported in detail as for azinphos methyl.
Ninety-one percent of the samples collected from the small agricultural watersheds contained at least two pesticides or pesticide breakdown products. Carbaryl was detected in 29% of tributary samples and 17% of mainstem Yakima River samples at a screening level of 21 nanogram/liter (Fuhrer *et al.* 2004). Methyl parathion and phorate were screened for, but not detected. The assessment did not screen for bensulide, dimethoate, fenamiphos, methamidophos, methidathion, naled, and phosmet. The median and

maximum number of chemicals in a mixture were 8 and 26, respectively {Fuhrer, 2004 #474}. The herbicide 2,4-D, occurred most often in the mixtures, along with azinphos methyl, the most heavily applied pesticide, and atrazine, one of the most aquatic mobile pesticides {Fuhrer, 2004 #474}. However, the most frequently detected pesticides in the Yakima River Basin are total DDTs [DDT and its breakdown products, dichloro-diphenyl-dichloroethylene (DDE)], dichloro-diphenyl-dichloroethane (DDD), and dieldrin {Johnson, 1983 #477;Joy, 2002 #478;Joy, 2002 #478;Fuhrer, 2004 #474}. Nevertheless, concentrations of total DDT in water have decreased since 1991. These reductions are attributed to erosion-controlling best management practices (BMPs).

Central Columbia Plateau: NAWQA Analysis

The Central Columbia Plateau is a prominent apple growing region. Based on 1992 NASS data, this NAWQA unit had the second highest azinphos methyl usage among the 20 NAWQA units initiated in 1991 and eight amongst all 60 NAWQA study units. There were 40 sampling sites for surface water on the Central Columbia Plateau with detections at seven of the sites or 17.5% of the sites. Of these, 13 sites were wasteways or drainage ways, and not suitable for use as a drinking water source. The maximum value found in the Central Columbia Plateau was 0.2 μg/L.

The USGS sampled 31 surface-water sites representing agricultural land use, with different crops, irrigation methods, and other agricultural practices for pesticides in Idaho and Washington from 1992 - 1995 {Williamson, 1998 #793}. Pesticides were detected in samples from all sites, except for the Palouse River at Laird Park (a headwaters site in a forested area). Many pesticides were detected in surface water at very low concentrations. Concentrations of six pesticides exceeded freshwater-chronic criteria for the protection of aquatic life in one or more surface-water samples. They include the herbicide triallate and five insecticides (azinphos methyl, chlorpyrifos, diazinon, *gamma*-HCH, and parathion). Carbaryl and carbofuran were detected in 6% and 5% of samples, respectively. Methomyl was screened for, but not detected in any samples (Williamson *et al.* 1998).

Detections at four sites were high, ranging from 12 to 45 pesticides. The two sites with the highest detection frequencies are in the Quincy-Pasco subunit, where irrigation and high chemical use combine to increase transport of pesticides to surface waters. Pesticide detection frequencies at sites in the dryland farming (non-irrigated) areas of the North-Central and Palouse subunits are below the national median for NAWQA sites. All four sites had at least one pesticide concentration that exceeded a water-quality standard or guideline.

Concentrations of organochlorine pesticides and PCBs are higher than the national median (50^{th} percentile) at seven of 11 sites; four sites were in the upper 25% of all NAWQA sites. Although most of these compounds have been banned, they still persist in the environment. Elevated concentrations were observed in dryland farming areas and irrigated areas.

Williamette Basin: NAWQA Analysis

From 1991 to 1995, the USGS also sampled surface waters in the Willamette Basin, Oregon. Wentz *et al.* {, 1998 #455} reported that 50 pesticides and pesticide degradates of the 86 were detected in streams. Ten of the pesticides exceeded criteria established by the EPA for the protection of freshwater aquatic life from chronic toxicity. Carbaryl exceeded protective criteria in 17 of its 46 detections, while carbofuran exceeded limits in three of 51 detections (Wentz et al. 1998). Azinphos methyl was detected in 3% of samples, though in every case the concentration exceeded protective criteria for aquatic life. Ethoprop was detected in 15% of samples (Wentz *et al.* 1998). Atrazine, simazine, metolachlor, deethylatrazine, diuron, and diazinon were detected in more than one-half of stream samples. Disulfoton, methomyl, methyl parathion, and phorate were tested for but not detected. Forty-nine pesticides were detected in streams draining predominantly agricultural land. About 25 pesticides were detected in streams draining mostly urban areas. The highest pesticide concentrations generally occurred in streams draining predominately agricultural land.

Marion County is within the Willamette Valley watershed and is a major agricultural region. According to EPA, the highest measured concentration of ethoprop (3.1 μg/L) was sampled on Mill Creek in Marion County, Oregon on October 31, 1994 {EPA, 2006 #1703}. At that time, ethoprop was registered for use on turf for sod and seed, which is grown in the area and may have been treated at a maximum application rate of 20 lbs a.i./acre. Other uses of ethoprop in this region include beans, sweet corn and ornamentals, which may have contributed as the source of these detections. A subsequent sampling (second) on November 4, 1994, at the same location measured ethoprop at 1.7 μg/L and 1.9 μg/L. At locations on other streams in Marion County, ethoprop was measured at up to 1.95 μg/L. Nearly all samples collected from streams in Marion County measured ethoprop above the detection limit of 0.003 μg/L.

Additionally, the USGS NAWQA database reported 2,549 samples analyzed from 1991 to 1995 in 20 major watersheds within the U.S. The highest reported concentration of 0.009 μg/L occurred in an agricultural watershed. All other samples were reported at less than the limit of detection (LOD) of 0.003 μg/L. The 1991 - 1995 NAWQA database further reported a maximum concentration of 2 μg/L from a sample collected in the Willamette River Basin, Oregon. A 1996 NAWQA study in this same basin showed that 21 of the 95 samples colleted had detectable levels of ethoprop. However, the maximum concentration reported was 0.44 μg/L.

Lower Clackamas River Basin: NAWQA Analysis

During 2000 - 2005, ultra low detection level analysis for 86 - 190 pesticides in 119 water samples collected from sites in the lower mainstem Clackamas River, its tributaries, and in pre- and post-treatment drinking-water from the study water-treatment plant. In all, 63 pesticide compounds: 33 herbicides, 15 insecticides, 6 fungicides, and 9 pesticide degradates were detected in samples collected during storm and nonstorm conditions. From 2000 - 2005, water samples were analyzed for azinphos methyl and its degradate, dimethoate, disulfoton and its degradates, ethoprop, fenamiphos and its degradates, methidathion, naled, phorate and its degradate, and phosmet. However, none of these compounds were detected. Fifty-seven pesticides or degradates were detected in the tributaries (mostly during storms), whereas fewer compounds (26) were detected in samples of source water from the lower mainstem Clackamas River, with fewest (15) occurring in drinking water. The two most commonly detected pesticides were the triazine herbicide simazine and atrazine, which occurred in abut one- half of samples. Deethylatrazine (a degradate of atrazine) commonly was detected along with atrazine in about 30% of samples. The a.i. in common household herbicides RoundUP (glyphosate) and Cross bow (triclopyr and 2,4-D) were also frequently detected together. These three herbicides often made up most of the total pesticide concentration in tributaries throughout the study area. According to Carpenter *et al.* (2008) some concentrations of insecticides (diazinon, chlorpyrifos, azinphos methyl, and *p',p'*-DDE) exceed EPA's aquatic life benchmarks in Carli, Sieben, Rock, Noyer, Doane, and North Fork Deep Creeks. One azinphos methyl concentration in Doane Creek (0.21 μg/L) exceeded federal and state of Oregon benchmarks for the protection of fish and benthic invertebrates.

Snake River Basin: NAWQA Analysis

The USGS conducted a water quality study from 1992 - 1995 in the upper Snake River basin, Idaho and Wyoming {Clark, 1998 #1504}. In basin wide stream sampling in May and June 1994, Eptam [EPTC] (used on potatoes, beans, and sugar beets), atrazine and its breakdown product desethylatrazine (used on corn), metolachlor (used on potatoes and beans), and alachlor (used on beans and corn) were the most commonly detected pesticides. These same compounds accounted for 75% of all detections. Seventeen different pesticides were detected downstream from American Falls Reservoir. Dissolved concentrations of 87 pesticides were detected and all detected concentrations were at less than 0.01 μg/L. They include azinphos methyl, ethoprop, and phorate. Disulfoton and methyl parathion were screened for but not detected. Carbaryl and carbofuran were each detected in only 1% of samples; methomyl was screened for but not detected {Clark, 1998 #1504}.

Hood River Basin

The Hood River Basin ranks fourth in the state of Oregon in total agricultural pesticide usage {Jenkins, 2004 #796}. The land in Hood River basin is used to grow five crops: alfalfa, apples, cherries, grapes, and pears. About 61 a.i.s, totaling 1.1 million lbs, are applied annually to roughly 21,000 acres. Of the top nine, three are carbamates and three are organophosphate insecticides (Table). These compounds will have a similar mode of action as the 12 a.i.s under consultation, but will have different toxicities.

Table 15. Amount of most common a.i.s applied to crops in Hood River Basin 1990-1996 (Jenkins et al. 2004).

Active Ingredient	Class	Lbs applied
Oil	-	624,392
Lime Sulfur	-	121,703
Mancozeb	Carbamate	86,872
Sulfur	-	60,552
Ziram	Carbamate	45,965
Azinphos methyl	Organo-phosphate	22,294
Metam-Sodium	Carbamate	17,114
Phosmet	Organo-phosphate	15,919
Chlorpyrifos	Organo-phosphate	14,833

The Hood River basin contains approximately 400 miles of perennial stream channel, of which an estimated 100 miles is accessible to anadromous fish. These channels are important rearing and spawning habitat for salmonids, making pesticide drift a major concern for the area.

Other Land Use in the Pacific Northwest Region

Urban and Industrial Development

The largest urban area in the basin is the greater Portland metropolitan area, located at the mouth of the Willamette River. Portland's population exceeds 500,000 {Hinck, 2004 #472}. Although the basin's land cover is about 8% of the U.S. total land mass, its human population is one-third the national average (about 1.2% of the U.S. population) {Hinck, 2004 #472}.

Discharges from sewage treatment plants, paper manufacturing, and chemical and metal production represent the top three permitted sources of contaminants within the lower basin according to discharge volumes and concentrations {Rosetta, 1996 #481}. Rosetta and Borys {, 1996 #256} review of 1993 data indicate that 52% of the point source waste water discharge volume is from sewage treatment plants, 39% from paper and allied products, 5% from chemical and allied products, and 3% from primary metals. However, the paper and allied products industry are the primary sources of the suspended sediment load (71%). Additionally, 26% of the point source waste water discharge volume comes from sewage treatment plants and 1% is from the chemical and allied products industry. Nonpoint source discharges (urban stormwater runoff) account for significant pollutant loading to the lower basin, including most organics and over half of the metals. Although rural nonpoint sources contributions were not calculated, Rosetta and Borys {, 1996 #256} surmised that in some areas and for some contaminants, rural areas may contribute a

large portion of the nonpoint source discharge. This is particularly true for pesticide contamination in the upper river basin where agriculture is the predominant land use.

Water quality has been reduced by phosphorus loads and decreased water clarity, primarily along the lower and middle sections of the Columbia River Estuary. Although sediment quality is generally very good, benthic indices have not been established within the estuary. Fish tissue contaminant loads (PCBs, DDT, DDD, DDE, and mercury) are high and present a persistent and long lasting effect on estuary biology. Health advisories have been recently issued for people eating fish in the area that contain high levels of dioxins, PCBs, and pesticides.

Habitat Modification

This section briefly describes how anthropogenic land use has altered aquatic habitat conditions for salmonids in the Pacific Northwest Region. Basin wide, critical ecological connectivity (mainstem to tributaries and riparian floodplains) has been disconnected by dams and associated activities such as floodplain deforestation and urbanization. Dams have flooded historical spawning and rearing habitat with the creation of massive water storage reservoirs. More than 55% of the Columbia River Basin that was accessible to salmon and steelhead before 1939 has been blocked by large dams {NWPPC, 1986 #513}. Construction of the Grand Coulee Dam blocked 1,000 miles (1,609 km) of habitat from migrating salmon and steelhead {Wydoski, 1979 #519}. Similarly, over one third (2,000 km) of coho salmon habitat is no longer accessible {Good, 2005 #574}. The mainstem habitats of the lower Columbia and Willamette rivers have been reduced primarily to a single channel. As a result, floodplain area is reduced, off-channel habitat features have been eliminated or disconnected from the main channel, and the amount of LWD in the mainstem has been reduced. Remaining areas are affected by flow fluctuations associated with reservoir management for power generation, flood control, and irrigation. Overbank flow events, important to habitat diversity, have become rare as a result of controlling peak flows and associated revetments. Portions of the basin are also subject to impacts from cattle grazing and irrigation withdrawals. Consequently, estuary dynamics have changed substantially.

Habitat loss has fragmented habitat and human density increase has created additional loads of pollutants and contaminants within the Columbia River Estuary {Anderson, 2007 #343}. About 77% of swamps, 57% of marshes, and over 20% of tree cover have been lost to development and industry. Twenty four threatened and endangered species occur in the estuary, some of which are recovering while others (*i.e.*, Chinook salmon) are not.

Stream habitat degradation in Columbia Central Plateau is relatively high {Williamson, 1998 #793}. In the most recent NAWQA survey, a total of 16 sites were evaluated - all of which showed signs of degradation {Williamson, 1998 #793}. Streams in this area have an average of 20% canopy cover and 70% bank erosion. These factors have severely affected the quality of habitat available to salmonids. The Palouse subunit of the Lower Snake River exceeds temperature levels for the protection of aquatic life {Williamson, 1998 #793}.

The Willamette Basin Valley has been dramatically changed by modern settlement. The complexity of the mainstem river and extent of riparian forest have both been reduced by

80% (PNERC 2002). About 75% of what was formerly prairie and 60% of what was wetland have been converted to agricultural purposes. These actions, combined with urban development, extensive (96 miles) bank stabilization, and in-river and nearshore gravel mining, have resulted in a loss of floodplain connectivity and off-channel habitat (PNERC 2002).

Habitat Restoration

Since 2000, land management practices included improving access by replacing culverts and fish habitat restoration activities at Federal Energy Regulatory Commission (FERC)-licensed dams. Habitat restoration in the upper (reducing excess sediment loads) and lower Grays River watersheds may benefit the Grays River chum salmon population as it has a sub-yearling juvenile life history type and rears in such habitats. Short-term daily flow fluctuations at Bonneville Dam sometimes create a barrier (*i.e.*, entrapment on shallow sand flats) for fry moving into the mainstem rearing and migration corridor. Some chum fry have been stranded on shallow water flats on Pierce Island from daily flow fluctuations. Coho salmon are likely to be affected by flow and sediment delivery changes in the Columbia River plume. Steelhead may be affected by flow and sediment delivery changes in the plume {Casillas, 1999 #689}.

In 2006, NOAA Fisheries completed consultation on issuance of a 50-year incidental take permit to the State of Washington for its Washington State Forest Practices Habitat Conservation Plan (HCP). The HCP is expected to improve habitat conditions on state forest lands within the action area. Improvements include removing barriers to migration, restoring hydrologic processes, increasing the number of large trees in riparian zones, improving stream bank integrity, and reducing fine sediment inputs {FCRPS, 2008 #658}.

Mining

Most of the mining in the basin is focused on minerals such as phosphate, limestone, dolomite, perlite, or metals such as gold, silver, copper, iron, and zinc. Mining in the region is conducted in a variety of methods and places within the basin. Alluvial or glacial deposits are often mined for gold or aggregate. Ores are often excavated from the hard bedrocks of the Idaho batholiths. Eleven percent of the nation's output of gold has come from mining operations in Washington, Montana, and Idaho. More than half of the nation's silver output has come from a few select silver deposits.

Many of the streams and river reaches in the basin are impaired from mining. Several abandoned and former mining sites are also designated as superfund cleanup areas {Stanford, 2005 #473;Anderson, 2007 #343}. According to the U.S. Bureau of Mines, there are about 14,000 inactive or abandoned mines within the Columbia River Basin. Of these, nearly 200 pose a potential hazard to the environment {Quigley, 1997 #495 in Hincke et al. 2004}. Contaminants detected in the water include lead and other trace metals.

Hydromodification Projects

More than 400 dams exist in the basin, ranging from mega dams that store large amounts of water to small diversion dams for irrigation. Every major tributary of the Columbia River except the Salmon River is totally or partially regulated by dams and diversions. More than 150 dams are major hydroelectric projects. Of these, 18 dams are located on the mainstem Columbia River and its major tributary, the Snake River. The FCRPS encompasses the operations of 14 major dams and reservoirs on the Columbia and Snake rivers. These dams and reservoirs operate as a coordinated system. The Corps operates 9 of 10 major federal projects on the Columbia and Snake rivers, and the Dworshak, Libby and Albeni Falls dams. The BOR operates the Grand Coulee and Hungry Horse dams. These federal projects are a major source of power in the region. These same projects provide flood control, navigation, recreation, fish and wildlife, municipal and industrial water supply, and irrigation benefits.

BOR has operated irrigation projects within the basin since 1904. The irrigation system delivers water to about 2.9 million acres of agricultural lands. About 1.1 million acres of land are irrigated using water delivered by two structures, the Columbia River Project (Grand Coulee Dam) and the Yakima Project. The Grand Coulee Dam delivers water for the irrigation of over 670,000 acres of croplands and the Yakima Project delivers water to nearly 500,000 acres of croplands {Bouldin, 2007 #31}.

The Bonneville Power Administration (BPA), an agency of the U.S. Department of Energy, wholesales electric power produced at 31 federal dams (67% of its production) and non-hydropower facilities in the Columbia-Snake Basin. The BPA sells about half the electric power consumed in the Pacific Northwest. The federal dams were developed over a 37-year period starting in 1938 with Bonneville Dam and Grand Coulee in 1941, and ending with construction of Libby Dam in 1973 and Lower Granite Dam in 1975.

Development of the Pacific Northwest regional hydroelectric power system, dating to the early 20th century, has had profound effects on the ecosystems of the Columbia River Basin {ISG, 1996 #483}. These effects have been especially adverse to the survival of anadromous salmonids. The construction of the FCRPS modified migratory habitat of adult and juvenile salmonids. In many cases, the FCRPS presented a complete barrier to habitat access for salmonids. Approximately 80% of historical spawning and rearing habitat of Snake River fall-run Chinook salmon is now inaccessible due to dams. The Snake River spring/summer run has been limited to the Salmon, Grande Ronde, Imnaha, and Tuscanon rivers. Damming has cut off access to the majority of Snake River Chinook salmon spawning habitat. The Sunbeam Dam on the Salmon River is believed to have limited the range of Snake River sockeye salmon as well.

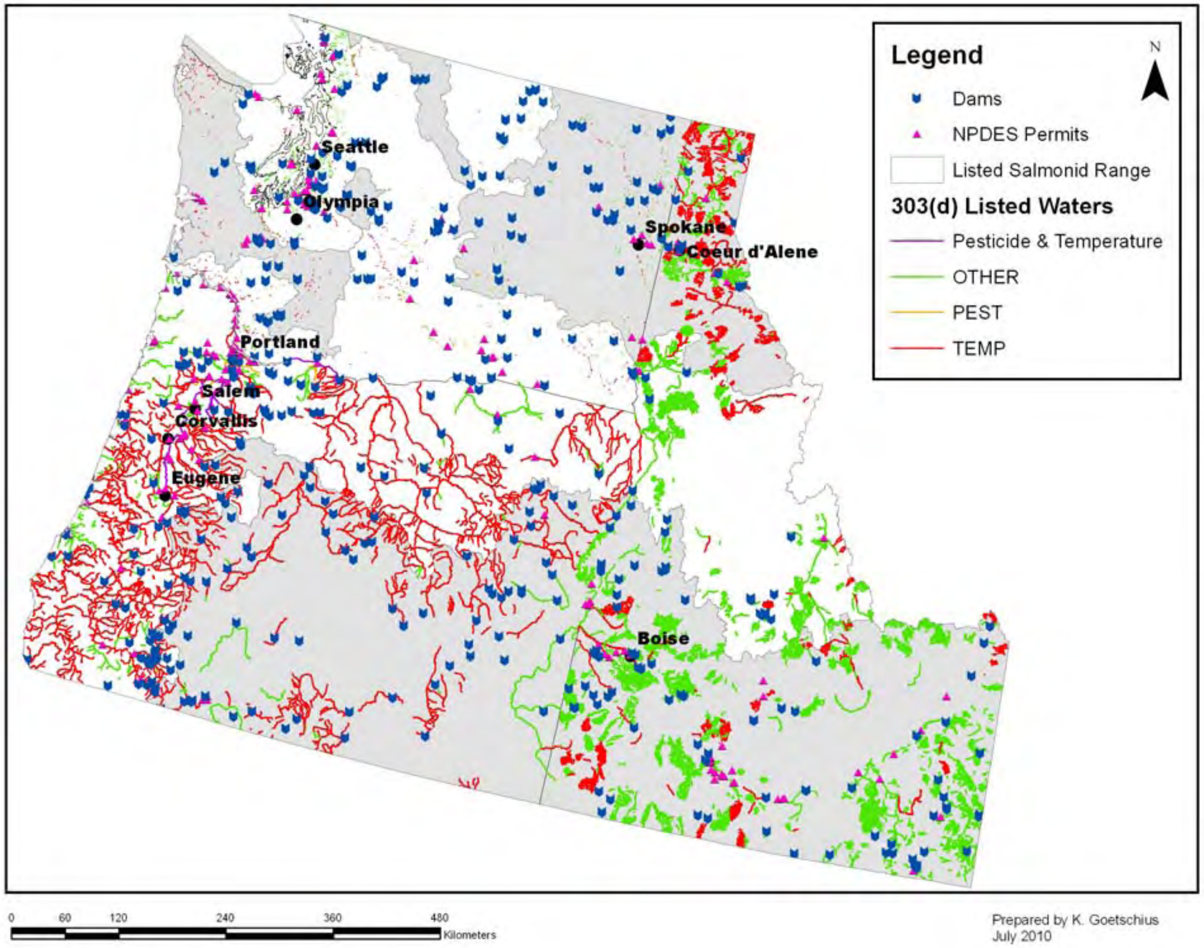
Both upstream and downstream migrating fish are impeded by the dams. Additionally, a substantial number of juvenile salmonids are killed and injured during downstream migrations. Physical injury and direct mortality occurs as juveniles pass through turbines, bypasses, and spillways. Indirect effects of passage through all routes may include disorientation, stress, delay in passage, exposure to high concentrations of dissolved gases, warm water, and increased predation. Non-federal hydropower facilities on Columbia River tributaries have also partially or completely blocked higher elevation spawning.

Qualitatively, several hydromodification projects have improved the productivity of naturally produced SR Fall-run Chinook salmon. Improvements include flow augmentation to enhance water flows through the lower Snake and Columbia Rivers [USBR 1998 *in* {FCRPS, 2008 #658}]; providing stable outflows at Hells Canyon Dam during the fall Chinook salmon spawning season and maintaining these flows as minimums throughout the incubation period to enhance survival of incubating fall-run Chinook salmon; and reduced summer temperatures and enhanced summer flow in the lower Snake River [see {Corps, 2007 #987}, Appendix 1 *in* {FCRPS, 2008 #658}]. Providing suitable water temperatures for over-summer rearing within the Snake River reservoirs allows the
expression of productive "yearling" life history strategy that was previously unavailable to SR Fall-run Chinook salmon.

The mainstem FCRPS corridor has also improved safe passage through the hydrosystem for juvenile steelhead and yearling Chinook salmon with the construction and operation of surface bypass routes at Lower Granite, Ice Harbor, and Bonneville dams and other configuration improvements {Corps, 2007 #987}.

For salmon, with a stream-type juvenile life history, projects that have protected or restored riparian areas and breached or lowered dikes and levees in the tidally influenced zone of the estuary have improved the function of the juvenile migration corridor. The FCRPS action agencies recently implemented 18 estuary habitat projects that removed passage barriers. These activities provide fish access to good quality habitat.
The Corps *et al.* {, 2007 #987} estimated that hydropower configuration and operational improvements implemented from 2000 to 2006 have resulted in an 11.3% increase in survival for yearling juvenile LCR Chinook salmon from populations that pass Bonneville Dam. Improvements during this period included the installation of a corner collector at Powerhouse II (PH2) and the partial installation of minimum gap runners at Powerhouse 1 (PH1) and of structures that improve fish guidance efficiency at PH2. Spill operations have been improved and PH2 is used as the first priority powerhouse for power production because bypass survival is higher than at PH1. Additionally, drawing

water towards PH2 moves fish toward the corner collector. The bypass system screen was removed from PH1 because tests showed that turbine survival was higher than through the bypass system at that location.



Legend
N
Dams
NPDES Permits
Listed Salmonid Range
303(d) Listed Waters
Pesticide & Temperature
OTHER
PEST
TEMP
Seattle
Olympia
Spokane
Coeur d'Alene
Portland
Salem
Corvallis
Eugene
Boise
0
60
120
240
360
480
Kilometers
Prepared by K. Goetschius
July 2010

Figure 36 Pacific Noethwest 303(d) waters, dams, and NPDES permit sites.

Artificial Propagation

There are several artificial propagation programs for salmon production within the Columbia River Basin. These programs were instituted under federal law to lessen the effects of lost natural salmon production within the basin from the dams. Federal, state, and tribal managers operate the hatcheries. For more than 100 years, hatcheries in the Pacific Northwest have been used to produce fish for harvest and replace natural production lost to dam construction. Hatcheries have only minimally been used to protect and rebuild naturally produced salmonid populations (*e.g.*, Redfish Lake sockeye salmon). In 1987, 95% of the coho salmon, 70% of the spring Chinook salmon, 80% of the summer Chinook salmon, 50% of the fall-run Chinook salmon, and 70% of the steelhead returning to the Columbia River Basin originated in hatcheries {CBFWA, 1990 #485}.
More recent estimates suggest that almost half of the total number of smolts produced in the basin come from hatcheries {Beechie, 2005 #328}.

The impact of artificial propagation on the total production of Pacific salmon and steelhead has been extensive {Hard, 1992 #487}. Hatchery practices, among other factors, are a contributing factor to the 90% reduction in natural coho salmon runs in the lower Columbia River over the past 30 years {Flagg, 1995 #488}. Past hatchery and stocking practices have resulted in the transplantation of salmon and steelhead from non-native basins. The impacts of these hatchery practices are largely unknown. Adverse effects of these practices likely included: loss of genetic variability within and among populations {Busack, 1990 #489;Hard, 1992 #487;Riggs, 1990 #490;Reisenbichler, 1997 #491}, disease transfer, increased competition for food, habitat, or mates, increased predation, altered migration, and the displacement of natural fish {Steward, 1990 #492;Hard, 1992 #487;Fresh, 1997 #493}. Species with extended freshwater residence may face higher risk of domestication, predation, or altered migration than species that spend only a brief time in freshwater {Hard, 1992 #487}. Nonetheless, artificial propagation may also contribute to the conservation of listed salmon and steelhead. However, it is unclear whether or how much artificial propagation during the recovery process will compromise the distinctiveness of natural populations {Hard, 1992 #487}.

The states of Oregon and Washington and other fisheries co-managers are engaged in a substantial review of hatchery management practices through the Hatchery Scientific Review Group (HSRG). The HSRG was established and funded by Congress to provide an independent review of current hatchery program in the Columbia River Basin. The HSRG has completed its work on Lower Columbia River populations and provided its recommendations. A general conclusion is that the current production programs are inconsistent with practices that reduce impacts on naturally-spawning populations, and will have to be modified to reduce adverse effects on key natural populations identified in the Interim Recovery Plan. The adverse effects are caused by hatchery-origin adults spawning with natural-origin fish or competing with natural-origin fish for spawning sites {FCRPS, 2008 #658}. Oregon and Washington initiated a comprehensive program of hatchery and associated harvest reforms {WDFW, 2005 #789;ODFW, 2007 #790}. The program is designed to achieve HSRG objectives related to controlling the number of hatchery-origin fish on the spawning grounds and in the hatchery broodstock.

Coho salmon hatchery programs in the lower Columbia have been tasked to compensate for impacts of fisheries. However, hatchery programs in the LCR have not operated specifically to conserve LCR coho salmon. These programs threaten the viability of natural populations. The long-term domestication of hatchery fish has eroded the fitness of these fish in the wild and has reduced the productivity of wild stocks where significant numbers of hatchery fish spawn with wild fish. Large numbers of hatchery fish have also contributed to more intensive mixed stock fisheries. These programs largely overexploited wild populations weakened by habitat degradation. Most LCR coho salmon populations have been heavily influenced by hatchery production over the years.

Commercial, Recreational, and Subsistence Fishing

Despite regulated fishing programs for salmonids, listed salmonids are also caught as bycatch. There are several approaches under the ESA to address tribal and state take of ESA-listed species that may occur as a result of harvest activities. Section 10 of the ESA provides for permits to operate fishery harvest programs. ESA section 4(d) rules provide exemptions from take for resource, harvest, and hatchery management plans. Furthermore, there are several treaties that have reserved the right of fishing to tribes in the North West Region.

Management of salmon fisheries in the Columbia River Basin is a cooperative process involving federal, state, and tribal representatives. The Pacific Fishery Management Council sets annual fisheries in federal waters from three to 200 miles off the coasts of Washington, Oregon, and California. Salmon and steelhead fisheries in the Columbia River and its tributaries are co-managed by the states of Washington, Oregon, Idaho, four treaty tribes, and other tribes that traditionally have fished in those waters. A federal court oversees Columbia River harvest management through the U.S. v. Oregon proceedings. Inland fisheries are those in waters within state boundaries, including those extending out three miles from the coasts. The states of Oregon, Idaho, and Washington issue salmon fishing licenses for theses areas.

Fisheries in the Columbia River basin are managed within the winter/spring, summer, and fall seasons. There are Treaty Indian and non-Treaty fisheries which are managed subject to state and tribal regulation, consistent with provisions of a U.S. v. Oregon 2008 agreement. The winter/spring season extends from January 1 to June 15. Commercial, recreational, and ceremonial subsistence fisheries target primarily upriver spring Chinook stocks and spring Chinook salmon that return to the Willamette and lower Columbia River tributaries. Some steelhead are also caught incidentally in these fisheries. The summer season extends from June 16 to July 31. Commercial, recreational, and ceremonial and subsistence fisheries are managed primarily to provide harvest opportunity directed at unlisted UCR summer Chinook salmon. Summer fisheries are constrained primarily by the available opportunity for UCR summer Chinook salmon, and by specific harvest rate limits for SR sockeye salmon and harvest rate limits on steelhead in non-Treaty fisheries. Fall season fisheries begin on August 1 and end on December 31. Commercial, recreational, and ceremonial and subsistence fisheries target primarily harvestable hatchery and natural origin fall Chinook and coho salmon. Fall season fisheries are constrained by specific ESA related harvest rate limits for listed SR fall Chinook salmon, and SR steelhead.

Treaty Indian fisheries are managed subject to the regulation of the Columbia River Treaty Tribes. They include all mainstem Columbia River fisheries between Bonneville Dam and McNary Dam, and any fishery impacts from tribal fishing that occurs below Bonneville Dam. Tribal fisheries within specified tributaries to the Columbia River are included.

Non-Treaty fisheries are managed under the jurisdiction of the states. These include mainstem Columbia River commercial and recreational salmonid fisheries at the river mouth of Bonneville Damn, designated off channel Select Area fisheries, mainstem recreational fisheries between Bonneville Dam and McNary Dam, recreational fisheries between McNary Dam and Highway 305 Bridge in Pasco, Washington, recreational and Wanapum tribal spring Chinook fisheries from McNary Dam to Priest Rapids Dam, and recreational spring Chinook fisheries in the Snake River upstream to Lower Granite Dam.

Archeological records indicate that indigenous people caught salmon in the Columbia River more than 7,000 years ago. One of the most well known tribal fishing sites within the basin was located near Celilo Falls, an area in the lower river that has been occupied by Dalles Dam since 1957. Salmon fishing increased with better fishing methods and preservation techniques, such as drying and smoking. Salmon harvest substantially increased in the mid-1800s with canning techniques. Harvest techniques also changed over time, from early use of hand-held spears and dip nets, to riverboats using seines and gill nets. Harvest techniques eventually transitioned to large ocean-going vessels with trolling gear and nets and the harvest of Columbia River salmon and steelhead from California to Alaska {Beechie, 2005 #328}.

During the mid-1800s, an estimated 10 to 16 million adult salmon of all species entered the Columbia River each year. Large annual harvests of returning adult salmon during the late 1800s ranging from 20 million to 40 million lbs of salmon and steelhead significantly reduced population productivity {Beechie, 2005 #328}. The largest known harvest of Chinook salmon occurred in 1883 when Columbia River canneries processed 43 million lbs of salmon {Lichatowich, 1999 #498}. Commercial landings declined steadily from the 1920s to a low in 1993. At that time, just over one million lbs of Chinook salmon were harvested {Beechie, 2005 #328}.

Harvested and spawning adults reached 2.8 million in the early 2000s, of which almost half are hatchery produced {Beechie, 2005 #328}. Most of the fish caught in the river are steelhead and spring/summer run Chinook salmon. Ocean harvest consists largely of coho and fall-run Chinook salmon. Most ocean catches are made north of Cape Falcon, Oregon. Over the past five years, the number of spring and fall salmon commercially harvested in tribal fisheries has averaged between 25,000 and 110,000 fish {Beechie, 2005 #328}. Recreational catch in both ocean and in-river fisheries varies from 140,000 to 150,000 individuals {Beechie, 2005 #328}.

Non-Indian fisheries in the lower Columbia River are limited to a harvest rate of 1%. Treaty Indian fisheries are limited to a harvest rate of 5 to 7%, depending on the run size of upriver Snake River sockeye stocks. Actual harvest rates over the last 10 years have ranged from 0 to 0.9%, and 2.8 to 6.1%, respectively [see TAC 2008, Table 15 *in* FCRPS {, 2008 #144}].

Columbia River chum salmon are not caught incidentally in tribal fisheries above Bonneville Dam. However, Columbia River chum salmon are incidentally caught occasionally in non-Indian fall season fisheries below Bonneville Dam. There are no fisheries in the Columbia River that target hatchery or natural-origin chum salmon. The species' later fall return timing make them vulnerable to relatively little potential harvest in fisheries that target Chinook salmon and coho salmon. CR chum salmon rarely take the sport gear used to target other species. Incidental catch of chum amounts to a few tens of fish per year (TAC 2008). The harvest rate of CR chum salmon in proposed state fisheries in the lower river is estimated to be 1.6% per year and is less than 5%.

LCR coho salmon are harvested in the ocean and in the Columbia River and tributary freshwater fisheries of Oregon and Washington. Incidental take of coho salmon prior to the 1990s fluctuated from approximately 60 to 90%. However, this number has been reduced since its listing to 15 to 25% {LCFRB, 2004 #648}. The exploitation of hatchery coho salmon has remained approximately 50% through the use of selective fisheries.

LCR steelhead are harvested in Columbia River and tributary freshwater fisheries of Oregon and Washington. Fishery impacts of LCR steelhead have been limited to less than 10% since implementation of mark-selective fisheries during the 1980s. Recent harvest rates on UCR steelhead in non-Treaty and treaty Indian fisheries ranged from 1% to 2%, and 4.1% to 12.4%, respectively {FCRPS, 2008 #658}.

Non-native Species

Many non-native species have been introduced to the Columbia River Basin since the 1880s. At least 81 non-native species have currently been identified, composing one-fifth of all species in some areas. New non-native species are discovered in the basin regularly; a new aquatic invertebrate is discovered approximately every 5 months {Sytsma, 2004 #786}. It is clear that the introduction of non-native species has changed the environment, though whether these changes will impact salmonid populations is uncertain {Sytsma, 2004 #786}.

Puget Sound Region

Puget Sound is the second largest estuary in the U.S. It has about 1,330 miles of shoreline and extends from the mouth of the Strait of Juan de Fuca east. Puget Sound includes the San Juan Islands and south to Olympia, and is fed by more than 10,000 rivers and streams.

Puget Sound is generally divided into four major geographic marine basins: Hood Canal, South Sound, Whidbey Basin, and the Main Basin. The Main Basin has been further subdivided into two subbasins: Admiralty Inlet and Central Basin. About 43% of the

Puget Sound's tideland is located in the Whidbey Island Basin. This reflects the large influence of the Skagit River, which is the largest river in the Puget Sound system and whose sediments are responsible for the extensive mudflats and tidelands of Skagit Bay.

Habitat types that occur within the nearshore environment include eelgrass meadows, kelp forest, mud flats, tidal marshes, sub-estuaries (tidally influenced portions of river and stream mouths), sand spits, beaches and backshore, banks and bluffs, and marine riparian vegetation. These habitats provide critical functions such as primary food production and support habitat for invertebrates, fish, birds, and other wildlife.

Major rivers draining to Puget Sound from the Cascade Mountains include the Skagit, Snohomish, Nooksack, Puyallup, and Green rivers, as well as the Lake Washington/Cedar River watershed. Major rivers from the Olympic Mountains include the Hamma Hamma, the Duckabush, the Quilcene, and the Skokomish rivers. Numerous other smaller rivers drain to the Sound, many of which are significant salmonid production areas despite their small size.

The Puget Sound basin is home to more than 200 fish and 140 mammalian species. Salmonids within the region include coho, Chinook, sockeye, chum, and pink salmon, kokanee, steelhead, rainbow, cutthroat, and bull trout {Wydoski, 1979 #519;Kruckeberg, 1991 #500}. Important commercial fishes include the five Pacific salmon and several rockfish species. A number of introduced species occur within the region, including brown and brook trout, Atlantic salmon, bass, tunicates (sea squirts), and a saltmarsh grass (*Spartina* spp.). Estimates suggest that over 90 species have been intentionally or accidentally introduced in the region {Ruckelshaus, 2007 #501}. At present, over 40 species in the region are listed as threatened and endangered under the ESA.

Puget Sound is unique among the nation's estuaries as it is a deep fjord-like structure that contains many urban areas within its drainage basin {Collier, 2006 #520}. Because several sills limit entry of oceanic water into Puget Sound, it is relatively poorly flushed compared to other urbanized estuaries of North America. Thus, toxic chemicals that enter Puget Sound have longer residence times within the system. This entrainment of toxics can result in biota exposure to increased levels of contaminant for a given input, compared to other large estuaries. This hydrologic isolation puts the Puget Sound ecosystem at higher risk from other types of populations that enter the system, such as nutrients and pathogens.

Because Puget Sound is a deep, almost oceanic habitat, the tendency of a number of species to migrate outside of Puget Sound is limited relative to similar species in other large urban estuaries. This high degree of residency for many marine species, combined with the poor flushing of Puget Sound, results in a more protracted exposure to contaminants. The combination of hydrologic and biological isolation makes the Puget Sound ecosystem highly susceptible to inputs of toxic chemicals compared to other major estuarine ecosystems {Collier, 2006 #520}.

An indication of this sensitivity occurs in Pacific herring, one of Puget Sound's keystone forage fish species {Collier, 2006 #520}. These fish spend almost all of their lives in pelagic waters and feed at the lower end of the food chain. Pacific herring should be among the least contaminated of fish species. However, monitoring has shown that herring from the main basins of Puget Sound have higher body burdens of persistent chemicals (*e.g.*, PCBs) compared to herring from the severely contaminated Baltic Sea. Thus, the pelagic food web of Puget Sound appears to be more seriously contaminated than previously anticipated.

Chinook salmon that are resident in Puget Sound (a result of hatchery practices and natural migration patterns) are several times more contaminated with persistent bioaccumulative contaminants than other salmon populations along the West Coast {Collier, 2006 #520}. Because of associated human health concerns, fish consumption guidelines for Puget Sound salmon are under review by the Washington State Department of Health.

Extremely high levels of chemical contaminants are also found in Puget Sound's top predators, including harbor seals and ESA-listed southern resident killer whales {Collier, 2006 #520}. In addition to carrying elevated loads of toxic chemicals in their tissues, Puget Sound's biota also show a wide range of adverse health outcomes associated with exposure to chemical contaminants. They include widespread cancer and reproductive impairment in bottom fish, increased susceptibility to disease in juvenile salmon, acute die-offs of adult salmon returning to spawn in urban watersheds, and egg and larval mortality in a variety of fish. Given current regional projections for population growth and coastal development, the loadings of chemical contaminants into Puget Sound will increase dramatically in future years.

Land Use

The Puget Sound Lowland contains the most densely populated area of Washington. The regional population in 2003 was an estimated 3.8 million people, with 86% residing in King, Pierce, and Snohomish counties (Snohomish, Cedar-Sammamish Basin, Green-Duwamish, and Puyallup River watersheds). The area is expected to attract 4 to 6 million new human residents in the next 20 years {Ruckelshaus, 2007 #501}. The Snohomish River watershed, one of the fastest growing watersheds in the region, increased about 16% in the same period.

Land use in the Puget Sound lowland is composed of agricultural areas (including forests for timber production), urban areas (industrial and residential use), and rural areas (low density residential with some agricultural activity). Pesticides are regularly applied to agricultural and non-agricultural lands and are found virtually in every land use area. Pesticides and other contaminants drain into ditches in agricultural areas and eventually to stream systems. Roads bring surface water runoff to stream systems from industrial, residential, and landscaped areas in the urban environment. Pesticides are also typically found in the right-of-ways of infrastructure that connect the major landscape types. Right-of-ways are associated with roads, railways, utility lines, and pipelines.

In the 1930s, all of western Washington contained about 15.5 million acres of "harvestable" forestland. By 2004, the total acreage was nearly half that originally surveyed {PSAT, 2007 #508}. Forest cover in Puget Sound alone was about 5.4 million acres in the early 1990s. About a decade later, the region had lost another 200,000 acres of forest cover with some watersheds losing more than half the total forested acreage. The most intensive loss of forest cover occurred in the Urban Growth Boundary, which encompasses specific parts of the Puget Lowland. In this area, forest cover declined by 11% between 1991 and 1999 {Ruckelshaus, 2007 #501}. Projected land cover changes indicate that trends are likely to continue over the next several decades with population changes {Ruckelshaus, 2007 #501}. Coniferous forests are also projected to decline at an alarming rate as urban uses increase.

According to the 2001 State of the Sound report {PSAT, 2007 #508}, impervious surfaces covered 3.3% of the region, with 7.3% of lowland areas (below 1,000 ft elevation) covered by impervious surfaces. From 1991 to 2001, the amount of impervious surfaces increased 10.4% region wide. Consequently, changes in rainfall delivery to streams alter stream flow regimes. Peak flows are increased and subsequent base flows are decreased and alter in-stream habitat. Stream channels are widened and deepened and riparian vegetation is typically removed which can cause increases in water temperature and will reduce the amounts of woody debris and organic matter to the stream system.

Pollutants carried into streams from urban runoff include pesticides, heavy metals, PCBs, polybrominated diphenyl ethers (PBDEs) compounds, PAHs, nutrients (phosphorus and nitrogen), and sediment (Table 68). Other ions generally elevated in urban streams include calcium, sodium, potassium, magnesium, and chloride ions where sodium chloride is used as the principal road deicing salt {Paul, 2001 #729}. The combined effect of increased concentrations of ions in streams is the elevated conductivity observed in most urban streams.

Table 68. Examples of Water Quality Contaminants in Residential and Urban Areas

Contaminant groups	Select constituents	Select example(s)	Source and Use Information
Fertilizers	Nutrients	Phosphorus Nitrogen	lawns, golf courses, urban landscaping
Heavy Metals	Pb, Zn, Cr, Cu, Cd, Ni, Hg, Mg	Cu	brake pad dust, highway and parking lot runoff, rooftops
Pesticides including- Insecticides (I) Herbicides (H) Fungicides (F) Wood Treatment chemicals (WT) Legacy Pesticides (LP) Other ingredients in pesticide formulations (OI)	Organophosphates (I) Carbamates (I) Organochlorines (I) Pyrethroids (I) Triazines (H) Chloroacetanilides (H) Chlorophenoxy acids (H) Triazoles (F) Copper containing fungicides (F) Organochlorines (LP) Surfactants/adjuvants (OI)	Chlorpyrifos (I) Diazinon (I) Carbaryl (I) Atrazine (H) Esfenvalerate (I) Creosote (WT) DDT (LP) Copper sulfate (F) Metalaxyl (F) Nonylphenol (OI)	golf courses, right of ways, lawn and plant care products, pilings, bulkheads, fences
Pharmaceuticals and personal care products	Natural and synthetic hormones soaps and detergents	Ethinyl estradiol Nonylphenol	hospitals, dental facilities, residences, municipal and industrial waste water discharges
Polyaromatic hydrocarbons (PAHs)	Tricyclic PAHs	Phenanthrene	fossil fuel combustion, oil and gasoline leaks, highway runoff, creosote-treated wood
Industrial chemicals	PCBs PBDEs Dioxins	Penta-PBDE	utility infrastructure, flame retardants, electronic equipment

Many other metals have been found in elevated concentrations in urban stream sediments including arsenic, iron, boron, cobalt, silver, strontium, rubidium, antimony, scandium, molybdenum, lithium, and tin {Wheeler, 2005 #522}. The concentration, storage, and transport of metals in urban streams are connected to particulate organic matter content and sediment characteristics. Organic matter has a high binding capacity for metals and both bed and suspended sediments with high organic matter content frequently exhibit 50 - 7,500 times higher concentrations of zinc, lead, chromium, copper, mercury, and cadmium than sediments with lower organic matter content.

Although urban areas occupy only 2% of the Pacific Northwest land base, the impacts of urbanization on aquatic ecosystems are severe and long lasting {Spence, 1996 #523}. O'Neill *et al.* {, 2006 #171} found that Chinook salmon returning to Puget Sound had significantly higher concentrations of PCBs and PBDEs compared to other Pacific coast salmon populations. Furthermore, Chinook salmon that resided in Puget Sound in the winter rather than migrate to the Pacific Ocean (residents) had the highest concentrations of persistent organic pollutants (POPs), followed by Puget Sound fish populations believed to be more ocean-reared. Fall-run Chinook salmon from Puget Sound have a more localized marine distribution in Puget Sound and the Georgia Basin than other populations of Chinook salmon from the west coast of North America. This ESU is more contaminated with PCBs (2 to 6 times) and PBDEs (5 to 17 times). O'Neill *et al.* {, 2006 #171} concluded that regional body burdens of contaminants in Pacific salmon, and Chinook salmon in particular, could contribute to the higher levels of contaminants in federally-listed endangered southern resident killer whales.

Endocrine disrupting compounds are chemicals that mimic natural hormones, inhibit the action of hormones and/or alter normal regulatory functions of the immune, nervous and endocrine systems and can be discharged with treated effluent {King County, 2002 #525}. Endocrine disruption has been attributed to DDT and other organochlorine pesticides, dioxins, PAHs, alkylphenolic compounds, phthalate plasticizers, naturally occurring compounds, synthetic hormones and metals. Natural mammalian hormones such as 17β-estradiol, are also classified as endocrine disruptors. Both natural and synthetic mammalian hormones are excreted through the urine and are known to be present in wastewater discharges.

Jobling *et al.* {, 1995 #555} reported that ten chemicals known to occur in sewage effluent interacted with the fish estrogen receptor by reducing binding of 17β-estradiol to its receptor, stimulating transcriptional activity of the estrogen receptor or inhibiting transcription activity. Binding of the ten chemicals with the fish endocrine receptor indicates that the chemicals could be endocrine disruptors and forms the basis of concern about WWTP effluent and fish endocrine disruption.

Fish communities are impacted by urbanization {Wheeler, 2005 #522}. Urban stream fish communities have lower overall abundance, diversity, taxa richness and are dominated by pollution tolerant species. Lead content in fish tissue is higher in urban areas. Furthermore, the proximity of urban streams to humans increases the risk of non-native species introduction and establishment. Thirty-nine non-native species were collected in Puget Sound during the 1998 Puget Sound Expedition Rapid Assessment Survey {Brennan, 2004 #627}. Lake Washington, located within a highly urban area, has 15 non-native species identified {Ajawani, 1956 #699}.

PAH compounds also have distinct and specific effects on fish at early life history stages {Incardona, 2004 #528}. PAHs tend to adsorb to organic or inorganic matter in sediments, where they can be trapped in long-term reservoirs {Johnson, 2002 #529}. Only a portion of sediment-adsorbed PAHs are readily bioavailable to marine organisms, but there is substantial uptake of these compounds by resident benthic fish through the diet, through exposure to contaminated water in the benthic boundary layer, and through direct contact with sediment. Benthic invertebrate prey are a particularly important source of PAH exposure for marine fishes, as PAHs are bioaccumulated in many invertebrate
species {Varanasi, 1992 #531;Varanasi, 1989 #530;Meador, 1995 #532}.

PAHs and their metabolites in invertebrate prey can be passed on to consuming fish species, PAHs are metabolized extensively in vertebrates, including fishes {Johnson, 2002 #529}. Although PAHs do not bioaccumulate in vertebrate tissues, PAHs cause a variety of deleterious effects in exposed animals. Some PAHs are known to be immunotoxic and to have adverse effects on reproduction and development. Studies show that PAHs exhibit many of the same toxic effects in fish as they do in mammals {Johnson, 2002 #529}.

Habitat Modification

Much of the estuarine wetlands in Puget Sound have been heavily modified, primarily from agricultural land conversion and urban development {NRC, 1996 #470}. Although most estuarine wetland losses result from conversions to agricultural land by ditching, draining, or diking, these wetlands also experience increasing effects from industrial and urban causes. By 1980, an estimated 27,180 acres of intertidal or shore wetlands had been lost at 11 deltas in Puget Sound {Bortleson, 1980 #506}. Tidal wetlands in Puget Sound amount to roughly 18% of their historical extent {Collins, 2005 #507}. Coastal marshes close to seaports and population centers have been especially vulnerable to conversion with losses of 50 - 90%. By 1980, an estimated 27,180 acres of intertidal or shore wetlands had been lost at eleven deltas in Puget Sound {Bortleson, 1980 #506}. More recently, tidal wetlands in Puget Sound amount to about 17 - 19% of their historical extent {Collins, 2005 #507}. Coastal marshes close to seaports and population centers have been especially vulnerable to conversion with losses of 50 - 90% common for individual estuaries. Salmon use freshwater and estuarine wetlands for physiological transition to and from salt water and rearing habitat. The land conversions and losses of Pacific Northwest wetlands constitute a major impact. Salmon use marine nearshore areas for rearing and migration, with juveniles using shallow shoreline habitats {Brennan, 2004 #627}.

About 800 miles of Puget Sound's shorelines are hardened or dredged {PSAT, 2004 #502;Ruckelshaus, 2007 #501}. The area most intensely modified is the urban corridor (eastern shores of Puget Sound from Mukilteo to Tacoma). Here, nearly 80% of the shoreline has been altered, mostly from shoreline armoring associated with the Burlington Northern Railroad tracks {Ruckelshaus, 2007 #501}. Levee development within the rivers and their deltas has isolated significant portions of former floodplain habitat that was historically used by salmon and trout during rising flood waters.

Urbanization has caused direct loss of riparian vegetation and soils and has significantly altered hydrologic and erosion rates. Watershed development and associated urbanization throughout the Puget Sound, Hood Canal, and Strait of Juan de Fuca regions have increased sedimentation, raised water temperatures, decreased LWD recruitment, decreased gravel recruitment, reduced river pools and spawning areas, and dredged and filled estuarine rearing areas (Bishop and Morgan 1996 in {NMFS, 2008 #975}). Large areas of the lower rivers have been channelized and diked for flood control and to protect agricultural, industrial, and residential development.

The principal factor for decline of Puget Sound steelhead is the destruction, modification, and curtailment of its habitat and range. Barriers to fish passage and adverse effects on water quality and quantity resulting from dams, the loss of wetland and riparian habitats, and agricultural and urban development activities have contributed and continue to contribute to the loss and degradation of steelhead habitats in Puget Sound {NMFS, 2008 #975}.

Industrial Development

More than 100 years of industrial pollution and urban development have affected water quality and sediments in Puget Sound. Many different kinds of activities and substances release contamination into Puget Sound and the contributing waters. According to the State of the Sound Report {PSAT, 2007 #508} in 2004, more than 1,400 fresh and marine waters in the region were listed as "impaired." Almost two-thirds of these water bodies were listed as impaired due to contaminants, such as toxics, pathogens, and low dissolved oxygen or high temperatures, and less than one-third had established cleanup plans. More than 5,000 acres of submerged lands (primarily in urban areas; 1% of the study area) are contaminated with high levels of toxic substances, including polybrominated diphenyl ethers (PBDEs; flame retardants), and roughly one-third (180,000 acres) of submerged lands within Puget Sound are considered moderately contaminated. In 2005 the Puget Sound Action Team (PSAT) identified the primary pollutants of concern in Puget Sound and their sources listed below in Table 69.

Table 69. Pollutants of Concern in Puget Sound {PSAT, 2005 #701}

Pollutant	Sources
Heavy Metals: Pb, Hg, Cu, and others	vehicles, batteries, paints, dyes, stormwater runoff, spills, pipes.
Organic Compounds: Polycyclic aromatic hydrocarbons (PAHs)	Burning of petroleum, coal, oil spills, leaking underground fuel tanks, creosote, asphalt.
Polychlorinated biphenyls (PCBs)	Solvents electrical coolants and lubricants, pesticides, herbicides, treated wood.
Dioxins, Furans	Byproducts of industrial processes.
Dichloro-diphenyl-trichloroethane (DDTs)	Chlorinated pesticides.
Phthalates	Plastic materials, soaps, and other personal care products. Many of these compounds are in wastewater from sewage treatment plants.
Polybrominated diphenyl ethers (PBDEs)	PBDEs are added to a wide range of textiles and plastics as a flame retardant. They easily leach from these materials and have been found throughout the environment and in human breast milk.

Puget Sound Basin: NAWQA Analysis

The USGS sampled waters in the Puget Sound Basin between 1996 and 1998. {Ebbert, 2000 #505} reported that 26 of 47 analyzed pesticides were detected. A total of 74 manmade organic chemicals were detected in streams and rivers, with different mixtures of chemicals linked to agricultural and urban settings. NAWQA results reported that the herbicides atrazine, prometon, simazine and tebuthiuron were the most frequently detected herbicides in surface and ground water {Bortleson, 2000 #505}. Herbicides were the most common type of pesticide found in an agricultural stream (Fishtrap Creek) and the only type of pesticide found in shallow ground water underlying agricultural land {Bortleson, 2000 #505}. The most commonly detected VOC in the agricultural land use study area was associated with the application of fumigants to soils prior to planting {Bortleson, 2000 #505}. One or more fumigant-related compounds (1,2-dichloropropane, 1,2,2-trichloropropane, and 1,2,3-trichloropropane) were detected in over half of the samples. Insecticides, in addition to herbicides, were detected frequently in urban streams {Bortleson, 2000 #505}. Sampled urban streams showed the highest detection rate for the three insecticides: carbaryl, diazinon, and malathion. Carbaryl was detected at over 60% of urban sample sites {Ebbert, 2000 #505}. The insecticide diazinon was also frequently detected in urban streams at concentrations that exceeded

EPA guidelines for protecting aquatic life {Bortleson, 2000 #505}. Insecticides screened for included both carbofuran and methomyl. Carbofuran was detected, while methomyl was not. No insecticides were found in shallow ground water below urban residential land {Bortleson, 2000 #505}. Ethoprop was detected in agricultural streams. Concentrations of azinphos methyl, disulfoton, methyl parathion, and phorate were tested for but not detected.

Habitat Restoration

Positive changes in water quality in the region are evident. One of the most notable improvements was the elimination of sewage effluent to Lake Washington in the mid-1960s. This significantly reduced problems within the lake from phosphorus pollution and triggered a concomitant reduction in cyanobacteria {Ruckelshaus, 2007 #501}. Even so, as the population and industry has risen in the region a number of new and legacy pollutants are of concern.

Mining

Mining has a long history in Washington. In 2004, the state was ranked 13th nationally in total nonfuel mineral production value and 17th in coal production {Palmisano, 1993 #511;NMA, 2007 #464}. Metal mining for all metals (zinc, copper, lead, silver, and gold) peaked between 1940 and 1970 {Palmisano, 1993 #511}. Today, construction sand and gravel, Portland cement, and crushed stone are the predominant materials mined. Where sand and gravel is mined from riverbeds (gravel bars and floodplains) it may result in changes in channel elevations and patterns, instream sediment loads, and seriously alter instream habitat. In some cases, instream or floodplain mining has resulted in large scale river avulsions. The effect of mining in a stream or reach depends upon the rate of harvest and the natural rate of replenishment, as well as flood and precipitation conditions during or after the mining operations.

Artificial Propagation

The artificial propagation of late-returning Chinook salmon is widespread throughout Puget Sound {Good, 2005 #574}. Summer/fall Chinook salmon transfers between watersheds within and outside the region have been commonplace throughout this century. Therefore, the purity of naturally spawning stocks varies from river to river. Nearly 2 billion Chinook salmon have been released into Puget Sound tributaries since the 1950s. The vast majority of these have been derived from local late-returning adults. Returns to hatcheries have accounted for 57% of the total spawning escapement. However, the hatchery contribution to spawner escapement is probably much higher than that due to hatchery-derived strays on the spawning grounds. The genetic similarity between Green River late-returning Chinook salmon and several other late-returning Chinook salmon in Puget Sound suggests that there may have been a significant and lasting effect from some hatchery transplants {Marshall, 1995 #727}.

Overall, the use of Green River stock throughout much of the extensive hatchery network in this ESU may reduce the genetic diversity and fitness of naturally spawning populations {Good, 2005 #574}.

Hydromodification Projects

More than 20 dams occur within the region's rivers and overlap with the distribution of salmonids. A number of basins contain water withdrawal projects or small impoundments that can impede migrating salmon. The resultant impact of these and land use changes (forest cover loss and impervious surface increases) has been a significant modification in the seasonal flow patterns of area rivers and streams, and the volume and quality of water delivered to Puget Sound waters. Several rivers have been modified by other means including levees and revetments, bank hardening for erosion control, and agriculture uses. Since the first dike on the Skagit River delta was built in 1863 for agricultural development {Ruckelshaus, 2007 #501}, other basins like the Snohomish River are diked and have active drainage systems to drain water after high flows that top the dikes. Dams were also built on the Cedar, Nisqually, White, Elwha, Skokomish, Skagit, and several other rivers in the early 1900s to supply urban areas with water, prevent downstream flooding, allow for floodplain activities (like agriculture or development), and to power local timber mills {Ruckelshaus, 2007 #501}.

Over the next few years, however, a highly publicized and long discussed dam removal project is expected to begin in the Elwha River. The removal of two dams in the Elwha River, a short but formerly very productive salmon river, is expected to open up more than 70 miles of high quality salmon habitat {Ruckelshaus, 2007 #501;Wunderlich, 1994 #510}. Estimates suggest that nearly 400,000 salmon could begin using the basin within 30 years after the dams are removed {PSAT, 2007 #508}.

In 1990, only one-third of the water withdrawn in the Pacific Northwest was returned to the streams and lakes {NRC, 1996 #470}. Water that returns to a stream from an agricultural irrigation is often substantially degraded. Problems associated with return flows include increased water temperature, which can alter patterns of adult and smolt migration; increased toxicant concentrations associated with pesticides and fertilizers; increased salinity; increased pathogen populations; decreased dissolved oxygen concentration; and increased sedimentation {NRC, 1996 #470}. Water-level fluctuations and flow alterations due to water storage and withdrawal can affect substrate availability and quality, temperature, and other habitat requirements of salmon. Indirect effects include reduction of food sources; loss of spawning, rearing, and adult habitat; increased susceptibility of juveniles to predation; delay in adult spawning migration; increased egg and alevin mortalities; stranding of fry; and delays in downstream migration of smolts {NRC, 1996 #470}.

Commercial and Recreational Fishing

Despite regulated fishing programs for salmonids, listed salmonids are also caught as bycatch. There are several approaches under the ESA to address tribal and state take of ESA-listed species that may occur as a result of harvest activities. Section 10 of the ESA provides for permits to operate fishery harvest programs. ESA section 4(d) rules provide exemptions from take for resource, harvest, and hatchery management plans. Furthermore, there are several treaties that have reserved the right of fishing to tribes in the North West Region.

Management of salmon fisheries in the Puget Sound Region is a cooperative process involving federal, state, tribal, and Canadian representatives. The Pacific Fishery

Management Council sets annual fisheries in federal waters from three to 200 miles off the coasts of Washington, Oregon, and California. The annual North of Falcon process sets salmon fishing seasons in waters such as Puget Sound, Willapa Bay, Grays Harbor, and Washington State rivers. Inland fisheries are those in waters within state boundaries, including those extending out three miles from the coasts. The states of Oregon, Idaho, and Washington issue salmon fishing licenses for theses areas. Adult salmon returning to Washington migrate through both U.S. and Canadian waters and are harvested by fishermen from both countries. The 1985 Pacific Salmon Treaty helps fulfill conservation goals for all members and is implemented by the eight-member bilateral Pacific Salmon Commission. The Commission does not regulate salmon fisheries, but provides regulatory advice.

Most of the commercial landings in the region are groundfish, Dungeness crab, shrimp, and salmon. Many of the same species are sought by Tribal fisheries and by charter and recreational anglers. Nets and trolling are used in commercial and Tribal fisheries. Recreational anglers typically use hook and line, and may fish from boat, river bank, or docks. Entanglement of marine mammals in fishing gear is not uncommon and can lead to mortality or serious injury.

Harvest impacts on Puget Sound Chinook salmon populations average 75% in the earliest five years of data availability and have dropped to an average of 44% in the most recent five-year period {Good, 2005 #574}. Populations in Puget Sound have not experienced the strong increases in numbers seen in the late 1990s in many other ESUs. Although more populations have increased than decreased since the last BRT assessment, after adjusting for changes in harvest rates, trends in productivity are less favorable. Most populations are relatively small, and recent abundance within the ESU is only a small fraction of estimated historic run size.

Oregon-Washington-Northern California Coastal Drainages

This region encompasses drainages originating in the Klamath Mountains, the Oregon Coast Mountains, and the Olympic Mountains. More than 15 watersheds drain the region's steep slopes including the Umpqua, Alsea, Yaquina, Nehalem, Chehalis, Quillayute, Queets, and Hoh rivers. Numerous other small to moderately sized streams dot the coastline. Many of the basins in this region are relatively small. The Umpqua River drains a basin of 4,685 square miles and is slightly over 110 miles long. The Nehalem River drains a basin of 855 square miles and is almost 120 miles long. However, systems here represent some of the most biologically diverse basins in the Pacific Northwest {Belitz, 2004 #456;Kagan, 1999 #513;Carter, 2005 #457}.

Land Use

The rugged topography of the western Olympic Peninsula and the Oregon Coastal Range has limited the development of dense population centers. For instance, the Nehalem River and the Umpqua River basins consist of less than 1% urban land uses. Most basins in this region have long been exploited for timber production, and are still dominated by forest lands. In Washington State, roughly 90% of the coastal region is forested {Palmisano, 1993 #511}. Roughly 80% of the Oregon Coastal Range is forested as well {Gregory, 2000 #980}. Approximately 92% of the Nehalem River basin is forested, with only 4% considered agricultural {Belitz, 2004 #456}. Similarly, in the Umpqua River basin, about 86% is forested land, 5% agriculture, and 0.5% is considered urban lands. Roughly half the basin is under federal management {Carter, 2005 #457}.

Habitat Modification

While much of the coastal region is forested, it has still been impacted by land use practices. Less than 3% of the Oregon coastal forest is old growth conifers {Gregory, 2000 #980}. The lack of mature conifers indicates high levels of habitat modification. As such, overall salmonid habitat quality is poor, though it varies by watershed. The amount of remaining high quality habitat ranges from 0% in the Sixes to 74% in the Siltcoos {ODFW, 2005 #979}. Approximately 14% of freshwater winter habitat available to juvenile coho is of high quality. Much of the winter habitat is unsuitable due to high temperatures. For example, 77% of coho salmon habitat in the Umpqua basin exceeds temperature standards.

Reduction in stream complexity is the most significant limiting factor in the Oregon coastal region. An analysis of the Oregon coastal range determined the primary and secondary life cycle bottlenecks for the 21 populations of coastal coho salmon {Nicholas, 2005 #983}. Nicholas *et al.* {, 2005 #983} determined that stream complexity is either the primary (13) or secondary (7) bottleneck for every population. Stream complexity has been reduced through past practices such as splash damming, removing riparian vegetation, removing LWD, diking tidelands, filling floodplains, and channelizing rivers.

Habitat loss through wetland fills is also a significant factor. Table 18 summarizes the change in area of tidal wetlands for several Oregon estuaries {Good, 2000 #982}.

Table 70. Change in total area (acres^2) of tidal wetlands in Oregon (tidal marshes and swamps) due to filling and diking between 1870 and 1970 {Good, 2000 #982}.

Estuary	Diked or Filled Tidal Wetland	Percent of 1870 Habitat Lost
Necanicum	15	10
Nehalem	1,571	75
Tillamook	3,274	79
Netarts	16	7
Sand Lake	9	2
Nestucca	2,160	91
Salmon	313	57
Siletz	401	59
Yaquina	1,493	71
Alsea	665	59
Siuslaw	1,256	63
Umpqua	1,218	50
Coos Bay	3,360	66
Coquille	4,600	94
Rogue	30	41
Chetco	5	56
Total	20,386	72%

The only listed salmonid population in coastal Washington is the Ozette Lake sockeye. The range of this ESU is small, including only one lake (31 km^2) and 71 km of stream. Like the Oregon Coastal drainages, the Ozette Lake area has been heavily managed for logging. Logging resulted in road building and the removal of LWD, which affected the nearshore ecosystem {NMFS, 2008 #632}. LWD along the shore offered both shelter from predators and a barrier to encroaching vegetation {NMFS, 2008 #632}. Aerial photograph analysis shows near-shore vegetation has increased significantly over the past 50 years {Ritchie, 2005 #984}. Further, there is strong evidence that water levels in Ozette Lake have dropped between 1.5 and 3.3 ft from historic levels [Herrera 2005 *in* {NMFS, 2008 #632}]. The impact of this water level drop is unknown. Possible effects include increased desiccation of sockeye redds and loss of spawning habitat. Loss of LWD has also contributed to an increase in silt deposition, which impairs the quality and quantity of spawning habitat. Very little is known about the relative health of the Ozette Lake tributaries and their impact on the sockeye salmon population.

Mining

Oregon is ranked 35th nationally in total nonfuel mineral production value in 2004. In that same year, Washington was ranked 13th nationally in total nonfuel mineral production value and 17th in coal production {Palmisano, 1993 #511;NMA, 2007 #464}. Metal mining for all metals (*e.g.*, zinc, copper, lead, silver, and gold) peaked in Washington between 1940 and 1970 {Palmisano, 1993 #511}. Today, construction sand, gravel, Portland cement, and crushed stone are the predominant materials mined in both Oregon
and Washington. Where sand and gravel is mined from riverbeds (gravel bars and floodplains) changes in channel elevations and patterns, and also changes in instream sediment loads, may result and alter instream habitat. In some cases, instream or floodplain mining has resulted in large scale river avulsions. The effect of mining in a stream or reach depends upon the rate of harvest and the natural rate of replenishment. Additionally, the severity of the effects is influenced by flood and precipitation conditions during or after the mining operations.

Hydromodification Projects

Compared to other areas in the greater Northwest Region, the coastal region has fewer dams and several rivers remain free flowing (*e.g.*, Clearwater River). The Umpqua River is fragmented by 64 dams, the fewest number of dams on any large river basin in Oregon {Carter, 2005 #457}. According to Palmisano *et al.* {, 1993 #565} dams in the coastal streams of Washington permanently block only about 30 miles of salmon habitat. In the past, temporary splash dams were constructed throughout the region to transport logs out of mountainous reaches. The general practice involved building a temporary dam in the creek adjacent to the area being logged, and filling the pond with logs. When the dam broke the floodwater would carry the logs to downstream reaches where they could be rafted and moved to market or downstream mills. Thousands of splash dams were constructed across the Northwest in the late 1800s and early 1900s. While the dams typically only temporarily blocked salmon habitat, in some cases dams remained long enough to wipe out entire salmon runs. The effects of the channel scouring and loss of channel complexity resulted in the long-term loss of salmon habitat {NRC, 1996 #470}.

Commercial and Recreational Fishing

Despite regulated fishing programs for salmonids, listed salmonids are also caught as bycatch. There are several approaches under the ESA to address tribal and state take of ESA-listed species that may occur as a result of harvest activities. Section 10 of the ESA provides for permits to operate fishery harvest programs. ESA section 4(d) rules provide exemptions from take for resource, harvest, and hatchery management plans.

Management of salmon fisheries in the Washington-Oregon-Northern California drainage is a cooperative process involving federal, state, and tribal representatives. The Pacific Fishery Management Council sets annual fisheries in federal waters from three to 200 miles off the coasts of Washington, Oregon, and California. Inland fisheries are those within state boundaries, including those extending out three miles from state coastlines. The states of Oregon, Idaho, California and Washington issue salmon fishing licenses for theses areas.

Most commercial landings in the region are groundfish, Dungeness crab, shrimp, and salmon. Many of the same species are sought by Tribal fisheries, as well as by charter, and recreational anglers. Nets and trolling are used in commercial and Tribal fisheries. Recreational anglers typically use hook and line and may fish from boat, river bank, or docks.

Atmospheric Deposition in the Pacific Northwest Region

Atmospheric transport and deposition may be important for some pesticides addressed in this Opinion. Atmospheric transport is discussed in detail in the *Targeted Monitoring Studies* subsection within the *Effects of the Proposed Action* chapter.

Integration of the Environmental Baseline on Listed Resources

Collectively, the components of the environmental baseline for the action area include sources of natural mortality as well as influences from natural oceanographic and climatic features in the action area. Climatic variability may affect the growth, reproductive success, and survival of listed Pacific salmonids in the action area. Temperature and water level changes may lead to: (1) Reduced summer and fall stream flow, leading to loss of spawning habitat and difficulty reaching spawning beds; (2) increased winter flooding and disturbance of eggs; (3) changes in peak stream flow timing affecting juvenile migration; and (4) rising water temperature may exceed the upper temperature limit for salmonids at 64ºF (18ºC) {JISAO, 2007 #550}. Additional indirect impacts include changes in the distribution and abundance of the prey and the distribution and abundance of competitors or predators for salmonids. These conditions will influence the population structure and abundance for all listed Pacific salmonids.

The baseline also includes human activities resulting in disturbance, injury, or mortality of individual salmon. These activities include hydropower, hatcheries, harvest, and habitat degradation, including poor water quality and reduced availability of spawning and rearing habitat for all 28 ESUs/DPSs. As such, these activities degrade salmonid habitat, including all designated critical habitat and their PCEs. While each area is affected by a unique combination of stressors, the two major impacts to listed Pacific salmonid critical habitat are habitat loss and decreased prey abundance. Although habitat restoration and hydropower modification measures are ongoing, the long-term beneficial effects of these actions on Pacific salmonids, although anticipated, remain to be realized. Thus, we are unable to quantify these potential beneficial effects at this time.

Listed Pacific salmonids and designated critical habitat may be adversely affected by the proposed registration of azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, methamidophos, methidathion, methyl parathion, naled, phorate, and phosmet in California, Idaho, Oregon, and Washington. These salmonids are and have been exposed to the components of the environmental baseline for decades. The activities discussed above have some level of effect on all 28 ESUs/DPSs in the proposed action area. They have also eroded the quality and quantity of salmonid habitat – including designated critical habitat. We expect the combined consequences of those effects, including impaired water quality, temperature, and reduced prey abundance, may increase the vulnerability and susceptibility of overall fish health to disease, predation, and competition for available suitable habitat and prey items. The continued trend of anthropogenic impairment of water quality and quantity on Pacific salmonids and their habitats may further compound the declining status and trends of listed salmonids, unless measures are implemented to reverse this trend.

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. Exposure of salmonids to stressors of the action consists of two separate components, one of which is addressed in this section, and the other of which is addressed in the *Integration and Synthesis* sections. Exposure estimates determined in this section assume salmon habitats are proximate to use sites. In the *Integration and Synthesis for Listed Species* section we evaluate the co-occurrence of use sites and salmonid populations. Designated critical habitat is analyzed separately (see *Effects of the Proposed Action to Designated Critical Habitat* and *Integration and Synthesis for Designated Critical Habitat*).

Exposure Analysis

In this section, we identify and evaluate potential exposure of salmonids to the stressors of the action (Figure 1). We begin by presenting general life history information of vulnerable life stages of Pacific salmon and steelhead. Next, we present a general discussion of the physical and chemical properties of the 12 OPs and their degradation products that influence exposure of listed species and designated critical habitat to these stressors of the action. We then summarize EPA exposure estimates presented in the 12 BEs and present other sources of information, including other modeling estimates and monitoring data to further characterize EECs. 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 ensure that the agency action is not likely to jeopardize listed species or destroy or adversely modify critical habitat, NMFS considers a variety of scenarios in addition to those presented in EPA's BEs. These scenarios provide estimates for the range of habitats used by listed salmonids.

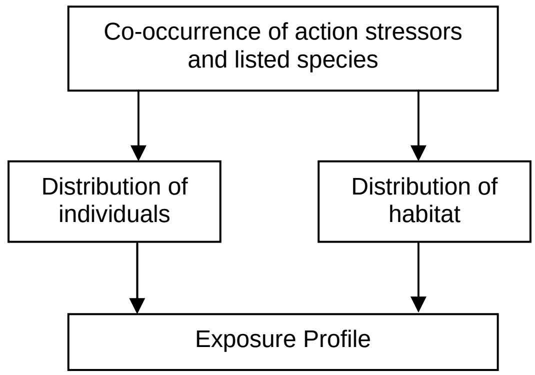


Figure 37 Exposure analysis

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 (Table 71). Many migrate hundreds or thousands of miles during their lifetime, increasing the likelihood that they will come in contact with aquatic habitats contaminated with pesticides.

Table 71 General life histories of Pacific salmonids.

Species (number of listed ESUs or DPSs)	General Life History Descriptions		
	Spawning Migration	Spawning Habitat	Juvenile Rearing and Migration
Chinook (9)	Mature adults (usually four to five years old) enter rivers (spring through fall, depending on run). Adults migrate and spawn in river reaches extending from above the tidewater to as far as 1,200 miles from the sea. 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.
Coho (4)	Mature adults (usually two to four years old) enter the rivers in the fall. The timing varies depending on location and other variables. Coho salmon are semelparous.	Spawn throughout smaller coastal tributaries, usually penetrating to the upper reaches to spawn. Spawning takes place from October to March.	Following emergence, fry move to shallow areas near stream banks. As fry grow they distribute up and downstream and establish territories in small streams, lakes, and off-channel ponds. Here they rear for 12-18 months. In the spring of their second year juveniles rapidly migrate to sea. Initially, they remain in nearshore waters of the estuary close to the natal stream following downstream migration.
Chum (2)	Mature adults (usually three to four years old) enter rivers as early as July, with arrival on the spawning grounds occurring from September to January. Chum salmon are semelparous.	Generally spawn from just above tidewater in the lower reaches of mainstem rivers, tributary stream, or side channels to 100 km upstream.	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 migrate downstream to estuarine areas. They reside in estuaries near the shoreline for one or more weeks before migrating for extended distances, usually in a narrow band along the Pacific Ocean's coast.

Species	General Life History Descriptions		
(number of listed ESUs or DPSs)	Spawning Migration	Spawning Habitat	Juvenile Rearing and Migration
Sockeye (2)	Mature adults (usually four to five years old) begin entering rivers from May to October. Sockeye are semelparous.	Spawn along lakeshores where springs occur and in outlet or inlet streams to lakes.	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 migrate to nursery lakes or intermediate feeding areas along the banks of rivers. Populations that migrate directly to nursery lakes typically occupy shallow beach areas of the lake's littoral zone; a few cm in depth. As they grow larger they disperse into deeper habitats. Juveniles usually reside in the lakes for one to three years before migrating to off shore habitats in the ocean. Some are residual, and complete their entire lifecycle in freshwater.
Steelhead (11)	Mature adults (typically three to five years old) may enter rivers any month of the year, and spawn in late winter or spring. Migration in the Columbia River system extends up to 900 miles from the ocean in the Snake River. 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 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 1979). The transition from yolksac fry to exogenous feeding is a critical life stage for all salmon species requiring availability of prey. Diverse, abundant communities of invertebrates (many of which are salmonid prey items), also populate floodplain habitats and, in part, are responsible for juvenile salmonids reliance on them. Juvenile coho salmon, stream-type Chinook salmon, and steelhead use floodplain habitats for extended durations (several months). Although these 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. Thus, rearing and migrating juvenile salmonids use these habitats extensively (Beechie and Bolton 1999, Beechie and others 2005, Caffrey 1996, Henning 2006, Montgomery 1999, Morley and others 2005, Opperman and Merenlender 2004, Roni 2002).

Exposure Pathways to Salmonids Habitats

Aquatic habitats can be contaminated by pesticides applied to terrestrial target sites through several alternative pathways. For example, spray drift or primary drift refers to the off-target deposition of droplets from spray-applied pesticides 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. Some pesticides are applied directly to surface water for control of plants, mosquitoes, and other aquatic pests. Other pathways of surface water contamination are influenced primarily 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 the 12 a.i.s to characterize the relative importance of these exposure pathways in terms of the potential for the active ingredients and their toxic degradates to contaminate salmonid bearing habitats and designated critical habitats.

Summary of Chemical Fate of the 12 Active Ingredients

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. The 12 a.i.s are primarily registered for use in terrestrial habitats although naled is registered for use over swamps and tidal marshes to control airborne mosquitoes. Fish are most likely exposed to the 12 a.i.s from the water column where the OPs enter the fish during respiration, (*i.e.*, across the gills), or where fish sensory systems come in direct contact with contaminated water (*i.e.*, olfactory sensory neurons). Other secondary 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 the 12 a.i.s reported by EPA in the salmon BEs and red-legged frog BEs. Where discrepencies existed between the two documents, we deferred to the more recent document.

Azinphos Methyl

The chemical fate parameters of azinphos methyl indicate it is relatively persistent under both aerobic and anaerobic soil conditions with weak adsorption to soil (Table 72). Once in the aquatic environment, azinphos methyl is moderately persistent at acid and neutral pH but is hydrolyzed fairly rapidly at high pH. It degrades rapidly by direct aqueous photolysis but rather slowly by soil photolysis. Overall, these parameters suggest that azinphos methyl is mobile in terrestrial environments and has a high potential to contaminate salmonid habitats through runoff and possibly leaching in permeable soils with high recharge (EPA 2003a). Spray drift is also expected to be a primary pathway of exposure to salmonid habitats since azinphos methyl can be spray-applied by ground boom and airblast applications in close proximity to aquatic habitats (60 ft). The persistence in neutral and acidic environments contributes to the likelihood of exposure, whereas alkaline conditions and aquatic habitats conducive to photolysis will reduce persistence and consequently reduce the likelihood and/or duration of exposure.

Table 72. Environmental fate characteristics of azinphos methyl

Parameter	Value
Water solubility[1]	25.10 mg/L at 25° C
Vapor pressure [1]	2.2×10^{-7} mm Hg
Henry's law constant[1]	3.66×10^{-9} atm m^3 mol^{-1}
Octanol/Water partition coefficient[2]	K_{ow} = 543
Hydrolysis ($t_{1/2}$) pH 5, pH 7, & pH 9[1]	39.4 d, 37.5 d, 6.6 d
Aqueous photolysis ($t_{½}$)[1]	3.2 d
Soil photolysis ($t_{½}$)[2]	180 d
Aerobic soil metabolism ($t_{½}$)[1]	31.8 d
Anaerobic soil metabolism ($t_{½}$)[1]	66.7 d
Aerobic aquatic metabolism ($t_{½}$)	Not Specified
Anaerobic aquatic metabolism ($t_{½}$)	Not Specified
Soil partition coefficient[1]	K_d = 7.6 L/kg_{soil}

1 EPA 2007d
2 EPA 2003a

Bensulide

The environmental fate parameters of bensulide indicate it is very persistent in both terrestrial and aquatic environments with half-lives of more than 1 year (Table 73). It is expected to dissolve in water as well as bind to sediments and be transported in runoff to surface waters (EPA 2002). Abiotic hydrolysis and photolysis appear to be minor degradation processes in water and soil surfaces. The primary route of degradation appears to be by aerobic soil metabolism (EPA 2002).

Bensulide's high application rates (up to 16 lbs a.i./A), persistence in terrestrial and aquatic environments, and lack of required buffers to aquatic habitats suggest that runoff of bensulide is likely to occur in salmonid habitats, and exposure may continue for extended durations. Given its solubility and adsorption to soils, run-off of dissolved bensulide and aquatic deposition of bensulide through soil erosion are highly likely. Additionally, contamination of surface water through primary drift is highly likely for spray-applied products when applications occur in close proximity to salmonid habitats. Once in the aquatic system, it is expected to partition primarily to sediments where it will be relatively persistent.

Table 73. Environmental fate characteristics of bensulide.

Parameter	Value
Water solubility[1]	56 mg/L at 25° C
Vapor pressure [1]	8.2 x 10^{-7} mm Hg
Henry's law constant[1]	7.7 x 10^{-3} atm m^3 mol^{-1}
Octanol/Water partition coefficient	Not specified
Hydrolysis ($t_{½}$) [1]	220 d
Aqueous photolysis ($t_{½}$)[1]	200 d
Soil photolysis ($t_{½}$)	Not Specified
Aerobic soil metabolism ($t_{½}$)[1]	363 d
Anaerobic soil metabolism ($t_{½}$)	Not Specified
Aerobic aquatic metabolism ($t_{½}$)[1]	693 d
Anaerobic aquatic metabolism ($t_{½}$)	Not Specified
Soil partition coefficient[1]	K_{oc} = 1,422 to 4,326 L/kg_{oc}

1 EPA 2007e

Dimethoate

Environmental fate studies indicate limited persistence of dimethoate in aerobic aquatic habitats and aerobic soils (Table 74). Photolysis is not a significant route of degradation. Dimethoate hydrolysis rates and persistence vary depending on pH. Dimethoate is hydrolyzed relatively quickly under acidic conditions, but hydrolysis occurs slowly in alkaline environments (hydrolysis half-lives 4.4-156 d). Dimethoate is primarily broken down through microbial degradation. Low soil partition coefficients and high water solubility indicate dimethoate is highly mobile in soil. In a soil column leaching study, 72-100% of applied radioactivity was eluted from the columns in different soils (EPA 2008d). The results of five terrestrial field studies found that dimethoate degraded with half-lives of 5-15 d (EPA 2008d). The toxic degradate, omethoate, was detected in the top layer of soil in all five studies. In a study conducted in California, omethoate was found through day 159 of the study (EPA 2008d). Dimethoate is applied by a variety of ground and aerial spray application methods and there are no required application buffers to aquatic habitats. Therefore, spray drift to salmonid habitats is highly likely and is an expected pathway of exposure to listed salmon and their designated critical habitat. Additionally, given its mobility in soils, dimethoate has a high potential to contaminate salmonid habitats through runoff.

Table 74. Environmental fate characteristics of dimethoate

Parameter	Value
Water solubility [1]	32,000 mg/L at 20° C
Vapor pressure	Not specified
Henry's law constant [1]	8.0 x 10 $^{-11}$ atm m^{3} mol $^{-1}$
Octanol/Water partition coefficient	Not Specified
Hydrolysis (t½) pH 5, pH 7, & pH 9 [1]	4.4 d, 68 d, 156 d
Aqueous photolysis (t½) [1]	353 d (estimate)
Soil photolysis (t½)	No significant degradation
Aerobic soil metabolism (t½)[2]	2.4 days
Anaerobic soil metabolism (t½) [1]	22 days
Aerobic aquatic metabolism (t½)[1]	16.4 days (estimate)
Anaerobic aquatic metabolism (t½)[1]	40.9 days (estimate)
Soil partition coefficient[1]	Kd = 0.06-0.66 L/kg$_{soil}$

1 EPA 2008d
2 EPA 2004a

Disulfoton

Environmental fate studies suggest disulfoton is moderately mobile (Table 75). It degrades fairly rapidly through aquatic and soil photolysis, but it is relatively stable to hydrolysis (EPA 2008e). It is readily broken down in aerobic environments through microbial degradation. However, the total toxic residues of disulfoton (disulfoton + d. sulfoxide + d. sulfone) are essentially stable (EPA 2008e). The d. sulfoxide and d. sulfone degradates, which are formed through microbial degradation and abiotic processes, are more mobile than the parent in soil, and more stable than the parent in soils and water (EPA 2008e). Exposure to these degradates is an important factor in assessing the ecological risk of disulfoton because they are of similar toxicity to the parent compound (EPA 2008e). Environmental fate studies indicate disulfoton sulfoxide is formed at maximum levels of 15% to 95% (of the applied parent compound) through all microbial and abiotic processes excluding hydrolysis. Disulfoton sulfone is the major product of aerobic metabolism in soil and aquatic environments, reaching maximum levels of 19% to 72% (EPA 2008e).

Transformation of disulfoton could lead to formation of three toxic oxons: disulfoton oxon, disulfoton sulfoxide oxon, and disulfoton sulfone oxon (EPA 2008e). Environmental fate studies have shown disulfoton oxon can be formed through hydrolysis, and disulfoton sulfoxide oxon can be formed through aquatic and soil photolysis. However, evaluation of the maximum amounts of these degradates that may be formed in the environment remains an uncertainty because some likely routes of transformation to these oxons have not been evaluated.

Disulfoton can be spray-applied in close proximity to salmonid habitats (0-25 ft). Therefore, spray drift to salmonid habitats is highly likely and is an expected pathway of exposure to listed salmon and their designated critical habitat. Surface water runoff is also expected to be a major pathway for exposure to salmonids and their designated habitat given the mobility and persistence of disulfoton and its toxic degradates. D. sulfoxide and d. sulfone have the potential to move vertically down through the soil profile, and potentially into groundwater, as these degradates form primarily in the shallow subsurface. Groundwater that contains disulfoton residues may then be discharged into surface waters as baseflow (EPA 2008e).

Table 75. Environmental fate characteristics of disulfoton

Parameter	Value
Water solubility[1]	15 mg/L at 20° C
Vapor pressure [1]	1.8 x 10^{-4} mm Hg
Henry's law constant[1]	2.6 x 10^{-6} atm m^3 mol^{-1}
Octanol/Water partition coefficient	Not specified
Hydrolysis ($t_{½}$) pH 5, pH 7, & pH 9[1]	1174 d, 323 d, 231 d
Aqueous photolysis ($t_{½}$)[1]	3.9 d (disulfoton); 385 d (TTR[2])
Soil photolysis ($t_{½}$)[1]	2.8 d; 385 d (TTR[2])
Aerobic soil metabolism ($t_{½}$)[1]	2.4 -15.6 d; 120 – 408 d (TTR[2])
Anaerobic soil metabolism ($t_{½}$)	No Data
Aerobic aquatic metabolism ($t_{½}$)[1]	10.7 d (disulfoton); 51 d (TTR[2])
Anaerobic aquatic metabolism ($t_{½}$)[1]	275 d (disulfoton); 385 d (TTR[2])
Soil partition coefficient	K_{oc} = sand: 888 L/kg_{oc}; sandy loam: 483 L/kg_{oc}; silt loam: 449 L/kg_{oc}; clay loam: 386 L/kg_{oc}

1 EPA 2008e
2 EPA estimated half-life for Total Toxic Residues (TTRs), in this case TTRs were based on the estimated sum of disulfoton, disulfoton sulfoxide, and disulfoton sulfone.

Ethoprop

The environmental fate characteristics of ethoprop suggest it is not significantly degraded by abiotic processes (Table 76). Ethoprop is stable to hydrolysis at pH 5, 7, and 9, and photodegradation does not occur in water or soil. Soil metabolism studies indicate it is relatively persistent under aerobic and anaerobic conditions. Ethoprop is highly soluble and is considered mobile in some soils given relatively low soil partition coefficients (EPA 2003c). The O-ethyl-S-propylphosphorodithioate degradate is highly mobile by comparison to ethoprop (EPA 2003c). In field dissipation studies, ethoprop's dissipation half-life was determined to be approximately 40 days in a potato field in Washington (EPA 2003c). Ethoprop monitored in artificial outdoor ponds showed steady declines in concentrations post-application with a dissipation half-life of 28.7 days in water and 29.1 days considering the combined dissipation in water and sediment (Bruns 2008). Whole fish concentrations ranged from 31 to 290 μg/kg in bluegill exposed to 2μg ethoprop/L in a 49 day study (EPA 2003c). The accumulation of ethoprop in prey other than fish is unknown. However, ethoprop accumulation in fish suggests the dietary route of exposure as a potential source of exposure to listed salmonids. The chemical fate parameters of ethoprop suggest it has a high potential to contaminate salmonid habitats through runoff. Spray drift is also expected to be a primary pathway of exposure to salmonid habitats since ethoprop can be spray-applied by ground application in close proximity to aquatic habitats (0-140 ft).

Table 76. Environmental fate characteristics of ethoprop

Parameter	Value
Water solubility[1]	843 ppm at 21° C
Vapor pressure [1]	3.5×10^{-4} mm Hg at 26° C
Henry's law constant[1]	1.5×10^{-4} atm m^3 mol^{-1}
Octanol/Water partition coefficient	$K_{ow} = 2.43$
Hydrolysis ($t_{½}$) [1]	Stable
Aqueous photolysis ($t_{½}$)[1]	Stable
Soil photolysis ($t_{½}$)[1]	Stable
Aerobic soil metabolism ($t_{½}$)[1]	100 d
Anaerobic soil metabolism ($t_{½}$)[1]	100 d
Aerobic aquatic metabolism ($t_{½}$)	Not specified
Anaerobic aquatic metabolism ($t_{½}$)	Not specified
Soil partition coefficient[2]	K_d= 2.1 L/kg_{soil} K_{oc}= 109 L/kg_{oc}

1 EPA 2003c
2 EPA 2006g

Fenamiphos

Fenamiphos readily photodegrades when exposed to natural light on the soil surface (Table 77). Fenamiphos further dissipates in soil by microbial degradation to fenamiphos sulfoxide and fenamiphos sulfone. The half-lives for the degradates in soil were determined to be 62 days for fenamiphos sulfoxide and 29 days for fenamiphos sulfone. Fenamiphos and its degradates are mobile in soils and have a high potential to leach into ground water and to contaminate runoff into surface waters due to their high water solubility. Field dissipation studies conducted on turf in California confirmed that both degradates leach further into the soil than the parent compound. Information on the persistence of fenamiphos in the aquatic environment was not reported. Overall, these parameters suggest that fenamiphos and its sulfoxide and sulfone degradates are mobile in terrestrial environments and have a high potential to contaminate salmonid habitats through runoff. Additionally, spray drift is also expected to be a primary pathway of exposure to salmonid habitats if fenamiphos is applied in close proximity to aquatic habitats. However, all uses of fenamiphos have been canceled and there are no active fenamiphos labels.

Table 77. Environmental fate characteristics of fenamiphos

Parameter	Value
Water solubility [1]	400 mg/L at 20° C
Vapor pressure [1]	1.3×10^{-6} mm Hg
Henry's law constant [1]	1.0×10^{-9} atm m^3 mol^{-1}
Octanol/Water partition coefficient	Not specified
Hydrolysis ($t_{½}$) pH 5, pH 7, & pH 9 [1]	245 d, 301 d, 235 d
Aqueous photolysis ($t_{½}$)	Not specified
Soil photolysis ($t_{½}$) [1]	0.13 d
Aerobic soil metabolism ($t_{½}$) [1]	15.7 d
Anaerobic soil metabolism ($t_{½}$) [1]	87.9 d
Aerobic aquatic metabolism ($t_{½}$)	Not specified
Anaerobic aquatic metabolism ($t_{½}$)	Not specified
Soil partition coefficient	Not specified

1 EPA 1999

Methamidophos

Information from laboratory studies indicates that methamidophos is not persistent in aerobic terrestrial environments but is more persistent in anaerobic aquatic environments (Table 78). Methamidophos photodegrades rapidly on soil surfaces but slowly in water. Its persistence under aerobic aquatic conditions is unknown. It is stable against hydrolysis in acidic environments, but degrades at neutral and alkaline pHs. Observed major degradates in sandy loam sediment were O,S-dimethyl phophorothioate (DMPT) and O-desmethyl methamidophos (EPA 2007f). Methamidophos is very mobile given its solubility and its soil partition coefficient (K_{oc} = 0.88). EPA indicated that methamidophos was adsorbed in only one of the five soils (a clay loam) tested in batch equilibrium studies. The methamidophos degradate DMPT is also very mobile (K_{oc} = 1.6). Although there are no data available for O-desmethyl methamidophos, it is expected to have similar mobility as its parent compound (EPA 2007f). There is a high potential that methamidophos will contaminate salmonid habitat through spray drift since methamidophos products can be spray-applied by ground and aerial methods and there are no requirements for buffers between application sites and aquatic habitats. Additionally, the mobility of methamidophos and its degradates indicate there is a high potential to contaminate surface water through runoff.

Table 78. Environmental fate characteristics of methamidophos

Parameter	Value
Water solubility [1]	200,000 mg/L
Vapor pressure [1]	1.73×10^{-5} mm Hg
Henry's law constant [1]	1.6×10^{-11} atm m^3 mol^{-1}
Octanol/Water partition coefficient [1]	1.5
Hydrolysis ($t_{½}$) pH 5, pH 7, & pH 9	Stable, 27 d, 3.2 d
Aqueous photolysis ($t_{½}$) [1]	>200 d
Soil photolysis ($t_{½}$) [1]	2.6 d
Aerobic soil metabolism ($t_{½}$) [1]	5.8 d
Anaerobic soil metabolism ($t_{½}$)	Not specified
Aerobic aquatic metabolism ($t_{½}$)	Not specified
Anaerobic aquatic metabolism ($t_{½}$)[1]	41 d
Soil partition coefficient [1]	K_{oc} = 0.88 L/kg_{oc}

1 EPA 2007f

Methidathion

Environmental fate studies indicate methidathion is moderately mobile (Table 79). Under aerobic soil conditions, methidathion initially breaks down quickly. However, dissipation may slow over time and methidathion can persist at lower concentrations in soil for several months (EPA 2009b). Photodegradation of methidathion on soil surfaces may occur, but appears to be relatively slow in terrestrial environments. Photodegradation may be more rapid in water, particularly in clear, shallow water. Methidathion is moderately susceptible to hydrolysis in acidic and neutral conditions, but rates increase in alkaline conditions. EPA reports there are no data available for metabolism rates in aquatic environments (EPA 2009b). Therefore, the persistence of methidathion in aquatic environments is highly uncertain. Spray drift is expected to be a primary pathway of exposure to salmonid habitats since methidathion can be spray-applied by a variety of ground and aerial application methods in close proximity to aquatic habitats (25-150 feet). Additionally, solubility and soil partition coefficients suggest runoff is a likely pathway of exposure to salmonid habitats. Although vapor pressures and Henry's law constants suggest limited volatility, EPA reports that detection of methidathion and its oxon in air at distances up to 20 km away from any use sites suggest volatilization could be a significant route of transport that may result in ecological effects (EPA 2009b).

Table 79. Environmental fate characteristics of methidathion

Parameter	Value
Water solubility [1]	250 mg/L at 20° C
Vapor pressure [1]	2.48×10^{-6} mm Hg
Henry's law constant [1]	3.97×10^{-9} atm m^3 mol^{-1}
Octanol/Water partition coefficient [1]	295
Hydrolysis ($t_{½}$) pH 5, pH 7, & pH 9 [1]	27-37 d, 25-48 d, 8-13 d
Aqueous photolysis ($t_{½}$) [1]	10 d
Soil photolysis ($t_{½}$) [1]	40 d
Aerobic soil metabolism ($t_{½}$) [1]	8.5 - 11.3 d
Anaerobic soil metabolism ($t_{½}$) [1]	<30 d
Aerobic aquatic metabolism ($t_{½}$)	Not specified
Anaerobic aquatic metabolism ($t_{½}$)	Not specified
Soil partition coefficient [1]	194 - 589 L/kg_{oc} Avg = 364 L/kg_{oc}

1 (EPA 2009b).

Methyl parathion

Methyl parathion degradation occurs through microbial and abiotic processes (photolysis and hydrolysis). Methyl parathion degrades fairly rapidly in soil and water under aerobic conditions (Table 80). Although microbial degradation of methyl parathion is not known for anaerobic soil conditions, it occurs relatively quickly in sediments based on aquatic metabolism studies. Other degradation processes appear to be less important routes of dissipation. Methyl parathion degradation through hydrolysis and photolysis occurs slowly. Solubility and soil adsorption suggest methyl parathion can be mobile in soils. Methyl parathion is spray-applied by aerial or ground methods, or applied though irriation (chemigation). Methyl parathion can be applied in close proximity to salmonid habitat. There are no buffer zones required between target application sites and aquatic habitats. Overall, the environmental fate parameters suggest that methyl parathion has a high potential to contaminate salmonid habitats through runoff. Spray drift is also expected to be a primary pathway of exposure to salmonid habitats given the allowable application methods. Although laboratory studies suggest that methyl parathion volatilization is not a major route of dissipation, EPA indicates that secondary drift through volatilization may be an important transport mechanism because methyl parathion has been detected in air and rain samples across the United States (EPA 2008f).

Table 80. Environmental fate characteristics of methyl parathion

Parameter	Value
Water solubility [1]	60 mg/L (ppm) at 20° C
Vapor pressure [2]	9.7×10^{-6} mm Hg
Henry's law constant [1]	6.12×10^{-7} atm m^3 mol^{-1}
Octanol/Water partition coefficient [2]	3300
Hydrolysis ($t_{½}$) pH 5, pH 7, & pH 9 [1]	68 d, 40 d, 33 d
Aqueous photolysis ($t_{½}$) [1]	2 d
Soil photolysis ($t_{½}$) [2]	61 d
Aerobic soil metabolism ($t_{½}$) [1]	3.75 d
Anaerobic soil metabolism ($t_{½}$)	Not specified
Aerobic aquatic metabolism ($t_{½}$) [1]	4.1 d
Anaerobic aquatic metabolism ($t_{½}$) [1]	11.1 d
Soil partition coefficient [1]	K_{oc} = 230 - 670 L/kg_{oc}

1 EPA 2008f
2 EPA 2006k

Naled

Naled is readily transformed to dichlorvos (DDVP), another cholinesterase-inhibiting OP insecticide. Both naled and DDVP are broken down through biotic and abiotic degradation (Table 81). Hydrolysis rates are pH dependent in both compounds, with hydrolysis occurring rapidly under alkaline conditions, and more slowly under acidic conditions. Both compounds are quickly metabolized under aerobic conditions in terrestrial soils. Metabolism also appears to occur relatively quickly under anaerobic conditions based on evaluations of DDVP in soil and naled in sediments. Environmental fate tests indicate volatility and mobility in the soil can be important factors in the transport and dissipation of naled and DDVP. Although mobile, the compounds lack persistence suggesting significant transport to groundwater will not occur (EPA 2008g). Overall, these parameters suggest that naled and DDVP have a high potential to contaminate salmonid habitats through runoff. These compounds are also expected to contaminate salmonid habitats from direct application and spray drift. Naled labels allow direct application to some aquatic habitats for some uses (*e.g.*, swamps and tidal marshes for mosquito abatement). Where direct applications to aquatic habitat are allowed, there are no specific prohibitions to direct application of naled to aquatic habitats that contain listed salmonids. Spray drift is also expected to be a primary pathway of exposure to salmonid habitats since naled can be spray-applied by a variety of ground and aerial application methods and there are no buffer zone requirements between the target treatment areas and aquatic habitats. Spray drift is expected to be a major transport mechanism for Ultra Low Volume (ULV) applications which are designed to prolong the pesticides suspension in the air for control of flying insects. Naled can be found in the atmosphere in the absence of local naled use, suggesting atmospheric transport is a potential pathway of exposure to salmonids (EPA 2008g). Incidents of direct exposure of naled and DDVP in listed salmonids are expected to be limited to acute episodes given rapid environmental degradation rates.

Table 81. Environmental fate characteristics of naled and dichlorvos

Parameter	Value
Water solubility [1]	1 mg/L (parent), 15600 mg/L (DDVP)
Vapor pressure [1]	2×10^{-3} mm Hg (parent), 1.2×10^{-2} mm Hg (DDVP)
Henry's law constant [1]	1×10^{-4} atm m^3 mol^{-1} (parent) 5×10^{-8} atm m^3 mol^{-1} (DDVP)
Octanol/Water partition coefficient	Not specified
Hydrolysis (t½) pH 5, pH 7, & pH 9 [1]	4 d , 0.642 d, 0.067 d (parent) 11.65 d, 5.19 d, 0.88 d (DDVP)
Aqueous photolysis (t½) [1]	4.4 - 4.7 d (parent) 10.2 d (DDVP)
Soil photolysis (t½) [1]	0.4 d (parent) 0.65 d (DDVP)
Aerobic soil metabolism (t½) [1]	1 d (parent) 0.42 d (DDVP)
Anaerobic soil metabolism (t½)	Not specified (parent) 6.3 d (DDVP)
Aerobic aquatic metabolism (t½)	Not specified(parent) Not specified (DDVP)
Anaerobic aquatic metabolism (t½) [1]	4.5 days(parent) Not specified (DDVP)
Soil partition coefficient [1]	K_{oc} = 180 L/kg_{oc} (parent), K_d = 0.3 L/kg_{soil} (DDVP)

1 EPA 2008g
DDVP- dichlorvos, a toxic degradate

Phorate

Phorate is metabolized relatively slowly in the field under aerobic and anaerobic conditions (Table 82). It forms two major degradates which are toxic and relatively persistent in soils, phorate sulfoxide and phorate sulfone. Significant transport of phorate through secondary drift is not expected because phorate is formulated exclusively as a granule and is not particularly volatile (EPA 2008h). Runoff and deposition of whole granules in surface water are expected to be the major routes of exposure for phorate and its degradates. Phorate can be applied in close proximity to habitats occupied by listed salmonids and there is a high potential that phorate will contaminate habitats used by listed salmonids. A vegetative filter strip of 66 feet to surface water is required on highly erodible land, which is expected to reduce surface water deposition from runoff. Otherwise, no setbacks are required between application sites and aquatic habitats. Phorate and its toxic degradates can persist on treated areas for several days to several weeks following application. Based on soil partitioning and solubility, phorate can be mobile in soils. Field studies suggest phorate sulfoxide and phorate sulfone may be more persistent and mobile than the parent compound (EPA 2008h). Once in the aquatic habitat, parent phorate is expected to break down rapidly through hydrolysis and photolysis. Abiotic transformation of the sulfoxide and sulfone degradates was not reported and remains unknown. Metabolism studies suggest they may persist in the aquatic environment for days to weeks.

Table 82. Environmental fate characteristics of phorate

Parameter	Value
Water solubility [1]	50 mg/L
Vapor pressure [1]	7.5×10^{-4} mm Hg
Henry's law constant [1]	5.8×10^{-6} atm m^3 mol^{-1}
Octanol/Water partition coefficient	Not specified
Hydrolysis ($t_{½}$) pH 5,7, & 9 at 25° C[1]	2.6 d, 3.2 d, & 3.9 d
Aqueous photolysis ($t_{½}$) [1]	1.1 d
Soil photolysis ($t_{½}$)	Not specified
Aerobic soil metabolism ($t_{½}$) [1]	82 d
Anaerobic soil metabolism ($t_{½}$) [1]	32 d
Aerobic aquatic metabolism ($t_{½}$)[1]	21 d
Anaerobic aquatic metabolism ($t_{½}$) [1]	42 d
Soil partition coefficient [1]	K_{oc} = 450 - 705 L/kg_{oc}
Parameter for Phorate sulfoxide and/or sulfone degradates	**Value**

Parameter	Value
Aerobic soil metabolism ($t_{½}$) [1]	65 d (sulfoxide); 137 d (sulfone)
Aerobic aquatic metabolism ($t_{½}$) [1]	7.5 d (sulfoxide); 20.9 d (sulfone)
Anaerobic aquatic metabolism ($t_{½}$)	Not specified
Field dissipation ($t_{½}$) [1]	14 d – 126 d (sulfoxide + sulfone)
Partition coefficient [1]	K_{oc} = 506 –106 L/kg_{oc} (sulfoxide); 50 – 138 L/kg_{oc} (sulfone)

1 (EPA 2008h)

Phosmet

Phosmet is stable to photolysis (Table 83). Degradation through hydrolysis occurs rapidly (minutes to hours) under acidic and neutral conditions, but more slowly (days to weeks) under alkaline environments (Lopez 2009). Degradation is also influenced by temperature, occuring more slowly at colder temperatures. Biotic degradation occurs moderately fast under both aerobic and anaerobic conditions in the soil. Metabolism in aquatic environments was not reported and remains uncertain. Phosmet is expected to be slightly to moderately mobile in soils based on its solubility and soil partition coefficients (EPA 2008i). Phosmet oxon has been identified as a minor metabolic product (≤ 0.5% of applied parent) under aerobic and anaerobic soil conditions. Environmental fate characteristics of phosmet oxon were not reported and remain uncertain. Phosmet can be spray-applied by a variety of ground and aerial application methods. Several labels do not require buffer zones between target application sites and aquatic habitats. Once transported to the water column, phosmet is expected to rapidly degrade through hydrolysis. Overall, these parameters suggest that phosmet and its degradates have a high potential to contaminate salmonid habitats through runoff and spray drift.

Table 83. Environmental fate characteristics of phosmet

Parameter	Value
Water solubility [1]	25 mg/L at 20° C
Vapor pressure [1]	4.5 x 10^{-7} mm Hg
Henry's law constant [1]	7.5 x 10^{-9} atm m^3 mol^{-1}
Octanol/Water partition coefficient [1]	Log K_{ow} = 2.78 – 3.04
Hydrolysis (t½) pH 5, pH 7, & pH 9 [1]	0.0038, 0.39, and 7.5 d
Aqueous photolysis (t½) [1]	Stable
Soil photolysis (t½) [1]	Stable
Aerobic soil metabolism (t½) [1]	3-27 d
Anaerobic soil metabolism (t½) [1]	3-22 d
Aerobic aquatic metabolism (t½)	Not specified
Anaerobic aquatic metabolism (t½)	Not specified
Soil partition coefficient [1]	K_{oc}= 716– 10,400 L/kg_{oc}

1 (EPA 2008i)

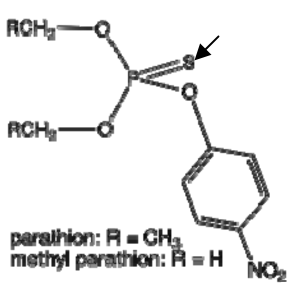
Exposure of salmonid habitats to the stressors of the action

Considering that all listed Pacific salmonid ESUs/DPSs use watersheds where the use of the 12 a.i.s are authorized, the 12 a.i.s are permitted for use in close proximity to salmonid habitats (0-140 ft), and that organophosphate pesticides detections are wide-spread in freshwater habitats across the U.S. (Gilliom 2006) we expect it is possible all listed Pacific salmonid ESUs/DPSs and their designated critical habitats may be exposed to the stressors of the action. We evaluate co-occurrence of use sites and salmon habitat for each a.i. and species pair in *Integration and Synthesis*.

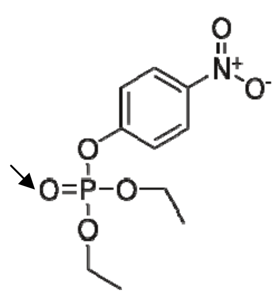
Degradates and Other Compounds of Concern

In evaluating the effects of the OP pesticides addressed in this Opinion on listed salmonids, we also consider degradates of the compounds, especially those which might also inhibit AChE (Table 84). Phosphorothionate pesticides (*i.e.*, those containing a P=S bond) can form oxygen analogues ("oxons", where an oxygen atom replaces the sulfur atom to form P=O, Figure 38). Eight of the pesticides addressed in this Opinion form oxons in varying amounts (azinphos methyl, bensulide, dimethoate, disulfoton, methidathion, methyl parathion, phorate, and phosmet). Transformation of the parent compound can occur via metabolism within an organism (vertebrate or invertebrate) or via environmental processes such as photo oxidation and microbial oxidation. It also occurs during chlorination of drinking water, which led EPA to do an analysis of oxon toxicity and stability in water for the OP cumulative drinking water assessment (EPA 2006b). Based on EPA's analysis, the oxons were estimated at 10 to 100 fold more toxic than the parent compounds. This finding agrees with other toxicity data available, although these data are scarce and are not available for every compound. Some oxons, including those from bensulide, methidathion, and methyl parathion, are stable for 72 hours in water (EPA 2006b). Based on the greater toxicity and persistence of oxons compared to parent compounds, we evaluate potential exposure of salmonid habitats.

Two of the currently registered phosphorothiones, disulfoton and phorate, degrade to sulfoxides and sulfones (the S in the CH_2-S- CH_2 portion of the molecule oxidizes to S=O or O=S=O, respectively, Figure 39) that are more persistent in the environment than the parent (Table 75, Table 82). Disulfoton sulfoxides and sulfones, in particular, have an aquatic half-life of close to a year. Fenamiphos, which has no active labels, also degrades to sulfoxides and sulfones. For these two chemicals, both sulfoxides and sulfones also form oxons in the environment, creating a total of six possible AChE-inhibiting chemicals per parent (parent, parent oxon, sulfoxide, sulfoxide oxon, sulfone, and sulfone oxon). The dimethoate oxon, also known as omethoate, is a commercially available pesticide, although it currently does not appear to be registered in the U.S. An additional degradate of methyl parathion is 4-nitrophenol, which does not inhibit AChE, but is considered a polar narcotic.

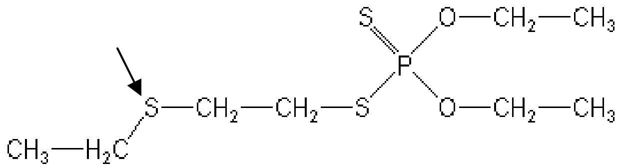


Parathion

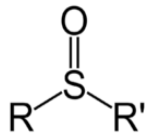


Paraoxon

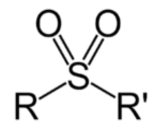
Figure 38 Example of Oxygen Analogue (Oxon) Showing Affected Sulfur and Oxygen Atoms



Disulfoton



Sulfoxide



Sulfone

Figure 39 Example of Sulfoxide and Sulfone Showing Affected Sulfur Atom

Ethoprop, fenamiphos, methamidophos, and naled already have a P=O bond and do not form oxons (Figure 40). However, methamidophos and naled are associated with other OP pesticides currently registered in the U.S., but not included in this action. Methamidophos is a degradate of acephate, and naled degrades to dichlorvos (DDVP). Thus, the toxicity, concurrent use, and environmental concentrations of acephate and DDVP must also be considered when assessing these chemicals. Naled also degrades to dichloroacetic acid (DCAA), a substance regulated under drinking water standards, but for which no toxicity data were located. Ethoprop is more persistent in aquatic ecosystems than many of the other OPs addressed in this Opinion (EPA modeled aquatic half-life as stable). Commonly occurring degradates of ethoprop are also phosphorodithoates, with variations on the alkane sidechains.

When EPA modeled aquatic Estimated Environmental Concentrations (EECs), some consideration was given to the degradates, either in the actual modeling, or in the risk description. Oxons were sometimes addressed in the BEs, and were generally evaluated at least qualitatively in the CA red-legged frog (RLF) determinations used to develop the fate profile for this Opinion. For disulfoton and phorate, EECs were modeled as total toxic residues (TTR)[3] in surface waters. Although these EECs did not account for the

[3] Total toxic residues (TTR) modeling, as done by EPA, sums the residues of the parent compound with its sulfoxide and sulfone degradates. However, it may not take into account all degradates of toxicological concern in developing EECs. Depending on the specific characteristics of the chemical, adjustments may

toxic oxon degradates (a significant missing degradate considering their toxicity), they did provide values for the sum of the parent, sulfoxide, and sulfone breakdown products. TTRs were not provided for fenamiphos, another a.i. that forms sulfoxide and sulfone degradates. Contributions from acephate applications to methamidophos EECs and the effect of dichlorvos applications on concentrations of naled degradates were not taken into consideration in salmonid BEs. Naled and dichlorvos were both included in TTRs in the RLF BE. Ethoprop was modeled as essentially stable, which would account for effects of the phosphorodithoates degradates, presuming toxicity of the degradates is similar to the parent.

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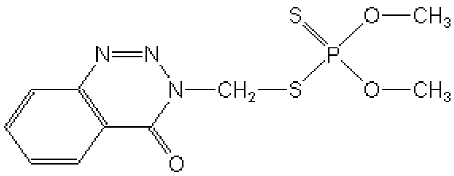
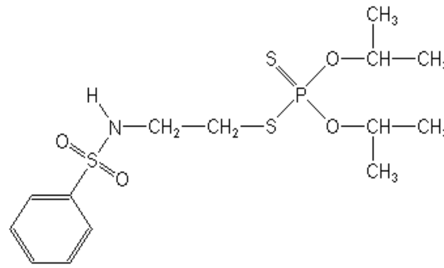
be made to the model inputs by using the most conservative fate characteristics (*e.g.,* the longest half-life), and/or summing expected concentrations of the parent and the degradates.

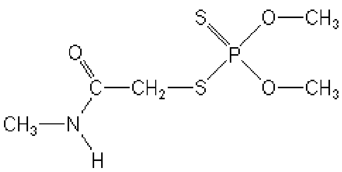
Oxon Formers



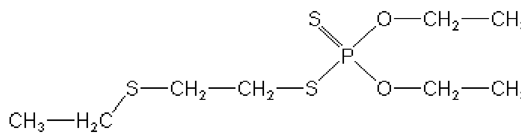
Azinphos methyl



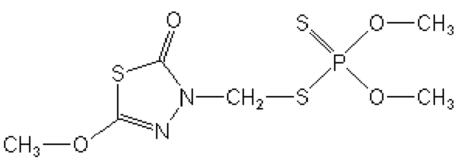
Bensulide



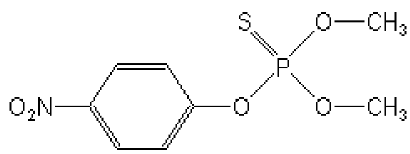
Dimethoate



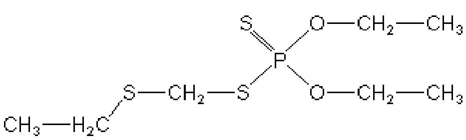
Disulfoton



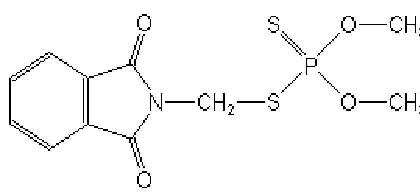
Methidathion



Methyl Parathion

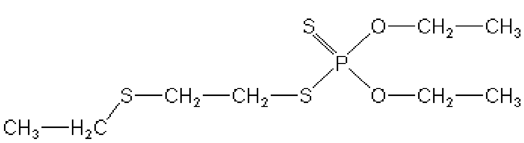


Phorate

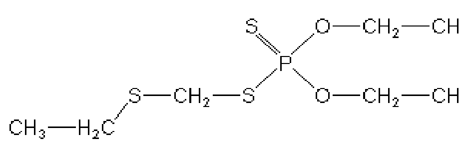


Phosmet

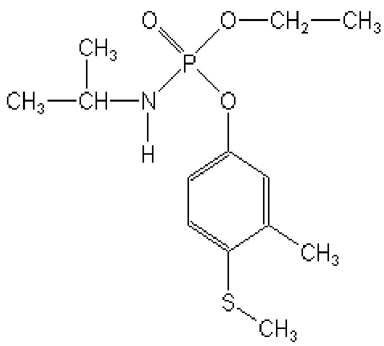
Sulfoxide and Sulfone Formers



Disulfoton

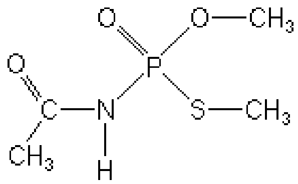


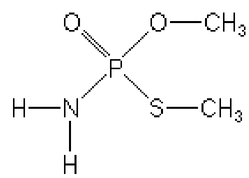
Phorate



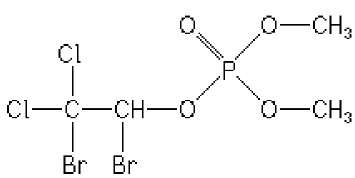
Fenamiphos

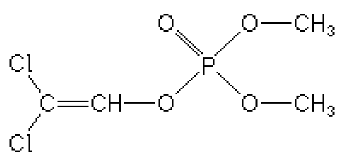
Degrade to or from other pesticides





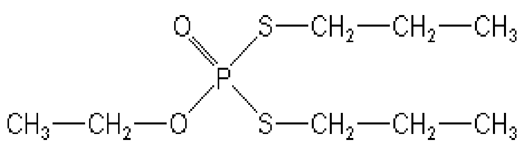
Acephate to Methamidophos





Naled to Dichlorvos

Does not degrade to or from other pesticide, nor form oxon



Ethoprop

Figure 40 Chemical Structures for all A.I.s

Table 84 Degradates of the 12 a.i.s

Active Ingredient	Parent Fate	Degradates of Toxicological Concern	Other Degradates Identified	Modeling Method(s) used in EPA Assessment (Direction of Uncertainty)
Oxon Former				
Azinphos methyl	Parent is mobile and fairly persistent in the environment. Degrades slowly by soil photolysis ($t_{1/2}$ 180 days) but more rapidly by hydrolysis ($t_{1/2}$ 77 hr) (RED)	Azinphos methyl oxon (minor degradate <10% of parent (RLF))	anthranilic acid, methyl anthranilate, azinphos methyl oxygen analog, mercaptomethyl benzazimide, hydroxymethyl benzazimide, benzazamide, and *bis*-methyl benzazamide sulfide, and methyl benzazimide sulfonic acid (all are considered minor degradates (<10%))	Risk Assessment based solely on parent material due to “limited concentrations” of degradates anticipated to be present (BE) Assumed 5% increase of parent for oxon (RLF) (Underestimate)
Bensulide	Parent is persistent and not mobile in soil. Aerobic soil $t_{1/2}$ is 1/year, and neither hydrolysis nor photolysis is a major pathway in soil or water (BE, RED)	Bensulide oxon (major degradate) is more mobile in soil than parent (BE, RED)	Benzenesulphonamide (BE, RED, RLF)	Parent only appears to have been modeled. Parent is highly persistent in both soil and water (BE, RED, RLF) (Underestimate or no difference)
Dimethoate	Parent is mobile and non-persistent. Primary degradation pathways are microbial metabolism and oxidative degradation (BE, RED)	Dimethoxon, also known as omethoate, was noted in field dissipation tests, but not in laboratory fate studies. Unknown what percentage of parent may convert to oxon in environment. Omethoate is recognized as an OP insecticide in and of itself, but does not appear to be currently registered in the U.S., based on a search of OPP website (PDD 6/9/09)	dimethyldimethoate and dimethylthiophosphoric acid in lab tests (RED) Not evaluated; described as 1) not likely to persist in field based on aerobic field studies, and 2) not toxicologically significant (RED).	Parent only in ecological risk assessment. Omethoate not anticipated to be major degradate in water except as product of chlorination (RED). (Underestimate or no difference)

Active Ingredient	Parent Fate	Degradates of Toxicological Concern	Other Degradates Identified	Modeling Method(s) used in EPA Assessment (Direction of Uncertainty)
Disulfoton	Parent is moderately mobile, and generally non-persistent. Its two major degradates, disulfoton sulfoxide and disulfoton sulfone are more stable than the parent, and so total residues are of concern (RLF).	Parent, sulfoxide, and sulfone all form oxons. No toxicity data available for oxons.	Parent, sulfoxide and sulfone are all toxic to fish.	EECs from both the BE and RLF assessment evaluated total toxic residues (parent+sulfoxide+sulfone). Oxons were not specifically included in either, as sulfoxide and sulfone were modeled as essentially stable (RLF) (Underestimate or no difference)
Methidathion	Parent is relatively non-persistent ($t_{1/2}$ 10-11 days in both soil and water) (RED) and moderately mobile (RLF).	Methidathion oxon identified as minor degradate (0.2-4.9%) in soil environments (aerobic soil metabolism, soil photolysis, terrestrial field dissipation) (RLF)	Other degradates include GS-12956 (major, soil and water), Des-methyl GS-13007[1] (major, in hydrolysis GS -12956, GS-28369, GS-28370, GS-20865, minor, aerobic soil metabolism) (RLF)	Parent only. (Underestimate) Potential impact of oxon considered in risk description (RLF).
Methyl Parathion	Parent is not persistent and mobile to relatively mobile in soil. Major dissipation routes are microbial degradation, aqueous photolysis, and hydrolysis ($t_{1/2}$ <5 days in soil and water). Aqueous photolysis is rapid ($t_{1/2}$ 49 hr). It may volatilize and has been detected in air and rainfall, but volatilization is not expected to be a major route of dissipation (BE, RED).	Methyl paraoxon 2.1% of applied 4-nitrophenol ~10% of applied (polar narcotic) (RLF)	Other minor degradates identified in environmental fate studies are monodesmethyl parathion, phosphorothioic acid, O,S-dimethyl o-(4-nitrophenyl) ester, nitrophenyl phosphoric acid, mono (4-nitrophenyl) ester and CO_2 (RLF)	Parent only (BE). (Underestimate) Separate toxicity evaluations on paraoxon and 4-nitrophenol (RLF).

Active Ingredient	Parent Fate	Degradates of Toxicological Concern	Other Degradates Identified	Modeling Method(s) used in EPA Assessment (Direction of Uncertainty)
Phorate	Parent is not persistent and moderately mobile in soil. It degrades by chemical and microbial activity, and field dissipation within 2-15 days. It is formulated exclusively as granules and unlikely to volatilize (BE[2]). Hydrolysis $t_{1/2}$ 2-3 days for all pHs tested, likely will not persist in water column (RLF[3]).	Parent, sulfoxide, and sulfone all form oxons {Roberts, 1999 #2067}. No toxicity data available for oxons.	Degrades to a sulfoxide and a sulfone, both of which are more persistent and more mobile than the parent	EECs from both the BE and RLF assessment evaluated total toxic residues (parent+sulfoxide+sulfone). Oxons were not specifically included (RLF) (Underestimate or no difference)
Phosmet	Parent degrades quickly via aqueous photolysis ($t_{1/2}$ 9.4hr, pH7) and aerobic soil metabolism ($t_{1/2}$ 3 d) (BE). Moderately mobile to mobile in soil (RED[4]).	Phosmet oxon minor degradate (<0.5% of applied) of soil metabolism	Phthalamic acid (major degradate in hydrolysis study) Other minor degradates in hydrolysis studies phthalic acid, pthalimide in soil metabolism studies phthalamic acid, n-hydroxymethyl phthalimide,n-methoxymethyl phthalimide)	Parent only - concentrations of oxon not anticipated to be high enough to warrant inclusion and fate data not available (RED). (Underestimate)
Forms Sulfoxide and Sulfone, but not Oxon				
Fenamiphos		Parent, sulfoxide, and sulfone Parent, sulfoxide, and sulfone also form phenolic derivatives. Limited toxicity data available for phenols in open literature.	Sulfoxide and sulfone appear similar to or less toxic to fish than parent. Sulfoxide is equally toxic to aquatic invertebrates. (BE)	EECs do not appear to include sulfoxide and sulfone degradates. (Science Chapter) (Underestimate)

Degrade to or from Another OP Insecticide				
Methamidophos	Parent is a degradate of the pesticide acephate (in soil, 77% of applied acephate is converted) in addition to being a registered pesticide (BE). Parent is not persistent in aerobic systems, with aerobic soil metabolism of 14 hrs. Photodegrades slowly and is stable to hydrolysis in aquatic systems. May also persist in anaerobic sediments (RED).	No oxon. No other degradate identified as of toxicological concern, largely due to stability of parent.	The identified major degradates of methamidophos are S-methyl phosphoramidothioate, O,S-dimethyl phosphorothioate, methyl mercaptan, dimethyl disulfide, and methyl disulfide (RLF).	Degradates were not considered in this assessment due to rapid dissipation in the environment, volatility, and lack of toxicity data (RLF). (Underestimate or no difference)
Naled	Parent degrades relatively quickly via aquatic hydrolysis (96 hrs at pH 5, 15.4 hr at pH 7, 1.6 hr at pH 9). It also degrades quickly on soil and in air (1-96 hrs). (BE) Degrades to dichlorvos (DDVP). Both parent and DDVP degrade and volatilize quickly. They may be subject to atmospheric transport, either as drift (applied in ultra-fine droplets) or due to secondary volatilization (RLF).	DDVP is a major degradates and registered OP insecticide (20% of applied) (BE). Dichloroacetic acid (DCAA) (Drinking water standards exist (MCL), no other toxicity data located).	None mentioned	Parent only (BE). (Underestimate)

Does not Degrade To or From Another OP Insecticide, nor Form Oxon				
Ethoprop	Parent is relatively persistent in soil ($t_{1/2}$ 100 days), and mobile. Degradates occur in low concentrations, and are primarily variants on the parent. dipropylphosphorodithioate (BE).	No	Ethyl alcohol and S,S-dipropylphosphorodithioate in aquatic studies (BE) In aerobic soil metabolism studies O-ethyl-S-methyl-S-propylphosphorodithioate (SME) <4 % O-ethyl-O-methyl-S-propylphosphorodithioate (OME) <1 % O-ethyl-S-propylphosphorodithioate (M1) <1 %	Appears to be parent only. Modeled as stable to hydrolysis and photolysis (BE). (Underestimate or no difference)

[1] In guideline environmental fate studies, companies will sometimes give an alpha–numeric designator to compounds that 1) are not readily otherwise identified, 2) transient in nature, or 3) occur as small percentages of the parent pesticide.

[2]BE = EPA Biological Evaluation on Pacific Salmonids

[3]RLF = EPA Biological Evaluation on California red-legged frog

[4]RED = EPA Reregistration Eligibility Decision

Modeling: Estimates of Exposure to the 12 a.i.s

EPA exposure estimates for non-crop pesticide applications

EPA's BEs indicate that pesticides containing the 12 a.i.s are approved for a variety of uses (Table 85). All are approved for use on agricultural crops. Some of the OPs are also approved for use on other sites including forestry, golf courses, nurseries, parks, residential areas, noncrop land adjacent to vineyards, margins of agricultural fields, aquatic habitats (*e.g.* swamps), rangeland, and livestock. As previously indicated, many of the uses identified in the BEs have since been modified, or are scheduled to be phased out or cancelled.

Table 85. Summary of use sites approved on active labels

Active Ingredient	Use Site*				
	Agricultural		Residential/ Urban/ Industrial/ Nursery	Forestry	Other
	Orchard	Field crops			
Azinphos methyl	x				Bee beds
Bensulide		x	x		Golf course, turf
Dimethoate	x	x	x	x	Alfalfa, grass, ornamental, adjacent to vineyards
Disulfoton		x	x		Christmas trees
Ethoprop		x	x		Ornamental
Fenamiphos		NA[1]			
Methamidophos		x			Alfalfa seed
Methidathion	x	x	x		Alfalfa, timothy, clover seed
Methyl parathion	x	x			Alfalfa, forage grass
Naled	x	x	x	x	Mosquito/fly, pastures, marshes, woodlands, etc.
Phorate		x			Lilies, daffodils
Phosmet	x	x	x	x	Livestock, fire ants, field margins, ornamental landscapes

x- Use site listed on active label
1- No active labels. Use of existing stocks is authorized and most recent CDPR data indicates fenamiphos is still used in vineyards and orchard crops in California.
*Use site categories correspond to GIS data layers

Azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, and methidathion all have current labels that authorize product use in non-crop areas (Table 86). The BEs did not provide exposure estimates for non-crop uses. The only exception was an estimate for bensulide associated with its use in turf. In general, the BEs provided few estimates of exposure given the number and variety of uses currently authorized.

Table 86. Examples of current registered uses of the 12 a.i.s and the exposure method used by EPA in BEs.

Active Ingredient	Examples of Registered Uses	Exposure Methods Applied in BEs
Azinphos methyl	Crops: almonds, apples, crab apples, cherries, pears, pistachios, walnuts	PRZM-EXAMS Estimates for 7 crops
	Other use sites: border around alkali bee beds	No estimates provided
Bensulide	Crops: fruiting vegetables, dry bulb vegetables, cantaloupes, cucumbers, muskmelons, watermelons, field grown flowers, bulbs	PRZM-EXAMS Estimates for onions, cole crops, and cucurbits
	Other use sites: ornamentals, residential lawns, golf course turf	PRZM-EXAMS turf
Dimethoate	Crops: mustard greens, pears, peas, pecans, peppers, potatoes, safflower, sorghum, soybean, Swiss chard, endive, tomatoes, turnips, watermelon, and wheat	PRZM-EXAMS Estimates for citrus, cotton, corn, Brussels sprouts
	Other use sites: woody ornamentals and Christmas trees, nurseries, meadowfoam, non-cropland areas adjacent to vineyards	No estimates provided
Disulfoton	Crops: asparagus, beans, broccoli, Brussels sprouts, cabbage, cauliflower, cotton, Easter lilies, lettuce, radish for seed	PRZM-EXAMS Estimates for barley, cotton, potatoes, wheat
	Other use sites: azaleas, camellias, flower beds, bulbs, and bedding plants, rhododendrons, Christmas trees, ornamental shrub, roses	No estimates provided
Ethoprop	Crops: bananas, beans, cabbage, corn, cucumber, hops, mint, plantains, potatoes, sugarcane, tobacco	PRZM-EXAMS Estimates for beans, cabbage, corn, cucumber, potatoes, sweet potatoes
	Other use sites: ornamentals, Easter lilies	No estimates provided
Fenamiphos	No active labels	PRZM-EXAMS 5 crops
Methamidophos	Crops: alfalfa for seed, cotton, potatoes, tomatoes	PRZM-EXAMS Estimates for cotton and potatoes
Methidathion	Crops: alfalfa, alfalfa for seed, almonds, apples, apricots, artichokes, cherries, citrus, clover for seed, cotton, mango, nectarines, olives, pears, peaches, plums, kiwi, safflower, sunflower, walnut	PRZM-EXAMS Estimates for fruits, almonds, wheat
	Other use sites: nursery woody ornamentals and herbaceous plants	No estimates provided
Methyl parathion	Crops: alfalfa, corn, cotton, onion, canola, rice, soybeans, sunflower, wheat, oats, rye, barley, white potatoes, walnut	PRZM-EXAMS Estimates for alfalfa, peas, walnut
	Other use sites: grass (forage)	No estimates provided
Naled	Crops: alfalfa grown for seed, almonds, beans, lima	PRZM-EXAMS

Active Ingredient	Examples of Registered Uses	Exposure Methods Applied in BEs
	beans, peas, broccoli, cabbage, carrots grown for seed, cauliflower, cotton, Brussels sprouts, kale, collards, cantaloupes, muskmelons, hops, melons for seed, celery, cotton, eggplant, peppers, grapes, oranges, lemons, grapefruit, tangerines, peaches, safflower, strawberries, sugar beets, summer squash, Swiss chard, walnuts	Estimates for alfalfa, beans, broccoli, cotton, sugar beets, walnut
	Other use sites: forest and shade trees, ornamentals, flowering plants, greenhouse, swamps, tidal marshes, pastures, residential, agricultural, woodlands, in and around food processing plants, loading docks, refuse area, corrals, holding pens, feed lots, rangeland, residential areas, woodlands	PRZM-EXAMS Estimates for rangeland
Phorate	Crops: beans, corn, cotton, peanuts, potatoes, radishes, sorghum, soybeans, sugar beets	PRZM-EXAMS Estimates for corn, cotton, sorghum
	Other use sites: lilies and daffodils	PRZM-EXAMS Estimates for lilies and daffodils
Phosmet	Crops: alfalfa, almonds, apples, apricots, blueberries, cherries, citrus, clover grown for seed, cotton, crab apples, cranberries, filberts, grapes, nectarines, peaches, pears, peas, pecans, pistachios, plums, potatoes, prunes, sweet potatoes, walnuts	PRZM-EXAMS Estimates for alfalfa, almonds, apples, berries, cherries, citrus, cotton, grapes, kiwis, peaches, pears, pecans, plums, potatoes, sweet potatoes, walnuts
	Other use sites: Christmas trees, conifer trees, deciduous trees, cattle and swine, fire ant mounds, ornamentals, agricultural field margins	No estimates provided

EPA exposure estimates for crop applications

EPA's BEs provided EECs for the 12 a.i.s in surface water. These EECs were generated using the PRZM-EXAMS model and used as expected concentrations of the 12 a.i.s for all aquatic habitats where listed salmonids and their prey reside. However, no exposure estimates were provide for other identified stressors of the action including inert/other ingredients, other active ingredients with formulations, and for all of the toxic degradates identified. These missing estimates introduce substantial uncertainty into the exposure analysis. The PRZM-EXAMS model generates pesticide concentrations for a generic "farm pond". The pond is assumed to represent all aquatic habitats including rivers, streams, floodplain habitats, estuaries, and near shore ocean environments. EPA's BEs indicate that the PRZM-EXAMS scenarios provide "worst-case" estimates of salmonid exposure and EPA "believes that the EECs from the farm pond model do represent first order streams, such as those in headwaters areas" used by listed salmon. However, listed salmonids use aquatic habitats with physical characteristics that would be expected to yield higher pesticide concentrations than would be predicted with the "farm pond" based model. Juvenile salmonids rely upon a variety of floodplain habitats that are critical to rearing. All listed salmonids use shallow, low flow habitats at some point in their life cycle (Table 71). Below, we discuss the utility of the EECs for the current consultation. NMFS presents information that indicates the EECs do not represent worst-case environmental concentrations to which listed Pacific salmonids may be exposed. Finally, NMFS provides additional modeling estimates to evaluate potential exposure in floodplain habitats used by salmonids.

Table 87. PRZM-EXAMS exposure estimates from EPA's BEs.

Scenario: crop, state	Application: rate (lbs a.i./A)/ method/ number of applications	Acute EEC peak (μg/L)	Chronic EEC 60-d average (μg/L)
Azinphos methyl			
Almonds, CA	3/foliar/2	8.3	4.8
Apples, NY/OR	1.5/foliar/4	13.9	9.0
Cherries, MI	0.75/foliar/2	10.7	6.7
Potatoes, ME	0.75/foliar/3	13.6	7.6
Peaches, GA	2/foliar/4	40.6	25.5
Pears, OR	1.5/foliar/4	8.9	4.9
Walnuts, OR	2/foliar/3	12	7.5
Bensulide			
Vegetables[1]	6/ground spray/1; 6/soil incorporated/1	36; 19	24; 13[1]
Onion, WA	9/ground spray/1; 9/soil incorporated/1	54; 28	36; 17[1]
Vegetables[1]	6/banded ground spray/1; 6/banded soil incorporated/1	27; 15	12; 6.5[1]
Onion, WA	9/banded ground spray/1; 9/banded soil incorporated/1	40; 23	18; 9.7[1]
Golf course greens[1]	12.5/granular broadcast/2; 12.5/ground spray/1	7.2; 4	3.9; 2.2[1]
Golf course fairways and home lawns[1]	12.5/granular broadcast/2; 12.5/ground spray/1	180; 100	98; 55[1]
Vegetables[5]	6/ground spray/1; 6/soil incorporated/1	93; 60	88; 55 [2]
Onion, WA	9/ground spray/1; 9/soil incorporated/1	140; 90	132; 83[2]
Vegetables[5]	6/banded ground spray/1;	42; 30	40; 28[2]

Scenario: crop, state	Application: rate (lbs a.i./A)/ method/ number of applications	Acute EEC peak (μg/L)	Chronic EEC 60-d average (μg/L)
	6/banded soil incorporated/1		
Onion, WA	9/banded ground spray/1; 9/banded soil incorporated/1	63; 45	60; 42[2]
Dimethoate			
Citrus[5]	0.5/aerial-ground/6,4 /aerial-ground/1,	9.6; 58.3	0.8; 1.3
Cotton[5]	0.5/aerial-ground/2	24.4	2.0
Corn[5]	0.5/aerial-ground/3	6.4	0.6
Brussels sprouts[5]	1/aerial-ground/6	19.0	3.0
Disulfoton residues *(parent+d. sulfoxide+d. sulfone)*			
Barley[5]	1/foliar/2; 0.83/granular/2	21.0; 19.3	17.4; 15.0
Cotton[5]	1/ground spray/3	43.5	34.4
Potatoes[5]	4/ground spray/2; 1/foliar/3	14.9; 26.7	12.2; 20.9
Spring Wheat[5]	0.75/foliar/2	16.4	12.6
Disulfoton (parent only)			
Barley[5]	1/foliar/2; 0.83/granular/2	9.2; 7.1	3.8; 2.4
Cotton[5]	1/ground spray/3	14.8	4.9
Potatoes[5]	4/ground spray/2; 1/foliar/3	7.1; 15.0	2.6; 6.9
Spring Wheat[5]	0.75/foliar/2	8.9	3.8
Ethoprop			
Beans, MI	8/soil incorporated/1	75.0	71.0
Cabbage, CA	5/soil incorporated/1	17.0	16.0
Corn, OH	6/soil incorporated/1	26.0	24.0
Cucumbers, FL	2/soil incorporated/1	15.0	14.0
Potatoes, ME	1/soil incorporated/1	29.0	27.0
Sweet Potatoes, LA	8/soil incorporated/1	18.2	17.4
Fenamiphos			
Peaches[5]	7.5/soil incorporated/1	17.4	11.9
Citrus[5]	7.5/ soil incorporated/1	0.3	0.2
Grapes[5]	6.0/ soil incorporated/1	14.4	9.9
Cabbage[5]	4.5/ soil incorporated/1	35.4	23.5
Raspberries[5]	6.0/ soil incorporated 1 in/1; 6.0/ soil incorporated 2 in/1	32.8; 16.4	19.4; 9.7
Methamidophos			
GENEEC[3]	4/aerial/1	65	35[2]
GENEEC[3]	4/ground/1	61	33[2]
Cotton, MS	4/aerial/1	40	2.9
Potatoes, ID	4/aerial/1	30	3.7
Methidathion			
Fruits, CA	3/aerial/1; 3/ground/1	15.5; 10.6	9.0; 6.1
Almonds, CA	3/aerial/1; 3/ground/1	14.6; 9.9	8.4; 5.3
Wheat[4], CA	1/aerial/1; 1/ground/1	9.8; 8.9	6.1; 5.3
Methyl parathion			
Walnuts, CA	4/aerial/4; 4/ground/4	18.2; 10.2	4.0; 1.8
Alfalfa, OR	1/aerial/2; 1/ground/2	3.9; 2.0	1.2; 0.5
Peas, ID	0.5/aerial/2; 0.5/ground/2	2.6; 1.3	0.4; 0.3
Naled			
Alfalfa, CA	1.4/aerial/3	4.0	0.2
Cotton, CA	0.94/aerial/5	2.6	0.2
Sugarbeets, CA	0.94/aerial/5	2.6	0.2

Scenario: crop, state	Application: rate (lbs a.i./A)/ method/ number of applications	Acute EEC peak (μg/L)	Chronic EEC 60-d average (μg/L)
Broccoli, CA	1.88/aerial/10	5.3	0.6
Walnuts, CA	1.88/aerial/2	5.3	0.1
Rangeland, OR	0.4/aerial/1	1.1	0.01
Alfalfa, CA	1.41/aerial/3	4.0	0.1
Beans, CA	1.41/ground/1	0.8	0.01
Phorate residues *(parent+p. sulfoxide+p. sulfone)*			
Sweet corn[5]	1.3/T-banded/1	26.9	5.9
Cotton[5]	Not reported/in-furrow/1	27.6	8.2
Field corn[5]	1.3/T-banded/1	7.7	2.5
Sorghum[5]	1.3/T-banded/1	12.7	4.2
Lilies/daffodils[5]	8/soil incorporated/1	138	41
Phorate (parent only)			
Sweet corn[5]	1.3/T-banded/1	21.3	1.2
Cotton[5]	Not reported/in-furrow/1	23.1	1.4
Field corn[5]	1.3/T-banded/1	4.6	0.2
Sorghum[5]	1.3/T-banded/1	7.5	0.4
Lilies/daffodils[5]	8/soil incorporated/1	115	7
Phosmet			
Alfalfa, OR	1/aerial/8	3.0	0.1
Almonds, CA	3.7/air blast/3	10.3	0.2
Apples, NY	4/ air blast/5; 1.5/ air blast/10	26.7; 15.6	0.8; 0.3
Apples, OR	4/ air blast/5; 1.5/ air blast/10	11.2; 14.0	0.5; 0.4
Berries, MI	1/ground boom/5	11.8	0.2
Cherries, WI	1.75/ air blast/4	9.5	0.3
Citrus, FL	2/ air blast/3	12.9	0.3
Cotton, MS	1/ground boom/5	29.9	0.4
Grapes, NY	1.5/ air blast/4	18.7	0.6
Kiwi, CA	2/ air blast/6	19.7	0.3
Peaches, GA	3/ air blast/4; 2/ air blast/5	16.2; 8.9	0.5; 0.2
Pears, OR	5/ air blast/3	14.0	0.4
Pecans, GA	2.25/ air blast/5	23.7	0.4
Plums/prunes, OR	3/ air blast/3	8.4	0.4
Potatoes, ME	1/aerial/5	7.9	0.2
Sweet potatoes, LA	1/aerial/5	20.6	0.4
Walnuts, OR	6/ air blast/5	8.4	0.3

1- Chronic EEC reported is a 21-d average, rather than a 60-d average
2- Chronic EEC reported is a 56-d average, rather than a 60-d average
3- EPA standard scenario, Generic Estimated Environmental Concentration
4- Oregon wheat scenario used as a surrogate for alfalfa in the methidathion BE
5- State not specified for scenario

Very few non-crop uses of pesticides were evaluated in the salmonid BEs. However, NMFS also reviewed aquatic exposure estimates developed by EPA within the red legged frog BEs (EPA 2007d, EPA 2007e, EPA 2007f, EPA 2008d, EPA 2008e, EPA 2008f, EPA 2008g, EPA 2008h, EPA 2008i, EPA 2009b).

Although these estimates were specific to registered uses in California only, they provided surface water estimates for pesticides authorized for non-crop uses that were not included in the BEs for listed salmon (Table 88).

Table 88. Estimates of pesticide concentrations in surface waters from California red legged frog BEs.

a.i.	PRZM-EXAMS exposure estimates for surface water (µg/L)		
		Non-Crop	Crops
Azinphos methyl	peak	Not reported	1.9 - 6.8
	60-d avg	Not reported	1.0 - 3.4
Bensulide	peak	Ornamentals: 168-231	42 -135[1]
	60-d avg	Ornamentals: 138-191	33-113[1]
Dimethoate	peak	Ornamentals, turf, nurseries, forestry, etc: 0.1-20.3	1.1-16.5
	60-d avg	Ornamentals, turf, nurseries, forestry, etc: 0.1-13.9	0.4-9.4
Disulfoton	peak	Residential: 3.7-15	1.8-67[2]
	60-d avg	Residential: 3.2-12	1.3-54[2]
Ethoprop	No assessment		
Fenamiphos	No assessment		
Methamidophos	peak	Not reported	1.7-12
	60-d avg	Not reported	0.2-2.7
Methidathion	peak	Nursery ornamentals: 116	0.45-49
	60-d avg	Nursery ornamentals: 54	0.22-27
Methyl parathion	peak	9-30	7-67
	60-d avg	5-13	2-26
Naled	peak	Swamps, mosquitos: 4-33 Flies (AgDrift, 10 foot buffer): 5.3	0.9-25
	60-d avg	Swamps, mosquitoes: 1-25	0.4-8.0
Phorate	peak	Ornamentals: 4.6	0.3-4.6[2]
	60-d avg	Ornamentals: 0.3	0.02-0.3[2]
Phosmet	peak	Forestry: 24	3.5-78
	60-d avg	Forestry: 1.5	0.2-3.7

1- Assumed a 25 foot no-application buffer to surface water
2- Estimate is for total residues of parent, the sulfone and the sulfoxide degradates for soil incorporated applications

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 highest concentrations to which any individuals of the population may be exposed. Several lines of evidence discussed below suggest that EECs in the BEs underestimate exposure of some listed salmonids and designated critical habitat.

Although EPA characterized these exposure estimates as “worst case” in the BEs, it has also acknowledged that measured concentrations in the environment sometimes exceed PRZM-EXAMS EECs (EPA 2007b). EPA has subsequently clarified that rather than providing worst case estimates, PRZM-EXAMS estimates are high end estimates for the vast majority of applications and aquatic habitats (EPA 2007b). NMFS agrees that the model is designed to produce upper bound exposure estimates for some, but not all aquatic habitats.

Recent formal consultation and reviews of EPA informal consultations by the Services found that concentrations measured in surface water sometimes exceed 24 hour average concentrations predicted with PRZM/EXAMS modeling (NMFS 2007a, NMFS 2008c, NMFS 2009b, USFWS 2008). These findings demonstrate that the EECs generated using PRZM-EXAMS can underestimate peak concentrations of active ingredients that occur in salmonid habitats. Consequently, underestimation of exposure and subsequent risk to species is likely. Below, we discuss the primary reasons why EPA’s exposure estimates do not represent worst case exposures to salmonid habitats.

Model outputs are 90th percentile time-weighted averages and are not worst case exposures for all salmonid habitats. It is important to recognize that the PRZM-EXAMS model predicts concentrations based on site-specific assumptions (*e.g.*, rainfall) and that environmental concentrations provided for the estimate do not represent the highest aquatic concentrations predicted given the assumptions. Rather, the exposure estimates provided in the BEs are time-weighted average concentrations over one day (*i.e.*, averaged over a 24 hr period), 21 d, and 60 d. Although EPA refers to the one day averages as peak concentrations, they do not represent the maximum concentration predicted by the model. Rather, they represent the average concentration over a 24-hour period. Additionally, the concentrations reported represent the 90^{th} percentile of the estimates derived using PRZM-EXAMS (Lin and others 1998). Although NMFS agrees this is a relatively protective approach for evaluating exposure in some aquatic habitats, it does not represent the possible "worst case" exposure in many salmonid habitats.

Model inputs did not use maximum application rates, maximum number of applications, and minimum application intervals. We compiled maximum use rates (single and seasonal), maximum number of applications, and minimum application intervals from current labels (Table 2 through Table 12). Several of the PRZM-EXAMS inputs within the BEs (Table 87) were not consistent with the maximum application rates and maximum number of applications allowed. For example, the maximum application rate assessed for azinphos methyl was 10-70% lower than the maximum single application rate allowed on active labels (10 lbs a.i./A). The same was true for bensulide (16 lbs a.i./A), dimethoate (4.5 lbs a.i./A), disulfoton (9 lbs a.i./A), methidathion (10 lbs a.i./A), and naled (2.12 lbs a.i./A). In several instances PRZM-EXAMS inputs did not account for the greatest number of allowable applications. For example, multiple applications of methidathion are specified for 10 crops (*e.g.*, eight applications in artichokes) but all of the model simulations run for methidathion only considered a single application. Some scenarios also failed to account for maximum seasonal or annual application rates specified on active labels. For example, up to 32 a.i. lbs/A of bensulide can be applied to golf courses annually versus the 25 assessed. Thus, NMFS does not rely strictly on EPA exposure estimates.

Few crop scenarios were assessed relative to the number of approved uses. No exposure estimates were provided for any of the 15 crops currently approved on active dimethoate product labels. Similarly, exposure estimates were provided for disulfoton use in cotton but were lacking for the other 14 use sites listed on active disulfoton labels. Exposure estimates were provided for two of the four crops currently listed on methamidophos labels, and two of 15 crops on active methyl parathion labels.

Crop scenarios provided are not representative of the entire action area. The regional scale that the modeled scenarios are intended to represent is unclear. Many of the scenarios were conducted for states outside the distribution of listed salmonids (*e.g.*, the azinphos methyl, methamidophos, and phosmet BEs). Others did not provide information on geographic locations simulated (*e.g.*, county, state, region, etc.). The assumed rainfall and other site-specific input assumptions can have large impacts on predicted exposure. For example, the phosmet BE illustrates this point with two scenarios for apples, one in OR and one in NY. The peak EEC for OR apples was 26.7, while the peak EEC for NY was 11.2 μg/L - a more than two fold difference (Table 87) NMFS also questions whether input values were adequate to represent the geographic variability throughout the action area. Site-specific meteorological and soil conditions vary greatly throughout the four states where listed salmonids are distributed and crops are grown. The BEs did not indicate how sites were selected and how well scenarios represented the range of conditions throughout the four states. Without a description of EPA's scenario selection, it is difficult to determine the representativeness of scenario estimates for the complete range of crop uses.

Crop scenarios do not consider application of more than one pesticide. Many products that include one of the 12 a.i.s are approved for use on the same crops (Table 86). Active pesticide labels include few restrictions regarding either the co-application (*i.e.,* tank mixture applications) or sequential applications of other pesticide products, including those containing ingredients that share a common mode of action. Examples of fish kill incidents discussed in a recent NMFS Opinion indicate combinations of AChE-inhibiting insecticides are sometimes applied on the same day or over a short interval, increasing the likelihood of salmonid exposure to chemical mixtures that may have additive or synergistic effects (NMFS 2008c, NMFS 2009b). Some labels encourage the use of more than one AChE -inhibiting insecticide. For example, phosmet labels recommend tank mixing with dimethoate products for pest control in alfalfa (EPA Reg. No. 10163-175 and 10163-215) or tank mixing with methomyl products to control insects in apple crops (Reg. No. 10163-169 and 10163-184). Tank mixing or applying multiple applications of pesticides to the same site increase the likelihood of cumulative exposure and effects.

NMFS exposure estimates for pesticide mixtures

We generated exposure concentrations for some of the 12 a.i.s using the GENEEC model[4] to estimate potential cumulative exposure resulting from tank mixtures or sequential applications of active ingredients. Values were generated for three separate crops to evaluate concentrations that might be present in a water body adjacent to that crop (Table 89). The GENEEC model was chosen to estimate potential exposure on sites vulnerable to runoff. The input values used were consistent with recent EPA assessments and restrictions specified on active pesticide labels (EPA Reg. No. 264-457, 70506-193, 5481-526, 10163-244, 5481-479, 10163-169/CA060002, and 10163-78).

Table 89. GENEEC estimated concentrations of pesticides in surface water adjacent to crops.

Chemical use	Rate	No.*	Interval	Buffer	EEC (µg/L)				
Foliar/ ground application	lbs/ acre		days	Ft	Peak	4-d avg	21-d avg	60-d avg	90-d avg
Potatoes									
ethoprop	12	1	-	0	127	127	125	122	120
methyl parathion	1.5	4	7	0	120	115	88	52	38
phorate	2.31	1	-	0	98	95	80	56	44
Oranges									
methidathion	5	2	45	50	201	197	178	142	122
naled	1.875	3	7	50	83	76	47	22	15
phosmet	2.1	2	7	0	90	36	6.9	2.4	1.6
Cherries									
azinphos methyl	0.75	1	-	60	21	21	20	18	17
methidathion	3	1	-	25	104	102	92	73	63
phosmet	0.93	1	-	0	21	8.2	1.6	0.6	0.4

*Number of applications
-Not applicable

NMFS exposure estimates for floodplain habitats

Model inputs used in BEs are not representative of most vulnerable salmonid habitats. The EECs within EPA's BEs were derived primarily using the PRZM-EXAMS model. The EPA "farm pond" scenario is likely a poor surrogate of many habitats used by listed salmonids that are more susceptible to higher pesticide concentrations given their

[4] EPA characterizes GENEEC as a tier-1 screening model (EPA 2004c). GENEEC is a meta-model of the PRZM-EXAMS model that incorporates assumptions that are intended to model exposure estimates on a site vulnerable to runoff. The size of the treated area and aquatic habitat (farm pond) are the same as described for PRZM-EXAMS.

physical characteristics. 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. The PRZM-EXAM estimates assume that a 10-hectare (approximately 25 acres) drainage area is treated and the aquatic habitat is assumed to be static (no inflow or outflow). Pesticide treatment areas of 10-hectares and larger occur frequently in agricultural crops, particularly under pest eradication programs. Additionally, aquatic habitats used by salmon vary in volume and recharge rates and consequently have different dilution capacities to spray drift and runoff events. The assumed drainage area to water volume ratio (100,000 m^2: 20,000 m^3) is easily exceeded for small water bodies. For example, a one acre pond with an average depth of 1 m would exceed this ratio for treated drainage areas of approximately five acres in size and larger. The assumed aquatic habitat and size of the treated area for the PRZM-EXAMS scenarios suggest that exposure is underestimated for listed salmonids that use relatively small aquatic habitats with low dilution capacities.

Direct over-spray of pesticides to aquatic habitats

To estimate potential exposure of salmon to pesticides in floodplain and other shallow-water habitats we first determined the initial average concentrations that will result from a direct overspray of shallow surface water (Table 90). The active labels for the 12 a.i.s do not authorize direct application of pesticides to surface water. The only exception is methyl parathion use on rice. When pesticides are applied directly to aquatic habitats the resulting initial concentration is a function of the amount applied and the volume of the water body. Methyl parathion can be applied twice in rice at a rate of 0.75 lbs a.i./A. A single application at that rate would result in an average initial methyl parathion concentration of 841 μg/L in 10 cm of water. The active labels for methyl parathion do not place any restrictions on rice paddy discharges to water bodies that contain listed species. Active labels indicate that “shrimp, crabs, and crayfish may be killed” by the use of methyl parathion products in rice indicating that this labeled use may be harmful to nontarget aquatic species and should not be used “where these are important resources.”

Table 90. Average initial concentration of any a.i. in surface water resulting from a direct overspray of aquatic habitat.

Application Rate (lbs a.i. / acre)	Water Depth (meters)	a.i. Concentration in Surface Water (µg/L)
0.25	2	14
0.5	2	28
1	2	56
3	2	168
10	2	560
0.25	1	28
0.5	1	56
1	1	112
3	1	336
10	1	1,121
0.25	0.5	56
0.5	0.5	112
1	0.5	224
3	0.5	673
10	0.5	2,242
0.25	0.3	93
0.5	0.3	187
1	0.3	374
3	0.3	1,121
10	0.3	3,736
0.25	0.1	280
0.5	0.1	560
1	0.1	1,121
3	0.1	3,363
10	0.1	11,208

Aerial ULV applications for mosquito and fly control

Although labels specify not to apply naled directly to surface water, they do allow for drift applications to be made over a variety of salmonid habitats such as streams, rivers, lakes and tidal marshes. For example, one label specifies not to apply over bodies of water except when "weather conditions will facilitate movement of the applied material away from the water in order in minimize incidental deposition into the water body (EPA Reg. No. 5481-480)." These applications typically occur at higher elevations (e.g. 200 feet) and smaller drop spectrums than those common to agricultural applications. Simulations using the AgDisp model where run to evaluate potential deposition of naled below and downwind from naled release sites (AgDisp version 8.17). Simulations were consistent with label requirements (Appendix 6). Results at varying release heights, distances, wind speeds, and application rates are presented in (Table 91). Under the condition of the simulation deposition predicted was influenced by release height and windspeed. The greatest deposition occurred within 30 feet of the application site for applications made at elevations of 10 and 50 feet. When applied at 200 feet, peak deposition predicted occurred 30 - 300 feet downwind at windspeeds of 2 mpg, and 300-3000 feet downwind at windspeeds of 10 mph. Naled labels are not consistent in the maximum amount that can be applied for treatment of flys and mosquitoes with allowable application rates that range from 0.1– 1.25 lbs a.i./acre (EPA Reg. No. 5481-480 and 10163-46, respectively). The simulations suggest mosquito application may result in aquatic concentrations that exceed 7 μg/L for the lower labeled rate, and 90 μg/L for the maximum labeled rate.

Table 91. Predicted average initial concentration (μg/L) in a floodplain habitat that is 2 m wide and 0.1 m deep using AgDisp (version 8.17).

Feet from edge of release site	Release Height (feet) : Wind Speed (mph)					
	10 : 2	10 : 10	50 : 2	50 : 10	200 : 2	200 : 10
0.10 lbs a.i./A rate						
0	2.80	5.63	7.50	7.01	3.05	2.33
30	2.27	5.17	7.03	7.06	3.11	2.34
150	1.04	3.35	5.29	6.48	3.22	2.39
300	0.52	2.11	3.27	6.07	3.16	2.40
1500	0.06	0.41	0.34	2.95	1.20	2.57
3000	0.00	0.02	0.04	0.52	0.58	1.80
15000	0.00	0.00	0.01	0.03	0.10	0.11
1.25 lbs a.i./A rate						
0	35.03	70.40	93.71	87.61	38.10	29.13
30	28.40	64.64	87.91	88.31	38.84	29.24
150	12.98	41.89	66.09	80.94	40.27	29.84
300	6.46	26.42	40.89	75.91	39.48	29.98
1500	0.69	5.07	4.30	36.84	14.96	32.14
3000	0.05	0.22	0.46	6.52	7.28	22.49
15000	0.01	0.02	0.14	0.42	1.27	1.38

Application of pesticides to adjacent terrestrial habitat

Some of the pesticides may be applied at the immediate edge of surface water while others require a buffer zone between the application area and aquatic habitats (Table 92). No-application buffer zones are a standard tool to reduce runoff and drift to aquatic habitats. Vegetated buffer zones are considered more effective than buffers that are not vegetated because plants can intercept drift and help to reduce runoff by slowing sheet flow, entraining sediments, binding pesticides, and increasing soil infiltration.

Table 92 Buffer zone requirements on active pesticide labels

Active Ingredient	Buffer Type	Use Sites	Distance from Aquatic Habitat (ft)
Azinphos methyl	Vegetated filter strip	Apples, cherries, pears[1]	60
Disulfoton	Vegetated filter strip	Agricultural uses	25
Ethoprop	No spray	All spray applications[2]	140
Methidathion	No spray	>3 lbs a.i./acre ≤3 lbs a.i./acre	50 25
Naled	No spray	Agricultural	25 ground boom, 100 air blast, 150 aerial
Phorate	Vegetated filter strip	All uses where highly erodible land is adjacent to aquatic bodies	66
Phosmet	No spray	Cotton	1 mile from coastal or estuarine waters, 100 ft from other aquatic habitats
Phosmet	No spray	Washington potatoes	25 ground 50 chemigation 150 rill

1 One active phorate label (EPA Reg. No. 10163-78) does not specify a buffer zone requirement for pears
2 Granular formulations do not require buffers

As identified earlier, no exposure concentrations were provided in the BEs for vulnerable floodplain habitats where juvenile salmonids rear and shelter. To fill this gap, we derived exposure estimates for floodplain habitats that incorporated label-specified buffer zone requirements (Table 93). These estimates were derived using the AgDrift model and estimate downwind deposition from pesticide drift (Teske 2001). This method does not incorporate additional contributions that may occur through the runoff pathway. 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 simulations assumed an aquatic habitat that was 0.1 m deep and 2 m wide. These dimensions are consistent with some of the smaller, and potentially more vulnerable floodplain habitats used by salmonids.

Table 93 Estimated average initial pesticide concentrations in a floodplain habitat that is 2 m wide and 0.1 m deep using AgDrift.

a.i.	Rate Lbs a.i./acre	Simulation	Buffer	Average Initial Concentration in Surface Water (μg/L)
Azinphos methyl	0.75 – 1.5	Airblast[1]	0 60	5.7 – 11.4 0.8 – 1.7
Bensulide	6 – 16	Ground boom[2]	0	1,100 – 2,940
Dimethoate	0.25 – 1.33 1 – 4.5 0.25 – 1.33	Ground boom[2] Airblast[3] Aerial[4]	0 0 0	46 – 245 105 – 436 123 – 652
Disulfoton	1 – 2 1	Ground boom[5] Aerial[4]	25 25	16 – 32 237
Ethoprop	3 – 12	Ground boom[2]	140	6.0 – 24
Fenamiphos	No active labels			
Methamidophos	1	Ground boom[5] Aerial[4]	0 0	267 490
Methidathion	1 2 3 – 10 2 3 – 10	Ground boom[5] Air blast[3] Air blast[3] Aerial[4] Aerial[4]	25 25 50 25 50	267 66 49 – 164 473 559 – 1,860
Methyl parathion	0.5 – 1.5 0.5 – 2	Ground boom[5] Aerial[4]	0 0	134 – 401 245 – 980
Naled	0.94 – 1.88 0.63 – 1.88 0.63 – 1.88	Ground boom[5] Air blast[6] Aerial[4]	25 100 150	15 – 30 6.8 – 20 44 – 132
Phorate	1.31 – 1.88	Ground methods	0 66	NA NA
Phosmet	0.75 – 2 1.75 1.02 2.5 – 5.95 0.75 – 5.95 1.02 1.75	Ground boom[5] Ground boom[5] Ground boom[5] Air blast[1] Aerial[4] Aerial[4] Aerial[4]	0 25 100 0 0 100 150	200 – 534 28 5.0 19 – 45 368 – 2,920 108 123

1 – Normal orchard
2 – Low ground boom, fine-medium/course distribution, 50th percentile
3 – Dense orchard
4 – EPA default (ASAE fine-medium droplet size distribution)
5 – Low ground boom, very fine-fine distribution, 50th percentile
6 – Sparse orchard
NA – Spray drift calculation is not applicable because only granular formulations are available

Monitoring Data: Measured Concentrations of Parent Compounds and Degradates in Surface Waters

We reviewed two types of pesticide monitoring data: 1) ambient data that measure concentrations of pesticides and other contaminants in surface waters, but are not targeted at any specific pesticide application or land cover type, and 2) data from more targeted studies (frequently found in published scientific literature and gray literature), which may be collected in waters near or downstream of agricultural or other pesticide uses. We evaluated data from three central sources: USGS' NAWQA database, state databases maintained by California and Washington and targeted monitoring studies which may not be included in monitoring databases. Neither Oregon nor Idaho currently maintains a state database. Data from Washington include studies conducted by the Washington State Departments of Ecology and Agriculture. The NAWQA data typically are general monitoring data, and sampling stations are distributed across a range of land uses, although some data may be from investigations into specific uses. The California and Washington databases contain data from studies that fall into both categories.

The OPs considered in this Opinion have relatively short aquatic half-lives, generally in the range of days to weeks, with the exceptions of ethoprop and the disulfoton sulfoxide and sulfone degradates. Thus, we do not necessarily expect a high frequency of detection for these chemicals. However, a low frequency of detection should not be interpreted as no exposure, but more an inability of the existing programs to detect brief pulses which might occur between sampling dates. Because OPs are both fast-acting (in a matter of hours) and irreversibly-binding, the inability of many monitoring programs to detect peak concentrations and/or short pulses is a recognized data gap and a concern to NMFS.

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 habitats 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). Additionally, the monitoring sampling designs and sites do not represent many salmonid species' ranges (see figure 5).

Of the monitoring programs discussed, only the Washington State Department of Ecology program was designed to evaluate potential exposure in selected urban and agricultural areas that do overlap with some listed Pacific salmonid habitats. This sampling program was intended to evaluate pesticide occurrence in a limited number of salmonid-bearing streams during the pesticide application seasons (Johnson and Cowles 2003). Sample sites for this study are best characterized as integrator sites selected based on the presence of listed salmonid populations and high diversity and intensity of agriculture (Johnson and Cowles 2003). The study design included sampling during the pesticide application season but did not target specific applications of pesticides nor did it target salmonid habitats that would be expected to produce the highest concentrations of pesticides (*e.g.*, shallow off-channel habitat in close proximity to pesticide application sites). Sampling was generally conducted on a weekly basis, so it is likely peak concentrations associated with drift and runoff events were not captured. This monitoring program is discussed in more detail in *Monitoring Data from Washington State.*

Other available monitoring data are also applicable to assessing exposure in listed salmon, but to varying degrees. 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; 3) lack of representativeness of current and future pesticide uses and conditions; and 4) lack of information on actual pesticide use to correlate with observed surface water concentrations.

Protocols not designed to capture peak exposure. The NAWQA monitoring studies contain the largest data set evaluated. However, these studies were designed to evaluate trends in water quality and were not designed to characterize exposure of pesticides to listed salmonids (Hirsch 1988). The NAWQA design does not result in an unbiased representation of surface waters which limits the ability to make statistical extrapolations to unsampled waters. Also, some agricultural activities and related pesticide uses that may be very important in a particular region may not be represented in the locations sampled. Sampling from the NAWQA studies and other studies reviewed was typically not conducted in coordination with specific applications of OP pesticides addressed in this Opinion. Similarly, sampling was not designed with consideration to salmon distribution or to target the salmonid habitats most likely to contain the greatest concentrations of pesticides. Given the relatively rapid dissipation of these 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.

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. The use of AChE-inhibiting insecticides has declined in California over recent decades. However, pesticide use patterns change annually and may result in either increases or decreases in use of pesticide products for specific uses. There is considerable uncertainty regarding the representativeness of monitoring conditions to forecast future use of products containing these a.i.s.

Lack of information on actual use to correlate with observed concentrations. A common constraint in the monitoring data was lack of information on actual use of pesticides containing the 12 a.i.s. For example, the ability to relate surface water monitoring data to the proposed action was severely hampered because information on application rates, setbacks/buffers, and applications methods associated with the monitoring were generally not reported. In most cases, the temporal and spatial aspect of pesticide use relative to sampling was not reported, further limiting the utility of the information.

Data Described in USEPA's Biological Evaluations

EPA summarized monitoring data in the BEs, derived mostly from the same sources we have considered. As we considered information from these databases, including the more recent data, we do not reiterate the BE summaries herein.

USGS NAWQA Data for California, Idaho, Oregon, and Washington

We obtained updated data from the USGS NAWQA database to evaluate the occurrence of azinphos methyl, disulfoton, dimethoate, ethoprop, fenamiphos, methidathion, methyl parathion, phosmet, and phorate in surface waters monitored in California, Idaho, Oregon, and Washington. Bensulide, methamidophos, and naled are not on the USGS list of analytes. Additionally, we obtained all available data for identified degradates of the sampled OPs. Data were available for dichlorvos, the oxons of azinphos methyl, methyl parathion, phorate, and phosmet; and the sulfoxides and sulfones of disulfoton and fenamiphos. Land uses associated with the sampling stations included agriculture, forest, rangeland, urban, and mixed use. The database query resulted in approximately 5,200 samples in which one or more of the a.i.s or a degradate was an analyte. Approximately 350 unique sampling locations were represented, with sample sites located in 11 NAWQA basins located throughout California, Idaho, Oregon and Washington (Figure 41). Some waterbodies and/or basins in this dataset do not contain listed salmonids and several of the species have had no sampling within their freshwater and coastal habitats (Table 94). Most notable are those ESUs/DPSs along the coasts of Oregon and California as well as listed salmonid habitats within Idaho. Available data included samples collected from 1991-2009. More than one third of the stations were sampled only once during the span of 18 years, and a relatively small number of sites accounted for the majority of the data; approximately 75% of the data was collected from 35 sites, including nine sites that fell outside the distribution of listed salmonids (Figure 42). The temporal and spatial distribution of sampling is inconsistent with temporal and spatial aspects of salmonid distribution. Consequently, we do not expect the data set to be representative of exposure distributions for listed salmonids.

Table 94. Number of NAWQA sample sites within the distribution of listed Pacific salmonids.

Species	ESU	Kilometers of Stream Inhabited	Sites in Spawning and Rearing Habitat	Sites in Migratory Corridor
Chinook	California Coastal	2,422.44	0	0
	Central Valley Spring - Run	2,212.94	5	0
	Lower Columbia River	2,443.29	17	0
	Upper Columbia River Spring - Run	1,646.75	0	4
	Puget Sound	3,639.65	38	0
	Sacramento River Winter - Run	546.84	5	0
	Snake River Fall - Run	1,370.44	1	2
	Snake River Spring/Summer - Run	5,288.23	1	2
	Upper Willamette River	3,013.85	41	3
Chum	Columbia River	1,162.18	11	0
	Hood Canal Summer - Run	141.89	2	0
Coho	Central California Coast	1,287.78	0	0
	Lower Columbia River	3,307.78	17	0
	Southern Oregon and Northern California Coast	5,619.58	0	0
	Oregon Coast	10,220.00	0	0
Sockeye	Ozette Lake	70.98	0	0
	Snake River	1,493.94	0	3
Steelhead	Central California Coast	4,620.72	0	0
	California Central Valley	4,273.66	42	0
	Lower Columbia River	4,302.03	16	1
	Middle Columbia River	10,196.80	81	2
	Northern California	5,324.31	0	0
	Puget Sound	3,849.64	38	0
	Snake River	13,423.40	1	2
	South-Central California Coast	5,104.56	0	0
	Southern California	3,015.86	2	0
	Upper Columbia River	2,143.15	9	2
	Upper Willamette River	3,063.07	25	3

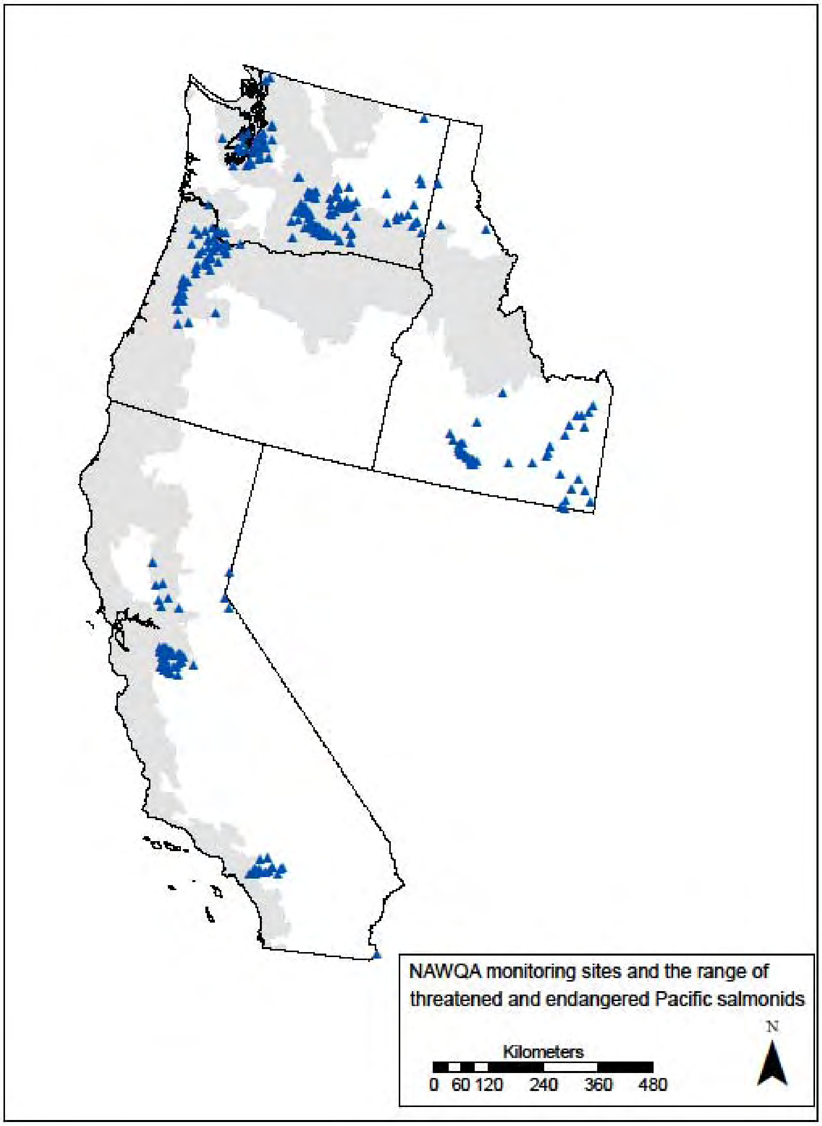


Figure 41. Distribution of NAWQA monitoring sites

The frequency of detection is a combination of the actual occurrence of pesticides in the water and the sampling intensity. NAWQA surface water detections represent the dissolved phase, as the water sample is filtered through a 0.7 micron glass fiber filter. Chemicals transported primarily in the particulate phase would be underreported in this data set. No sediment or tissue data were available from USGS for these compounds. This is also a recognized uncertainty and compromises our ability to determine toxicity of contaminated sediments. Because 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 immediately following drift or a runoff event.

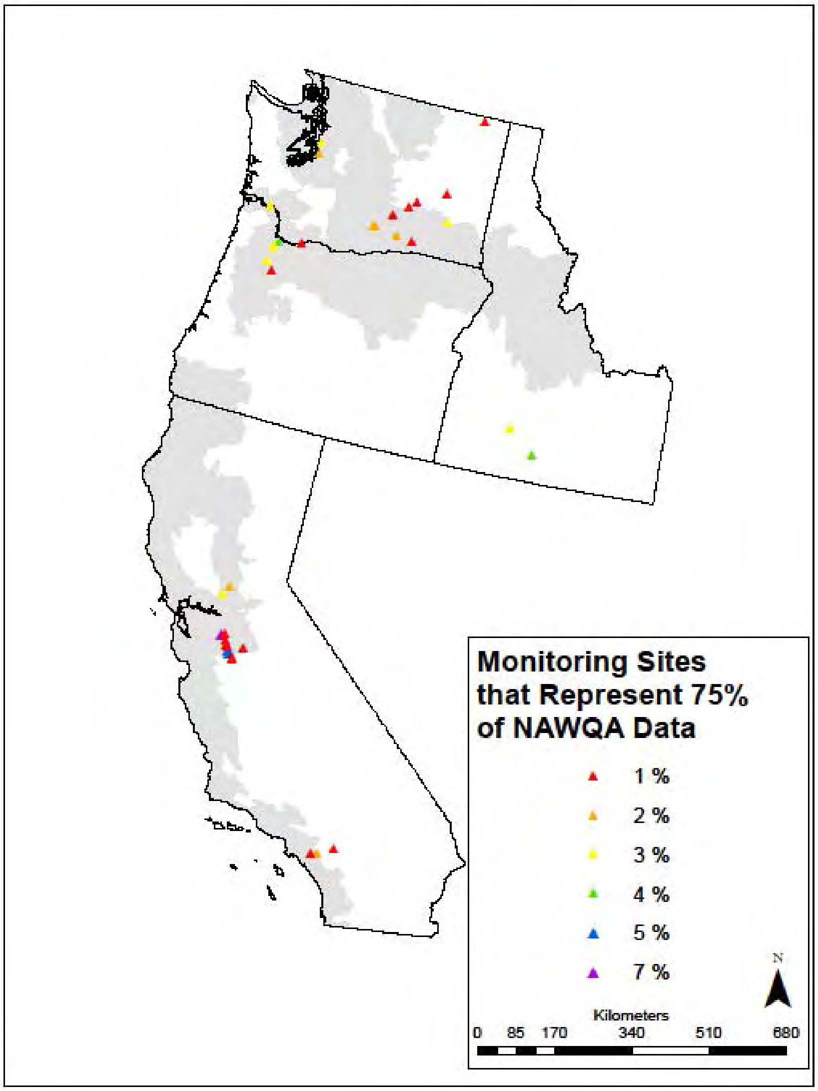


Figure 42. Location of NAWQA monitoring sites representing the majority of data

Summary information for quantifiable concentrations of the pesticides addressed in this Opinion (Table 95) and the degradates (Table 96) are presented below. In the USGS database, 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. These data are included in the summary statistics.

Five of the pesticides (azinphos methyl, disulfoton, ethoprop, methyl parathion, and phorate) were monitored for in approximately 5,000 samples. Of these five, the most commonly occurring were azinphos methyl (8.9%, range 0.002-7.35 μg/L, median 0.024 μg/L), and ethoprop (6.5%, range 0.001-5.75 μg/L, median 0.014 μg/L). Disulfoton, methyl parathion, and phorate were detected in <1% of samples. Dimethoate, fenamiphos, methidathion, and phosmet had smaller sample sets (~1,000 samples), and of these four, dimethoate (4.2%, range 0.004 - 0.158 μg/L, median 0.013 μg/L) and methidathion (3.0%, range 0.003 - 7.35 μg/L, median 0.311 μg/L), occurred most frequently. Fenamiphos and phosmet were detected in <1% of samples.
The available degradates sample sets (dichlorvos, azinphos methyl oxon, methyl paraoxon, phorate oxon, phosmet oxon, disulfoton sulfoxide, disulfoton sulfone, fenamiphos sulfoxide, and fenamiphos sulfone) consisted of approximately 1,000 samples each, with the exception of disulfoton sulfoxide (155 samples), and 4-nitrophenol. Only 14 samples, taken in 1993, included 4-nitrophenol in the data set, and all were non-detects. Of the degradates, only disulfoton sulfone occurred more than 1% of the time, but that was in a small data set (5% of 757 samples, range 0.006-0.235 μg/L, median 0.021 μg/L, 1.5% of parent concentration). These results show that degradates are detected and also highlight that degradates are not as frequently sampled as parent pesticides.

.

Table 95. Concentrations of Parent Pesticides in NAWQA Water Samples for California, Idaho, Oregon, and Washington

Statistic	Azinphos methyl	Dimethoate	Disulfoton	Ethoprop	Fenamiphos	Methidathion	Methyl Parathion	Phorate	Phosmet
Samples	5,138	1,135	4,765	4,765	1,136	1,135	5,141	5,144	1,003
Percent detections	8.9%	4.2%	0.3%	6.5%	0.2%	3.0%	0.7%	0.02%	No detects
LRL range (μg/L)	0.001-0.500	0.006-0.012	0.017-0.050	0.003-0.100	0.029	0.004-0.045	0.005-0.300	0.002-0.060	0.0079-0.210
Minimum concentration (μg/L)	0.002	0.004	0.004	0.001	0.007	0.003	0.003	0.012	N/A
Maximum concentration (μg/L)	7.350	0.158	3.810	5.750	0.010	0.311	0.524	0.012	N/A
Median concentration	0.024	0.013	0.019	0.014	N/A	0.008	0.010	0.012	N/A

Table 96. Concentrations of Degradates in NAWQA Water Samples for California, Idaho, Oregon, and Washington

Statistic	Dichlorvos	Azinphos methyl Oxon	Methyl Paraoxon	Phorate Oxon	Phosmet Oxon	Disulfoton Sulfoxide	Disulfoton Sulfone	Fenaminphos Sulfoxide	Fenamiphos Sulfone
Samples	1,135	1,124	1,135	1,135	909	155	757	1,109	1,120
Percent Detections	0.3%	0.2%	0.09%	No Detects	0.7%	No Detects	5.4%	0.2%	0.2%
LRL range (μg/L)	0.014-0.020	0.016-0.125	0.010-0.030	0.015-0.105	0.022-0.100	0.021	0.006-0.016	0.008-0.010	0.031
Minimum concentration (μg/L)	0.004	0.014	0.006	N/A	0.007	N/A	0.006	0.022	0.008
Maximum concentration (μg/L)	0.014	0.043	0.006	N/A	0.045	N/A	0.235	0.042	0.009
Median concentration	0.007	0.027	N/A	N/A	0.014	N/A	0.021	N/A	N/A

Monitoring Data from California Department of Pesticide Regulation

We evaluated monitoring data available from the CDPR, which maintains a public database of pesticide monitoring data for surface waters in California {Johnson, 2003 #1343}. Data in the database (www.cdpr.ca.gov/docs/emon/surfwtr/surfdata.htm) are from multiple sources, including monitoring conducted by CDPR, USGS (data from the NAWQA program, as well as other studies), state, city, and county water resource agencies; and some non-governmental or inter-governmental groups such as Deltakeeper. 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 majority of the studies, which are described in metadata available from CDPR, are not targeted at correlating water concentrations with specific application practices, with the exception of some studies evaluating rice pesticides. The database, last updated in June 2008, consists of approximately 270,000 data records. Each record reports a specific sampling site, date, and analyte. The number of records associated with a particular compound is indicative of monitoring intensity rather than actual occurrence in surface waters. In this database, detections below the LOQ are reported as 0 μg/L. Summary statistics were calculated on samples with values above the LOQ.

Some data were available for all parent compounds considered in this opinion, although there was considerably more monitoring effort associated with azinphos methyl, dimethoate, disulfoton, ethoprop, methidathion, methyl parathion, and phorate (~20 studies, 3,000-5,000 samples, 15 years of data) than with bensulide, fenamiphos, methamidophos, and naled (1-6 studies, 10-650 samples, 1-4 years of data). Data were also available for the degradates dichlorvos, azinphos methyl oxon, methidathion oxon, methyl paraoxon, and phosmet oxon. Oxon data were from 2-3 studies, conducted in 1991-1995, and were relatively small datasets (580-740 samples).

Table 97 Concentrations of Parent Pesticides in CDPR Database

Statistic	AZM	Bensulide	Dimethoate	Disulfoton	Ethoprop	Fenamiphos	Methamidophos	Methidathion	Methyl Parathion	Naled	Phorate	Phosmet
Samples	4,742	12	3,535	4,304	3,735	603	261	5,176	6,008	656	4,575	2,782
Percent Detections	0.4%	0%	11%	1.6%	1.2%	0.2%	0	4.6%	1.2%	0%	0.2%	0.1%
LOQ range (μg/L)	0.001-1.00	NR	0.010-1.00	0.010-1.00	0.003-0.100	0.050	NR	0.024-0.100	0.005-1.000	0.500	0.002-0.100	0.010-1.00
Minimum concentration (μg/L)	0.006	N/A	0.030	0.011	0.003	1.500	N/A	0.001	0.006	N/A	0.016	0.300
Maximum concentration (μg/L)	0.826	N/A	11.310	0.418	1.110	1.500	N/A	15.100	1.700	N/A	0.220	0.630
Median concentration	0.060	N/A	0.160	0.062	0.021	1.500	N/A	0.069	0.056	N/A	0.088	0.465
Dates	1991-2006	2004-2005	1991-2006	1991-2006	1992-2006	2002-2006	2005-2006	1991-2006	1991-2006	1992-2006	1991-2006	1991-2006
# of Studies	20	2	26	19	19	6	1	29	41	6	20	21

Summary information is reported in Table 97 and Table 98. Dimethoate and methidathion were the most commonly detected pesticides, with, respectively, detections in 11% and 4.6% of the samples. Concentrations ranged from 0.030-11.310 μg/L for dimethoate, and 0.001-15.100 μg/L for methidathion. Disulfoton, ethoprop, and methyl parathion were the next most commonly occurring compounds (1.2-1.6% detections), and all detections were <2μg/L. Azinphos methyl, phorate, and phosmet were detected in less than 0.5% of samples, in datasets ranging from 2,800-4,700 samples. Bensulide, fenamiphos, methamidophos, and naled were not detected in any of the samples, but it should be noted these are much smaller data sets. Dichlorvos was detected in 0.2% of samples with a maximum concentration of 0.542 μg/L. Three studies, conducted from 1991-1995, analyzed for oxons. No detections were reported.

Table 98. Concentrations of Degradates in CDPR Database

Statistic	Dichlorvos	AZM Oxon	Methidathion Oxon	Methyl Paraoxon	Phosmet Oxon
Samples	2,244	581	741	580	635
Percent Detections	0.2%	0%	0%	0%	0%
LOQ range (μg/L)	0.010-0.200	0.050-0.500	0.050	0.050-0.200	0.050-0.500
Minimum concentration (μg/L)	0.015	N/A	N/A	N/A	N/A
Maximum concentration (μg/L)	0.542	N/A	N/A	N/A	N/A
Median concentration	0.101	N/A	N/A	N/A	N/A
Avg percent of parent	N/A	N/A	N/A	N/A	N/A
Dates	1991-1996	1991-1995	1991-1995	1991-1995	1991-1995
# of Studies	14	3	2	3	3

Monitoring Data from Washington State

Data from monitoring studies conducted in the state of Washington are included in Department of Ecology's Environmental Information Management (EIM) database (http://www.ecy.wa.gov/eim/). Data in the database are from multiple sources, including state agencies, and may contain whole water samples as well as filtered samples. The EIM 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. Some of the studies contained in this database are targeted with respect to specific pesticide uses, while others are more generalized water quality surveys. Some data for all pesticides considered in this Opinion were available, and are shown inTable 99. Data were also available for the degradates dichlorvos, disulfoton sulfone, and methyl paraoxon.

The procedure for reporting in the EIM database includes reporting non-detects as the reporting limit for that particular sample, and adding a “U” data qualifier. The reporting limit was not specified in the data accessed by NMFS, thus LOQ ranges were estimated based on “U”-qualified data. Summary statistics were calculated on samples with values above the LOQ (*i.e.*, not qualified with a “U”). Statistics include data qualified with a “J” (analyte positively identified, resulting value an estimate) and data qualified with an “NJ” (analyte tentatively identified, resulting value an estimate).

In the complete dataset, most sample sets consisted of 1,200-1,737 samples. Bensulide, methamidophos, methidathion, and naled were analytes in only 4-6 studies. All other parent compounds were analytes in 24-25 separate studies. Data were available for the degradates dichlorvos, disulfoton sulfone, and methyl paraoxon. Azinphos methyl (13.7%), dimethoate (3.8%), and disulfoton (1.8%) were the most commonly detected pesticides. Maximum concentrations for these three compounds were 0.74 μg/L, 0.45 μg/L, and 0.30 μg/L, respectively. Methamidophos was detected in 2.6% of the samples, but this was a smaller dataset (618 samples) and all the detections were from a single study conducted on cranberry bog effluent in 1996. The maximum detected concentration of methamidophos was 0.13 μg/L . Disulfoton sulfone (3.7%) was another of the more frequently detected compounds, with a maximum concentration of 0.28 μg/L. Bensulide, methidathion, methyl parathion, naled, and phorate were not detected in any of the samples. The degradates dichlorvos and methyl paraoxon were not detected either. Ethoprop, fenamiphos, and phosmet were detected in <1% of the samples. Maximum concentrations for these three compounds were 0.06 μg/L, 0.05 μg/L, and 0.08 μg/L, respectively.

Included in the EIM database is a subset of recent monitoring efforts conducted by the Washington Department of Ecology in some of Washington's salmon-bearing streams. Final reports for 2003-2007 seasons are publically available on their website (http://agr.wa.gov/PestFert/natresources/SWM/default.htm). Monitoring was conducted in 2008, but the report is not yet available. A separate summary of data from those investigations is provided below (Table 101 and Table 103). Water samples are not filtered, and thus concentrations reported include pesticides in both dissolved and particulate phases, although the sampling protocol specifies an attempt to avoid collection of excessive particulates (Johnson and Cowles 2003). Whole water concentrations for compounds that sorb significantly to the particulate phase will appear higher than those for a filtered sample, which represents only the dissolved phase.

The Washington program sampled between 6 and 17 sites, depending on the year (Figure 43) (Anderson et al 2007a, Burke et al 2006a, Burke et al 2006b, Johnson and Cowles 2003). Sampling stations were located primarily in agricultural-dominated watersheds. A single watershed, the Cedar-Sammamish (Thornton Creek) represented the urban sites. Three sites were sampled in Thornton Creek in 2003, and 2 sites from 2004-2007. Agricultural sites were distributed in four watersheds (Lower Yakima, Skagit/Samish, Wenatchee and Entiat), but only the Lower Yakima sites have been sampled since 2003. Sites in the Skagit/Sammish watershed were added in 2006 and sites in the Wenatchee and Entiat were added in 2007. Sampling favored the detection of multiple pesticides, rather than peak concentrations in some habitats used by listed salmonids. Generally, samples were taken weekly between March and September at the various sites, but the specific sampling design has changed somewhat over the years. The limited number and spatial distribution of samples sites does not reflect the distribution of listed salmonids in the state. Additionally, NMFS does not believe these sites represent the full range of habitats and potential exposure to pesticides for the ESUs/DPSs located in Washington State and therefore should not be used to represent distribution of pesticide exposure in a probabilistic assessment for salmonids.

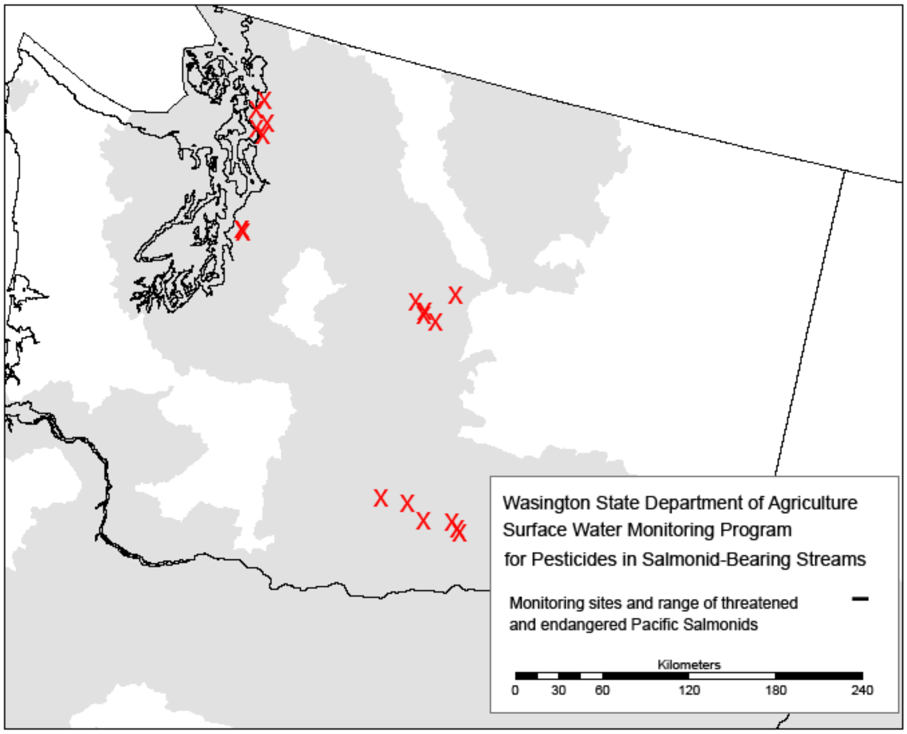


Figure 43. Washington DOE sample sites compared to listed salmon ESUs/DPSs

Table 99 Washington DOE sample sites within the distribution of listed Pacific salmonids.

Species	ESU	Kilometers of Stream Inhabited	Sites in Spawning and Rearing Habitat	Sites in Migratory Corridor
Chinook	California Coastal	2,422.44	NA	NA
	Central Valley Spring - Run	2,212.94	NA	NA
	Lower Columbia River	2,443.29	0	0
	Upper Columbia River Spring - Run	1,646.75	5	0
	Puget Sound	3,639.65	8	0
	Sacramento River Winter - Run	546.84	NA	NA
	Snake River Fall - Run	1,370.44	NA	0
	Snake River Spring/Summer - Run	5,288.23	0	0
	Upper Willamette River	3,013.85	NA	0
Chum	Columbia River	1,162.18	0	0
	Hood Canal Summer - Run	141.89	0	0
Coho	Central California Coast	1,287.78	NA	NA
	Lower Columbia River	3,307.78	0	0
	Southern Oregon and Northern California Coast	5,619.58	NA	NA
	Oregon Coast	10,220.00	NA	NA
Sockeye	Ozette Lake	70.98	0	0
	Snake River	1,493.94	NA	0
Steelhead	Central California Coast	4,620.72	NA	NA
	California Central Valley	4,273.66	NA	NA
	Lower Columbia River	4,302.03	0	0
	Middle Columbia River	10,196.80	6	0
	Northern California	5,324.31	NA	NA
	Puget Sound	3,849.64	8	0
	Snake River	13,423.40	NA	0
	South-Central California Coast	5,104.56	NA	NA
	Southern California	3,015.86	NA	NA
	Upper Columbia River	2,143.15	5	0
	Upper Willamette River	3,063.07	NA	0

With the exception of methamidophos (599 samples), most of these sample sets contain ~1,100 samples. Several pesticides were not detected in any year, including bensulide, methamidophos, methidathion, methyl parathion, naled, and phorate. There were also no detections of the degradates dichlorvos and methyl paraoxon (Table 101) The most commonly detected pesticides were azinphos methyl (5.6%), dimethoate (2.0%), and ethoprop (1.7%). Maximum concentrations for these three compounds were 0.53 μg/L, 0.45 μg/L, and 0.18 μg/L, respectively. Disulfoton, fenamiphos, and phosmet were each detected 1-2 times (≤0.2%). Maximum concentrations for these three compounds were 0.02 μg/L, 0.05 μg/L, and 0.02 μg/L, respectively. However, disulfoton sulfone, which is more persistent than the parent disulfoton, was detected in 1.3% of the 509 samples collected in 2007. Disulfoton sulfone was not an analyte in any other year. Maximum concentration for disulfoton sulfone was 0.06 μg/L.

When these occurrence data are examined more closely, both seasonal and locational patterns begin to emerge. The majority of the detections are in May through September, with a peak in May. Azinphos methyl, dimethoate, and ethoprop were detected in all years. Disulfoton, disulfoton sulfone, fenamiphos, and phosmet were detected in 1-2 years. The majority of detections were in the three waterbodies located in the Yakima watershed: Marion Drain, Sulphur Creek Wasteway, and Spring Creek. This may indicate widespread use of these compounds in this watershed, or may be due to the fact that these are the sites with the most years of data. Azinphos methyl, dimethoate, disulfoton, disulfoton sulfone, and ethoprop were all detected in Marion Drain, and for all these pesticides, this was the waterbody where it was most commonly detected. Azinphos methyl, dimethoate, disulfoton, and disulfoton sulfone were all detected in Sulphur Creek Wasteway. Azinphos methyl, dimethoate, fenamiphos, and phosmet were detected in Spring Creek, and fenamiphos and phosmet were not detected anywhere else. Marion Drain, Sulphur Creek Wasteway, and Spring Creek are all occupied by listed Mid-Columbia steelhead. Azinphos methyl was also detected in Brender Creek and Peshastin Creek. Ethoprop was also detected in Big Ditch and Thornton Creek. Big Ditch and Thornton Creek are occupied by listed Puget Sound Chinook and steelhead.

Table 100. Concentrations of Parent Pesticides in Washington EIM Database.

Statistic	AZM	Bensulide	Dimethoate	Disulfoton	Ethoprop	Fenamiphos	Methamidophos	Methidathion	Methyl Parathion	Naled	Phorate	Phosmet
Samples	1,737	1,102	1,664	1,703	1,691	1,602	618	1,143	1,173	1,136	1,703	1,618
Percent Detections	13.7%	No detects	3.8%	1.8%	0.9%	0.06%	2.6%	No detects	No detects	No detects	No detects	0.2%
LRL range (µg/L)	0.0032-0.347	0.030-17.0	0.014-0.250	0.0032-0.250	0.012-0.300	0.023-0.250	0.030-3.33	0.017-3.33	0.011-0.250	0.030-3.33	0.011-1.200	0.015-0.250
Minimum concentration (µg/L)	0.0003	N/A	0.001	0.019	0.002	0.049	0.001	N/A	N/A	N/A	N/A	0.005
Maximum concentration (µg/L)	0.740	N/A	0.450	0.300	0.064	0.049	0.130	N/A	N/A	N/A	N/A	0.076
Median concentration	0.037	N/A	0.077	0.300	0.025	0.049	0.008	N/A	N/A	N/A	N/A	0.015
Dates	1992-2007	2004-2007	1992-2007	1992-2007	1992-2007	1992-2007	1996-2006	2004-2007	1992-2007	1992-2007	1992-2007	1992-2007
# of Studies	24	5	25	26	26	25	4	6	28	5	25	24

Table 101. Concentrations of parent pesticides detected in recent studies by Washington Department of Ecology (2003-2007)[1]

Statistic	AZM	Bensulide	Dimethoate	Disulfoton	Ethoprop	Fenamiphos	Methamidophos	Methidathion	Methyl Parathion	Naled	Phorate	Phosmet
Samples	1,278	1,102	1,269	1,279	1,279	1,260	599	1,102	1,277	1,108	1,279	1,279
Percent Detections	5.6%	No detects	2.0%	0.2%	1.7%	0.08%	No detects	No detects	No detects	No detects	No detects	0.2%
LRL range (μg/L)	0.003-0.051	0.030-17.0	0.024-0.250	0.0032-0.250	0.012-0.250	0.030-0.250	0.030-3.33	0.017-3.33	0.011-0.250	0.030-3.33	0.011-1.2	0.015
Minimum concentration (μg/L)	0.0003	N/A	0.023	0.023	0.002	0.049	N/A	N/A	N/A	N/A	N/A	0.022
Maximum concentration (μg/L)	0.530	N/A	0.160	0.160	0.180	0.049	N/A	N/A	N/A	N/A	N/A	0.076
Median concentration	0.029	N/A	0.092	0.092	0.033	0.049	N/A	N/A	N/A	N/A	N/A	0.049
Dates	2003-2007	2004-2007	2003-2007	2003-2007	2003-2007	2003-2007	2004-2006	2004-2007	2003-2007	2004-2007	2003-2007	2003-2007

1 Data in this table are a subset of data used to create Table 97

Table 102. Concentrations of degradates in Washington EIM Database

Statistic	Dichlorvos	Disulfoton sulfone	Methyl Paraoxon
Samples	872	562	833
Percent Detections	No detects	3.7%	No detects
LRL range (μg/L)	0.016-0.053	0.017-0.200	0.028-0.344
Minimum concentration (μg/L)	N/A	0.004	N/A
Maximum concentration (μg/L)	N/A	0.280	N/A
Median concentration	N/A	0.014	N/A
Dates	1992-2007	1990-2007	1992-2007
# of Studies	22	5	20

Table 103. Concentrations of degradates detected in recent studies by Washington Department of Ecology (2003-2007)[1]

Statistic	Dichlorvos	Disulfoton sulfone	Methyl Paraoxon
Samples	529	509	529
Percent Detections	No detects	1.3%	No detects
LRL range (μg/L)	0.025-0.051	0.017-0.250	0.031-0.250
Minimum concentration (μg/L)	N/A	0.014	N/A
Maximum concentration (μg/L)	N/A	0.056	N/A
Median concentration	N/A	0.039	N/A
Dates	2003 & 2007	2007	2003 & 2007

1 Data in this table are a subset of data used to create Table 102

Targeted Monitoring Studies

In some cases, EPA documents reported targeted monitoring studies, where water concentrations or the percentage of runoff is associated with particular application rates and/or methods. We describe those in this section, along with some of the information included in more recent EPA documents regarding ambient air concentrations and atmospheric transport of pesticides. Generally, EPA's exposure assessment is focused on concentrations occurring on or near the application site, and does not address transport to more distant habitats. Several of the RLF Effects Determinations included some analysis of atmospheric deposition and/or ambient air concentrations which are described in the *Atmospheric Deposition* section.

A survey of the open literature showed that although the pesticides addressed in this Opinion have been registered for a number of years, few are subject to the intensive research efforts we have seen for the other OPs (NMFS 2008c) and carbamates (NMFS 2009b) addressed in previous Opinions. In some cases, we have located targeted monitoring studies for these pesticides, which are presented below. In other cases, we have located monitoring and/or fate studies for pesticides used in a similar way or with similar physico-chemical properties. We have focused on specific uses and/or modes of transport that may not be adequately addressed by EPA's assessment method and/or the untargeted monitoring databases discussed previously.

Targeted Water Monitoring Reported in EPA Documents

In some cases, registrant submitted studies and/or open literature monitoring studies are reviewed in the EPA RED documents, the EPA BEs, or the EPA California Red-legged Frog (RLF) Effects Determinations, all of which were considered in developing this Opinion

Azinphos methyl

The RED chapter (EPA2006d) references two runoff studies on cotton fields, one conducted in Mississippi (referred to as MRID 425167-01) and one conducted in Georgia (referred to as MRID 425167-02). In the Mississippi study, "a total of 14.9 g of azinphos methyl ran off the 5.2 acre plot in a storm of 3.08 inches on August 9, 1989." Due to differences in the nominal application rate, and the application rate as determined from spray cards in the field, EPA noted that it was impossible to determine exactly how much of the applied azinphos methyl ran off the field. Also, measurements do not specify whether they account for only dissolved phase or also include pesticide sorbed to particulates. However, the EPA reviewer concluded “the runoff event in the study represents a less than one in seven year event on a typical site” and that “between 0.9% and 3.5% of the applied azinphos methyl” was in the runoff. Presumably the storm described was the first rain following application, but that is not clearly stated and there is no indication of length of the time between application and runoff.

The study in Georgia was on a 9 acre plot, isolated from the rest of the watershed in order to measure pesticide deposition and runoff. The watershed itself was 50 acres large, draining into a 3.5 acre pond (depth not reported). Forty-nine acres of the watershed (including the study plot) were cropped field. The field was sprayed with azinphos methyl 8 times, at 3 day intervals, starting August 1. Four storms occurred during the monitoring period (Aug 8, 26, 31, and Oct 1). Specific application rate of azinphos methyl was not reported in any of the EPA documents. Based on two separate calibration methods, 75% or greater of the nominal application rate was deposited on the field. Azinphos methyl in the dissolved phase of the runoff was 0.18-0.24% of applied, resulting in mean concentrations of 2-3 μg/L in the pond. Concentrations are described as highly variable, but no other measurements are given. Following the August 26 storm, a fish kill occurred in the pond, with mortalities reported as 500-1,000 fish (no species noted).

Bensulide

The RED for bensulide (EPA 2006e) mentions a study on runoff and groundwater conducted in Japan, following application of bensulide to turf on a golf course underlain by volcanic ash soils (referred to as (Odanaka 1994, open literature). After fairways were treated with 15 kg/ha (16.8 lb/A) bensulide, concentrations in surface water and leachate water were measured at 3, 10, and 38 days post-treatment. This application rate is comparable with the maximum single application rate of 12-16 lbs a.i./acre permitted on active product labels. Specific concentrations at these times were not reported in the summary, but maximum concentration in the surface runoff was 2,840 μg/L, and maximum concentration in the leachate was 282 μg/L.

Disulfoton

The disulfoton RED (EPA 2006f) reports on a study evaluating the effects of agricultural Best Management Practices (BMPs) in a Virginia watershed (no specific reference given). The summary does not provide application rates, timing, or details of sampling. Based on the table in the summary, it appears only three samples were collected (or perhaps only three contained disulfoton). Measured concentrations ranged from 0.37 - 6.11 μg/L. The watershed is 3,616 acres, half of which is agricultural, and half of which is forested. No other details were provided.

Ethoprop

The only monitoring data reported in the ethoprop RED (EPA 2006g) in addition to sources already discussed in the Opinion are water quality monitoring conducted by the South Florida Water Management District (SFWMD), in 1988-1993. At that time, ethoprop was used on sugarcane, all of which is grown within the SFWMD. No ethoprop was detected during the sampling program (detection limits 0.06-0.73 μg/L).

Fenamiphos

The fenamiphos environmental risk assessment (EPA 1999) did not report water concentrations for any targeted monitoring studies. It does, however, report some incidents, and notes the following:

"The screening level risk assessment indicates that for all current registered fenamiphos uses and application rates, aquatic communities (fish and invertebrates) downgradient of runoff from the application site are expected to be adversely affected," and,

"Based on these incidents, EFED concludes that use of Nemacur 10G on golf courses can cause fish kills even when the product is used in accordance with current label directions and restrictions."

Dimethoate, Methamidophos, Methidathion, Methyl Parathion, Naled, Phorate, and Phosmet

No targeted monitoring studies were described in either the BE, RED chapter or the RLF effect determination that was not previously discussed in this Opinion under either the monitoring databases section or in this targeted monitoring section.

Open Literature Targeted Water Monitoring Studies

This section includes targeted studies located in open literature, and analyses on some of the more targeted studies contained in the Washington and California databases. Many of these studies are associated with a particular agricultural practice.

Rice Chemicals

Of the 12 pesticides addressed in this Opinion, only methyl parathion currently has rice uses. Within the action area, rice is grown in several California counties, including Colusa, Glenn, Butte, Sutter, Yuba, Placer, and Yolo counties in the Sacramento River Basin, and San Joaquin, Merced, and Fresno counties in the San Joaquin River Basin (www.nass.usda.gov/Charts_and_Maps/Crops_County/index.asp#ar, Accessed 8/19/09, PDD). The highest concentration of rice production is in Colusa County. Rice is generally flood irrigated, and pesticides applied to the crop may be contained in the effluent when it is released. Generally, direct water application rates are higher than concentrations derived from field runoff.

The CDPR database contained data from a number studies evaluating concentrations of rice chemicals in the Colusa Basin Drain, Butte Slough, and the Sacramento River at Village Marina (Table 104), which receive the irrigation returns from rice-growing areas (referenced in their metadata as studies 17, 30, 34, 40, 53, 67, 73 and 75). California has established water holding times for most chemicals used on rice. At the time many of these studies were conducted, the holding time for methyl parathion was 24 days (Gorder and Lee 1995). A study on water from the Colusa Basin Drain using *Ceriodaphnia dubia* as the test organism in a Toxicity Indicator Evaluation (TIE) procedure found that an extract of Colusa Basin Drain water caused toxicity in laboratory tests (Norberg-King et al 1991). The procedure indicated that methyl parathion (4.1 μg/L) and carbofuran (8.2 μg/L) accounted for the toxicity of the sample, although other chemicals such as molinate and thiobencarb were present as well. Detectable concentrations of methyl parathion, ranging from 0.09-0.14 μg/L were found in samples from the Colusa Basin Drain in 1995-1998, primarily in the months of May and June. A single detectable concentration (0.19 μg/L) occurred in Butte Slough in June of 1995. Methyl parathion was not detected in any of the samples from any of the sites in 1999-2002. For these studies, sampling appears to have been conducted twice a week from May to July. LOD for the studies ranged from 0.05-0.10 μg/L.

Table 104 Methyl Parathion Concentrations in CDPR Studies of Rice Effluents (1995-2002)

Statistic	Colusa Basin Drain	Butte Slough	Sacramento River at Village Marina
Samples in data set	203	114	114
Quantifiable samples	11	1	0
% of quantifications in data set	5.4%	0.9%	0%
LOQ range (μg/L)	0.05-1.0	0.05-1.0	0.05-1.0
Minimum concentration (μg/L)	0.059	0.187	N/A
Maximum concentration (μg/L)	0.137	0.187	N/A
Median concentration	0.079	0.187	N/A

Colusa Basin Drain empties into the Sacramento River near Knight's Landing. In a study conducted on other rice chemicals (carbofuran, molinate, and thiobencarb) in 1990-1993, investigators found that chemical concentrations were 1-2 orders of magnitude lower at the sampling site in the Sacramento River near Sacramento (downstream of both the Feather River and American River confluences with the Sacramento River) than they were at the Knight's Landing sampling site. Although no listed salmonids occupy Colusa Basin Drain, Sacramento winter-run and Central Valley spring-run Chinook, Central Valley steelhead are found in the Sacramento Basin. The Colusa Basin Drain is likely a conduit for agricultural pesticides. We found no studies specifically addressing rice chemicals that were conducted in the San Joaquin River Basin.

Cranberry Chemicals

Cranberries are grown in bogs, which are periodically flooded for pest control and harvesting. Of the 12 a.i.s addressed in this Opinion, phosmet is the only one currently approved for use on cranberries, although methamidophos was previously registered for this use. Cranberries are grown in both Oregon and Washington. It is worthwhile to note that in the monitoring data for methamidophos, the only time it was detected was during a study in Washington State (1996), specifically addressing cranberry bog pesticide effluents. Methamidophos is no longer registered for cranberries, but the detections may be illustrative of how the bog effluents can contain higher concentrations of chemicals than might be in typical agricultural field runoff. This study did not include phosmet, and there are no other studies specifically directed at cranberry uses.

Back Sloughs and Small Upland Drainages

A study conducted in the Sacramento-San Joaquin river delta region used *Ceriodaphnia dubia* (cladoceran) bioassays to determine water toxicity at various types of sites (Werner et al, 2000). A TIE procedure was then used to determine what pesticides were causing the toxicity. In this study, they evaluated five types of sites: main-stem rivers, representative delta island drains, back sloughs and small upland drainages, urban runoff-receiving water bodies, and points along the pathways of water movement across the delta. They found the greatest numbers of toxic samples occurred in back sloughs and small upland drainages. These samples caused both acute and chronic (reproductive) toxicity to the *C. dubia* test organisms. Toxic samples occurred throughout the year, based on two years (1993-1995) of sampling the sites once a month. In this particular case, TIEs identified nonpolar organic pesticides as the primary toxicants, specifically OPs (chlorpyrifos, diazinon, malathion) and carbamates (carbaryl, carbofuran). The key finding of this study is that OPs were responsible for toxicity in the field, particularly in small streams and sloughs. These pesticides were addressed in previous Opinions produced by NMFS (NMFS 2008c, NMFS 2009b).

An extensive review of exposure and effects studies conducted by Schulz (Schulz 2004) came to a similar conclusion regarding the vulnerability of small catchments after doing a (log-transformed) linear regression of the relationship between aqueous-phase insecticide concentrations and catchment size (n=133, p=0.0025). Schulz also noted that some studies suggest a correlation between the amount of a chemical applied in a catchment and occurrence in water samples. The studies he evaluated specifically addressed agricultural uses rather than urban runoff. His conclusions can be applied to salmonids in two ways. First, small, intensively cultivated catchments represent a situation where high concentrations of pesticides are likely to occur in the water. Second, since concentrations are associated in part with usage, specific chemicals and amounts of the chemicals in the water will change over time, as pest pressures and market forces modify the suite of compounds used.

With the exception of azinphos methyl (n=13), most of the OPs addressed in this Opinion were not well represented in Schulz's fate data set. There were four studies each for dimethoate and methyl parathion, two for methidathion, and one each for disulfoton and dichlorvos. Table 105, below, summarizes his findings.

Table 105 Concentrations of Pesticides Measured in Agricultural Use Area (after Schulz 2004)

Pesticide	Concentration Range (μg/L)	Pesticide Source	Location	Original Reference
Azinphos methyl	0.001-0.016	Nonpoint	Ioannina Lake, Greece	Albanis et al 1986
	0.001-0.025	Nonpoint	Kalamas River, Greece	Albanis et al 1986
	0.06-1.0	Nonpoint	Orchard wetlands, Ontario	Harris et al 1998
	0.02-0.1	Runoff	Berg and Franschock Rivers, South Africa	Schulz 2003
	0.07-0.38	Runoff	Lourens River, South Africa	Schulz and Liess 2001a
	0.06-1.5	Runoff	Lourens River and tributaries, South Africa	Schulz and Liess 2001b
	0.39-0.60	Runoff	Lourens River subcatchments	Dabrowski et al 2002a
	0.14-0.8	Runoff	Lourens River tributary, South Africa	Schulz and Peall 2001
	0.1-7.0	Runoff	Estuarine sites, South Carolina	Scott et al 1999
	0.002-21	Runoff	Estuarine sites, South Carolina	Finley et al 1999
	1.1-2.6	Spray drift	Lourens River tributary, South Africa	Schulz et al 2001b
	0.03-0.05	Spray drift	Lourens River, South Africa	Schulz et al 2001b
	0.36-0.87	Spray drift	Lourens River tributary, South Africa	Schulz et al 2001c
Dichlorvos	0.1-0.3	Nonpoint	Tama River, Japan	Kikuchi et al 1999
Dimethoate	0.2	Rice fields	Shinano River, Japan	Tanabe et al 2001
	0.05-0.1	Nonpoint	San Joaquin River and tributaries, California	Pereira et al 1996
	0.1-30	Nonpoint	Vemmenhög subcatchment, Sweden	Kreuger 1998
	0.01-11.6	Nonpoint	Farm ditches, British Columbia	Wan et al 1994
Disulfoton	0.1-0.4	Runoff	Shell Creek, Nebraska	Spalding and Snow 1989
Methidathion	0.03-9.2	Runoff	San Joaquin tributaries, California	Domgalski et al 1997
	0.01-0.6	Runoff	Sacramento-San Joaquin catchment, California	Kuivila and Foe 1995
Methyl parathion	0.001-0.12	Nonpoint	Ioannina Lake, Greece	Albanis et al 1986
	0.002-0.032	Nonpoint	Kalamas River, Greece	Albanis et al 1986
	0.4-213	Aerial application	Vineyard catchments, Southwestern Germany	Aufsess et al 1989
	0.01-0.49	Runoff	Moon Lake catchment, Mississippi	Cooper 1991b

Several of the studies Schulz considered addressed pesticides in the San Joaquin and Sacramento River basins, which contain listed salmonids including Sacramento winter-run and Central

Valley spring-run Chinook, and Central Valley steelhead. Pereira *et al.,* evaluated concentrations of current use pesticides, legacy pesticides and PAHs in water, sediments, and clams at locations in the San Joaquin Valley (Pereira et al 1996). Dimethoate was one of the analytes in the study, and was not detected in either sediments or clams, but was detected in the water at two of the sites. Samples were collected only once, so no temporal trends can be discerned. Sample date was not provided, other than year (1992). Dimethoate was detected in Orrestimba Creek (0.101 μg/L), which drains a primarily agricultural watershed, and in the mainstem San Joaquin River at Patterson (0.051 μg/L). In both cases, atrazine and simazine were also detected (0.025-0.035 μg/L) as well as various PAHs (t-PAH 0.016-0.019 μg/L). Diazinon, another OP, was also detected in the San Joaquin River (0.012 μg/L). Presence of dimethoate was not linked with any particular crop or agricultural practice.

While many of these studies do not provide sufficient information to correlate concentrations in the streams and rivers to pesticide applications on specific crops or at specific rates, they do illustrate a number of important points regarding the exposure of listed salmonids to pesticides. Based on the studies we reviewed, and Schulz' (2004) review of other papers we draw the following conclusions: 1) small waterbodies or waterbodies in small catchments, especially if they are intensively cultivated areas, may receive high concentrations of pesticides in runoff or drift, and 2) mixtures of OPs and mixtures containing OPs and other toxic chemicals have been shown to occur in areas occupied by listed salmonids. Some of these small waterbodies are important areas for rearing or spawning.

Dormant Orchard Spray Pesticides

Two other studies discussed by Schulz targeted pesticides used in dormant orchard sprays. Dormant sprays were targeted because they are applied during the winter (rainy season in California), and concentrations in runoff were anticipated to be higher than might occur during drier months. Both of these studies were conducted in 1993, the same year as the Pereira study described above.

The Domgalski study (Domgalski et al 1997) was concerned with the differences between pesticide runoff in the eastern and western portions of the San Joaquin Valley, and how the different land uses, soil types, and inputs from non-agricultural upstream basins affect pesticide concentrations in storm water runoff. As the small watersheds in this study did not occur in the orchard areas where methidathion was heavily used, they did not test for it at several sites. At sites where methidathion was sampled, detectable concentrations were found, and ranged from 0.3 μg/L (Spanish Grant Drain) to 9.2 μg/L (Central California Irrigation District Canal (CCIDC)). This study noted that methidathion concentrations in the CCIDC remained high for nearly two weeks following the rain event, unlike diazinon concentrations, which peaked quickly and then decreased.

Kuivila and Foe (1995) measured dissolved phase concentrations of pesticides at a series of sites along the Sacramento and San Joaquin Rivers, through the Sacramento-San Joaquin Delta, and into San Francisco Bay. These areas contain listed salmonids, including Sacramento winter-run and Central Valley spring-run Chinook, and Central Valley steelhead. In addition to methidathion, they measured the concentrations of other OPs, including ethyl parathion (no longer registered for use), diazinon, chlorpyrifos, and malathion. Methidathion is the most water-soluble of these a.i.s (250 mg/L @20°C), increasing the likelihood that it will be found in the dissolved phase. Water samples were taken daily throughout January and February of 1993 and pulses were followed from the agricultural areas to the estuary, making this one of the few studies where clear temporal and transport trends can be discerned. Sampling sites included tidally influenced areas in the delta. Discharge data from USGS gaging stations and additional discharge sensors were used to correlate streamflow and pesticide concentrations with rainfall. Discharge at tidally influenced areas was corrected for tidal flux. The method detection limit for methidathion was 0.040 μg/L. Two groups of rainstorms occurred during or before the sampling period, some in late December and early January, and others in February. Pesticide applications were made before the first set of storms, and in the two dry weeks in late January. Sampling was not linked to a specific application, and the CDPR application data for 1993 were not yet available at the time the paper was published.

Methidathion and diazinon were the only compounds detected in the Sacramento River at Sacramento, and they were detected in February but not January. The maximum measured concentration of methidathion was 0.212 μg/L, occurring on February 12. A second, smaller peak of 0.071 μg/L occurred in late February. Both peaks correlated roughly with peaks in discharge. Diazinon peaks occurred on the same date, but concentrations were greater (maximum 0.393 μg/L, also occurring on February 12) (Kuivila and Foe 1995).

Some of the target pesticides were detected in the San Joaquin River in both January and February. Methidathion, chlorpyrifos and diazinon were detected in February, but only diazinon was detected in January. The peak methidathion concentration was 0.586 μg/L, occurring on February 10. Diazinon peaks of 0.733μg/L (Feb 8) and 1.070 μg/L preceded and followed the methidathion peak (Feb 11). Chlorpyrifos peaked at 0.042 μg/L, on February 12 (Kuivila and Foe 1995).

Using discharge data, the authors estimated dissolved phase pesticide loading moving toward the San Francisco Bay. From the Sacramento River, integrated loads for each peak produced estimated input of 177 kg for methidathion and 290 kg for diazinon in the month of February. Total estimated input from the San Joaquin River (January-February) was 12 kg for methidathion and 92 kg for diazinon. The authors did not calculate chlorpyrifos input. The methidathion and diazinon pulses from the Sacramento River were followed into the delta, and showed detectable, although much lower concentrations even at the furthest sample point, Martinez (119 river km from the Sacramento River at Sacramento station). Movement of the pulse from river into the delta ranged from 5 to 8 days. The pulse moved more slowly and the speed of movement became more variable as it moved seaward (Kuivila and Foe 1995).

As a final component of the study, the authors conducted a *Ceriodaphnia dubia* 48-hr acute bioassay on samples collected from the Sacramento River at Rio Vista and the San Joaquin River at Vernalis in the month of February. Samples were split, and pesticide analysis was also conducted. Multiple days of 100% mortality events were noted in both rivers (Sacramento River Feb 12-14; San Joaquin River Feb 8-19). Mortality events corresponded with higher pesticide concentrations, especially of diazinon. Authors note the toxicity in the bioassays “appears to be slightly higher than would be predicted from the diazinon concentrations alone” (Kuivila and Foe 1995). They do not attempt to apportion the toxicity using either a TIE or TEQ approach, but the presence of additional OPs and/or the herbicide simazine are likely contributory.

These data show clearly that dormant orchard spray chemicals are in the mainstem rivers in California at measurable concentrations during the winter months, and that binary and tertiary mixtures of OPs can occur. Concentrations actually occurring in the water column are related to the amount of chemical applied, physico-chemical properties of the pesticide, and proximity/connectivity of the application site to the sample site. These pesticides were transported into the estuary, and occurred at sufficiently high concentrations to cause mortality to water column organisms. Data are relatively dated (1993) and changes in chemicals used, application rates, and management practices may have occurred. However, we have not located a more recent study of this quality, thus it remains relevant for characterizing pesticide transport and toxicity in the California agricultural area.

Targeted Monitoring in ESA-listed Salmonid Habitats: Hood River Oregon

A group of field studies evaluated macroinvertebrate community responses in the orchard-dominated Hood River Basin, Oregon and correlated results with azinphos methyl and chlorpyrifos use and detections (Grange 2002, St. Aubine 2004, Vander Linde 2005). Hood River Basin contains several listed anadromous salmonids, including Lower Columbia River steelhead.

Two sets of field experiments directly investigated effects on juvenile steelhead (hatchery-reared) and aquatic invertebrate communities in exposed streams (Grange 2002, St. Aubin 2004). The studies analyzed water samples for chlorpyrifos, azinphos methyl, and malathion before, during, and after orchard spray periods. None of the pesticides were detected at reference sites. Both azinphos methyl (0.03 - 0.27 μg/L) and chlorpyrifos (0.08 - 0.20 μg/L) were frequently detected at orchard stream and river sites (Grange 2002, St. Aubin 2004). One site showed chlorpyrifos ranging from 0.032 - 0.183 μg/L over an eight day period (Van der Linde 2005), indicating that extended pulses can occur.

Atmospheric Deposition

A percentage of pesticide applied to a crop may volatilize and be transported a significant distance from the application site by various atmospheric processes. The fate of a chemical in the atmosphere depends greatly on its physico-chemical properties (especially the Henry's Law constant, $t_{1/2}$, and K_{ow}), prevailing winds, season and amount of application, and temperatures at the application site and receiving areas. Currently, EPA's standard assessment process does not account for this atmospheric transport and therefore exposure and subsequent risk to salmonids may be underestimated. Some of the California RLF BEs do consider atmospheric transport, generally in terms of air monitoring or concentrations in rain, snow, or fog. However, connecting atmospheric concentrations to surface water concentrations is difficult.

In 2002, chlorpyrifos, diazinon, trifluralin, and other pesticides were detected in air samples collected from Sacramento, California (Majewski and Baston 2002). Azinphos methyl was detected in two CDPR studies in Kern and Glenn counties (CA) conducted in 1987 and in 1994, but not detected in rainwater or snow (EPA 2007d). EPA concluded that since Kern and Glenn counties are major agricultural locations, and azinphos methyl has not been detected in studies conducted at higher elevations, and it has relatively low volatility, these detections are "likely reflective of near field (spray drift) exposure and are not indicative of long-range transport" (EPA 2007d). Atmospheric background levels (samples taken prior to application) of naled have been detected (EPA 2008g). EPA concluded they were likely the result of other uses in the area, or drift from a neighboring air shed with recent aerial uses. Air samples obtained immediately after local spraying have much higher naled and diclorvos (DDVP) concentrations (EPA 2008g).

Majewski *et al.* evaluated the role of atmospheric deposition in the San Joaquin Valley (Majewski et al 2006). In the 2004-2006 sample years, they considered both wet and dry depositions, and evaluated 6 sites, including both urban and agricultural areas. Of the OPs considered in this Opinion, methidathion and azinphos methyl were most commonly detected in the wet (rainfall) samples (n=137). Methidathion occurred in 39% of the samples (max 0.317 μg/L, mean 0.043 μg/L). Azinphos methyl (max 0.322 μg/L, mean 0.043 μg/L) occurred in 26%. Others also detected, included phosmet (20%), methyl parathion (17%), and dimethoate (5%). Several degradates of the OPs appeared in a number of samples; dichlorvos (10%), methyl paraoxon (2%), phosmet oxon (1%), and azinphos methyl oxon (1%). Fenamiphos, phorate, fenamiphos sulfone, fenamiphos sulfoxide, and phorate oxon were all analyzed for but not detected. Importantly, 5 pesticides (dacthal, simazine, diazinon, chlorpyrifos, and pendimethalin) were detected in ≥85% of samples, indicating that rainfall inputs a mixture of compounds into aquatic systems.

Pesticides were detected in rain (Capel et al 1998) and snow samples from Mount Rainier National Park, Washington (Hagemen et al 2006). Three of the four most frequently detected pesticides were found in the Mount Rainier snow (dacthal, chlorpyrifos, and endosulfan).

Transport from California's Central Valley

Beginning in the early 1990's, some research efforts have been directed at elucidating the fate and transport processes of OP pesticides applied to California's Central Valley, and their potential impacts on ecosystems in the Sierra Nevada mountains. Aston and Seiber (1997) evaluated the fate of OP pesticides applied during the summer, and found measureable quantities of methidathion, and methidathion oxon, as well as chlorpyrifos and chlorpyrifos oxon in the air and pine needles of two sites in Sequoia National Park. Concentrations were compared to baseline concentrations at a site in the Central Valley approximately 15-20 miles from the sites in Sequoia National Park. Park sites were at higher elevations (533 m and 1,920 m) than the baseline site. While they found concentrations of the parent compounds, one of their conclusions was that the oxons were generally transported further than the parents. This is due, in part, to the fact that the oxons are formed in the atmosphere by photodegradation, and also because they have longer half-lives in the atmosphere than the parents.

A study of wet deposition (rain and snow) conducted at the same locations in Sequoia National Park, and an additional location near Lake Tahoe in December 1995 through April 1996 found "residues of all of the currently used pesticides (trifluralin, chlorothalonil, chlorpyrifos, endosulfan, diazinon, and malathion) on our analytical target list as well as some of the organo-chlorine pesticides, α- and γ-HCH, were observed in at least some rain and snow samples" (McConnell et al 1998). Chlorothalonil, chlorpyrifos, α-endosulfan, α- and γ-HCH were also detected in water samples taken in Lake Tahoe in June. Chlorothalonil and chlorpyrifos were most commonly detected, and were detected in the highest concentrations. Based on an analysis of isomer patterns for the legacy organochlorines, authors concluded a global background loading in the air was the source. Legacy pesticides are substances banned from current use, but which still persist and cycle globally as the parent compound or a readily identifiable degradate.

In summary, they note “results of this study have serious implications for the Sierra Nevada Mountains, as this area receives the majority of water inputs through rain and snow.”

Following the evaluation of winter wet deposition (McConnell et al 1998), a second study considered summer dry deposition (chemicals sorbed to airborne particulate matter) (LeNoir et al 1999). This study included measurement of concentrations in air, concentrations in dry deposition, and concentrations in the surface water of Moro Creek. Air samples were taken monthly at three elevations between May and September of 1996. Particulate samples were taken at least twice at each location over the summer months. Surface water samples were taken along an elevational transect, at eight points extending from the upper creek at 3,322 m down to an elevation of 118 m. “Residues of all currently used pesticides analyzed (trifluralin, chlorothalonil, chlorpyrifos, endosulfan, diazinon, and malathion) were observed at once during the summer” in air samples (LeNoir et al 1999). They measured concentrations the chlorpyrifos oxon, and usually it was present in air at concentrations greater than the parent chlorpyrifos. Concentrations are similar to but differ slightly from Aston and Sieber (1997), which LeNoir, *et al*. (1999) attribute to differences in the sampling protocols. Chlorothalonil, chlorpyrifos, chlorpyrifos oxon, α-endosulfan, and β-endosulfan were all present at measurable concentrations, although the endosulfans occurred in lower concentrations than the other compounds. Similar to air samples, the chlorpyrifos oxon was consistently present in higher concentrations than the parent chlorpyrifos. All pesticides were present in measurable quantities (LODs ranging from 0.00002 – 0.0023 μg/L, depending on the compound) at nearly all stream locations sampled, although some were not detected at elevations >3,200 m. Concentrations of all compounds “dropped significantly in surface water above the 2,040 m elevation.” Consistently, chlorpyrifos and chlorothalonil appeared in the highest concentrations. The chlorpyrifos/chlorpyrifos oxon ratio in the surface water was the reverse of the air, which authors attribute to differences between aquatic and atmospheric reaction processes.

In 1997, researchers investigated the possibility that atmospheric deposition of current-use pesticides, including two OPs (chlorpyrifos and diazinon) was a contributor to declines in populations of mountain yellow-legged frogs (*Rana muscosa*) in the Sierra Nevada Mountains (Fellers et al 2004). Two populations of frogs were compared: a population in the Tablelands, which is exposed to prevailing winds from the Central Valley, and a population in the Sixty Lakes Basin, which is protected from those winds. Frog populations in Tablelands were extirpated in the late 1980s or early 1990s, and reintroduced populations have failed to thrive. Populations in the Sixty Lakes Basin have not suffered the same declines. Investigators measured concentrations of several current use pesticides known to transport atmospherically (chlorpyrifos, diazinon, α- and β-endosulfan, and endosulfan sulfate) in both water and frog tissue at both sites. They also measured concentrations of several legacy pesticides, including α- and γ-chlordane, α- and γ-hexachlorocyclohexane, *trans*-nonachlor, two parent DDT isomers and the DDT degradate dichlorodiphyldichloroethylene (DDE). Presence of these compounds in an area not expected to otherwise contain pesticide residues from direct application or runoff is an indicator of contamination via an atmospheric background source. Investigators found concentrations of current use pesticides in the water ranging from 0.40 ng/L (β-endosulfan) to 12 ng/L (chlorpyrifos) in the Tablelands sites and ranging from 0.17 ng/L (β-endosulfan, chlorpyrifos) to 1.8 ng/L (β-endosulfan, diazinon) in the Sixty Lakes sites. Generally, water concentrations of most of the current use pesticides were 2-3 times higher at the Tablelands sites. The legacy pesticides were detected in water far less frequently, and in lower concentrations (maximum 0.46 ng/L) than the current use pesticides. However, the tissues of 20 frogs from each location were analyzed, and most contained residues of from 1 to 3 of the legacy pesticides at concentrations of 0.5-12 ng/g wet weight. In addition, all frogs from both sites contained residues of DDE (13-100 ng/g wet weight in Tablelands, 3.4-27 ng/g wet weight in Sixty Lakes). While the decline of the frogs cannot be specifically attributed to pesticides, this study does present clear evidence that atmospherically transported pesticides can be deposited in locations that might not otherwise be exposed to pesticides, that these pesticides may be present in a variety of mixtures, and that legacy pesticides continue to be present in environmental matrices and biota.

While only one of the chemicals addressed in this Opinion was measured and detected in the studies discussed above, we find this body of work to provide convincing evidence that regional atmospheric processes can provide sufficient pesticide input into aquatic systems remote from the application site to warrant concern.

Summary of Monitoring Data

NMFS did not locate many edge-of-field studies for the compounds addressed in this Opinion. The open literature evaluated for the Opinions rendered on other OPs (NMFS 2008c) and carbamates (NMFS 2009b) and the general state of knowledge regarding field runoff from pesticide applications, leads us to anticipate the following:

- edge-of-field runoff concentrations will be higher than concentrations measured in waterbodies with substantial diluting volume,
- low-flow or runoff-dominated systems likely contain the highest concentrations (approaching or exceeding modeled concentrations), and
- measured concentrations in general monitoring programs are likely to be lower than peak runoff concentrations, as sampling may not coincide with initial application and/or runoff events.

Based on studies evaluated for this Opinion, we also note that small waterbodies or water bodies in small catchments, especially if they are intensively cultivated areas, may receive high concentrations of pesticides in runoff or drift. Several studies also showed mixtures of OPs and mixtures containing OPs and other toxic chemicals in water bodies, some of which did contain listed salmonids. In some cases, concentrations of pesticides remained elevated for several days. Additionally, pesticides applied in one location may be regionally transported via rivers or the atmosphere to more distant salmonid habitats in ecologically significant concentrations.

Table 106 Monitoring Data Concentrations

Chemical	From Databases[1]				From Targeted Studies	
Parent compounds	Min Conc. µg/L	Monitoring Database	Max Conc. µg/L	Monitoring Database	Max Conc. µg/L	Study Reference
Azinphos methyl	0.0003	WEIM	7.35	NAWQA	21	Finley et al 1999
Bensulide	ND	ND	ND	ND	2,840	EPA 2006e
Dimethoate	0.001	WEIM	11.31	CDPR	30	Schulz 2004
Disulfoton	0.004	NAWQA	3.81	NAWQA	6.1	EPA 2006f
Ethoprop	0.001	NAWQA	5.75	NAWQC	NL	NL
Fenamiphos	0.007	NAWQA	1.50	DPRA	NL	NL
Methamidophos	0.001	WEIM	0.13	WEIM	NL	NL
Methidathion	0.001	CDPR	15.10	CDPR	9.2	Domgalski et al 1997
Methyl parathion	0.003	NAWQA	1.70	CDPR	213	Schulz 2004
Naled	ND	ND	ND	ND	NL	NL
Phorate	0.012	NAWQA	0.22	CDPR	NL	NL
Phosmet	0.005	WEIM	0.63	CDPR	NL	NL
Degradates						
Dichlorvos	0.004	NAWQA	0.54	CDPR	0.3	
AZM Oxon	0.014	NAWQA	0.04	NAWQA	NL	NL
Methidathion Oxon	ND	ND	ND	ND	NL	NL
Methyl Paraoxon	0.006	NAWQA	0.01	NAWQA	NL	NL
Phorate Oxon	ND	ND	ND	ND	NL	NL
Phosmet Oxon	0.007	NAWQA	0.05	NAWQA	NL	NL
Disulfoton Sulfoxide	ND	ND	ND	ND	NL	NL
Disulfoton Sulfone	0.006	NAWQA	0.23	NAWQA	NL	NL
Fenamiphos Sulfoxide	0.022	NAWQA	0.04	NAWQA	NL	NL
Fenamiphos Sulfone	0.008	NAWQA	0.01	NAWQA	NL	NL

[1]Minimum and maximum based on detected values
ND Not detected
NL No targeted monitoring study reporting this compound located

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.

Metabolites and degradates of the 12 a.i.s

EPA identified major degradates, and degradates of toxicological concern for most of the 12 a.i.s. Quantitative exposure estimates for these compounds were generally not provided in the

BEs. Exceptions included a Total Toxic Residue (TTR) approach to estimate the combined residues of parent and some of the toxic degradates of disulfoton and phorate (Table 87). The combined residues of parent disulfoton and the sulfone and sulfoxide degradates in surface water were estimated to range from 14.9 – 43.5 μg/L for the modeled scenarios. A similar approach was taken in a more recent EPA assessment for California where peak estimated concentrations ranged 0.66 – 66.7 μg/L for TTR of disulfoton (EPA 2008e). The phorate BE estimated TTR for phorate and the sulfoxide and sulfone degradates at 7.7 - 138 μg/L for the modeled scenarios (EPA 2003e). A more recent EPA assessment estimates peak combined concentrations of phorate and its degradates at 0.293 – 11.17μg/L for pesticides uses registered in California (EPA 2008h).

Although the BE did not provide quantitative estimates of exposure to naled degradates, a more recent EPA assessment provides estimates of aquatic concentrations for the combined residues of naled and DDVP. This assessment estimates peak concentrations of total naled residues ranging from 1- 33 μg/L for uses approved in California (2008g).

The BEs recognized that listed salmonids are likely exposed to several other metabolites and degradates of the 12 a.i.s. However, estimates quantifying exposure to these transformation products were not provided and remain a considerable source of uncertainty. In general, failure to consider exposure to these breakdown products increases the likelihood that risk is underestimated (Table 84).

Other ingredients in formulated products

Registered pesticide products containing the 12 a.i.s generally include other ingredients such as carriers, surfactants, and synergists. NMFS reviewed active labels of the 12 a.i.s and found three active pesticide products that contain multiple a.i.s., including two bensulide products which contained oxadiazon, and one disulfoton product that contained pentachloronitrobenzene and etridiazole (Table 107). Several other products are also formulated with petroleum distillates and other solvents, but did not indicate the concentration in the formulation as needed to estimate potential exposure in aquatic environments. Other ingredients in the formulation were not specified.

Table 107 Examples of pesticide product ingredients.

EPA Product Registration Number	Active Ingredients	Other Ingredients
538-164, 9198-176	bensulide, oxadiazon	unspecified
400-408	Pentachloronitrobenzene, etridiazole, disulfoton	unspecified
432-1286	disulfoton	fertilizers and other unspecified
264-734	disulfoton	petroleum distillates and other unspecified
1063-196, 1063-200, 1063-205, 2217-696	bensulide	petroleum distillates and other unspecified
66330-244, 66330-245, 19713-232	dimethoate	xylene range solvents and other unspecified
9779-273	dimethoate	petroleum distillates and other unspecified
264-458	ethoprop	petroleum distillates and other unspecified
10163-238	methidathion	xylene range solvents, petroleum distillates, and other unspecified
4787-48, 70506-193, 67760-43	methyl parathion	petroleum distillates and other unspecified
5481-479, 5481-480, 5481-481,	naled	petroleum distillates and other unspecified
10163-215	phosmet	aromatic solvents, petroleum distillates, other unspecified

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. NP has been linked to endocrine disrupting effects in aquatic systems. 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. The median concentration of NP in streams surveyed was 0.8 μg/L and the maximum concentration detected was 40.0 μg/L (Table 108). Related compounds were also detected at a relatively high frequency (Koplin et al 2002).

Table 108 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 the 12 a.i.s 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, chloropictrin, 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.

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 (Table 109). Although mixtures with other pesticides products are not specifically recommended on many labels, 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.

Table 109. Examples of label recommended tank mixtures.

Pesticide Products containing	Tank mixture recommendation
Azinphos methyl	Compatible with summer oils, many registered pesticides, and liquid fertilizers.
Bensulide	Mix with another herbicide (ALANAP) for a broader spectrum of weed control.
Dimethoate	Compatible with other registered pesticides including: azinphos methyl, malathion, parathion, carbaryl, and diazinon; pyrethroids, dicofol, captan, thiram, zineb, dodine, endosulfan, and others.
Disulfoton	Compatible with many registered pesticides and liquid fertilizers.
Methyl parathion	Mix with a pyrethroid or other non-organophosphate insecticide to control whitefly. Mixing with other products is also recommended for control of *Heliothis* species.
Phosmet	Recommendations for mixing with dimethoate products for pest control in alfalfa, and mixing with various adjuvants such as stickers, extenders, and dormant spray oils.

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 2006) “More than 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,” (Gilliom et al 2006). The likelihood of exposure to multiple pesticides throughout a listed salmonids’ lifetime is great, considering their migration routes and habitats occupied for spawning and rearing. In a three-year monitoring study conducted by the Washington DOE, pesticide mixtures were found to be common in both urban and agricultural watersheds (Burke et al 2006b). An average of three pesticides was found in each sample collected on urban sampling sites, with as many as nine pesticides found in a single sample. Agricultural sites averaged three to five pesticides per sample, with as many as 14 pesticides being detected in a single sample (Burke et al 2006b). Mixtures of chemicals that share a common mode or mechanism of action are of particular concern to NMFS. Six to 11 million lbs of cholinesterase-inhibiting insecticides are used annually in California (CDPR 2007a). Potential effects of multiple co-occurring cholinesterase inhibitors and/or other pesticide mixtures which might have potentiating or synergistic effects were not addressed in the BEs. Cholinesterase inhibiting insecticides, including carbaryl, diazinon, chlorpyrifos, and malathion are the most frequently detected mixtures in urban streams across the U.S. (Gilliom et al 2006).

Gilliom and others (2006) suggested that assessment of pesticide mixture toxicity to aquatic life is needed given the widespread and common occurrence of pesticide mixtures, particularly in streams, because the total combined toxicity of pesticides in water is often greater than that of any single pesticide compound. 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. CDPR's most recent pesticide use report indicates 6,857,530 lbs of cholinesterase-inhibiting insecticides were applied in California during 2006. It also notes over 60 cholinesterase-inhibiting a.i.s are currently registered in California (CDPR 2007a). Exposure to these compounds and other baseline stressors (*e.g.,* thermal stress) was not a consideration in the BEs, which only considered effects from single a.i.s. Therefore, risk to listed species may be underestimated in EPA's assessments.

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. Many of the a.i.s and degradates addressed in this Opinion, especially azinphos methyl, dimethoate, ethoprop, and methidathion, are detected in freshwater habitats within the four western states where listed Pacific salmonids are distributed. Because the action of registering the for the next 15 years authorizes a number of the same uses, they will continue to be present in the action area. Therefore, we expect some individuals within all the listed Pacific salmon and steelhead ESUs/DPSs will be exposed to these chemicals and other stressors of the action. For those OPs that are being cancelled, exposure is expected to decline over time until all existing stocks are used. Given variable use of these pesticides across the landscape, and variable temporal and spatial distributions of listed salmonids, we expect exposure is also highly variable among individuals and populations of listed salmon. However, defining exposure and distributions of exposure among differing life stages of each independent population is complicated by several factors. Paramount among these is the uncertainty associated with the use of pesticide products containing these a.i.s. More specifically:

- Although the BEs and RED documents provide information on EPA regulatory decisions, they lack a full characterization of label-specific information needed to assess exposure (*e.g.*, application restrictions including application methods, rates, and intervals are lacking for many non-agricultural uses);

- EPA-authorized labels contain language that frequently does not provide clear distinctions on product use (*e.g.*, the maximum number of applications is commonly not specified and labels often instruct applicators to repeat applications "as necessary");

- Product labels authorize the application of chemical mixtures that are not specified or not clearly defined (*e.g.*, the ingredients of pesticide formulations are not fully disclosed, labels recommend tank mixture applications with other pesticides and adjuvants, and tank mixtures with other pesticides are permitted unless specifically stated otherwise);

- Defining actual use of these products is highly uncertain. Historical use information is limited and may not reflect future use.

A major limitation of these assessments is that the majority of monitoring data used were not designed to determine exposure to listed salmonids, with the exception of specific studies conducted in Washington, and even those studies were more focused on "integrator sites" rather than habitats receiving direct runoff. Therefore, caution should be exercised in using these data for that purpose especially when conducting probabilistic assessments.

Additionally, the assessments lack uncertainty analyses of the monitoring and toxicity data used, which limit the confidence in the given estimates (Warren-Hicks and Moore 1998). Given the complexity and scale of this action, we are unable to accurately define exposure distributions for the chemical stressors. We assume the highest probability of exposure occurs in freshwater, and nearshore estuarine/marine environments in close proximity to areas where pesticide products containing azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, methamidophos, methidathion, methyl parathion, naled, phorate, and phosmet are applied. We considered several sources of information to define the range of potential exposure to action stressors. These sources are summarized in (Table 110). Ranges of concentrations for the monitoring data are given. Ranges are given for EECs generated by EPA in the salmonid BEs and the more recent BEs generated for the California Red-legged Frog (RLF), which include more non-crop estimates and California specific scenarios. NMFS generated EECs for surface water runoff (using GENEEC) and spray drift (using AgDrift) and into floodplain habitats (2 m wide, 10 cm deep) (Table 89 through Table 91). Typically, the estimates for the floodplain habitat are higher than or near the high end of the range of EECs generated by EPA's PRZM-EXAMS modeling. Spray drift estimates for the off-channel habitats are also higher, with estimates for several chemicals approaching (methyl parathion, naled) or exceeding (bensulide, methidathion, phosmet) mg/L concentrations. In only a few cases were estimates available that included oxon or sulfoxide and sulfone degradates. These were typically modeled as total toxic residues (TTR), and are marked in Table 110.

Table 110 Chemical Concentration Ranges in Monitoring Data and Modeling

Chemical	Monitoring Data		EPA Estimates		NMFS Estimates	
Parent compounds	Range[1]. µg/L (Database)	Max Conc. µg/L (Targeted)	Salmonid BE Conc. Range µg/L	RLF BE Conc. Range µg/L	Surface Water Adjacent to Crops[2] µg/L	Spray Drift µg/L
Azinphos Methyl	0.003-7.35	21	8.3-40.6	1.9-6.8	21	0.8-11.4
Bensulide	ND	2,840	7.2-180	42-231	NE	1,100-2,940
Dimethoate	0.001-11.31	30	6.4-58.3	0.1-20.3	NE	46-652
Disulfoton	0.004-3.81	6.1	7.1-15.0	1.8-67	NE	16-237
Ethoprop	0.001-5.75	NL	15.0-75.0	NE	127	6.0-24
Fenamiphos	0.007-1.50	NL	0.3-35.4	NE	NE	NE
Methamidophos	0.001-0.13	NL	30-65	1.7-12	NE	267-490
Methidathion	0.001-15.10	9.2	8.9-15.5	0.45-116	201	66-1,860
Methyl parathion	0.001-1.70	213	1.3-18.2	7-67	120	134-980
Naled	ND	NL	0.8- 5.3	0.9-33	83	251-921
Phorate	0.012-0.22	NL	4.6-115	0.3-16	98	NE
Phosmet	0.005-0.63	NL	3.0-29.9	3.5-78	90	19-2,920
Degradates						
Dichlorvos	0.004-0.54	0.3	NE	NE	NE	NE
AZM Oxon	0.014-0.04	NL	NE	NE	NE	NE
Methidathion	ND	NL	NE	NE	NE	NE

Chemical	Monitoring Data		EPA Estimates		NMFS Estimates	
Parent compounds	Range[1]. μg/L (Database)	Max Conc. μg/L (Targeted)	Salmonid BE Conc. Range μg/L	RLF BE Conc. Range μg/L	Surface Water Adjacent to Crops[2] μg/L	Spray Drift μg/L
Oxon						
Methyl Paraoxon	0.006-0.01	NL	NE	NE	NE	NE
Phorate Oxon	ND	NL	7.7-138*	NE	NE	NE
Phosmet Oxon	0.007-0.05	NL	NE	NE	NE	NE
Disulfoton Sulfoxide	ND	NL	16.4-43.5*	NE	NE	NE
Disulfoton Sulfone	0.006-0.23	NL	16.4-43.5*	NE	NE	NE
Fenamiphos Sulfoxide	0.022-0.04	NL	NE	NE	NE	NE
Fenamiphos Sulfone	0.008-0.01	NL	NE	NE	NE	NE

* Modeling estimate for total toxic residues (TTR) includes parent and degradates.
ND – No detects, NE – Not estimated, NL – No targeted monitoring study for this chemical located
[1]Minimum and maximum based on detected values
[2] Highest value from GENEEC mixture estimates

Some targeted monitoring studies we discuss addressed issues not typically considered by EPA's near-field runoff modeling. A group of studies described higher pesticide concentrations in back sloughs and small upland drainages, both of which are ecologically important areas for salmonids. Several others showed concentrations of pesticides in larger rivers in California to be somewhat correlated with dormant orchard pesticide sprays, and that this pulse of pesticides may extend into estuarine environments. Studies conducted in the Hood River of Oregon showed invertebrate community effects and salmonid AChE inhibition following applications of azinphos methyl to nearby orchards. Finally, a group of studies conducted in the Sierra Nevadas indicated the concentrations of current use pesticides atmospherically transported to higher elevations may be sufficient to affect biota residing there.

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 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 species. Adult salmon and steelhead spend weeks to several months in freshwater habitats during their migration and spawning activities. Immediately after emerging from the gravel substrate and transitioning from alevins to fry, salmonids move to habitats where they can swim freely and forage. At this point in their development most salmon occupy freshwater habitats. Chum salmon are an exception. They immediately migrate downstream following emergence to nearshore environments in estuaries near the mouth of their natal stream. Upon arrival in the estuary the chum salmon fry inhabit nearshore areas at a preferred depth of 1.5-5 m. In Puget Sound, WA, surveys indicate chum salmon fry are distributed extremely close to the shoreline and concentrated in the top 15 cm of water. Therefore, chum salmon fry are less likely to be exposed to high concentrations of pesticides than other salmonids given they quickly migrate to larger estuaries with greater dilution potential. They may reside immediately next to the shore in estuaries for as little as one or two weeks before moving offshore or into deeper-water habitats within the nearshore environment. Sockeye salmon fry most frequently rear in lakes, where they distribute in the littoral zones. They initially occupy shoreline habitats of only a few centimeters in depth before moving further off-shore and taking on a more pelagic existence. Coho salmon, Chinook salmon, and steelhead fry typically select the stream's nearshore zone and floodplain habitats associated with their natal rivers and streams. These species are most likely to experience higher pesticide exposures given their use of shallow freshwater habitats for juvenile rearing. Coho salmon and steelhead have a greater preference for the shallow habitats and rear in freshwater for more than a year. Coho salmon fry rear in lower gradient river channels and often rear in pools of the river channels. They may also rear in ponds and lakes. Steelhead juveniles use riffles and faster flowing waters more than coho salmon, and are often found in steeper gradient channels. Coho salmon juveniles may make extensive migrations in fall to overwinter in floodplain habitats such as ponds, sloughs, oxbows, flooded wetlands, and other seasonally connected and inundated habitat. Spring foraging in these habitats often provides substantial growth before smoltification and juveniles in these habitats can grow significantly larger than mainstem overwintering coho salmon juveniles. Steelhead do not use channels with organic bottom substrate for overwintering and often seek refuge under larger stones in the flowing river as protection from strong winter flows. Chinook salmon commonly spawn and rear in larger rivers and tributaries than the other *Oncorhynchus* species. Juvenile Chinook salmon in California, with the exception of the Central Valley spring-run Chinook salmon, spend less than six months in freshwater and out-migrate as fry or sub-yearlings (ocean type). Juveniles in the Columbia River basin and in Puget Sound may outmigrate as fry/sub-yearlings (ocean type) or as yearlings (river type), depending on race and the river basin of origin. Fry of the ocean type life history typically rear in estuarine shallow waters, tidal wetlands, and sloughs for days to weeks before entering the ocean while yearling or older juveniles spend less time and use deeper water in the estuary.

Substantial data gaps in EPA's exposure characterization include exposure estimates associated with product uses on many crops and, particularly, on non-crop uses. The highest concentrations detected in surface waters were those associated with applications directly to aquatic habitats. Those types of applications (for naled, phosmet, methyl parathion), although mentioned, were not evaluated in EPA's BEs. Additionally, exposure estimates for other chemical stressors including other ingredients in pesticide formulations, other pesticide products authorized for co-application, adjuvants, degradates, and metabolites were not provided in BEs. 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. However, 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.

Response Analysis

In this section, we identify and evaluate toxicity information from the stressors of the action and organize the information under assessment endpoints (Figure 44). The endpoints target potential effects from the stressors of the action to individual salmonids and their supporting habitats. The assessment endpoints represent biological attributes that, when adversely affected, reduce fitness of individual salmonids or degrade PCEs (*e.g.*, prey abundance and water quality).

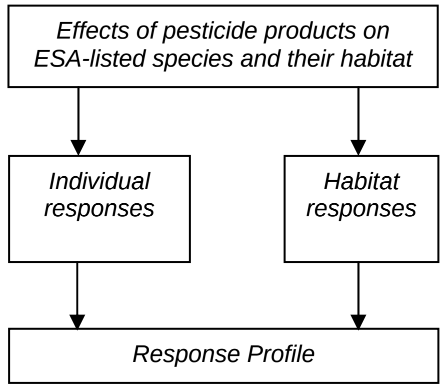


Figure 44 Response analysis

We constructed a visual conceptual model to guide development of risk hypotheses and assessment endpoints to highlight potential uncertainties uncovered by our analysis of the available information **(**Figure 2**).** We begin the response analysis by describing the toxic mode and mechanism of action of organophosphate insecticides. Next we summarize the toxicity data presented in the salmonid BEs, REDS, IREDs, California Red Legged Frog BEs, and EFED Science Chapters for the twelve active ingredients and organize the information according to applicable assessment endpoints (*e.g.*, survival, growth, etc.). The information provided by EPA addressed aspects of survival, growth and reproduction of aquatic species (freshwater and saltwater), and provided some discussion on other information found in the open literature, such as results from some field experiments and experiments that evaluated sublethal effects. Under Section 7 of the ESA and its implementing regulations, NMFS to evaluates all direct and indirect effects of an action. We therefore evaluate all aspects of an action that may reduce fitness of individuals or reduce primary constituent elements of designated critical habitat. The evaluation includes information that EPA provided on survival, growth, or reproduction, but also encompasses a broader range of endpoints including behaviors, endocrine disruption, and other physiological alterations such as impairment of olfactory-mediated behaviors. The information we evaluated is derived from published, scientific journals and information from government agency reports, theses, books, information and data provided by the registrants identified as applicants, and independent reports. Typically, the most relevant study results are those that directly measure effects to an identified assessment endpoint derived from experiments with salmonids, preferably listed Pacific salmonids or hatchery surrogates, exposed to one or more of the stressors of the action. We also indicate when we use information from studies with other pesticides that share the same mode of toxic action (see below).

Mode of Action

The twelve OPs share a similar mode of toxic action. Eleven are insecticides and one is an herbicide (bensulide), however all generally share a similar chemical structure and act as neurotoxicants by impairing nerve cell transmission in vertebrates and invertebrates. They inhibit the enzyme acetylcholinesterase (AChE), present in cholinergic synapses. In an irreversible[5] reaction OPs phosphorylate AChE, thereby inhibiting AChE's normal activity to hydrolyze the neurotransmitter acetylcholine at nerve synapses (Kennedy 1991). While AChE is the primary site of action, OPs can also interact with other cholinesterases. Phosphorothionate OPs (*i.e.*, those containing a P=S bond*)* can form oxygen analogues, referred to as oxons. Eight of the pesticides addressed in this Opinion, (bensulide, dimethoate, disulfoton, methidathion, methyl parathion, phorate, and phosmet) are biotransformed, (*i.e*., metabolized), by vertebrates and invertebrates into oxon metabolites that are more potent inhibitors of AChE than the parent compounds. Abiotic transformation in the environment can also lead to oxon formation (Wu and Laird 2003). The normal function of AChE is to breakdown (hydrolyze) the neurotransmitter, acetylcholine, thereby serving as an off switch to the electrochemical signal between nerve cells. The key result of AChE inhibition by OP insecticides is accumulation of the neurotransmitter, acetylcholine, in a nerve synapse. The buildup of acetylcholine causes continuous nerve firing and eventual failure of nerve impulse propagation. Pesticides that inhibit AChE are not necessarily selective to target pests and may have adverse effects on fish and other nontarget species. Acetylcholinesterase is prevalent in a variety of cell and organ types throughout the body of vertebrates and invertebrates (Walker and Thomposon 1991). Interference of normal nerve transmission by OPs may affect a wide array of physiological systems in fish. A variety of adverse effects to organisms can result, including death (Mineau 1991).

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[5] The inhibition may not be completely "irreversible" as phosphorylated ACHE can spontaneously dephosphorylate to its active form. Spontaneous de-alkylation of one of the alkyl groups can occur which results in permanent inactivation known as aging, reviewed in {Kennedy, 1982 #2084}

Incidences of acute poisoning from AChE inhibitors arecommont for wildlife, particularly for birds and fish (Mineau 1991). The following passage describes the classic signs of AChE-inhibiting pesticide poisonings of fish:

> Fish initially change normal swimming behavior to rapid darting about with loss of balance. This hyper excitability is accompanied by sharp tremors which shake the entire fish. The pectoral fins are extended stiffly at right angles from the body instead of showing the usual slow back and forth motion normally used to maintain balance. The gill covers open wide, and opercular movements become more rapid. With death the mouth is open and the gill covers are extended. Hemorrhaging appears around the pectoral girdle and base of the fins (Weiss and Botts 1957).

AChE inhibition following exposure to OPs has been documented in animals, including salmonids, in numerous reports, articles, reviews, text books, and cases of wildlife poisoning (Antwi 1985, Coppage and Matthews 1974, Haines 1981, Holland et al 1967, Rabeni and Stanley 1975, Williams and Sova 1966.)

Pesticides can exhibit more than one mechanism of action. Bensulide is classified as an organophosphate herbicide. EPA reported that the mode of action within plants involves "the inhibition of cell division in the roots and shoots of plants" (EPA 2002a). We therefore discuss the effects of bensulide not only on listed salmonids and their prey, but also on the aquatic plant community, *i.e.*, primary producers, within the action area.

Inhibition of AChE by the twelve OPs

The EPA concluded that the 12 a.i.s share a mode and mechanism of toxic action with other OPs pursuant to the Food Quality Protection Action of 1996. We also report on AChE inhibition experiments where juvenile coho salmon were exposed to each of the twelve OPs. This work was conducted by NOAA's Northwest Fisheries Science Center in the summer of 2009 and is of particular value because it directly addresses the toxicity of the 12 a.i.s (NOAA 2009). Other laboratory and field studies that measured AChE activity in aquatic invertebrates and fish following exposures to OPs, including several of the 12 a.i.s and some of their toxic degradates, are also discussed.

Activity of AChE in juvenile coho salmon was investigated for each of the 12 a.i.s following 96 h exposures (NOAA 2009). Activity of AChE was measured in brain, and in some cases in muscle tissue, of fish (n=8 per treatment). At each exposure concentration, three water samples were collected at either 48 or 72 h immediately following the day's water exchange and quantified using standard analytical chemistry techniques. Azinphos methyl, dimethoate, disulfoton, ethoprop, methidathion, methyl parathion, naled, phorate, and phosmet showed a clear trend of decreasing AChE activity with increasing concentration (Table 111). However, bensulide, fenamiphos, and methamidophos did not show the same relationship. Bensulide and fenamiphos showed no inhibition of AChE in either brain or muscle tissue at the concentrations tested. That said, exposed juvenile coho still showed classic symptoms of neurotoxicity including lethargy, excitability, and loss of orientation at 300, 400, and 500 µg/L of bensulide and at 10, 30, and 100 µg/L fenamiphos. Three of the 8 fish exposed to 100 µg/L fenamiphos died at 96 h. These results suggest that bensulide and fenamiphos are neurotoxic to salmonids, but by an unknown mechanism besides AChE-inhibition. Further study is needed to elucidate the mechanism of action for bensulide and fenamiphos. Methamidophos showed neither a consistent trend in reduction of AChE activity over the range of concentrations tested nor symptoms of cholinergic poisoning observed. Based on the acute lethality of methamidophos to salmonids, LC50 ~ 25,000 µg/L, the dose-response relationship may not have been captured by the range tested, 0-1,000 µg/L as percent of activity measured was between 90% and 115%. Further experiments are needed to confirm this. The results do show that concentrations of up to 1,000 µg/L do not affect AChE activity in juvenile coho.

Table 111 AChE activity of 12 OP pesticides in juvenile coho salmon

OP Pesticide	EC50 µg/L (95% CI)	Sigmoid Slope (95% CI)	Concentration range tested (µg/L)	Qualitative observations
Azinphos methyl	0.16 (0.10 - 0.26)	-1.91 (-3.44 to -0.40)	0, 0.05, 0.1, 0.2, 0.3, 0.45, 0.6, 1	na

OP Pesticide	EC50 µg/L (95% CI)	Sigmoid Slope (95% CI)	Concentration range tested (µg/L)	Qualitative observations
Bensulide	na	na	0, 5, 10, 50, 100, 200, 300, 400, 500 4 treatments replicated	At 300, 400, 500 µg/L lethargy, excitability, and loss of orientation
Dimethoate	273.4 (195.7 – 382.0)	-0.862 (-1.20 to -0.52)	0, 30, 60, 150, 300, 450, 600	na
Disulfoton	485.5 (99.35-2373)	-0.34 (-0.65 to -0.03)	0, 10, 30, 100, 300, 1000	na
Ethoprop	90.62 (69.46 – 118.2)	-1.33 (-1.76 to -0.89)	0, 10, 25, 50, 100, 250, 500	na
Fenamiphos	na	na	0, 0.3, 1, 3, 10, 30, 100	3/8 coho died at 96 h; at 10, 30, 100 µg/L lethargy, excitability, and loss of orientation
Methamidophos	na	na	0, 15, 30, 60, 120, 300, 600, 1000	na
Methidathion	1.12 (0.47 – 2.68)	-0.92 (-1.93 to -0.10)	0, 0.2, 0.6, 1.2, 2, 4, 6	na
Methyl parathion	28.75 (21.17 – 39.03)	-0.70 (-0.88 to -0.51)	0, 3, 10, 20, 30, 100, 300	na
Naled	7.85 (6.49 – 9.49)	-1.31 (-1.60 to -1.02)	0, 0.6, 2, 6, 20, 60	na
Phorate	0.57 (0.42 – 0.76)	-1.61 (-2.31 to -0.92)	0, 0.06, 0.2, 0.6, 1.2, 2, 6	na
Phosmet	3.25 (2.52 – 4.19)	-1.04 (-1.31 to -0.78)	0, 0.6, 2, 6, 20, 60	na

na indicates no values calculated or observations made.

Several studies documented AChE inhibition in a variety of aquatic organisms following exposure to azinphos methyl including rainbow trout (Ferrari et al 2007) , goldfish (Ferrari et al 2004 b), toads (Ferrari et al 2004a), and aquatic snails and oligochaetes (Kristoff et al 2006).

Activity of AChE in brain and muscle of juvenile rainbow trout was reduced by 50% following 96 h exposure to 0.42 µg/L and to 1 µg/L azinphos methyl (AZM), respectively (Ferrari et al 2007). No significant recovery of AChE activity in trout muscle occurred throughout the 21 days in uncontaminated water, indicating that recovery from acute exposures may take several weeks. This long-lasting inhibition of AChE activity observed is explained in part by the well-established slow rate of spontaneous reactivation of AChE after OP binding (in this case AZM), followed by the aging process which leads to irreversible inactivation. Thus, to recover from AChE-inhibiting pesticide exposure, affected animals must synthesize new AChE. The process to fully recover from inhibition can take several weeks. Consequently, sublethal effects to fish may persist after the OP has dissipated in the aquatic environment. A study with azinphos methyl showed a high degree of inhibition of brain AChE activity (85 – 88%) in rainbow trout compared to control activity when exposed for 48 hours to 1 ug/L. Following the exposures, rainbow trout were transferred into clean water where AChE activity was measured after 7 days. Only partial recovery occurred, showing statistically significant inhibition with respect to controls (41-58%). Repeated 48 h pulses of azinphos methyl at 1 µg/L, separated by 7 day recovery periods in clean water, showed that repeated exposures reduced AChE activity to similar levels following each exposure

A freshwater snail and an oligochaete also showed reductions in cholinesterase activity following 48 h exposures to azinphos methyl suggesting that a range of aquatic invertebrates are potentially susceptible to AZM's neurotoxic effects Kristoff et al 2006). Total cholinesterase activity was reduced by half compared to control levels following 48 h exposures to 5,960 µg/L for the snail and 6 µg/L for the oligochaete. The oligochaetes also showed sublethal behavioral responses including uncoordinated movements beginning at concentrations as low as 1 µg/L AZM, the statistical NOEC for cholinesterase inhibition. At 90% inhibition the oligochaetes showed total lack of movement. These studies demonstrate AZM inhibits AChE activity following short term exposures to a variety of aquatic organisms. The degree of inhibition is species and dose dependent. Juvenile rainbow trout were the most sensitive species based on a 50% reduction in brain AChE activity following 96 h of exposure to 0.42 µg/L AZM.

Two sets of field experiments directly investigated juvenile steelhead (hatchery-reared) AChE activity from caged-fish studies in an agricultural watershed in the Hood River Basin, OR (Grange 2002, St. Aubine 2004). The Hood River Basin contains several listed anadromous salmonids, including lower Columbia River steelhead. The studies analyzed water samples for azinphos methyl, chlorpyrifos, and malathion before, during, and after orchard spray periods. Steelhead from reference sites had statistically significant greater AChE activity than steelhead from orchard-dominated areas. The reductions in AChE activity in steelhead corresponded to the application seasons and detections of azinphos methyl and chlorpyrifos (Grange 2002). None of the pesticides were detected at reference sites and both chlorpyrifos (maxima ranged from 0.077-0.196 µg/L) and azinphos methyl (0.026-0.057 µg/L) were frequently detected at orchard stream and river sites. AChE activity was inhibited up to 21% in smolts, and 33% in juveniles relative to reference locations. Temperature was a confounding factor as lower temperatures showed lower AChE activity while higher temperatures showed higher AChE activity at reference sites. The authors normalized data to temperature and found statistically significant reductions in AChE in steelhead. Study results showed that steelhead in these systems exposed to OP insecticides lose AChE activity (up to 33%) and, depending on the percentage of inhibition, this impairment may lead to fitness level consequences (Grange 2002, St. Aubin 2004). These field studies show that salmonids' AChE activity was reduced in orchard-dominated streams during chlorpyrifos and azinphos methyl application seasons.

Dimethoate substantially reduced activity of AChE in marine shore crab haemolymph following 18 h of exposure at concentrations of 2 mg/L, however at concentrations below 2 mg/L (0.5 and 1.0 mg/L) statistically significant reductions were absent (Lundebyer et al 1997). In another experiment with dimethoate, AChE activity was reduced by 25% at 130 µg/L, by 34% at 260 µg/L, and by 66% at 1,300 µg/L in the common carp (*Cyprinus carpio*) following seven days of exposure (De Mel and Pathiratne 2005). Following 14 day exposures to dimethoate, AChE activity in carp was reduced by 42% at 130 µg/L, by 66% at 260 µg/L and by 69% at 1,300 µg/L, demonstrating that inhibition of AChE increases with increasing exposure duration (De Mel and

Pathiratne 2005). In comparison, methidathion at 2.0 mg/L reduced AChE activity in various organs of adult carp (*C. carpio*) by 70-92% following five days of exposure (Balint et al 1995). For example, a 90-92% reduction in AChE activity occurred in the brain, heart, and blood, while 70-75% inhibition occurred in the liver and skeletal muscle. Methidathion decreased during the five day exposure to approximately 30% of the initial concentration of 2 mg/L.

Temperature and toxicity

We reviewed the EPA-submitted information regarding the potential influence of temperature on the toxicity of the 12 a.i.s. One of the salmonid BEs, phosmet, reported a positive correlation between temperature and toxicity in fish, but no further analysis was conducted. For the remaining 11 OPs, ranges of LC50s were provided based on temperature (when available), however no discussion or further analysis was conducted. Mayer and Ellersieck (1988) reviewed the effects of temperature on toxicity to aquatic organisms and reported that of the 410 organic chemicals tested, most showed a 2- to 4-fold increase in toxicity for each 10 ˚C rise in temperature. Acute lethality bioassays with OPs showed a distinct, robust relationship between toxicity (measured by 96 h LC50) and temperature. The experiments were conducted with bluegill sunfish (phosmet, parathion, malathion, trichlorfon), rainbow trout (phosmet, chlorpyrifos, trichlorfon), yellow perch (azinphos methyl), Atlantic salmon (trichlorfon), and brook trout (trichlorfon). We note some effects may occur at temperatures outside the physiological optimum temperatures for salmon. However, we assume salmon that are already stressed by high temperatures will be more susceptible to the a.i.s and other stressors.

An increase in toxicity as temperature rises has also been observed for aquatic invertebrates. The midge larvae, (*Chironomus tentans*), showed increased lethality at 30 ˚C compared to 20 ˚C following 96 h exposure to methyl parathion, although no significant change was measured between 10 ˚C and 20 ˚C treatments (Lydy et al 1999). In the same experiment, chlorpyrifos showed a consistent relationship with temperature, i.e. at 10 ˚C 96 h LC50 = 0.58, at 20 ˚C 96 h LC50 = 0.33 and at 30 ˚C 96 h LC50 = 0.15 mg/L.

A recent test evaluated the effect of temperature on AChE activity in juvenile coho following 96 h exposures to AChE-inhibiting insecticides at four temperature regimes (NOAA 2009. Brain

AChE activity was measured in juvenile coho (mean weight = 5g; 8 fish per treatment) following exposure to carbofuran (a carbamate insecticide), chlorpyrifos (OP), and a diazinon-malathion mixture (OPs) at 8, 12, 14, and 15.5 °C. Fish were exposed to concentrations that were expected to reduce AChE activity by 50% (EC50). For the mixture of diazinon and malathion, one-half the respective EC50 was added for each pesticide (1.27 µg/L diazinon and 0.65 µg/L malathion). Both a control and a solvent control were run along with the pesticide treatments. Exposure to chlorpyrifos reduced brain AChE activity to 75-90% of control activity; however, no apparent trend in AChE activity occurred with temperature (linear regression, $p = 0.67$). The absence of a temperature effect on chlorpyrifos toxicity was unexpected, since temperature-dependent toxicity of chlorpyrifos and other OPs is well documented in acute survival experiments (Mayer and Ellersieck 1986). The concentrations may have reduced AChE so much that it hit a floor of effect. Further research is necessary to determine if these results were due to differences in fish age, size, or possible variations in pesticide stock solution concentrations (analytical verification of treatment concentrations was not conducted). Coho exposed to the diazinon and malathion mixture showed a significant trend of declining AChE activity (98 – 41%) with increasing temperature (linear regression, $p<0.0001$). This result suggests that as temperature increases, the toxicity of a mixture of two OPs is enhanced resulting in more pronounced adverse effects to juvenile coho. Whether this temperature effect is consistent across other OPs remains to be determined.

Multiple studies with fish and invertebrates, including salmonids, indicate that OPs can become more toxic as temperature increases. We therefore reviewed 303(d) lists for temperature to show where elevated temperature may be an issue within the *Environmental Baseline* and apply the information within the *Risk Characterization* section.

pH and toxicity

The available data from acute 96 h lethality assays showed no apparent trend between pH and toxicity of the 12 a.i.s (Mayer and Ellersieck 1986).

Studies with mixtures of AChE inhibiting insecticides

We discuss the available toxicity studies with fish exposed to mixtures of OPs because these compounds: 1) share a common mode and mechanism of action, 2) are authorized for use and frequently applied in the same watersheds, and 3) have demonstrated additive and synergistic effects in aquatic organisms. We present results from an analysis of combinations of the twelve OPs to salmonids and their habitats to common mixtures of OPs within the *Risk Characterization* section.

One of the earliest pesticide mixture studies evaluated bluegill survival following a range of exposure durations (24, 48, 72, or 96 h) to binary combinations of 19 insecticide mixtures, some of which were OPs (Macek 1975). The equation used to calculate mixture toxicity was, AB/(A+B) = X; where AB was the number of dead fish from a mixture of pesticides A and B, and A + B was the sum of dead fish from A and B alone. The resulting ratios, X, were designated as antagonistic for a ratio of less than 0.5, additive when the ratio fell between 0.5 and 1.5, and synergistic for a ratio of more than 1.5. Malathion containing mixtures resulted in additive (DDT, toxaphene), synergistic (with Baytex [OP], parathion [OP], carbaryl [carbamate], perthane) and antagonistic (with copper sulfate) responses. Caution should be placed on the difference between additive and synergistic designations, as the selected threshold was arbitrarily set at 1.5 and mixture results with DDT and toxaphene were at 1.31 and 1.14, respectively. Diazinon and parathion were synergistic to bluegill survival, (*i.e.*, more fish died than would have been predicted from additivity). Validation of chemical concentrations with analytical chemistry was not conducted. Although the lack of raw data makes it difficult to determine the precise concentrations tested, the study was one of the first to show that both additive and synergistic responses occurred with OPs. Although none of the pesticides addressed in this Opinion were tested in this experiment, we find it reasonable that additive and/or synergistic effects will also occur from these compounds.

Additive toxicity of binary combinations of OPs and carbamates was demonstrated from *in vitro* experiments with Chinook salmon (Scholz et al 2006). The oxons of diazinon, chlorpyrifos, and malathion in addition to the carbamates carbaryl and carbofuran caused additive toxicity as measured by AChE inhibition in salmonid brain tissue. Further, the joint toxicity of the mixtures could be accurately predicted from each insecticide's toxic potency, simply by adding the two potencies together at a given concentration. Since the experiments were conducted using *in vitro* exposures with the oxon degradates and not with the parent compounds, the authors conducted subsequent sets of experiments to investigate whether additive toxicity as measured by AChE inhibition also occurred when live fishes were exposed for 96 h to the parent compounds (*i.e., in vivo* exposures) (Laetz et al 2009).

The results from *in vivo* experiments showed both additivity and synergism (Laetz et al 2009). As with the *in vitro* study, brain AChE inhibition in juvenile coho salmon exposed to sublethal concentrations of chlorpyrifos, diazinon, and malathion as well as the carbamates carbaryl and carbofuran were measured. Dose-response data for individual chemicals were normalized to their respective EC50 concentrations and collectively fit to a non-linear regression. The regression line was used to determine whether toxicological responses to binary mixtures were antagonistic, additive, or synergistic. No binary mixtures resulted in antagonism. Additivity and synergism were both observed, with a greater degree of synergism at higher exposure concentrations. Moreover, certain combinations of OPs were lethal at concentrations that were sublethal in single chemical trials. Based on the null hypothesis of dose-addition, the five pesticides were combined in all possible pairings to yield target levels of AChE inhibitions in the brains of exposed coho salmon (Laetz et al 2009).

Two thirds (20/30) of pesticide pairs yielded significant synergistic inhibition of AChE compared to levels expected based on additive toxicity (t-test with Bonferroni correction, $p < 0.005$). The number of combinations that were statistically synergistic increased with increasing exposure concentrations. Additionally, pairings of two OPs produced a greater degree of synergism than mixtures containing one or two carbamates. This was particularly true for mixtures containing malathion coupled with either diazinon or chlorpyrifos. At the highest exposure treatment, 1.0 EC50 (malathion at 37.3, chlorpyrifos at 2, diazinon at 72.5 μg/L), binary combinations produced synergistic toxicity (Table 41). Many fish species die following acute brain AChE inhibition of 70-90% (Fulton and Key 2001). Coho salmon exposed to combinations of diazinon and malathion (1.0 and 0.4 EC50) as well as chlorpyrifos and malathion (1.0 EC50) all died. Fish exposed to these OP mixtures showed typical toxic signs of

AChE inhibition, including loss of equilibrium, rapid gilling, altered startle response, and increased mucus production before dying. OP combinations were also synergistic at the lowest concentrations tested. Diazinon and chlorpyrifos were synergistic when combined at 7.3 µg/L and 0.1 µg/L, respectively. The pairing of diazinon (7.3 µg/L) with malathion (3.7 µg/L) produced severe (> 90%) AChE inhibition, and classical signs of poisoning as well as death in some combinations. Thus, for binary combinations of malathion, diazinon, and chlorpyrifos synergism is likely to occur at exposure concentrations that were below the lowest ones tested in {Laetz, 2009 #1345}, *i.e.*, chlorpyrifos less than 0.1 µg/L; diazinon less than 7.3 µg/L; malathion less than 3.7 µg/L. The mechanism for synergistic toxicity in salmonids remains unknown.

We expect that some combinations of the 12 a.i.s will result in synergistic toxicity to exposed juveniles and in some situations, juvenile salmonids may die. We expect this because all 12 share the same mode of action *i.e.*, impairment of nerve cell transmissions. Additionally, data with other OPs and carbamates indicate synergism. Unfortunately, we are unable to create a predictive model of synergistic toxicity as dose-response relationships with multiple ratios of the 12 pesticides are not currently available and the mechanism of synergism remains to be determined. We conducted a mixture analysis based on additivity, (*i.e*., concentration addition) with combinations of OPs (see mixture analyses in the *Risk Characterization* section). We also infer that synergism remains a possibility for combinations of these pesticides as well as combinations of these pesticides with other OPs. If synergism does occur, toxicity estimates based on additivity would underestimate the toxicity.

Table 112. Mixture concentrations resulting in 100% mortality of juvenile coho following 96 h exposures (Laetz et al 2009).

OP mixture	Concentration, µg/L
diazinon + malathion	72.5 diazinon, 37.3 malathion 29.0 diazinon, 14.9 malathion
chlorpyrifos + malathion	1.0 chlorpyrifos, 37.3 malathion

Summary of Toxicity Information Presented in the BEs, EFED Science Chapters, REDs, CA Red-Legged Frog BEs

Each BE primarily summarized acute and chronic toxicity data from "standardized toxicity tests" submitted by pesticide registrants during the registration process, tests from government laboratories available in EPA databases, or from published, peer-reviewed scientific publications (books and journals). The assessment endpoints from these tests for an individual organism generally included aspects of survival (death), reproduction, and growth measured in laboratory dose-response experiments (EPA 2004g). Table 113 provides a summary of the available toxicity information for each of the 12 active ingredients compiled from EPA-provided documents including EPA's Pacific salmonid BEs, EPA's California Red-Legged Frog BEs, REDs, IREDs, and EFED science chapters.

Survival is typically measured in both acute and chronic tests. Fish reproduction and growth are generally measured using chronic tests. Population-level endpoints and analyses were generally absent in the BEs, other than a few measurements of fish and aquatic invertebrate reproduction. Adverse effects to organisms were not translated into consequences to populations For this Opinion, NMFS translates effects to individual salmonids into potential population-level consequences as explained in the *Risk Characterization* portion of the *Effects of the Proposed Action* section, and ultimately draws a conclusion on the likely risk to listed salmonids based on exposure and anticipated individual and population-level effects.

Survival Endpoints

Survival of individual fish is typically measured by incidences of death following 96 h exposures (acute test) and incidences of death following 21 d, 30 d, 32 d, and "full life cycle" exposures (chronic tests) to a subset of freshwater and marine fish species reared in laboratories under controlled conditions (temperature, pH, light, salinity, dissolved oxygen, etc.,) (EPA 2004g). Lethality of the pesticide (a technical product or formulated product) is usually reported as the median lethal concentration (LC50), the statistically-derived concentration sufficient to kill 50% of the test population. For aquatic invertebrates it may be reported as an EC50, because death of these organisms may be too difficult to confirm and therefore, immobilization is considered a terminal endpoint. An LC50 is derived from the number of surviving individuals at each concentration tested following a 96 h exposure and is typically estimated by probit or logit analysis and recently by statistical curve fitting techniques. In FIFRA guideline tests, LC50s are typically calculated by probit analysis. If the data are not normally distributed for a probit analysis, than either a moving average or binomial is used, resulting in no slope being reported. Ideally, to maximize the utility of a given LC50 study, a slope, variability around the LC50, and a description of the experimental design, such as experimental concentrations tested, number of treatments and replicates used, solvent controls, *etc.*, are reported. The slope of the observed dose-response relationship is particularly useful in estimating the magnitude of death at concentrations below or above an estimated LC50. The variability of an LC50 is usually depicted by a confidence interval (95% CI) or standard deviation/error and is illustrative of the degree of confidence associated with a given LC50 estimate *i.e.*, the smaller the range of uncertainty, the higher the confidence in the estimate. Without an estimate of 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 that are available for toxicity testing including hatchery reared coho salmon, Chinook salmon, steelhead, and chum salmon, as well as rainbow trout[6]. The available toxicity data include a variety of salmonids. Unfortunately, slopes, estimates of variability for an LC50, and experimental concentrations frequently are not reported. In our review of the salmonid BEs, we did not locate any reported slopes of dose-response curves, although some of this information was presented in some of the corresponding Science Chapters and the CRLF BEs. Consequently, we must err on the side of the species in the face of these uncertainties and select LC50s from the lower range of available salmonid studies. We selected salmonid LC50s and slopes as input in the population modeling exercises discussed later. We evaluate 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.

[6] 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 life stages that use estuarine and ocean environments.

Growth Endpoints

Growth of individual organisms is an assessment endpoint derived from standard chronic fish and invertebrate toxicity tests summarized in the BEs. It is difficult to translate the significance of impacted growth derived from a guideline study on fish growth in aquatic ecosystems. The health of the fish, availability and abundance of prey items, and the ability of the fish to adequately feed are not assessed in standard chronic fish tests. These are important factors affecting the survival of wild fish. What is generally assessed is size or weight of fish measured at several times during an experiment. The test fish are usually fed twice daily, *ad libitum*, (i.e., an over abundance of food is available to the fish). Therefore, any reductions in size are a result of fish being affected to such an extent that they are not feeding or are unable to metabolize food even when presented with an abundance of food. Subtle changes in feeding behaviors or availability of food would not be detected from these types of experiments. If growth is affected in these experiments, it is highly probable that growth of fish in natural aquatic systems would be severely affected. If effects to growth are likely, we assess salmonid population-level consequences based on reductions in juvenile growth and subsequent reduction in size prior to ocean entry.

Reproduction Endpoints

Reproduction, at the scale of an individual, can be measured by the number of offspring per female (fecundity), and at the scale of a population by measuring the number of offspring per females in a population over multiple generations. The BEs summarized reproductive endpoints at the individual scale from chronic freshwater fish experiments where hatchability and larval-juvenile survival is measured. NMFS also considers many other assessment measures of reproduction, including egg size, spawning success, sperm and egg viability, gonadal development, reproductive behaviors, and hormone levels, as these endpoints could have considerable effect on wild populations. These endpoints are not generally measured in standardized toxicity assays used in pesticide registration.

Sublethal Endpoints

Sometimes qualitative observations of sublethal effects are summarized from 96 h lethality dose-response bioassays in EPA's risk assessments. These observations generally were limited in the

BEs of the 12 a.i.s, and when noted, pertained to unusual swimming behaviors. None of these behaviors were rigorously measured and therefore are of limited value in assessing the effects of these OP pesticides on Pacific salmonids. We do, however note a few of the observations when they pertained to a relevant assessment endpoint, such as impaired swimming. Some BEs presented toxicity information on degradates, metabolites, and formulations. However, toxicity information on other or "inert" ingredients found in pesticide formulations was usually not presented.

Multi-species (Micro- and Mesocosm) Studies

Results from multiple species tests, called microcosm and mesocosm studies, were also discussed in the BEs to a varying degree. These types of experiments are likely closer approximations of potential ecosystem-level responses such as interactions among species (predator-prey dynamics), recovery of species, and indirect effects to fish. However, the interpretation of results is complicated by how well the results represent natural aquatic ecosystems and how well the studies apply to salmonid-specific assessment endpoints and risk hypotheses. These studies typically measured individual responses of aquatic organisms to contaminants in the presence of other species. Some are applicable to questions of trophic effects and invertebrate recovery, as well as providing pesticide fate information. The most useful mesocosm study results for this Opinion are those that directly pertain to identified assessment endpoints and risk hypotheses. We discuss study results in the context of salmonid prey responses, emphasizing the capacity of prey taxa to rebound following death of individuals as well as shifts from preferred taxa to other taxa if measured. One of the notable limitations of these study types is they do not represent real world aquatic ecosystems that are degraded from various stressors including contaminants and elevated water temperature.

Results from aquatic field studies were generally not discussed in great detail within the BEs. We discuss field studies that evaluated assessment endpoints, particularly those which address salmonid prey responses in systems with ESA-listed salmonids.

Toxicity values presented in the BEs for each pesticide active ingredient are summarized in Table 113. Toxicity values (μg/L) are organized by assessment endpoint and associated assessment measures. The BEs primarily provided toxicity information from EPA's EFED Pesticide Toxicity Database and from the ACQUIRE database. Although we did not conduct a review of each study found in these databases, we assume the information is accurate.

Table 113. Assessment endpoint toxicity values (µg/L) for aquatic organisms presented in salmonid BEs, CRLF BEs, REDs, IREDS, and EFED science chapters. Abbreviations as follows: NR = Not Reported; T= Technical grade; Formulation = formulated product; EC=Emusifiable concentrate, G=granular, sw = estuarine and/or marine species. [a]96 h test; [b]48 h test

Assessment Endpoint		Concentration (µg/L)		
		Azinphos methyl		degradate(s): unspecified
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
Survival	Freshwater, estuarine, and marine fish LC50 (96 h)	Striped mullet (*Mugil cephalus*) (sw) (96%; T) = 3.2[b] Spot (*Leiostomus xanthurus*) (sw) (96%; T) = 28[b]	Sheepshead minnow (*Cyprinodon variegatus*) (sw) (89%; T) = 2.7 Sheepshead minnow (*C. variegatus*) (sw) (% a.i. NR) = 2 Bluegill sunfish (*Lepomis macrochirus*) (93%; T) = 4.1 – 34, n = 7 Bluegill sunfish (*L. macrochirus*) (22% Guthion 2S) = 40 Bluegill sunfish (*L. macrochirus*) (% a.i. NR) = 2.2-22, n = 3 Bluegill sunfish (*L. macrochirus*) formulation (% a.i., NR) = 4.2-32 and 120 n = 10 Northern Pike (*Esox lucius*) (93%; T) = 0.36 (yolk-sac fry) Black crappie (*Pomoxis nigromaculatus*) (93%; T) = 3.0 Green sunfish (*Lepomis cyanellus*) (93%; T) = 52 Carp (*Cyprinus carpio*) (93%; T) = 695	Yellow perch (*Perca flavescens*) 7 d unspecified degradate = 24 Yellow perch (*P. flavescens*) 14 d unspecified degradate = 20 Yellow perch (*P. flavescens*) 21 d unspecified degradate = 33

Assessment Endpoint		Concentration (μg/L)		
		Azinphos methyl		degradate(s): unspecified
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
			Golden orfe (*Leuciscus idus*) (92.6%; T) = 120 Channel catfish (*Ictalarus punctatus*) (93%; T) = 3,290 Goldfish (*Carassius auratus*) (93%; T) = 4,270 Black bullhead (*Ictalarus melas*) (93%; T) = 3,500 – 4,810 n = 3 Fathead minnow (*Pimephales promelas*) (93%; T) = 148; 293 Fathead minnow (*P. promelas*) (% a.i. NR)= 64 – 3,260 n = 23 Mosquito fish (*Gambusia affinis*) (Formulation, % a.i. NR) = 68-78 Threespine stickleback (*Gasterosteus arculeatus*) (Formulation, % a.i. NR) = 4.8; 12.1 Red drum (*Sciaenops ocellatus*) (Formulation, % a.i. NR) = 6.2; 7.1 Largemouth bass (*Micropterus salmoides*) (93%; T) = 4.8 Yellow perch (*Perca flavescens*) (93%; T) = 2.4 – 40 n = 13	
Survival	Salmonid LC50 (96 h)		Coho salmon (*Oncorhynchus kisutch*) (93%; T) = 3.21 – 17 n = 4 Coho salmon (*O. kisutch*) (% a.i. NR) = 17	

Assessment Endpoint		Concentration (μg/L)		
		Azinphos methyl		degradate(s): unspecified
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
			Coho salmon (*O. kisutch*) (Formulation, % a.i. NR) = 4.2 Atlantic salmon (*Salmo salar*) (93%; T) = 2.1 – 3.6 and >15 n = 8; 1.18-18 (yolk sac fry n=5; >15 - >50 (green eggs) n = 3 Rainbow trout (*O. mykiss*) (93%) T= 2.9 – 14 n = 5 Rainbow trout (*O. mykiss*) (22% Guthion 2S) = 27.49 Rainbow trout (*O. mykiss*) (50%) = 8.8 Rainbow trout (*O. mykiss*) (% a.i. NR) = 4.3 – 14 n = 3 Rainbow trout (*O. mykiss*) formulation (% a.i. NR) = 3.2 – 7.1 n=5 Brown trout (*Salmo trutta*) (93%; T)= 3.5 – 6.6 n = 6 Brook trout (*Salvelinus fontinalis*) (93%; T) = 1.2	
Reproduction or larval survival	NOEC/LOEC		Rainbow trout (*O. mykiss*) (88.8%) = 0.44/0.98 Sheepshead minnow (*C. variegatus*) (sw) (92.5%) = 0.2/ 0.4	
Fish growth	NOEC/LOEC		Rainbow trout (*O. mykiss*) (88.8%) = 0.23/0.98; 0.44/0.98	

Assessment Endpoint		Concentration (μg/L)		
		Azinphos methyl		degradate(s): unspecified
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
Habitat-salmonid prey	invertebrate survival (48 h EC/LC50)	Brown shrimp (*Penaeus aztecus*) (sw) (96%) T= 2.4[b]	Mysid (*Mysidopsis bahia*) (sw) (89%) = 0.21[b] Mysid (*M. bahia*) (sw) (22% Guthion 2L) = 0.26[b] Water flea (*Daphnia magna*) (90.6%; T)= 1.13[b] Water flea (*D. magna*) (50%) = 4.8[b] Scud (*Gammarus fasciatus*) (93%; T) = 0.16; 0.25[b] Glass shrimp (*Palaemonetes kadiakensis) (93%; T)* = 1.2[a] Stonefly (*Pteronarcys californica*) (93%; T) =1.9[a] Sowbug (*Asellus brevicaudus*) (93%; T) = 21[a] Crayfish (*Procambarus* sp.)[2] (93%; T) = 56[a]	
	Invertebrate reproduction NOEC/LOEC (21 day)	Water flea (*D. magna*) (99.6%; T) = 0.25/0.4		

Assessment Endpoint		Concentration (μg/L)		
		Bensulide		degradate(s)
	Assessment measure	**> 95% a.i. (% a.i.)**	**< 95% a.i. (%a.i.)**	
Survival	Freshwater, estuarine, and marine fish LC50 (96 h)	Bluegill sunfish (*Lepomis macrochirus*) (95%; T) = 810	Channel catfish (*Ictalurus punctatus*) (% a.i. NR; T) = 380 Sheepshead minnow (*Cyprinodon variegatus*) (sw) (92%; T) = 560 Bluegill sunfish (L. *macrochirus*) (64%) = 1,780 Spot (*Leistomus xanthurus*) (sw) (95%; T) = 320	
Survival	Salmonid LC50 (96 h)	Rainbow trout (*Oncorhynchus mykiss*) (95%; T) = 720	Rainbow trout (*O. mykiss*) (92.9%; T) = 1,100[b]	
Fish growth	NOEC/LOEC		Fathead minnow (*Pimephales promelas*) (93.4%; T) = 374/789	
Habitat-salmonid prey	Invertebrate survival (48 h EC/LC50)		Water flea (*Daphnia magna*) (92.9%; T) = 580 Amphipod (*Gammarus fasciatus*) (95%; T) = 1,400[a]; 3,330[b] Mysid (*Americamysis bahia*) (92%; T) = 62.4[a]	
	Invertebrate survival and growth rate NOEC/LOEC (21 day)		Water flea (*D. magna*) (93.4%; T) = 4.2/10	
	Invertebrate reproduction NOEC/LOEC (21 day)		Water flea (*D. magna*) (93.4%; T) = <6.9/6.9	

Assessment Endpoint		**Concentration (μg/L)**		
		Bensulide		**degradate(s)**
	Assessment measure	**> 95% a.i. (% a.i.)**	**< 95% a.i. (%a.i.)**	
	Growth Inhibition of aquatic primary producers (algae sp.)	Green Algae (*Pseudokirchneriella subcapitata*) (% a.i. NR) = 1500; 1850; 1862; 2352; 2842	Green algae (*Anabaena floss-aquae*) (93.4%, T) = > 3580; Green algae (*Selenastrum capriconutum*) (93.4%, T) = 1800	

Assessment Endpoint		Concentration (μg/L)		
		Dimethoate		Degradate(s): omethoate
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
Survival	Freshwater, estuarine, and marine fish LC50 (96 h)	Bluegill sunfish (*Lepomis macrochirus*) (97.4%; T) = 6,000 Longnose killifish (*Fundulus similis*) (99.3%; T) (sw) = >1,000[b] Sheepshead minnow (*Cyprinodon variegatus*) (sw) (99.1%; T) = >111,000	Goldfish (*Carassius auratus*) (30.5%) = 180,000[b]	
Survival	Salmonid LC50 (96 h)	Rainbow trout (*O. mykiss*) (97.4%; T) = 6,200 Rainbow trout (*O. mykiss*) (95%; T) = 7,500		Rainbow trout (*O. mykiss*) (%degradate NR) = 9100
Fish growth	NOEC/LOEC	Rainbow trout (*O. mykiss*) (99.1%; T) = 430/840		
Habitat-salmonid prey	Invertebrate survival (48 h EC/LC50)	Water flea (*Daphnia magna*) (>95%; T) = 3,320 ; 5,040 Yellow fever mosquito (*Aedes aegypti*) (>95%; T) = 5,040 Salt marsh mosquito (*Aedes taeniorhychus*) (> 95%; T) = 31 Mysid shrimp (*Mysidopsis bahia*) (99.1%; T) = 15,000[a] Brown shrimp (*Paneaus. aztecus*) (99.3%; T) = >1,000 Brine shrimp (*Artemia* sp.) (>95%) =		Water flea (*Daphnia magna*) (% degradate NR) = 22

Assessment Endpoint		Concentration (μg/L)		
		Dimethoate		Degradate(s): omethoate
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
		15,730 Eastern oyster (*Crassostrea virginica*) (99.1%; T) = 113,000[a] Scud (*Gammarus lacustris*) (97.4%; T) = 200[b] Stonefly (*Pteronarcys californica*) (97.4%; T) = 43[b]		
	Invertebrate survival and growth rate NOEC/LOEC (21 day)	Water flea (*D. magna*) (99%; T) = 40/100	Water flea (*D. magna*) (94%; T)= 220/450	
	Invertebrate reproduction NOEC/LOEC (21 day)	Water flea (*D. magna*) (99%; T) = 40/100	Water flea (*D. magna*) (94% T) = 220/450	

Assessment Endpoint		Concentration (µg/L)		
		Disulfoton		**degradate (s):** disulfoton sulfone, disulfoton sulfoxide
	Assessment measure	**> 95% a.i. (%a.i.)**	**< 95% a.i. (%a.i.)**	
Survival	Freshwater, estuarine, and marine fish LC50 (96 h)	Channel catfish (*Ictalurus punctatus*) (98%) = 4,700 Bluegill sunfish (*L. macrochirus*) (% a.i. NR; T) = 39 Sheepshead minnow (*Cyprinodon variegatus*) (sw) (95.5%) = 520 Sheepshead minnow (*C. variegatus*) (sw) (97.8%) = >1,000 Largemouth bass (*Micropterus salmoides*) (98%) = 60 Fathead minnow (*Pimephales promelas*) (98%) = 4,300	Bluegill sunfish (*L. macrochirus*) (20% E, % a.i. NR) = 8.2 Bluegill sunfish (*L. macrochirus*) (15%G, % a.i. NR) = 250 Bluegill sunfish (*L. macrochirus*) (65% EC, % a.i. NR) = 59 Goldfish (*Carassius auratus*) (90%) = 7,200 Guppy (*Poecilia reticulata*) (90%) = 280	Bluegill sunfish (*L. macrochirus*) disulfoton sulfone (20% E) = 112 Bluegill sunfish (*L. macrochirus*) disulfoton sulfoxide metabolite (20%E) = 188
Survival	Salmonid LC50 (96 h)	Rainbow trout (*Oncorhynchus mykiss*): (% a.i. NR; T) = 3,000 (98%; T) = 1,850	Rainbow trout (*O. mykiss*) (65% EC, % a.i. NR) = 3,500 Rainbow trout (*O. mykiss*) (15% G, % a.i. NR) = 13,900	Rainbow trout (*O. mykiss*) (65%EC) disulfoton sulfone = 9,200 Rainbow trout (*O. mykiss*) (65%EC) disulfoton sulfoxide = 60,300
Reproduction or larval survival	NOEC/LOEC (21 day)	Sheepshead minnow (*C. variegatus*) (sw) (98%) = 0.96/2.9	Sheepshead minnow (*C. variegatus*) (sw) (94.7%) = 16.2/32.9	
Fish growth	NOEC/LOEC (21 day)	Rainbow trout (*O. mykiss*) (98%) = 220/420		

Assessment Endpoint		Concentration (μg/L)		
		Disulfoton		**degradate (s):** disulfoton sulfone, disulfoton sulfoxide
	Assessment measure	**> 95% a.i. (%a.i.)**	**< 95% a.i. (%a.i.)**	
Habitat-salmonid prey	Invertebrate survival (48 h EC/LC50)	Water flea (*Daphnia magna*) (98.6%) = 13 Scud (*Gammarus fasciatus*) (98%) = 52 Glass shrimp (*Palaemonetes kadiakensis*) (98%) = 3.9 Mysid (*Americamysis bahia*) (97.8%) = 100 Brown shrimp (*Penaeus aztecus*) (95.5%) = 15 Stonefly (*Pteronarcys californica*) (98%) = 5 Eastern oyster (*Crassotrea virginica*) (97.8%) = 720 Eastern oyster (*C. virginica*, (% a.i. NR; T) = 900 Eastern oyster (*C. virginica*) (95.5%) = 720	Stonefly (*Acroneuria pacifica*) (89%) = <8.2	Water flea (*D. magna*) disulfoton sulphone (87.4%) = 35.2 Water flea (*D. magna*) (85.3%) disulfoton sulfoxide = 64
	Invertebrate survival, length, and #young/adult NOEC/LOEC (21 day)	Water flea (*D. magna*) (98%) = 0.037/0.070		Water flea (*D. magna*) disulfoton sulfone (21d) = 0.14/0.27 Water flea (*D. magna*) disulfoton sulfoxide (21 d) = 1.53/2.97
	Invertebrate growth NOEC/LOEC	Mysid (*Americamysis bahia*) (98.5%) = 2.35/8.26		

Assessment Endpoint		Concentration(μg/L)		
		Ethoprop		degradate (s): OME, SME, M1
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (%a.i.)	
Survival	Freshwater, estuarine, and marine fish LC50 (96 h)	Bluegill sunfish (*Lepomis macrochirus*) (99.7%;T) = 300 Sheepshead minnow (*Cyprinodon variegatus*) (96.8%;T) (sw) = 180; 958 Pinfish (*Logodon rhomboides*) (95%; T) = 6.3 Spot (*Leiostomus xanthurus*) (95%; T) = 33		No aquatic data Acute rat data includes degradate info LD50 Parent 32.8 mg/kg SME 50.0 mg/kg OME 22.4 mg/kg M1 1608 mg/k
Survival	Salmonid LC50 (96 h)	Rainbow trout (*Oncorhynchus mykiss*) (% a.i. NR; T) = 13,800	Rainbow trout (*Oncorhynchus mykiss*) (92%; T) = 1,020; 1,150	
Reproduction or larval survival	NOEC/LOEC	Sheepshead minnow (*C. variegatus*) (sw) (95%;T) = 12/21 Fathead minnow (*Pimephales promelas*) (% a.i. NR; T) = 24/54		
Fish growth	NOEC/LOEC	Sheepshead minnow (*C. variegatus*) (sw) (96.8%; T) = 5.9/11		
Habitat-salmonid prey	Invertebrate survival (48 h EC/LC50)	Water flea (*Daphnia magna*) (99.7%; T)= 93[b] (% a.i. NR; T) = 44		

Assessment Endpoint		**Concentration(μg/L)**		
		Ethoprop		**degradate (s): OME, SME, M1**
	Assessment measure	**> 95% a.i. (% a.i.)**	**< 95% a.i. (%a.i.)**	
	Invertebrate survival and growth NOEC/LOEC	Mysid (*Americamysis bahia*) (95%; T) = 360/ 620 Mysid (*A. bahia*) (96.8%; T) = 1,400/ 2,700 Water flea (*Daphnia magna*) (96.8%; T) = 0.8/2.4		
	Invertebrate reproduction NOEC/LOEC (21 day)	Water flea (*D. magna*) (96.5%; T)= 2.4/5.4		

<table>
<tr><th rowspan="3">Assessment Endpoint</th><th></th><th colspan="3">Concentration(µg/L)</th></tr>
<tr><th></th><th colspan="2">Fenamiphos</th><th>degradate (s): fenamiphos sulfoxide, fenamiphos sulfone</th></tr>
<tr><th>Assessment measure</th><th>> 95% a.i.
(% a.i.)</th><th>< 95% a.i.
(%a.i.)</th><th></th></tr>
<tr><td>Survival</td><td>Freshwater, estuarine, and marine fish LC50 (96 h)</td><td></td><td>Bluegill Sunfish
(Lepomis macrochirus)=
(36%) = 4.5 (95% CI 3.9-5.1);
(88.0%; T) = 9.5 (95% CI 6.8-15.0);
(81%; T) = 17.7 (95% CI 14.4-21.6);
(15%) = 151 (95% CI 114-201)

Sheepshead minnow (sw)
(Cyprinodon variegatus)=
(88.7%; T) = 17.0;
(36.0%)= 320</td><td>Bluegill Sunfish
(Lepomis macrochirus)
fenamiphos sulfoxide (% degradate NR)=2000; 2653

Bluegill Sunfish
(Lepomis macrochirus)
fenamiphos sulfone (% degradate NR)=1173</td></tr>
<tr><td>Survival</td><td>Salmonid LC50 (96 h)</td><td></td><td>Rainbow Trout
(O. mykiss) =
(36%) = 68 (95% CI 59.6-77.1)
(81%; T) = 72.1 (95% CI 61.2-84.7)
(15%) = 563 (95% CI 454-698)</td><td></td></tr>
<tr><td>Fish growth</td><td>NOEC/LOEC</td><td></td><td>Rainbow Trout
(O. mykiss) (88.7%;T)= 3.8/7.4</td><td></td></tr>
<tr><td>Habitat-salmonid prey</td><td>Invertebrate survival (48 h EC/LC50)</td><td></td><td>Water flea
(Daphnia magna)
(88.7%;T) = 1.9 (95% CI 1.7-2.1)
(36.0%)= 1.3

Scud
(Gammarus italicus)=
(Formulation, %a.i. NR) = 20[a]

Mysid (Americamysis bahia) (sw)
(88.7 %; T) = 6.2;
(Formulation, %a.i. NR) = 6.8

Pink shrimp
(Panaeus duorarum)
(36.0%)= 150</td><td>Water flea
(Daphnia magna)
fenamiphos sulfoxide (% degradate NR)=
7.5</td></tr>
</table>

Assessment Endpoint		Concentration(μg/L)		
		Fenamiphos		**degradate (s):** fenamiphos sulfoxide, fenamiphos sulfone
	Assessment measure	**> 95% a.i. (% a.i.)**	**< 95% a.i. (%a.i.)**	
			Eastern oyster (*Crassostrea Virginica*) (36.0%) = >1000 (88.7%; T) =1,650[a] Rotifer (*Brachionus plicatilis*) (formulation, %a.i. NR) = 3,000; (formulation, %a.i. NR) = 10,000	
	Invertebrate reproduction and growth NOEC/LOEC 21 d	Water flea *(Daphnia magna)* (99.6%; T)= 0.12/0.24		

Assessment Endpoint		Concentration(µg/L)		
		Methamidophos		degradate (s)
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
Survival	Freshwater, estuarine, and marine fish LC50 (96 h)		Bluegill Sunfish (*Lepomis macrochirus*) (74.0%)= 34,000 (95% CI 30,000-38,000) (75.4%) = 45,000 (95% CI 35,000-58,000) (75.0%) = 46,000 (95% CI 34,000-62,000) (40%) = 31,000 (95% CI 21,000-46-000) Carp (*Cyprinpus carpio*)(90.0%)=68,000 Sheepshead minnow(sw) (*Cyprinodon variegatus*) (70.1%) = 5,630 (95% CI 4,130-6,890)	
Survival	Salmonid LC50 (96 h)		Rainbow Trout (*O. mykiss*) = (74.0%) = 25,000 (95% CI 21,000-29,000) (40.0%) = 37,000 (95% CI 28,000-49,000) (71.0%) = 40,000 (95% CI 35,000-46,000) (75.0%) = 51,000 (95% CI 36,000-72,000)	
Reproduction or larval survival			-	
Fish growth	-	-	-	-
Habitat-Salmon prey	Invertebrate survival (48 h EC/LC50)		Water Flea (*Daphnia magna*) (74.0%) = 26 (95% CI 20-34); (74.0%; T) = 27 (95% CI 14-53); (72.0%) = 50 (95% CI 40-70) Freshwater prawn (*Macrobrachium rosenbergii*)	

Assessment Endpoint		Concentration(µg/L)		
		Methamidophos		degradate (s)
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
			(% a.i. NR)= 0.042 Mysid (*Americamysis bahia*)(sw) (% a.i. NR; T) = 1,054 Blue shrimp (*Penaeus stylirostris*)(sw) (% a.i. NR) = 0.16 Eastern Oyster (*Crassostrea viginica*) (sw) (72.9%) T = 36,000 (30,000-47,000)	
	Invertebrate reproduction and growth NOEC/LOEC 21 d		Water flea *(D. magna)* (78.5.0%) = 4.49/5.32	

Assessment Endpoint		Concentration(µg/L)		
		Methidathion		degradate (s)
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
Survival	Freshwater, estuarine, and marine fish LC50 (96 h)	Bluegill Sunfish (*Lepomis macrochirus*) (95.0%) = 2.2 (98.5%; T) = 9 Goldfish (*Carassius auratus*) (NR)(97.7%) = 6.8	Bluegill Sunfish (*L. macrochirus*) (NR)(25.2%) = 8.2 Sheepshead minnow (sw) (*Cyprinodon variegatus*) (25.2%) = 7.8 - 111.9 n=3	
Survival	Salmonid LC50 (96 h)	Rainbow Trout (*O. mykiss*) (97.7%; T) = 10.0 (98.55%; T) = 14	Rainbow Trout (*O. mykiss*) (25.2%) = 6.6	
Reproduction or larval survival				
Fish growth	NOEC/LOEC 35 d	Fathead minnow (*Pimephales promelas*) (99.2%; T) = 6.1/12		
Habitat-Salmonid prey	invertebrate survival (48 h EC/LC50)	Water Flea (*D. magna*) (%a.i. NR; T) = 7.2 Mysid (SW) (*Americamysis bahia*) (97.2%)= 0.7 (95 % CI 0.44-0.99)	Water Flea (*D. magna*) (25.5%) = 3.0 Mysid (*A. bahia*) (sw) (25.2%)2.34; 0.59	
Habitat-Salmonid prey	invertebrate reproduction and growth NOEC/LOEC 21 d	Water Flea (*D. magna*) (96.1%) = 0.66/1.13; (97.2%)= 0.51/1.0	Mysid (*A. bahia*) (sw) (% a.i. NR) = 0.02/0.06	

Assessment Endpoint		Concentration (µg/L)		
		Methyl parathion		**degradate (s):** 4-nitrophenol, methyl paraoxon
	Assessment measure	**> 95% a.i. (% a.i.)**	**< 95% a.i. (% a.i.)**	
Survival	Freshwater, estuarine, and marine fish LC50 (96 h)	Spot (*Leiostomous xanthrus*) (sw) (99%; T) = 59 (45 – 74)	Channel catfish (*Ictalurus punctatus*) (90 %) = 5,240 (4,270-6,440) Fathead minnow (*Pimephales promelas*) (%a.i. NS; T)= 1,220 (80%) = 3,470 (80%) = 9,500 Sheepshead minnow (*Cyprinodon variegatus*) (sw) (43.2%) = 3,400 (2,800-4,100); (90%: T) =12,000 Bluegill Sunfish (*Lepomis macrochirus*) = (77%) = 1,000 (600-1,600); formulation (% a.i. NR) = 1,600 (80%) = 2,400; (90%; T) = 4,380; (44.6%) = 11,200 Green Sunfish (*L. cyanellus*) = (90%; T) = 6,860 Pumpkinseed (*L. gobbosus*)= formulation (%a.i.NR) =3600 Redear Sunfish (*L. microlophus*) formulation (%a.i. NR) = 5,170 (4,410 – 6,090) Striped Bass	

Assessment Endpoint		Concentration (µg/L)		
		Methyl parathion		**degradate (s):** 4-nitrophenol, methyl paraoxon
	Assessment measure	**> 95% a.i. (% a.i.)**	**< 95% a.i. (% a.i.)**	
			(*Morone saxatilis*) (80%) = 790 (170-1,400) Snakehead catfish (*Channa punctata*) formulation (% a.i. NS) = 2,150 Smooth breasted snakehead (*Channa orientalis*)= formulation (% a.i. NR) = 4,900	
Survival	Salmonid LC50 (96 h)		Brown Trout (*Salmo trutta*) (90%)= 4,700 Coho salmon (*Oncorhynchus kisutch*) (90%) = 5,300 Cutthroat Trout (*O. clarki*) (90%) = 1,850 (95% CI 390-2,470) Rainbow Trout (*O. mykiss*) = 2,200 – 3,700 n=4	Rainbow Trout (*O. mykiss)* 4-nitrophenol (% degradate NR) = 4,000
Fish growth	NOEC/LOEC		Fathead Minnow (*Pimephales promelas*) (% a.i. NR) = 310/380 Rainbow Trout (*O. mykiss*) (75.1%) = <10/10; NR/<80	
Habitat-Salmon prey	Invertebrate survival (48 h EC/LC50)	Copepod (*Acartia tonsa*) (sw) (99%; T) = 28.0 Mysid *(Americamysis bahia)* (sw) (% a.i. NR; T) = 0.77 (95 % CI 0.64-0.98)	Mosquito 4th instar *(Aedes nigromaculis)* formulation (% a.i NR) = 8.0 Mosquito 4th instar (*Culex tarsalis*)= formulation (% a.i. NR)= 3.6	Water flea (*D. magna*) Methyl paraoxon = 2.3 (95% CI 2.2-2.5) 24 h Water flea (*D. magna)* 4-nitrophenol

Assessment Endpoint		Concentration (µg/L)		
		Methyl parathion		**degradate (s):** 4-nitrophenol, methyl paraoxon
	Assessment measure	**> 95% a.i. (% a.i.)**	**< 95% a.i. (% a.i.)**	
		Mysid *(A. bahia)* (sw) = (99%; T) = 0.78 (95 % CI 0.58-1.1)	Mosquito 4th instar (*Culex tritaeniorhynchus*) formulation (%a.i. NR) = 0.54 Crayfish (*Orconectes nais*) (90%)=15 Scud (*Gammarus italics*) formulation (%a.i. NR) = 2.9 (95% CI 2.3-4.4) Water flea (*D. magna*) (90%; T) = 0.14 (95% CI 0.09-0.2) Water flea (*Ceriodaphnia dubia*) formulation (% a.i. NR) = 0.97 (95% CI 0.8-1.18) formulation (% a.i. NR) = 2.6 (95% CI 2.1-3.1) Water flea (*Simocephalus serrulatus*)= (90%; T) = 0.37 (95% CI 0.23-0.57) Mysid (*A. bahia*) (sw) (43.2%)= 0.35 (95% CI 0.31-0.39)	(%degradate NR) = 5,000
Habitat-Primary productivity	Aquatic plant EC50 (96 h)	Marine diatom (*Skeletonema costatum*) (99%; T) = 5,300 (4,300 – 5,700)		
Habitat-Salmonid prey	Invertebrate reproduction and	Water flea (*D. magna)* (96%; T) = 0.178/0.562	Water flea (*D. magna*) (80%) = 0.2/0.25 (75.1%; T) = 0.16/2.51	Water flea (*D. magna*) Methyl paraoxon (% degradate NR) = 1.0/1.5 24 h exposure, chronic effects noted

Assessment Endpoint		**Concentration (µg/L)**		
		Methyl parathion		**degradate (s):** 4-nitrophenol, methyl paraoxon
	Assessment measure	**> 95% a.i. (% a.i.)**	**< 95% a.i. (% a.i.)**	
	growthNOEC/ LOEC (21 day)		Water flea (*Ceriodaphnia dubia*) (% a.i. NR) = 0.99/1.37 Mysid (*A. bahia*) (sw) (% a.i. NR) = 0.11/0.37	following removal to clean media

Assessment Endpoint		Concentration(μg/L)		
		Naled		degradate(s): Dichlorvos
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
Survival	Freshwater, estuarine, and marine fish LC50 (96 h)	Bluegill sunfish (*L. macrochirus)* (% a.i. NR; T) = 600	Bluegill Sunfish *(L. macrochirus)* (90%; T) = 2,200 Bluegill Sunfish *(L. macrochirus)* (Formulation, 15%) = 4,000 Bluegill Sunfish *(L. macrochirus)* (Formulation, % a.i. NR) = 180 Fathead minnow *(Pimephales promelas)* (90%; T) = 3,300 Channel Catfish (*Ictalurus punctatus)* (90%; T) = 710 Largemouth bass (*Micropterus salmoides)* (90%; T)= 1,900 Sheepshead minnow (*Cyprinodon variegates)* (sw) (T, 90% a.i.) = 1,200 Sheepshead minnow (*C. variegates)* (sw) (Formulation, 59.5%) = 1,200	Bluegill Sunfish (*L. macrochirus) (*4-100% dichlorvos)= 800->180,000, n = 6 Fathead minnow (*Pimephales promelas)* (100% dichlorvos) = 11,600 Sheepshead Minnow *(Cyprinodon variegates)* (sw) = (98% dichlorvos) = 7350 (42.4% dichlorvos) = 14,400
Survival	Salmonid LC50 (96 h)		Lake Trout (*Salvelinus namaycush*) (90%; T) = 87 (95% CI 53-142); (%a.i. NR; T) = 92 Rainbow Trout *(Oncorhynchus mykiss)* (90%; T) = 195; 345 Rainbow Trout *(O. mykiss)* (% a.i. NR; T) = 160; 210 Rainbow Trout (*O. mykiss*)	Lake Trout (*Salvelinus namaycush)* (100% dichlorvos) = 187 (90% dichlorvos) = 183 Rainbow Trout *(O. mykiss)* = (98.1% dichlorvos) = 100 (100% dichlorvos) = 100 (42.4% dichlorvos) = 750 (100% dichlorvos) = 500 (24 h LC50)

Assessment Endpoint		Concentration(µg/L)		
		Naled		degradate(s): Dichlorvos
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
			(Formulation, 15%) = 900 Rainbow Trout *(O. mykiss)* (Formulation, % a.i. NR) = 132-340 n = 5 Cutthroat Trout *(O. clarki)* (90%; T) =127 (95% CI 115-139)	Cutthroat Trout (*O. clarki)* (100% dichlorvos)= 170 (90% dichlorvos) = 213
Fish growth	NOEC/LOEC (Length and weight)		Fathead minnow (*Pimephales promelas)* (94.4%) NOEC = 6.9; LOEC = 15 (35 d exposure)	Sheepshead minnow *Cyprinodon variegatus* (sw) (98% dichlorvos) NOEC = 960 LOEC = 1840 (34 days)
Habitat-salmonid prey	Invertebrate survival (48 h LC/EC50)	Scud (*Gammarus fasciatus)* (97%;T) = 0.14	Water flea (*Daphnia pulex*) (90%; T) = 0.4 Water flea (*Daphnia magna*) (91.6%; T) = 0.3 Water flea (*D. magna)* (Formulation, 15%-85%) = 0.5 – 2.9 n = 4 Water flea (*D. pulex)* (Formulation, % a.i. NR) =0.35 Scud (*Gammarus fasciatus)* (90%; T) = 18 [a] (95% CI 16-20) Scud (*G. fasciatus)* (% *a.i.* NR; T) = 14 [a] (95% CI 11-18) Scud (*G. lacustris)* (Formulation, % *a.i.* NR) = 110[a] Water flea (*Simocephalus serrulatus)* (90%; T) = 1.1 (95% CI 0.8-1.4) Stonefly (*Pteronarcys californica)* (90%;T)	Water flea (*Daphnia sp.)* (100% dichlorvos) = 0.066 (95% CI 0.05-0.09); (46% dichlorvos) = 1,000 (95% CI 800-1400) Scud (*Gammarus fasciatus)* (100% dichlorvos) = 0.50[a] (95% CI 0.37-0.7) (tech dichlorvos) = 400,000[a] (95% CI 320,000 – 490,000) Water flea (*Simocephalus serrulatus)* (100% dichlorvos) = 0.26 (95% CI = 0.16-0.42) Stonefly (*P. californica)* (100% dichlorvos) = 0.10[a] (95% CI 0.07-0.15) Mysid (*Mysidopsis bahia)* (sw) = (98% dichlorvos) = 19;

Assessment Endpoint		Concentration(µg/L)		
		Naled		degradate(s): Dichlorvos
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
			= 8[a] (95 % CI 6-11) Stonefly (*P. californica)* (Formulation, % a.i. NR) = 8 Sowbug (*Asellus brevicaudus*)(90%; T) = 41[a] Sowbug (*Asellus brevicaudus)* (% a.i. NR, T) = 230[a] Mysid (*Mysidopsis bahia)* (sw) (Formulation, 59.6%) = 8.8[a]	(42.4% dichlorvos) = 44
	Invertebrate NOEC/LOEC (length and weight)	Water flea *(D. magna)* (T, 97.3%) NOEC = 0.045 LOEC = 0.098	Mysid *Mysidopsis bahia* (sw) (89.2%; T) = NOEC < 0.2 LOEC = 0.2 (31 d)	Water flea (*D. magna*) (98% dichlorvos) NOEC = 0.0058 LOEC = 0.0122 Growth and # of young produced: (21 d) Mysid (*Mysidopsis bahia)* (sw) (98% dichlorvos) NOEC = 1.48 LOEC = 3.25 length, growth: (28 d)

Assessment Endpoint		Concentration(µg/L)		
		Phorate		**degradate(s):** Phorate sulfoxide, Phorate sulfone
	Assessment measure	**> 95% a.i. (% a.i.)**	**< 95% a.i. (% a.i.)**	
Survival	Freshwater, estuarine, and marine fish LC50 (96 h)	Bluegill sunfish (*Lepomis macrochirus*) (100%; T)= 1.0 Walleye (*Stizostedion vitreum v.*) (100%; T)= 57	Channel catfish (*Ictalurus punctatus*) (91%; T) T= 280 Channel catfish (*I. punctatus*) (20G, %a.i. NR) = 2.2 Largemouth bass (*Micropterus salmoides*) (91%) = 5.0 Bluegill sunfish (*L. macrochirus*) (91%) = 2 Bluegill sunfish (*L. macrochirus*) (20G, % a.i. NR) = 12 Bluegill sunfish (*L. macrochirus*) (66%) = <2.8 Northern pike (*Esox lucius*) (91%) = 110 Sheepshead minnow (*Cyprinodon variegatus*) (sw) (90%) = 1.3 Sheepshead minnow (*C. variegatus*) (sw) (89.5%) = 4 (95% CI 3.5 – 4.5) Sheepshead minnow (*C. variegatus*) (sw) (20%G, % a.i. NR) = 8.2 (95% CI 5.5 – 10) Longnose killifish (*Fundulus similis*) (sw) (90%) = 0.36[b] Spot (*Leistomus xanthurus*) (sw) (89.5%) = 5 (95% CI 4.2 – 5.6) Spot (*L. xanthurus*) (sw) (90%) = 3.9	Bluegill sunfish (*L. macrochirus*) Phorate sulfoxide (% degradate NR) = 22

Assessment Endpoint		Concentration(µg/L)		
		Phorate		**degradate(s):** Phorate sulfoxide, Phorate sulfone
	Assessment measure	**> 95% a.i. (% a.i.)**	**< 95% a.i. (% a.i.)**	
Survival	Salmonid LC50 (96 h)	Rainbow trout (*Oncorhynchus mykiss*) (100%;T) = 13 Cutthroat trout (*O. clarki*) (100%; T) = 66	Rainbow trout (*O. mykiss*) (20G%, % a.i. NR) = 45 Rainbow trout (*O. mykiss*) (66%) = 19	
Fish growth	NOEC/LOEC	Sheepshead minnow (*Cyprinodon variegatus*) (sw) (99%) = 96/190	Rainbow trout (*O. mykiss*) (92.1%) = 1.9/4.2	
Habitat-salmonid prey	Invertebrate survival (48 h EC/LC50)	Stonefly (*Pteronarcys californica*) (100%; T)= 4 (95%CI 2-6)	Water flea (*Daphnia magna*) (20G, % a.i. NR) = 37[b] (95% CI 30-44) Midge larvae (*Paratanytarsus parthenogenical*) (20G, % a.i. NR) = 41[a] (95% CI 38-45) Mayfly nymphs (*Hexagenia* sp.) (20G, % a.i. NR) = 65 (95% CI 7 – 74) Mysid (*Americamysis bahia*) (89% a.i.) = 1.9 (95% CI 1.0 – 3.2) Mysid (*A. bahia*) (90%) = 0.31 Mysid (*A. bahia*) (20G%, % a.i. NR) = 0.3; 1.4 Scud (*Gammarus fasciatus*) (% a.i. NR; T) = 4; 9	Water flea (*D. magna*) Phorate sulfoxide (% degradate NR)= 4.0 Water flea (*D. magna*) Phorate sulfone (% degradate NR) = 0.4
	Invertebrate survival and growth rate NOEC/LOEC (21 day)	Mysid (*A. bahia*) (99%; T) =9/21	Water flea (*D. magna*) (92.1%) = 0.29/0.44	

Assessment Endpoint		Concentration(µg/L)		
		Phorate		**degradate(s):** Phorate sulfoxide, Phorate sulfone
	Assessment measure	**> 95% a.i. (% a.i.)**	**< 95% a.i. (% a.i.)**	
	Invertebrate reproduction NOEC/LOEC (21 day)	Water flea (*D. magna*) (100%; T) = 0.21/0.41		

Assessment Endpoint		Concentration(µg/L)		
		Phosmet		degradate(s)
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
Survival	Freshwater, estuarine, and marine fish LC50 (96 h)	Bluegill sunfish (*Lepomis macrochirus)* (95.8%; T) = 70; (95.3%; T) = 120 Channel catfish (*Ictalurus punctatus)* (95.8%); T) =10,600 Fathead minnow (*Pimephales promelas)* (95.8%; T) = 7,300 Smallmouth bass (*Micropterus dolomieui*) (95.3%;T) = 150 Largemouth bass *(Micropterus salmoides)* (95.3%; T) = 160 Longnose killifish (*Fundulus similis)* (95% a.i.; T) = 32[b] Striped mullet (*Mugil cephalus)* (95%; T) = 32[b]	Channel catfish (*I. punctatus)* (Formulation, 50%) = 7,500 Channel catfish (*I. punctatus*) = 13,000 (Formulation, % a.i. NR Fathead minnow *P. promelas* (Formulation, 50%) =7,500; 9,000 Sheepshead minnow (*Cyprinodon variegates*) (sw) (94%;T)= 170	
Survival	Salmonid LC50 (96 h)	Chinook salmon *Oncorhynchus tshawytscha* (95.3%; T) = 150; 285[b] Rainbow Trout (*O. mykiss)* (95.8; T) = 230 (97%; T) = 560	Rainbow Trout (*O. mykiss*) (Formulation, 11.55%) = 1,560 (Formulation, 50%) = 290; 500	
Reproduction or larval survival	NOEC/LOEC		Rainbow Trout (*O. mykiss)* (94.3%; T) = 3.2/6.1	

Assessment Endpoint		Concentration(µg/L)		
		Phosmet		degradate(s)
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
Fish growth	NOEC/LOEC		Rainbow Trout *(O. mykiss*) (94.3%; T) = 3.2/6.1	
Habitat-salmonid prey	Invertebrate survival (48 h EC/LC50)	Water flea (*D. magna)* (95.8%; T) = 5.6 Mysid (*Mysidopsis bahia)* (sw) (94.3% T) = 1.6 Scud *(Gammarus fasciatus)* (95.8%; T) = 4.2 [a] ; 2 [a] Aquatic Sow bug (*Asellus brevicaudus)* (95.3%;T) = 72 [a] ; 90[a] Midge (*Chironomus plumosus)* (95.3%; T) = 3,150 Brown shrimp (*Penaeus aztecus)* (95%; T)= 2.5	Water flea (*D. magna)* (Formulation, 50%)= 10.9 (Formulation, 51% a.i.) = 24 Aquatic Sow Bug (*Asellus brevicaudus*) = 100[3] (Formulation, % a.i. NR) Scud (*Gammarus fasciatus*) (Formulation, % a.i. NR) = 2.4; (% a.i. NR) = 2 Midge (*C. plumosus*) (Formulation, 50% a.i.) = 3,400	
	Invertebrate reproduction and growth NOEC/LOEC (21 day)	Water flea (*D. magna)* (99%; T) = NOEC 0.75 LOEC 1.1 Mysid *Mysidopsis bahia* (sw) (95.5%; T) = NOEC 0.37 LOEC 0.69	Water flea (*D. magna*) (% a.i. NR) = NOEC 0.8 LOEC 1.1	

Abbreviations as follows: NR = Not Reported; T= Technical grade; Formulation = formulated product; EC=Emusifiable concentrate, G=granular, sw = estuarine and/or marine species. [a]96 h test; [b]48 h test

Identified data gaps and uncertainties in EPA-submitted toxicity information and analyses for the twelve OP pesticides

Our review of the toxicity information presented in EPA's risk assessments uncovered data gaps, uncertainties, and absence of analyses which hamper our ability to evaluate the risk posed by EPA's pesticide re-registration actions to listed salmonids and designated critical habitat. Overall the toxicity data provided in BEs, including BEs developed for ESA Section 7 consultation with the U.S. Fish and Wildlife Service for the California Red Legged Frog (CRLF), IREDs, REDs, and EFED Science Chapters were insufficient to allow for a thorough evaluation of assessment endpoints and measures identified as relevant to this consultation. Several relevant aspects of EPA's actions were either not addressed or dismissed, and in some cases processes employed by EPA screened out relevant toxicity information. The primary uncertainties arise from data gaps and from lack of analyses for mixtures, sublethal effects, and toxicity data on degradates, adjuvants, and other ingredients within formulations. The CRLF BEs contained the most robust review of toxicity information compared to the other EPA documents, although lack of discussion of sublethal effects and absence of a mixture analysis remained. Because of this, NMFS supplemented EPA–provided information with open literature data.

No assessment of mixture toxicity from label-recommended tank mixes, environmental mixtures containing other AChE-inhibiting insecticides, and current formulations was found in the EPA-provided information. Each of these mixture types may influence salmonid and habitat responses, but no assessments were conducted to characterize potential risk from mixtures to listed salmonids. NMFS is charged with using the best commercial and scientific data available when conducting Section 7 consultation which include historical and current information on mixture toxicity. BEs that do not use available scientific information in respect to mixtures likely underestimate risk to listed salmonids, particularity in the case of OP pesticides where data show additive and synergistic toxicity.

Although there is inherent difficulty in addressing the potential for mixture toxicity of every possible combination of co-occurring chemical with the 12 OP active ingredients, established methods exist for evaluating mixture toxicity for chemicals that share a common mode of action *e.g.*, inhibition of AChE. As with the first two NMFS' Opinions that evaluated other AChE-inhibiting insecticides (three OPs [chlorpyrifos, diazinon, malathion] (NMFS 2008c) and three carbamates [carbaryl, carbofuran, methomyl] (NMFS 2009b)) NMFS reviews and evaluates the risk posed by combinations of the 12 a.i.s and their oxon degradates (as applicable) within the present Opinion. To determine the potential for mixture toxicity of EPA's actions, we compiled, reviewed, and applied study results in both qualitative and quantitative mixture analyses within the *Risk Characterization Section*.

Information on sublethal toxicity was not used within the BEs to determine potential risk to listed salmonids or their habitat. Inhibition of AChE, effects on swimming, olfactory-mediated behavior impairment, endocrine disruption, and other sublethal endpoint data were not compiled, reviewed, or used. These types of data are apparently screened out and discounted due to lack of available "quantitative" links between the sublethal effect and an individual's fitness[7]. NMFS reviews all available effects data including sublethal effects to determine whether individual and habitat endpoints are potentially affected.

We also found discrepancies between toxicity values within the public version of ECOTOX and toxicity values presented in EPA documents (BEs, CRLF BEs, REDS/IREDs and EFED Science Chapters). Because we did not have primary sources or the resources to validate ECOTOX, we relied on the EPA-supplied information and not the values within ECOTOX.

Below are examples of specific data gaps and uncertainties identified, although not all apply to each of the 12 a.i.s.

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[7] Language on sublethal effects from the CRLF dimethoate BE (similar language is also found in the other CRLF BEs).*"Open literature is useful in identifying sublethal effects associated with exposure to dimethoate. However, no data are available to link the sublethal measurement endpoints to direct mortality or diminished reproduction, growth and survival that are used by OPP as assessment endpoints. OPP acknowledges that a number of sublethal effects have been associated with diemthaote exposure; however, at this point there are insufficient data to definitively link the measurement endpoints to assessment endpoints."*

- Reported LC50s generally not accompanied by slopes, experimental design (number of treatments and replicates, life stage of organism, concentrations tested), measures of variability such as confidence intervals or standard deviations/errors;
- No analysis of the degree or magnitude of inhibition of acetylcholinesterase by the 12 a.i.s and associated responses by listed salmonids or aquatic, salmonid prey communities;
- No spatial analysis of influence of environmental factors (*e.g.,* temperature) on exposure and toxicity although more fish die when exposed to elevated temperatures than when exposed to lower temperature when exposed to OPs;
- Limited or no toxicity data on current formulations containing the 12 a.i.s;
- Limited or no toxicity data presented for identified surface water degradates of the 12 a.i.s;
- Sensitivity of surrogate lab strains compared to wild, listed fish, particularly comparisons between warm and cold water fish species used in chronic guideline tests;
- No data summarized for mixture toxicity including tank mixtures and environmental mixtures to assessment endpoints;
- No toxicity data presented on "inert" or other ingredients present in formulations containing each of the a.i.s.

Summary of Toxicity Information from Open Literature

To organize the available toxicity information on listed salmonids and habitat, we developed risk hypotheses with associated assessment endpoints as described in the *Approach to the Assessment* section. Recall that assessment endpoints are biological attributes of salmonids and their habitat potentially susceptible to the stressors of the action. In addition to toxicity data presented in the BEs, EFED's science chapters, and REDs, we also considered information from other sources to evaluate both individual and population-level endpoints. The results of those studies are summarized below under corresponding assessment endpoints. We qualitatively assigned the most significance to study results that were: 1) derived from experiments using salmonids (preferably listed Pacific salmonids or hatchery surrogates); 2) measured an assessment endpoint of concern (*e.g.,* survival, growth, behavior, reproduction, abundance), as identified in a risk hypothesis; 3) resulted from exposure to stressors of the action or relevant chemical surrogates (*i.e.,* other AChE inhibitors); and 4) had no substantial flaws in the experimental design. When a study did not meet these criteria, we highlighted the issue(s) and discussed how the information was used or why the information could not be used.

Assessment endpoint: Swimming

Assessment measures: Burst swimming speed, distance swam, rate of turning, baseline speed, tortuosity of path, acceleration, swimming stamina, and spontaneous swimming activity

Swimming is a critical function for anadromous salmonids that is necessary to complete their life cycle. Impairment of swimming may affect feeding, migrating, predator avoidance, and spawning (Little and Finger 1990). 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 AChE-inhibiting insecticides such as OPs and carbamates, as described in previous biological opinions issued by NMFS (NMFS 2008c, NMFS 2009b). A review paper on methyl parathion and other OPs summarized a number of experimental swimming behavioral studies and concluded effects to swimming activity generally occur at lower concentrations than effects to swimming capacity and that effects to both occur at exposure concentrations below the LC50 (Little and Finger 1990). Swimming-mediated behaviors are frequently adversely affected at 0.3 – 5.0% of reported fish LC50s[8], and 75% of reported adverse effects to swimming occurred at concentrations lower than reported LC50s (Little and Finger 1990). Swimming capacity is a measure of orientation to flow as well as the physical capacity to swim against it (Dodson and Mayfield 1979, Howard 1975). Evaluations of swimming activity include 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. In a study on rainbow trout (*O. mykiss*), Little and others (1990) reported swimming activity, amount of prey consumed, and percent survival from predation were all affected by 96 h exposures to methyl parathion at 10 μg/L. The lowest rainbow trout LC50 reported in EPA documentation on methyl parathion (EPA 2004d, EPA 2006k, EPA 2008f) was 2,200 μg a.i./L (EPA 2008f, Appendix B). In bluegill sunfish (*Lepomis macrochirus*), methyl parathion adversely affected burst swimming behavior at 300 μg/L (Henrt and Atchison 1984). Respiratory disruptions, comfort movements, and aggression behaviors in bluegill were all adversely affected by 24 h exposures to methyl parathion at 3.5 μg/L. The

[8] The current hazard quotient-derived threshold for effects to threatened and endangered species used by EPA is 5 % (1/20th) of the lowest fish LC50 reported. If the exposure concentration is less than 5 % of the LC50 a no effect determination is made which likely underestimates risk to listed salmonids based on swimming behaviors.

lowest bluegill LC50 reported in the salmonid BE on methyl parathion (EPA 2004 d) was 4,380 μg a.i./L.

Iannacone and others (2007) reported a 96 h impaired swimming EC50 and reduced opercular movement in rainbow trout alevins at 16.1 mg a.i./L of methamidophos when they tested the formulation Tamaron. While they did not determine an LC50 for the rainbow trout in their study, they did report both swimming and immobilization 96 h EC50s for neon tetra (*Paracheirodon innesi*) used in the same series of tests. For the tetra, the swimming impairment EC50s were 9.0 mg a.i./L for methamidophos from the formulation Monofos, and <4.9 mg a.i./L for methamidophos from the formulation Tamaron. Corresponding immobilization EC50s (essentially an LC50) were 15.4 mg a.i./L for methamidophos from the formulation Monofos, and 8.2 mg a.i./L for methamidophos from the formulation Tamaron. The lowest rainbow trout LC50 reported in the EPA salmonid BE (EPA 2004b) is 25 mg a.i./L.

A study of methidathion effects on carp (*Cyprinus carpio*) reported behavioral effects including increased ventilation, agitated movements of fins, and tremorous movements of the jaws at concentrations of 2 and 6 mg a.i./L of methidathion (test material Ultracid-WP40) (Hugehes et al 1997). The authors do not report an LC50 for carp, but they appear to be considerably less sensitive to the effects of methidathion than do rainbow trout. The most sensitive methidathion LC50 for rainbow trout in the EPA salmonid BE (EPA 2004c) is 10 μg a.i./L.

A review of literature on the OPs chlorpyrifos, diazinon, and malathion showed significant and persistent effects to a suite of swimming related behaviors in salmonids (NMFS 2008c). Effects included reductions in swimming speed (Brewer et al 2001), distance swam (Brewer et al 2001), acceleration (Tierney et al 2007) and food strikes (Sandahl et al 2005). One study examined effects of malathion on juveniles from several salmonid species (rainbow trout, brook trout, and coho salmon) (Post and Leasure 1974). The study evaluated swimming performance, brain AChE activity, and recovery time. Test protocol also included a second exposure to malathion to determine if prior exposure altered susceptibility. Based on AChE inhibition, brook trout were most sensitive, followed by rainbow trout and coho salmon. Swimming performance was affected at the lowest concentrations tested in each salmonid species. Response occurred in a dose-dependent fashion. The data indicate AChE inhibition of approximately 20-30% resulted in a ≤5% reduction in swimming performance. As AChE inhibition increased, swimming performance decreased. Several other studies showed similar significant correlations between reduced AChE activity and swimming impairment (Brewer et al 2001, Sandahl et al 2005).

We located no open literature studies on swimming behavior for azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, naled, phorate or phosmet. However, studies evaluated by NMFS for other OP and carbamate insecticides show effects similar to those described in the Little and Finger (1990) review. The available data support the conclusion that AChE inhibitors at concentrations below LC50s can affect swimming behaviors. In summary, the body of evidence on AChE inhibition and its effect on swimming behaviors provide a weight of evidence that OPs can reduce the fitness of affected salmonids by adversely affecting these behaviors.

Assessment endpoints: Olfaction and olfactory-mediated behaviors: Predator avoidance, prey detection and subsequent growth, imprinting of juvenile fish to natal waters, homing of adults returning from the ocean, and spawning/reproduction

Assessment measures: Olfactory recordings (electro-olfactogram [EOG]), behavioral measurements such as detection of predator cues and alarm response, adult homing success, AChE activity in olfactory rosettes and bulbs, and avoidance/preference

The olfactory sensory system in salmonids is particularly sensitive to toxic effects of metals and organic contaminants. This is likely a result of the direct contact of olfactory neurons with dissolved contaminants in surface waters. Olfactory-mediated behaviors play a critical role in the successful completion of anadromous salmonid life cycles, and include detecting and avoiding predators, recognizing kin, imprinting and homing in natal waters, and reproducing. It is well established that Pacific salmon lose navigation skills when olfactory function is lost and consequently are unable to return to natal streams (Wisby and Hasler 1954). Salmonids that do not successfully return to their place of birth are functional losses to their natal population.

Impairment of olfaction (*i.e.*, smell) can be measured by an electrophysiological technique called the electro-olfactogram (EOG) (Scott and Scott-Johnson 2002, Baldwin and Scholz 2005, Sandahl et al 2006). The EOG measures the olfactory response of receptor neurons in a fish's nose. Reductions in the EOG amplitude of exposed fish compared to unexposed fish reflect functional losses in sensory capacity. A contaminant's toxic effect to olfactory sensory neurons is observable as a reduction in or elimination of the EOG amplitude to a recognizable odor (Baldwin and Scholz 2005).

We located no studies that measured olfactory responses of fish to the 12 a.i.s addressed in this Opinion. We therefore broadened our search to encompass other OPs. We found and summarized studies with OP insecticides, some of which we previously reviewed in a NMFS Biological Opinion (NMFS 2008c). We use this information as a surrogate for the OPs in the present consultation to determine whether salmonid olfactory-mediated behaviors are impaired by the 12 a.i.s. If the information with other OPs suggests olfaction is affected, we understand that the magnitude of effects likely differ depending on the OP. We do not discuss or use studies with carbamates as surrogates for olfactory effects in salmonids because AChE inhibition does not appear to be the putative mode of action affecting olfaction, although more empirical data are needed to confirm this (Jarrard et al 2004, Sandahl et al 2004).

Chlorpyrifos (OP insecticide)-

Juvenile coho salmon lost 25, 50 and 50% of olfactory function following 7 d exposures to 0.625, 1.25, and 2.50 μg/L(nominal), respectively (Sandahl et al 2004). The concentrations used in this experiment are environmentally relevant to predicted and measured concentrations in salmonid-bearing waters. Olfaction response was reduced by 25% at the lowest concentration tested (0.625 μg/L). AChE activity in coho salmon olfactory rosettes was inhibited by 25% at the highest exposure level tested (2.5 μg/L). No significant correlation between AChE inhibition and olfactory impairment was found. These results indicate that olfaction is impaired by the OP chlorpyrifos at exposures below 1 μg/L, but that the response is not necessarily mediated by reduction in AChE. This study measured olfactory response of a listed salmonid species, coho, exposed to an OP insecticde using a well-executed experimental design and therefore is ranked as highly relevant for use as surrogate information.

Diazinon (OP insecticide)-

We located two studies that investigated effects of diazinon on salmonid olfaction and olfactory –mediated behaviors; both of which were briefly discussed in multiple BEs (Moore and Waring 1996, Scholz et al 2000). The first study investigated two aspects of diazinon's effect on olfaction in Atlantic salmon parr (Moore and Waring 1996). First, male parr were exposed to diazinon concentrations (0, 0.1, 1.0, 2.0, 5.0, 10, and 20μg/L) for 30 minutes and EOG recordings were analyzed to determine parr's ability to detect female-released priming odorant $PGF_{2\alpha}$, a prostaglandin involved in spawning synchronization that also has a role as a primer on male plasma steroids and gonadotropin production. At 1.0μg/L, diazinon significantly reduced the capacity for parr to detect $PGF_{2\alpha}$ by 22% compared to controls. At 20μg/L, diazinon inhibited olfaction by 79%. Following the 30 minute exposures, olfaction remained affected for up to 4-5 hrs post exposure. Second, diazinon's effect following 120 d exposures on male parr's plasma reproductive steroid levels was assessed following exposure to ovulating female's urine. Female urine, detected by males via olfaction, is important for a variety of male salmon reproductive priming behaviors including attraction and detection of an ovulating female and eliciting orientation behavior. Four male hormones (17, 20β-dihydroxy-4-pregnen-3-one [17,20βP], testosterone, 11-ketotestosterone [11-KT], and gonadotropin II [GtH II]) and milt were measured following diazinon exposures. Diazinon concentrations of 0.3 - 45 μg/L effectively abolished the priming effect. There was no significant difference in the plasma level of 17, 20βP when compared to fish not exposed to female urine. Similarly, diazinon at 0.3 - 45 μg/L showed no elevation of GtH II in plasma, effectively abolishing the priming effects. Testosterone and 11-KT levels were not significantly affected by diazinon. Milt production in parr was significantly reduced (~ 28%) at all concentrations of diazinon, 0.3 - 45μg/L. Recovery time of olfactory capacity following the 120 h exposure was not measured. In summary, the impairment of Atlantic salmon's ability to detect and respond to reproductive scents may lead to missed spawning opportunities. We infer that OPs that act similarly to diazinon would likely impair ESA-listed salmonid's spawning behaviors and result in missed spawning opportunities. In these cases, reproduction would be adversely affected.

The second study addressed two olfactory-mediated behaviors: predator avoidance behavior as measured by the alarm response of juveniles, and homing ability of adults as measured by the number of returning adults (Scholz et al 2000). Both of these endpoints are ecologically relevant behaviors and were assessed in Chinook salmon after short-term exposures. Following 2 h exposures to nominal concentrations (0.1, 1, and 10 μg/L diazinon), juvenile Chinook salmon showed reduced alarm response at 1 and 10 μg/L ($p = 0.05$) as measured by changes in swimming and feeding behaviors before and after exposure to an alarm odor. Compared with unexposed juveniles, diazinon-treated Chinook salmon remained more active and fed more frequently when exposed to the predator alarm signal (skin extract from another Chinook). The lack of response to the alarm cue indicates that olfaction was impaired, leaving Chinook salmon oblivious to a predator's presence, thereby increasing the likelihood of being eaten. Swimming and feeding (food strikes / minute) in the absence of the alarm cue were not affected by diazinon exposures (0.1- 10 μg/L). Homing of adult Chinook salmon was significantly affected at 10 μg/L diazinon, where 6 of 40 fish returned compared with 16 of 40 fish in control treatment. At 0.1 and 1.0 μg/L, fewer fish returned (12 of 40) compared to controls (16 of 40) although the effect was not statistically significant. In summary, diazinon significantly impaired responses by juvenile Chinook salmon (*O. tshawytscha*) to alarm scents, thereby increasing their susceptibility to predation, and also decreasing adult Chinook homing which may reduce their ability to locate their natal streams.

Collectively, these two studies show that exposure to diazinon, an OP, in the low μg/L range impairs predator avoidance behavior in juvenile Chinook salmon, homing in adult Chinook salmon, and reproductive priming and milt production in adult Atlantic salmon. Both studies' results are highly relevant for use as surrogate information on the effects of OPs to salmonid olfaction. These data indicate that OP insecticides can and do negatively affect salmonids' sense of smell at environmentally relevant concentrations. However, we located no dose-response studies that tested the any of12 active ingredients' capacity to affect olfaction. This is a current data gap.

Effects of mixtures containing one or more of the 12 OP insecticides on salmonid olfaction

We located one study that tested two of the 12 a.i.s contained in a mixture with an additional 7 other pesticide active ingredients (Tierney et al 2008b). The study measured olfaction in juvenile steelhead (*O. mykiss*) exposed for 96 h to an environmentally relevant pesticide mixture. Three treatment concentrations of a mixture containing 10 pesticides were tested. Treatments of 0.1x (low), 1x (realistic), and 10x (high) of the 10 most prevalent pesticides (including 6 OPs) detected in the Nicomekl River were used. The Nicomekl River is a salmon producing river in British Columbia, Canada. Within the three treatments, measured concentrations of dimethoate were 0.137, 0.486, and 6.620 μg/L (the control had 0.0032 μg/L); methamidophos was added to one treatment at 0.0672 µg/L; measured concentrations for parathion were 0.0231, 0.196, and 3.540 µg/L; for chlorpyrifos were 0.0017, 0.0134, 0.114 ng/L; for diazinon 0.0157, 0.157, 1.820 µg/L; and for malathion 0, 0.0463, and 0.926 µg/L. Juvenile steelhead exposed to these mixtures showed no significant reductions in olfactory response to a single odor, L-serine, known to elicit olfactory responses presented against a background with no L-serine. However when steelhead were exposed to increase in odor intensity from 10^{-5} to 10^{-3}M L-serine, olfactory responses were significantly reduced by the realistic (1x) and high (10x) treatments {Tierney, 2008 #1303}. These results indicate that at environmentally realistic concentrations of a mixture that includes methamidophos, dimethoate, parathion, chlorpyrifos, diazinon, and malathion, juvenile steelhead's ability to detect changes in odorant concentrations is compromised. Without properly functioning olfaction, behaviors that rely on smell such as homing and migration may be impaired. We ranked this study as highly relevant because it was conducted with juvenile steelhead, measured an ecologically relevant endpoint, used environmentally relevant concentrations detected in salmonid watersheds, and followed a rigorous experimental design. The degree to which salmonids' olfaction is affected by individual OPs within the mixture is impossible to discern from the experimental design; however the data suggest that olfaction is impaired following exposures to OPs.

The available literature with other OPs and with mixtures shows that olfaction can be impaired by OPs, and we infer that some of the 12 a.i.s may impair olfaction. However, we found no studies that measured fish olfaction or olfactory-mediated behaviors following exposures to any of the 12 a.i.s, a recognized data gap.

Assessment endpoints: Fish growth

Assessment measures: Growth rate, size in weight or length

Assessment endpoints: Fish reproduction

Assessment measures: Number of offspring, hatchability, number of fish that attained sexual maturity, and number of spawns per spawning pair

A multi-year experiment with bluegill sunfish (*Lepomis macrochirus*) exposed in outdoor, littoral enclosures investigated the effects of azinphos methyl on behavior, reproduction, embryo hatchability, larval survival, and young-of-the-year growth rate and biomass (Tanner and Knuth 1995). Inhibition of AChE was not measured. The overall objective of the studies was to measure the direct and indirect effects of azinphos methyl on indigenous organisms and estimate the impact on the whole littoral community. The experiments are described in greater detail as an EPA report from the Environmental Research Laboratory in Duluth, MN (EPA 1992). The series of experiments evaluated a range of effects on bluegill including effects to bluegill prey items. Bluegill were exposed to 0, 1, or 4 µg/L azinphos methyl from a single pulse and evaluated for 63 days. Mortalities occurred in all treatments including controls which were likely due to handling stress (EPA 1992). No significant long term effects on bluegill reproduction, embryo hatchability, larval survival, growth or biomass were detected, however few spawning events occurred. Abundances of cladocerans and copepods were significantly reduced by day 7 and recovered by day 35. In a preliminary test to determine lethal concentrations, 6 and 8 µg/L azinphos methyl treatments killed all exposed adult bluegill in eight days (Tanner and Knuth 1995). Although no significant reproductive effects were detected, several observational effects were described including a 90% reduction in spawning events following exposure to 4 µg/L azinphos methyl and a 60% reduction in spawning events control treatments compared to pre-exposure spawning events. The reduction in spawning, however, was not statistically significant as the control treatments had very poor reproductive success. The authors noted several reasons for this disparity including that skewed sex ratios between male and females were observed in all treatments, insufficient duration for bluegills to acclimate to enclosures prior to test initiation (12 d compared to 26 d in previous experiments), poor survival of control fish potentially from stocking mortality (*i.e.*, handling stress), and a combination of stocking mortality and azinphos methyl-induced mortality in the 1.0 and 4.0 ug/L treatments. For these reasons, the reproductive results are called into question and we do not use these data. In the same set of experiments growth of larval dace was measured in response to the 0, 1, and 4 µg/L azinphos methyl exposures. No significant changes in growth were detected, although the aquatic invertebrate community was substantially reduced (further discussed within the effects to prey section).

Assessment endpoints: Toxic effects in salmonids from consuming contaminated prey

Assessment measures: Survival, swimming performance

A current uncertainty in our assessment is the degree to which secondary poisoning of salmonids may occur from feeding on contaminated dead and dying drifting invertebrates. Secondary poisoning is a frequent occurrence with OPs and carbamates in bird deaths (Mineau 1991), yet is much less studied in fish. We are particularly concerned about juvenile salmonids, which are both more likely to be exposed to pulse concentrations in freshwater systems, and are more vulnerable due to their smaller size. In our literature survey, we did not locate any studies evaluating this type of effect for the 12 a.i.s. Physico-chemical properties of OPs indicate these compounds would not typically be expected to biomagnify, but do accumulate in fish. Bioconcentration by fathead minnows of azinphos methyl was measured at three hours (BCF = 3003), at 24 hours (BCF=1027), and at eight days (BCF = 2254) (EPA 1992). Given the acute toxicity of these compounds to aquatic invertebrates, juvenile salmonids could receive a sufficient dose of OPs from feeding on drifting, contaminated invertebrates in the event of a large kill or catastrophic drift event.

Typically, dietary toxicity studies for fish are not available, although we did locate one study on the OP fenitrothion. In this laboratory feeding study with the OP fenitrothion, brook trout (*S. fontinalis*) were fed contaminated pellets (1 or 10 mg/g fenitrothion for four wks) (Wildish and Lister 1973). Growth was reduced in both treatments. AChE inhibition was measured at 2, 12, and 27 d following termination of contaminated diet treatments. Exposed trout had lower AChE activity than unexposed fish in both treatments, and by 27 d following termination of contaminated diet, AChE levels recovered slightly. Although the experiment showed that AChE inhibition from the diet is possible in trout, the relative potency of the 12 a.i.s and the concentrations of the pesticides in prey items remains an uncertainty.

Some field studies show juvenile salmonids feed on pesticide-affected insects, producing effects in the fish which may be attributable to ingestion of the insecticide. Juvenile brook trout gorged on drifting insects following applications of carbaryl, and as a result, AChE activity was reduced (15-34%) in the trout (Haines 1981). However, it is not possible to differentiate the contribution to AChE inhibition from the aqueous and dietary routes in this study because concentrations were not measured in the water, prey, or fish. In another study on the pyrethroid cypermethrin, feeding on dying and dead drifting invertebrates caused a range of physiological symptoms in brook trout: loss of self-righting ability and startle response; lethargy; hardening and haemolysis of muscular tissue similar to muscle tetany; and anemic appearance of blood and gills (Davies and Cook 1993). The possibility the adverse effects in the trout manifested from exposure to the water column instead of from feeding on contaminated prey in this study was ruled out by the authors as measured field concentrations of pesticides did not produce known toxic responses. It should be noted that the physico-chemical properties of pyrethroids are significantly different than the OPs, and they may be more likely to bioconcentrate in invertebrates.

It is difficult to determine how likely it is that fish would receive a sufficient dose of insecticide on contaminated prey to cause modifications in behavior and/or decrease survival, but we note the possibility does exist, especially for a compound such as azinphos methyl, which is very highly toxic to both aquatic invertebrates and fish.

Habitat assessment endpoints:

Prey survival, prey drift, nutritional quality of prey, abundance of prey, health of aquatic prey community, and recovery of aquatic communities following OP exposures

Assessment measures: 24, 48, and 96 h survival of prey items from laboratory bioassays reported as EC/LC50s; sublethal effects to prey items; field studies on community abundance; indices of biological integrity (IBI); community richness; and community diversity.

Aquatic invertebrates are generally more sensitive to the effects of OP insecticides than fish, and data presented by EPA in the BEs and other documentation reflect this. The 12 a.i.s considered in this Opinion vary in toxicity, with 48 h EC50s for *Daphnia magna*, a standard invertebrate test organism, ranging from 0.3 μg/L (azinphos methyl, naled) to 430 μg/L (dimethoate). All are considered highly toxic or very highly toxic to aquatic invertebrates based on EPA's toxicity criteria. In our survey of literature, we located studies on aquatic invertebrates for azinphos methyl, dimethoate, fenamiphos, methamidophos, and methyl parathion. Prey information from these studies is summarized in Table 114. We did not locate any studies on bensulide, disulfoton, ethoprop, methidathion, naled, phorate, or phosmet. Although we did not locate any study results on naled, there was a substantial body of work on its degradate, dichlorvos.

The majority of the studies reported standard EC50 survival data, although there was a wider range of species evaluated than in pesticide guideline tests. Often, authors presented EC50 data and test descriptions as background for other components of the study, such as AChE inhibition or selection of the appropriate test organism(s) for a specific environment. We located few microcosm or community studies for the pesticides evaluated in this Opinion, thus data from other pesticides with the same mode of action (OP and carbamate AChE inhibitors) are incorporated in this section. Environmental concentrations at which these effects might occur for the pesticides in this Opinion are recognized as an uncertainty.

A review of field studies published from 1982-2003 on insecticide contamination concluded that "about 15 of the 42 studies revealed a clear relationship between quantified, non-experimental exposure and observed effects *in situ*, on abundance [aquatic invertebrate], drift, community structure, or dynamics" (Schulz 2004). Strikingly, the review focused on insecticide contamination of surface waters due to usual agricultural practices. The top three insecticides most frequently detected at levels expected to result in toxicity included the OPs azinphos methyl and chlorpyrifos, and the organochlorine endosulfan. Another OP, methyl parathion showed a clear relationship of exposure and effect within several field *in situ* experiments. The AChE –inhibiting *N*-methyl carbamates (carbaryl, carbofuran, oxymyl, and fenobucarb) also showed clear or assumed relationships between exposure and effect (Schulz 2004). Collectively, the effects from AChE-inhibiting insecticides on aquatic invertebrates included reductions in abundance and AChE activity, direct mortality, increased drift, and reduced community diversity.

Drift, feeding behavior, swimming activity, and growth are sublethal endpoints for aquatic prey negatively affected by exposure to AChE inhibitors (Beyers et al 1995, Coutemacnch and Gibbs 1980, Davies and Cook 1993, Haines 1981, Hatakeyama et al 1990, Schulz 2004). Drift of aquatic invertebrates is an evolutionary response to aquatic stressors. Insecticides, particularly OPs and carbamates, can trigger catastrophic drift of salmonid prey items (Beyers et al 1995, Coutemacnch and Gibbs 1980, Davies and Cook 1993, Haines 1981, Hatakeyama et al 1990, Schulz 2004). Some invertebrates may drift actively to avoid pesticides and settle further downstream, providing temporary spikes in available food items for feeding salmonids in that location while depleting resources in the affected location. Catastrophic depletion of benthic populations can result in long-term prey reduction in locations that are otherwise good quality habitat, possibly affecting salmonid growth at critical time periods. We located no studies addressing this line of reasoning for OPs directly with Pacific salmonids. Davies and Cook

(1993) did show aquatic invertebrate community changes, mortality of invertebrates, drift of dying and dead invertebrates, and affected trout following spraying of a pyrethroid pesticide, cypermethrin, an invertebrate and fish neurotoxicant. Effect concentrations were estimated at 0.1-0.5 μg/L cypermethrin. It is difficult to compare these effect concentrations to the range of OP insecticides considered in this Opinion but it is illustrative of how insecticides can damage multiple endpoints of an aquatic community, including creating a reduction in prey abundance.

Several scientific peer-reviewed publications and EPA documents have reviewed aspects of the available information on multi-organism microcosm, mesocosm, and field test results for the AChE-inhibiting insecticides (Barron and Woodburn 1995, Leewangh 1994, Schulz 2004, Van Wijingaarden et al 2005). Van Wijngaarden *et. al*, (2005) conducted a literature review listing ecological threshold values (*e.g.*, $NOEC_{eco}$ and $LOEC_{eco}$) for the OPs azinphos methyl, chlorpyrifos, diazinon, fenitrothion, methyl parathion, phorate and the *N*-methyl carbamates carbaryl, carbofuran, and bendiocarb from model ecosystems or "adequate" field studies. A $NOEC_{eco}$ represented "the highest tested concentration at which no, or hardly any, effects on the structure and functioning of the studied model ecosystem were observed. The $LOEC_{eco}$ is the lowest tested concentration at which significant treatment-related effects occurred" The majority of studies were conducted in littoral lentic systems, *i.e.,* ponds, and other static systems, but one study with carbaryl was conducted in a running water (lotic) system (Courtemanch and Gibbs 1980). Population densities of salmonid prey items (*i.e.*, Ephemeroptera, Diptera, Amphipoda, Cladocera, Copepoda, Isopoda, Ostracada, Trichoptera) declined following single exposures to AChE-inhibiting insecticide concentrations (Van Wijingaarden etal 2005). Effects were more severe in studies with repeated or chronic exposures. Adverse effects to these groups occurred at or above "0.1 toxic unit"-where a toxic unit equals field concentrations normalized by dividing them by the 48 h EC50 of *Daphnia magna* for a given AChE inhibitor (Van Wijngaarden et al 2005). For OPs in this Opinion reported $LOEC_{eco}$ were 0.72 -1 μg/L azinphos methyl, 10 μg/L methyl parathion, and 23 μg/L phorate.

Two studies examining the effects of dimethoate on invertebrate communities were available. One evaluated a constant concentration of dimethoate (1 μg/L) on wild-collected stream invertebrates maintained in a continuous-flow artificial stream (Baakken and Aanes 1994) .This study considered sublethal effects, evaluating drift, mobility, and community structure in the treated stream compared to a control stream, and to the community composition at the start the the experiment. Tests run for five weeks were conducted twice, once with communities collected in the autumn, and another time with communities collected in the spring . Authors concluded dimethoate at the applied concentration of 1μg/L resulted in higher drift rate, a greater proportion of population drifting, and higher non-drifting movements in the treated streams compared to the control streams. They also noted structural changes, with some populations responding differently in the treated stream. The abundance of stoneflies and caddisflies was proportionally lower in the treated stream than the control in the fall test. Mayflies, ostracods, and copepods were proportionally lower in the treated stream than the control in the spring test. In both cases, the total abundance was similar, but the community in the treated stream had shifted towards taxa generally considered more pollution tolerant, such as chironimids and oligochaetes.

Hessan *et. al* (2000) compared laboratory toxicity endpoints for *Daphnia magna* and *Daphnia pulex* to toxicity endpoints in an enclosure study in a Norwegian lake. Authors evaluated some community dynamics, but their experimental design was focused on comparing effects on primary productivity and food availability for the herbicides and fungicide they also tested (glypohosate, chlorsulfuron, and propiconazole). In this paper, there is some confusion regarding actual concentrations, as in some cases they report concentrations in μg/L and in other cases in mg/L, with no adjustment of the numerical value. Thus, we do not rely on concentrations presented in this study, but do note that authors concluded there was good agreement between their laboratory-derived endpoint, and mortality in the enclosures. They also noted that not all aquatic invertebrate taxa were affected equally by dimethoate. There were differences in population reductions for two types of crustaceans (cladocerans and copepods). Rotifer populatons, which are not of phylum Arthropoda, showed no effect from dimethoate.

The available literature from field experiments indicates that populations of aquatic insects and crustaceans are likely the first aquatic organisms impacted by exposures to OPs and other AChE-inhibiting insecticides. Benthic community shifts from sensitive mayfly, stonefly and caddisfly taxa, the preferred prey of salmonids, to worms and midges occur in areas with degraded water quality including from contaminants such as pesticides {Courtemanch, 1980 #889}. Reduced salmonid prey availability correlated to OP use in salmonid bearing watersheds {Van Wijngaarden, 2005 #93}.

We located two separate studies that addressed impacts on fish growth due to prey reduction from a single pulsed exposure to azinphos methyl (larval dace, bluegill sunfish) and to chlorpyrifos (fathead minnows) (Brazner and Kline 1990, Tanner and Knuth 1995). Both studies were conducted at EPA's Environmental Research Laboratory in Duluth, MN using the same experimental design with outdoor mesocosms (*i.e.*, littoral pond enclosures). Single one-time pulses of the OPs were sprayed over the mesocosms. The azinphos methyl study showed that of the eleven macroinvertebrate taxa analyzed, significant reductions were observed in the majority of taxa, while others showed little if any reductions in biomass. For example, macroinvertebrates collected on Day 15 revealed significant reductions (> 50%) in the biomass of Tanypodinae, Chironminae, Talitridae, and Hydracarina populations at 4 and 20 µg/L (EPA 1992). Within the microinvertebrate community (which contains the primary prey items for larval dace), cladoceran abundances were the most reduced at sampling times in the 20 µg/L treatments. Cladocerans recovered to pre-exposure levels by the last sampling date. Copepods and rotifer populations showed no consistent reductions from azinphos methyl in any of the concentrations tested. As discussed earlier, larval dace showed no significant difference in growth rates between azinphos methyl treatments and control treatments which differed from the chlorpyrifos study that showed significant reductions in fathead minnow growth. Bluegill sunfish also showed no significant differences in growth over the course of the azinphos methyl experiment and showed high variation of fish sizes within and among treatments. The microinvertebrate community, which initially serves as the primary food source for both bluegill and dace, was much less sensitive to azinphos methyl than the larger macroinvertebrates in the mesocosms.

In the study with chlorpyrifos, macro- and micro-invertebrates were significantly affected and recovered less quickly than in the azinphos methyl experiments. Although this study was conducted on chlorpyrifos, an insecticide not considered in this Opinion, we deemed it highly relevant due to the ecological context it provided. The study indicated that native fathead minnows exposed to chlorpyrifos had reduced growth due to reductions in prey abundance in pond mesocosms (Brazner and Kline 1990). The experiment tested the hypothesis that, “addition of chlorpyrifos would reduce the abundance of invertebrates and cause diet changes that would result in reduced growth rates.” Single, one-time pulses of chlorpyrifos at 0, 0.5, 5.0, and 20 μg/L were applied to littoral enclosures (chemical analysis of water concentrations provided at 0, 12, 24, 96, 384, 768 h) and resulted in statistically significant reductions in fathead minnow growth at 31 days. A single pulse of chlorpyrifos was introduced into each enclosure at day 0. Chlorpyrifos dissipated relatively quickly, since it has an aqueous half-life of 5-8 hours (Knuth and Heinis 1992). Invertebrate abundance was determined in each replicate on days -3, 4, 16, and 32. Fathead minnows were sampled from enclosures on days -2, 7, 15, and 31 and fish were weighed, measured, and dissected to determine gut content (dietary items identified). By day 7, significant differences in mean numbers of rotifers, cladocerans, protozoans, chironomids, mean total number of prey being eaten per fish, and mean prey species richness were greater in fish from the control enclosures than in some of the treatments. By day 15, control minnows were significantly larger than fish from treated levels. We note the apparent disparity between the two OPs, where chlorpyrifos exposures resulted in growth reductions of fathead minnows and azinphos methyl did not affect growth of larval dace or bluegill sunfish. The authors suggested the disparity was due to the greater duration of prey effects observed for chlorpyrifos compared to azinphos methyl. They suggest that this may be a result of differences in physical/chemical properties of the pesticide as azinphos methyl was less persistent than chlorpyrifos in the water and sediment after day 10 (EPA 1992).

These experimental results support the conclusion that reductions in abundance of prey result from single, one-time exposures to azinphos methyl and chlorpyrifos. Reductions in prey by chlorpyrifos resulted in reduced growth of juvenile fish. It is reasonable to assume that reductions in prey from AChE-inhibiting insecticides may result in reduced juvenile salmonid growth and ultimately reduced survival and productivity. The precise levels of prey reduction necessary to cause subsequent reductions in salmonid growth remain a recognized data gap for listed salmonids. We did not locate any microcosm, mesocosm, or field experiments measuring responses of aquatic communities that contained salmonids and salmonid prey simultaneously; this is a recognized data gap.

Recent declines in aquatic species in the Sacramento-San Joaquin River Delta in California have been attributed, in part, to toxic pollutants, including pesticides(Werner et al 2000). Significant mortality or reproductive toxicity in *C. dubia* was detected in water samples collected at 24 sites in the Sacramento-San Joaquin River Delta in California. Ecologically important sloughs had the largest percentage of toxic samples (14 - 19%). Toxicity Identification Evaluations (TIE) conducted on these samples identified OPs (including chlorpyrifos, diazinon, and malathion) and carbamates (including carbofuran and carbaryl) as the primary toxicants in these samples responsible for the adverse effects. The type of effects resulting from use of the OPs addressed in this Opinion is expected to be similar, although the extent of effect will vary widely depending on specific chemical toxicity values and use patterns.

Recovery of salmonid prey communities following acute and chronic exposures to AChE-inhibiting compounds depends on the organisms' sensitivity, life stage, and length of life cycle, among other characteristics. Univoltine species will take longer than multivoltine species to recover (Liess and Schulz 1999). Recovery of high quality salmonid prey items such as caddisflies, stoneflies, and mayflies will be slow, given they have relatively long life spans (1-2 years) and reproduce at the end of their lives. Additionally, these species require clean, cool waters to both recover and maintain self-sustaining populations. In several salmonid-supporting systems, habitats which might otherwise contain a diverse prey community are continually exposed to anthropogenic disturbances, including pesticide contamination, thereby limiting the potential for recovery of sensitive taxa, and potentially causing community shifts to more pollution tolerant invertebrate species. For example, urban environments are seasonally affected by stormwater runoff that introduces toxic levels of contaminants and scours stream bottoms with high flows. Consequently, urban environments do not typically support diverse communities of aquatic invertebrates (Morely and Karr 2002, Paul and Meyer 2001).

Similarly, yet due to a different set of circumstances, watersheds with intensive agriculture often have impacted invertebrate communities (Cuffney et al 1997). Indices of biological integrity (IBI) and other invertebrate community metrics are useful measures of the health of an aquatic community, as cumulative impacts of aquatic stressors are integrated over time. IBIs are also valuable because they convert relative abundance data of a species assemblage into a single index of biological integrity and provide an integrator for transient effects such as peak pesticide concentrations that might go undetected by a chemical monitoring program (Allen 1995). Salmonid-inhabited watersheds have been assessed using IBIs and other metrics of aquatic community health.

A study on the condition of Yakima River Basin's aquatic benthic community found that invertebrate taxa richness was directly related to the intensity of agriculture (*i.e.*, at higher agriculture intensities taxa richness declined significantly both for invertebrates) (Cuffney et al 1997). Locations with high levels of impairment were associated with high levels of pesticides and other agricultural activities. Salmonid ESUs and DPSs occurring in the Yakima River Basin, as well as other watersheds where invertebrate community measurements indicate severely compromised aquatic invertebrate communities, will be more difficult to recover. Other locations likely subject to such conditions include the Willamette River Basin, Puget Sound Basin, and the Sacramento-San Joaquin River Basin.

Table 114. Study designs and results with freshwater aquatic invertebrates

Chemical	Taxa/species	Assessment endpoints and measures	Effects	Data source
Azinphos methyl	*Chironomid* sp. (midge fly larvae)	Survival (24 h)	Concentrations between 0.36-0/.87 µg/L caused 41-46%mortalities	Schulz and Peall 2001
Azinphos methyl	*Corophium volutator* (estuarine amphipod)	Sediment toxicity test (48 h)	LC50 73 ng/g; LC20 41 ng/g; Sediment had 1.4% TOC No significant preference/avoidance for contaminated sediment	Hellou 2009
Azinphos methyl	*Hyalella azteca* (amphipod); *Chironomus tentans* (midge fly larvae); *Lumbriculus vareigatus* (oligochaete)	Survival (96 h)	*H. azteca* LC50 0.29 µg/L (95% CI 0.2-0.42); *C. tetans* LC50 0.37 µg/L (95% CI 0.27-0.51); *L. varigatus* LC50 Not determined	Ankley and Collyard 1995
Azinphos methyl (48 h pulse)	Microcosm taxa (no species provided): amphipod, cladocerans, copepods.	Microcosm: Densities (1, 3, 7,11,15,22,32,42 days), population density	Microcosm data: NOEC 0.2 µg/L; LOEC 0.8 µg/L cladocerans, NOEC 0.2 µg/L; LOEC 0.8 µg/L amphipods, NOEC 0.8 µg/L copepods @ 2.0 µg/L >95% reductions in cladoceran populations @ 8.0 µg/L >95% reductions in amphipod populations @ 20 µg/L 400% increase in copepods	Stay and Jarvinen 1995
	Mesocosm taxa: cladocerans, copepods, insects	Mesocosm: population density (7, 21,35, 42 days)	Mesocosm data:0.2 µg/L NOEC to population densities of cladocerans, copepods, insects @1 µg/L 40% of cladocerans reduced by 50-75%, 33% of copepods reduced by 40-70%, 70% of insects reduced by 70-100%. Recovery began on day 22 @4 µg/L 75% of cladocerans reduced by 60-100%, 30% of copepods reduced by 30-100%, 70% of insects reduced by 70-100%. Recovery began day 14- day 22 for copepods and day 22 for others @ 20 µg/L 100% of cladocerans eliminated, 60% of copepods reduced by 30-90%, 100% of insects eliminated. Recovery began day 14 - 22 for copepods.	Stay and Javinent 1995, Knuth 1992, EPA 1992)
Bensulide	*None located*			
Dimethoate	*Aedes taeniorhynchus* (Salt marsh mosquito) *Artemia sp.* (Brine shrimp)	Survival (48 h)	*A. taeniorhynchus* LC50 0.031 mg/L (95% CI 0.023- 0.041) Slope = 2.36 SD+/- 0.23 *Artemia* LC50 15.73 mg/L (95% CI 8.09-34.08) slope= 1.14 SD+/- 0.13 mg/L.	Song and Brown 1998

Chemical	Taxa/species	Assessment endpoints and measures	Effects	Data source
Dimethoate	*Daphnia magna* (cladoceran)	Survival (48 h) Reproduction (21d post exposure observation period) (included pulsed exposures from 0.5 – 8 h and multiple exposures at intervals of 7 d)	EC10 0.8 mg/L (EC50 not given) Growth significantly affected at 30 mg/L Reproduction- single pulses of dimethoate (30 mg/l) reduced number of offspring at 21 d; A single 3 h exposure to dimethoate (30 mg/L) increased time to reach maternity, reduced fecundity, and reduced sized of daphnids	Anderson et al 2006b
Dimethoate	*Daphnia magna* (cladoceran).	Oxygen consumption (15 and 240 min exposure)	Dimethoate was tested at 1.5 mg/L. No effect on oxygen consumption noted.	Martins et al 2007
Dimethoate	*Arctodiaptomus salinus* (estuarine copepod)	Survival (48 h), Reproduction (96 h)	EC50 9.56 µg/L (95% CI 4.08-7.14); Hatching rate (%) Control = 98%, 2 mg/L = 87.5 %; 20 mg/L 33.5% Neonate mortality Control = 0%, 2 mg/L = 21%; 20 mg/L 100%	Parra et al 2005
Dimethoate	*Neomysis integer* (mysid shrimp (adults))	Survival (96 h)	LC50 540 µg/L (95% CI 403-680)	Roast et al 1999
Dimethoate	*Chironomus riparius* (midge fly larvae) *Kiefferulus calligaster* (midge fly larvae)	AChE inhibition, Growth (body length), GST activity Emergence delay (48 h)	Reduced AChE activity 7.3 µg/L. No reduction in body length was observed. Reduced GST activity 31.7 µg/L Delayed emergence 7.3 µg/L	Domingues et al 2007
Disulfoton	*None located*			
Ethoprop	Numerous species of zooplankton, benthic and aquatic invertebrates	Abundance, species richness, diversity, evenness, similarity, and composition	Species specific, and community level impacts at 5.62 µg/L and 10 ug/L. The NOEAC was determined to be 3.16 µg/L based on changes to species composition	Bruns et al 2008
Fenamiphos	*Daphnia carinata* (cladoceran)	Survival (48 h)	Lab test water 2.19 µg/L ± 0.57 Natural water (DOC 6.9 mg/L) 3.26 µg/L ± 0.57	Caceres et al 2007
Methamidophos	*Chironomus calligraphus* (bloodworm), *Tetrapygus niger* (black sea urchin)	Survival (48 h-*C. calligraphus*) Fertilization (15 min *T. niger*) Two formulations	*C. calligraphus* Monofos EC50 1.32 mg a.i./L, Tamaron EC50 4.50 mg a.i./L *T. niger* Monofos EC50 1,423 mg a.i./L, Tamaron EC50 608.3 mg a.i./L	Iannacone et al 2007
Methamidophos	*Litopenaeus vannamei* (estuarine white shrimp)	Survival (72 h)	LC50 2.34 mg a.i./L (95% CI 1.98-3.06) AChE EC50 1.64 mg a.i./L (95% CI 0.98-2.49)	Gercia-de la Parra et al 2006

Chemical	Taxa/species	Assessment endpoints and measures	Effects	Data source
Methamidophos	*Daphnia magna* (cladoceran))	Survival (48 h)	Racemic methamidophos LC50 34.0 mg a.i. /L ± 5.2 (+) enantiomer LC50 33.8 mg a.i. /L ± 3.6 (-) enantiomer LC50 237.9 mg a.i. /L ± 3.6	Lin 2006
Methidathion			*None located*	
Methyl parathion	*Daphnia magna* (cladoceran)	Survival (48 h)	EC50 2.6 mg/L (95% CI1.2-4.5)	Baun 2008
Methyl parathion	*Daphnia magna* (cladoceran) *Ceriodaphnia cornuta* (cladoceran)	Survival (48 h)	*D. magna* EC1 1.9 µg/L (95% CI 1.3-2.8), EC50 4.9 µg/L (95% CI 8.3-6.8) *C. cornuta* EC1 0.3 µg/L (95% CI 0.2-0.4), EC50 1.4 µg/L (95% CI 0.8-2.4)	Do Hong et al 2004
Methyl parathion	*Chironomus tetans* (midge fly larvae)	Survival (96h)	Based on water concentration in water, larvae in silica sand. 10EC EC50 66.5 mg/L (95% CI 58-76.3) 20EC EC50 64.9 mg/L (95% CI 52.4-80.3) 20EC EC50 25.6 mg/L (95% CI 22.5-29.1) (significantly different)	Lydy et al 1999
Methyl parathion	*Hyalella azteca* (amphipod)	Survival (96 h)	Methyl parathion alone: LC1 0.3 µg/L (95% CI 0.2-0.4) LC5 0.5 µg/L (95% CI 0.4-0.8) LC15 0.7 µg/L (95% CI 0.4-1.0), LC50 2.1 µg/L (95% CI 1.0-2.9); Concurrent exposure with atrazine (AT) lowers LC50: 0 mg/L AT LC50 2.1 mg/L, 10 mg/L AT LC50 2.0 mg/L (synergistic ratio (SR) 1.0), 40 mg/L AT LC50 1.8 mg/L (SR 1.0) 80 mg/L AT LC50 1.2 mg/L (SR 1.7) 200 mg/LAT LC50 0.7 mg/L (SR 2.9)	Anderson and Lydy 2002
Methyl parathion	*Hyalella azteca* (amphipod)	Survival (48 h)	Natural water from wetland, DOC not specified EC50 9.0±0.3 µg/L	Schulz et al 2003
Methyl parathion	*Metapenaeus monoceros* (penaeid shrimp)	Survival (96 h)	LC50 0.886 mg/L	Reddy and Rao 1991
Naled			*None located*	
Phorate			*None located*	
Phosmet			*None located*	

Studies with other AChE inhibiting pesticides on salmonid prey items:

Robust evidence shows that salmonid prey taxa and communities can be substantially reduced following exposures to the OP (chlorpyrifos, diazinon, and malathion) and carbamate (carbaryl, carbofuran, and methomyl) insecticides. NMFS reviewed these data and presented its findings in recent Biological Opinions (NMFS 2008c, NMFS 2009b). We use these findings to show the types of aquatic community responses following exposures to AChE inhibiting insecticides. The toxic potency of a pesticide is a function of concentration and duration of exposure, which in turn is a function of a pesticide's physical properties and interactions of the pesticide with environmental variables such as temperature, pH, sunlight, soil micro-organisms, *etc*. With this in mind, if chemicals addressed in this Opinion are at concentrations individually (or together in mixtures) anticipated to cause lethal or sublethal effects to salmonid prey communities, we infer a similar magnitude of population-level effects (reductions in abundance from death and catastrophic drift) and similar recovery periods to those observed in affected aquatic communities treated with other OP insecticides.

Reviews of field, mesocosm, and microcosm studies with the three OPs addressed in the Opinion on chlorpyrifos, diazinon, and malathion (NMFS 2008c) document reductions in aquatic invertebrate populations and lengthy recovery times for populations of some taxa. A recent study found significant changes to macroinvertebrate assemblages of artificial stream systems following a 6 h exposure to chlorpyrifos at 1.2 μg/L (Colville et al 2008). The addition of chlorpyrifos to the artificial streams resulted in a rapid (6 h) change in the macroinvertebrate assemblages of the streams, which persisted for at least 124 d after dosing. Chlorpyrifos dissipated from the system within 48 h (Pablo et al 2008), however the macroinvertebrate community did not recover rapidly. Several species similar to salmonid prey items were significantly affected. As the OPs addressed in this Opinion share the same mode of action, we expect similar types of effects, although with available data we are unable to predict the relative extent of effect.

Zooplankton and insect taxa are generally more sensitive than fish to OPs. A diazinon study on the salmonid prey taxa Trichoptera, Diptera, and Cladocera showed they were highly sensitive (Giddings et al 2000). Other field studies in salmonid habitat also show reductions in salmonid prey abundances when the habitat is subject to regular input of insecticides. For example, in listed steelhead habitat in the Salinas River, California, abundances of the salmonid prey items including mayfly taxa, daphnids, and an amphipod (*Hyalella azteca*) were significantly reduced downstream of an irrigation return drain compared to upstream (Anderson et al 2003a, Anderson et al 2003b, Anderson et al 2006b). Several OPs were detected above acute toxicity thresholds in surface waters and sediments. Combined toxicity of chlorpyrifos and diazinon using a toxic unit approach correlated strongly with mortality of daphnids. For *H. azteca*, acute toxicity was attributed to sediment pore-water concentrations of chlorpyrifos (Andersone et al 2003b). Other pesticides were likely present and responsible for some of the toxicity in the Salinas River. In a subsequent study on the Salinas River, a TIE demonstrated that chlorpyrifos and diazinon were responsible for the observed death of the daphnid *C. dubia* (Hunt et al 2003). These data support the line of evidence that field concentrations of OPs can and do adversely affect aquatic invertebrates in salmonid habitats. It is reasonable to assume that the same situation occurs with some, if not all, of the compounds addressed in this Opinion.

Toxicity of Degradates

Based on data available in EPA documentation and open literature, we have evaluated degradates which we feel to be of toxicological concern. We discussed the environmental profile of these degradates in the exposure section.

Dichlorvos

Dichlorvos is a degradate of naled, and is also a registered pesticide. EPA provided toxicity data for this chemical in the BE (EPA 2004e), and we located additional information in open literature. Behavioral changes, including abnormal swimming, loss of equilibrium, and rapid opercular movement were noted within one hour in mirror carp, *Cyprinus carpio*) and European catfish, *Siluris glanis* exposed to concentrations of dichlorvos at or above the LC50 (mirror carp LC50 9.4 mg/L, European catfish LC50 16.7 mg/L) (Ural and Koprucu 2006). The LC50 for Iberian toothcarp (*Aphanis iberus*) was 3.2 mg/L (Varo et al 2008). The BE included LC50s for bluegill sunfish (*Lepomis macrochirus*, LC50 0.8-180 mg/L), fathead minnow (*Pimephales promelas* 11.6 mg/L), and sheepshead minnow (*Cyprinodon variegates*, LC50 7.4-14.4 mg/L), indicating these species have similar sensitivities (EPA 2004e). Data in the BE showed salmonid species are more sensitive to acutely toxic concentrations of dichlorvos than other freshwater fish, with LC50s for lake trout 183-187 μg/L (n=2), for rainbow trout 100-750 μg/L (n=4), and for cutthroat trout 170-213 μg/L (n=2) (EPA 2004e). In addition to use on crops, dichlorvos was previously registered for use to treat sea lice infestations (*Cerotothoa gaudichaudii)* in farmed salmon, and one study on this use reported no mortality in Atlantic salmon (*Salmo salar*) exposed to 5 mg/L for one hour (Sievers et al 1995). From the author's presentation, it is unclear whether the exposure concentration is for the formulated product, Nuvan 1000, or if 5 mg/L represents the concentration of dichlorvos. A single study reported a NOEC of 0.96 mg/L and a LOEC of 1.84 mg/L for sheepshead minnow (growth and reproductive effects) following chronic exposures to dichlorvos; (EPA 2004e).

The BE included EC50s for five species of invertebrates, ranging from 0.07 μg/L (*Daphnia* sp.) to 400,000 μg/L (*Gammarus fasciatus*) (EPA 2004e). EC50 data we located in open literature exhibited a similar range, with a low end of 0.23 μg/L (*Daphnia magna*, *(*Sturm and Hansen 1998))and a high end of 886 μg/L (*Metapenaeus monoceros*, panaeid shrimp, (Reddy and Rao 19910. One study showed a range of sensitivity in various life stages of *Tigriopus brevicornis* (marine copepod), with EC50s of 0.92 μg/L for nauplius, 2.9 μg/L for copepod, and 4.6 μg/L for ovigerous females. Another study evaluated inhibition of AChE and other enzymes at sublethal doses (~34% of LC50) in the hepatopancreas and muscle tissue of panaeid shrimp and found a decrease in activity of these enzymes ranging from 45-69% (Reddy and Rao 1991).

Although substantial variability between species and testing regimes exists, dichlorvos appears to be slightly more toxic than naled to salmonids and freshwater aquatic invertebrates.

Oxons

Oxygen analogues of the OPs (oxons) are the metabolically activated form of the parent OPs. They are also formed via environmental processes and during chlorination of drinking water, which led EPA to do an analysis of oxon toxicity for the OP cumulative drinking water assessment (EPA 2006b). Based on toxicity data from mammals, EPA estimated that the oxons were 10 to 100 times more toxic than the parent compounds. Although the estimate for oxon toxicity was generated for human health risk assessments, it is likely applicable to vertebrates and to some invertebrates as similar biotransformation enzymes have been characterized. However, there may be differences in toxicity based on route of exposure via the water column as compared to dietary exposure.

The relatively few toxicity data on aquatic species available for oxons hampered our ability to make robust comparisons with parent OPs. EPA reported LC50 data for rainbow trout (*O. mykiss*) of 6.2-7.5 mg/L for dimethoate as compared to 9.1 mg/L for omethoate (EPA 2008d). EPA data for *D. magna* included EC50 of 3.32 mg/L for dimethoate and 0.022 mg/L for omethoate, and chronic NOAEC/LOAECs of 0.04/0.01 mg/L and 0.042/0.14 mg/L, respectively, for the chemical and oxon (EPA 2008d). Open literature identified in EPA's assessment of methyl parathion (EPA 2008f) reported a methyl paraoxon EC50 of 2.3 μg/L for *D. magna*, as compared to the EC50 of 0.14 μg/L for parent methyl parathion. It should be noted that the data for methyl parathion were derived from a standard 48 hr test, whereas the data for methyl paraoxon was from a 24 hr test. Generally, longer tests produce lower EC50s.

Based on available toxicity data, we can draw no firm conclusion as to the differences in toxicity between the parent chemicals and the oxons.

Sulfoxides and Sulfones

The BE for disulfoton (EPA 2003b) presented toxicity data for both the sulfoxide and sulfone degradates for bluegill and rainbow trout. LC50s for the parent disulfoton, the sulfoxide, and the sulfone, respectively, for these species were: 39 μg/L and 3,000 μg/L;

118-188 μg/L and 60-300 μg/L; and 112 μg/L and 9,200 μg/L. From these data, it appears that the trout are less sensitive to all disulfoton compounds compared to bluegill. This is unusual, as salmonids are typically more sensitive on an acute lethality basis compared to warm water fish. The toxicity pattern for bluegill is parent>sulfone≥sulfoxide and for rainbow trout is sulfoxide>parent≥sulfone. For *D. magna*, the EC50 for the parent was 13 μg/L, for the sulfoxide it was 64 μg/L, and for the sulfone it was 35.2 μg/L. Toxicity pattern in this case is parent>sulfone>sulfoxide. Data for fenamiphos sulfoxide and sulfone were available both in EPA documentation (EPA 2003d) and open literature (Caceres et al 2007, Cacers et al 2008). In the BE, technical fenamiphos (88%) had an LC50 of 9.5 μg/L for bluegill sunfish, the sulfoxide had an LC50 of 2,000-2,653 μg/L for bluegill, and the sulfone had an LC50 of 1,173 μg/L (EPA 2003d). Toxicity pattern in this case is parent>>sulfone≥sulfoxide. For *D. magna*, sulfone data were not available, but the EC50 for the parent technical (88.7%) was 1.9 μg/L and the EC50 for the sulfoxide was 7.5 μg/L (EPA 2003d). In this case the toxicity pattern was parent>sulfoxide. Caceres *et. al* (2007) tested *Daphnia carinata*, another species of cladoceran, and determined an EC50 of 2.2 μg/L for fenamiphos, 5.8 μg/L for the sulfoxide, and 2.7 μg/L for the sulfone. They also evaluated toxicity in natural waters with higher dissolved organic carbon (DOC) content than lab water and reported the following toxicity trend: fenamiphos>fenamiphos sulfone>fenamiphos sulfoxide. Phenolic degradates of fenamiphos sulfoxide and sulfone were also evaluated and were shown to be non-toxic to invertebrates at concentrations up to 500 mg/L (Caceres et al 2007). In a separate study, the effects of fenamiphos and its phenolic derivative, and its sulfoxide and sulfone degradates were evaluated (Caceres et al 2008). Growth inhibition (reported as EC50s) of a green alga species (*Pseudokirchneriella subcaptata*) ranged from 1.05 mg/L (fenamiphos phenol) to 38.5 mg/L (fenamiphos). Fenamiphos sulfoxide and fenamiphos sulfone were non-toxic to both species of algae at concentrations up to 100 mg/L. We located no other discussion of phenolic-derivative compounds for any of the OPs.

While data are variable, it appears that generally the parent is more toxic than the sulfoxide and sulfone degradates, but that in most cases the toxic concentrations are in the same order of magnitude for the same species.

4-nitrophenol

4-nitrophenol, a degradate of methyl parathion, is generally considered a polar narcotic (Di Tor et al 2000. EPA 2008f) but, based on a QSAR analysis (Schultz et al 1986) it may also have a toxic effect via uncoupling of oxidative phosphorylation. Some toxicity data were available for this compound, including an LC50 of 4.0 mg/L for rainbow trout (EPA 2008f). Shuurmann *et. al*, (1997) reported LC50s for bluegill, rainbow trout, and fathead minnow of 8.4 mg/L, 7.8 mg/L and 44.7 mg/L. Lange *et. al*, (1995) reported a zebrafish (*Brachydanio rerio*) LC50 of 13.9 mg/L. The Lange *et. al*, paper reported effects of a 48 h exposure to zebrafish embryos, as an embryo of EC50 41.7 mg/L (lethal effects), an embryo EC50 of 27.8 mg/L (sublethal effects) and an embryo NOEC of 12.5 mg/L.

Data were located for prey species in both the EPA evaluation (EPA 2008f) and open literature. EPA cites an EC50 of 5.0 mg/L for *D. magna*. Literature sources (no original studies) cited values of 3.5 mg/L (Schuurmann et al 1997) and 28.2 mg/L (Goi et al 2004).

Because very few data on 4-nitrophenol are from original source, we have some concerns regarding reliability of the data. However, what data do exisit show 4-nitrophenol LC50s/EC50s for both fish and invertebrates to be in the 1-50 mg/L range.

Dichloroacetic Acid

In addition to dichlorvos, naled also degrades to dichloroacetic acid (DCAA), a substance regulated under drinking water standards. No data on this compound were presented in any of the EPA documents. We located some information from the open literature. A study evaluating biomarkers for oxidative stress in zebrafish found that at a concentration of 4,126 mg/L all embryos exposed developed edema, and 75-85% exhibited craniofacial malformations, skeletal muscle deformities, and swimming irregularities (Williams et al 2006a). DCAA is a member of the haloacetic acids, a chemical class known to have herbicidal properties. We located two studies (Hanson and Soloman 2004a, Hanson and Soloman 2004b) evaluating effects on duckweed (*Lemna gibba*, a standard toxicity testing organism) and two species of watermilfoil (*Myriophyllum spicatum, M. sibiricum*). The NOEC and LOEC for *L. gibba* are 10 mg/L and 30 mg/L, respectively (Hanson 2004). Calculated EC_{10}s for the most sensitive endpoints for all three species range from 0.8 to 31 mg/L. Authors concluded that the rooted dicots (*Myrioplhyllum* species) were generally more sensitive than *L. gibba* to this compound.

Table 115. Degradate Toxicity Data from Open Literature

Degradate (Parent)	Taxa/species	Assessment endpoints and measures	Effects	Data source
Dichlorvos (Naled)	*Cyprinus carpio* (Mirror carp)	Swimming (behavioral changes)	Behavioral and other sublethal effects noted at concentration of 8 mg/L after 24 h of exposure, and at 1 h in ≥16 mg/L. Abnormal behavior included rapid gill movement, erratic swimming, swimming at the water surface, gulping air at the water surface, and staying motionless on the bottom.	Ural and Koprucu 2006
Dichlorvos (Naled)	*Siluris glanis* (European catfish)	Swimming (behavioral changes)	Behavioral changes were noted in fish exposed to ≥24 mg/L dichlorvos during the test approximately 30 minutes after exposure. Abnormal behavior included loss of equilibrium, hanging vertically in the water, erratic swimming, swimming at the water surface, rapid gill movement, air gulping from the water surface, and staying motionless on the bottom.	Ural and Koprucu 2006
Dichlorvos (Naled)	*Cyprinus carpio* (Mirror carp)	Survival (96 h)	LC50 9.41 mg/L (95% CI 7.45-11.49)	Ural and Calta 2005
Dichlorvos (Naled)	*Siluris glanis* (European catfish)	Survival (96 h)	LC50 16.67 mg/L	Ural and Koprucu 2006
Dichlorvos (Naled)	*Aphanis iberus* (Iberian toothcarp)	Survival (96 h)	LC50 3.17 mg/L (95% CI 1.34-3.97)	Varo et al 2008

Degradate (Parent)	Taxa/species	Assessment endpoints and measures	Effects	Data source
Dichlorvos (Naled)	*Salmo salar* (Atlantic salmon)	Survival (1 h)	No mortality was reported when fish were exposed to up 5 mg/L for an hour as a treatment for the ecto parasite *Cerotothoa gaudichaudii* (sea lice). Authors tested the product Nuvan 1000, and it is uncertain if the 5 mg/L measurement is reported in mg/L a.i. or mg/L product.	Sievers et al 1995
Dichlorvos (Naled)	*Palaemonetes pugio* (Marsh grass shrimp)	Salmonid prey Survival (96 h)	UV regime, dose-dependent curve EC50 62 µg/L (95 %CI 39-109 µg/L); Dark regime, non-dose-dependent curve EC50 57 µg/L (95 %CI 26-216 µg/L); AChE inhibition NOEC 12 µg/L, LOEC 50 µg/L	Bolton-Warberg et al 2007
Dichlorvos (Naled)	*Metapenaeus monoceros* Penaeid shrimp	Salmonid prey Survival AChE and enzyme inhibition (96 h)	LC50 0.886 mg/L. 96 h exposure to sublethal dose (0.3 mg/L) of dichlorvos reduced levels of oxidative metabolic enzymes in addition to AChE inhibition when compared to control. Hepatopancreas results AChE -57% Succinate dehydrogenase (SDH) -55% Isocitrate dehydrogenase (ICDH) -63% Pyruvate dehydrrogenase (PDH) -67% Lactate dehydrogenase (LDH) -56% Cytochrome-c-oxidase -69%. Muscle results: AChE -48% Succinate dehydrogenase (SDH) -45% Isocitrate dehydrogenase (ICDH) -48% Pyruvate dehydrrogenase (PDH) -46% Lactate dehydrogenase (LDH) -63% Cytochrome-c-oxidase -50%	Reddy and Rao 1991
Dichlorvos (Naled)	*Lymnaea acuminata* Freshwater snail	Salmonid prey Survival (48 h)	LC10 0.002 mg/L, LC50 0.014 mg/L (95 %CI 0.011-0.017 mg/L) LC90 0.083 mg/L	Tripathi and Agarwal 1998

Degradate (Parent)	Taxa/species	Assessment endpoints and measures	Effects	Data source
Dichlorvos (Naled)	*Daphnia magna* (Cladoceran) *Chironomus riparius* (Midge fly larvae)	Salmonid prey Survival (24 h)	*D. magna* EC50 (lethality, immobility) 0.233 µg/L (95 %CI 0.225-0.242), ChE IC50 0.17 µg/L ± 0.04 *C. riparius* 0.10<EC50 <20 µg/L ChE IC50 6.2 µg/L ± 3.1	Sturm and Hansen 1999
Dichlorvos (Naled)	*Tigriopus brevicornis* (Marine copepod)	Salmonid prey Survival (96 h)	Nauplius EC50 0.92 µg/L (95% CI 0.7-1.1 µg/L) Copepodid EC50 2.9 µg/L (95% CI 0.9-4.9 µg/L) Ovigerous female 4.6 µg/L (95% CI 2.6-6.6)	Forget et al 1998
Dichlorvos (Naled)	*Aphanis iberus* (Iberian toothcarp)	AChE inhibition (96 h)	AChE inhibition determined separately in heads and in muscle tissue. Multifactorial analysis showed significant differences for concentration, sex, and tissue variables, with concentration having the most effect, followed by tissue type and then sex. EC50 AChE inhibition Head, female 0.30 mg/L (0.175-0.421 mg/L), male 0.27 mg/L (95% CI 0.138-0.402 mg/L); Muscle tissue female 0.78 mg/L (95% CI0.646-0.918 mg/L), males 0.56 mg/L (95% CI 0.001-0.080).	Varo et al 2008
Fenamiphos sulfoxide and sulfone (Fenamiphos)	*Daphnia carinata* (Cladoceran)	Salmonid prey Survival (48 h)	Fenamiphos LC50 2.19 ± 0.57 µg/L; Fenamiphos sulfoxide LC50 5.82 ± 1.41 µg/L Fenamiphos sulfone LC50 2.69 ± 0.1 µg/L Fenamiphos sulfoxide phenol and fenamiphos sulfone phenol non-toxic to test species at a concentration of 500 µg/L. Authors also tested in natural waters (DOC 6.9 mg/L) and found decreased toxicity for all compounds (toxicity ratio of natural water to test water ranged from 1.3-1.5). Toxicity results consistently showed the following trend fenamiphos>fenamiphos sulfone>fenamiphos sulfoxide	Caceres et al 2007

Degradate (Parent)	Taxa/species	Assessment endpoints and measures	Effects	Data source
Fenamiphos phenol, Fenamiphos sulfoxide phenol, and Fenamiphos sulfone phenol (Fenamiphos)	*Pseudokirchneriella subcaptata* (FW green alga) *Chlorococcum* sp. (terrestrial green alga isolated from soil culture)	Habitat Primary production-growth inhibition (96 h)	P*. subcapitata* Fenamiphos EC50 38.49 mg/L, EC20 10.28 mg/L Fenamiphos phenol EC50 10.54 mg/L, EC20 2.16 mg/L Fenamiphos sulfoxide phenol EC50 30.33 mg/L EC20 12.47 mg/L Fenamiphos sulfone phenol EC50 16.25 mg/L, EC20 0.79 mg/L *Chlorococcum sp*. Fenamiphos EC50 73.26 mg/L, EC20 30.56 mg/L Fenamiphos phenol EC50 13.64 mg/L, EC20 1.87 mg/L Fenamiphos sulfoxide phenol EC50 30.06 mg/L EC20 10.17 mg/L Fenamiphos sulfone phenol EC50 27.04 mg/L, EC20 4.36 mg/L Fenamiphos sulfoxide and fenamiphos sulfone were non-toxic to both species of algae at concentrations up to 100 mg/L.	Caceres et al 2008
4-Nitrophenol (Methyl parathion)	*Lepomis macrochirus* (Bluegill) *Pimephales promelas* (Fathead minnow) *Oncorhynchus mykiss* (Rainbow trout)	Survival (96 h)	Bluegill LC50 8.38 mg/L Fathead minnow LC50 44.7 mg/L, Rainbow trout LC50 7.82 mg/L (Note: as reported by author from other sources)	Schuurman 1997
4-Nitrophenol (Methyl parathion)	*Brachydanio rerio* (Zebrafish)	Survival (96 h)	Zebrafish LC50 13.9 mg/L (Note: as reported by author from other sources)	Lange et al 1995
4-Nitrophenol (Methyl parathion)	No species	QSAR analysis of mode of action	Analysis based on log Kow and existing toxicity tests (fathead minnow and *Tetrahymmean pyriformis*, a ciliated protozoan) indicated that the toxic mode of action for 4-nitrophenol could be either polar narcosis or uncoupling of oxidative phosphorylation.	Schultz 1986
4-nitrophenol (Methyl parathion)	*Brachydanio rerio* (Zebrafish)	Reproduction (48 h embryo exposure)	Embryo EC50 (lethal effects) 41.7 mg/L, Embryo EC50 (sublethal effects) 27.8 mg/L Embryo NOEC 12.5 mg/L (Note: as reported by author from other sources)	Lange et al 1995
4-nitrophenol (Methyl parathion)	*Daphnia magna* (Cladoceran)	Salmonid prey Survival (48 h)	EC50 3.49 mg/L (Note: as reported by author from other sources)	Schuurmann 1997

Degradate (Parent)	Taxa/species	Assessment endpoints and measures	Effects	Data source
4-nitrophenol (Methyl parathion)	*Daphnia magna* (Cladoceran)	Salmonid prey Survival (48 h)	EC50 28.2 mg/L ±1.5 (Note: as reported by author from other sources)	Goi et al 2004
DCAA (Naled)	*Danio rerio* (Zebrafish)	Biomarkers for oxidative stress (114 h embryo exposure)	At treatment level of 4,126 mg/L all embryos developed edema, 75-85% exhibited craniofacial malformations, skeletal muscle deformities, and were lethargic and swimming at the bottom of the test vessel. Heart rates in the embryos were significantly different from the controls, as was production of the superoxide anion and nitrous oxide	Williams et al 2006a
DCAA (Naled)	*Lemna gibba* (Duckweed), *Myriophyllum spicatum* (Eurasian watermilfoil), *Myriophyllum sibiricum* (Common watermilfoil)	Habitat: Primary production (14 d)	*L. gibba* NOEC 10 mg/L (plant number, frond growth rate, plant growth rate), LOEC 30 mg/L. EC10s for L. gibba range from 4.5 mg/L-10.5 mg/L for the various endpoints. EC10s for *M. spicatum* range from 1.5-64.7 mg/L, with most of the endpoints in the 30 mg/L-60mg/L range. EC10s for *M. sibiricum* range from 0.8 -38.1 mg/L, and were bimodally distributed, with 4 values <10 mg/L and 3 values >30 mg/L	Hanson and Solomon 2004a
DCAA (Naled)	*Lemna gibba* (Duckweed), *Myriophyllum spicatum* (Eurasian watermilfoil), *Myriophyllum sibiricum* (Common watermilfoil)	Habitat: primary production (14 d)	EC10 values for most sensitive endpoints *M. sibiricum* (3.4, 5.8, 6.0,7.1 mg/L; dry mass, wet mass, plant length, node number) *M. spicatum* (3.0, 4.5, 5.8, 5.6; root length, dry mass, root number, wet mass) *L. gibba* (31.6, 38.6 mg/L; frond mass, frond number) Rooted dicots were generally more sensitive than *Lemna* to this compound.	Hanson and Solomon 2004a

Adjuvant toxicity

Assessment endpoints: Survival of fish and aquatic prey items, endocrine disruption in fish

Assessment measures: 24, 48, 96 h LC50s, and vitellogenin levels in fish plasma

Although no data were provided in the BEs 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, NP ethoxylates and octylphenol ethoxylates, degrade in aquatic environments to the more persistent, toxic, and bioaccumulative degradates, NP and octylphenol, respectively. 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.

We queried EPA's ECOTOX online database and retrieved 707 records of nonylphenol's (NP) acute toxicity to freshwater and saltwater species. The lowest reported LC50 for a salmonid was 130 μg/L for Atlantic salmon. Aquatic invertebrates, particularly crustaceans, were killed at low concentrations of NP, with the lowest reported LC50 = 1 μg/L for *H. azteca*. These data indicate that a wide array of aquatic species are killed by NP at 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. Additionally, salmonid prey species are sensitive to sublethal effects of NP at low μg/L concentrations. The amphipod, *Corophium volutator,* grew less and had disrupted sexual differentiation (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 and Fairchild 2003, Hutchinson et al 2006, Jardine et al 2005, Lerner et al 2007a, Lerner et al 2007b, Luo et al 2005, Madsen et al 2004, McCormick 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 w002, Ishibashi et al 2006, Schoenfuss et al 2008). Vitellogenin is an egg yolk protein produced by mature females in response to 17-β 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 in specific tissues. Additionally, processes involved in sea water adaptation of salmonid smolts are impaired by NP(Jardine et al 2005, Lerner et al 2007a, Lerner et al 2007b, Luo et al 2005, Madsen et al 2004, McCormick et al 2005). 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).

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 formulations containing the 12 a.i.s. 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.

Summary of Response Analysis:

We summarize the available toxicity information by assessment endpoint in Table 45. 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 the 12 a.i.s was prevalent 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, in part, due to the lack of formulation information provided in the BEs as well as the statutory mandate under FIFRA for toxicity data on the a.i.s to support registration. We did locate a substantial amount of data on one group of adjuvants/surfactants, the NP ethoxylates. However, we received and located minimal information for the majority of tank mixes and other ingredients within formulations.

Table 116. Summary of assessment endpoints and effect concentrations

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)
Azinphos methyl (AZM)			
Fish:			
-survival (LC50)	yes	0.36 – 4810	high
-growth	yes	0.98	moderate
-reproduction	yes	0.4 - 0.98	moderate
-swimming	-	-	-
-olfactory-mediated behaviors	-	-	-
-AChE inhibition (EC50)	yes	0.16	high
Habitat: -prey survival (LC50)	yes	0.16 - 56	high
Degradate toxicity	yes	20 - 33	high
Bensulide			
Fish:			
-survival (LC50)	yes	720 – 1780	high
-growth	yes	789	high
-reproduction	-	-	-
-swimming	-	-	-
-olfactory-mediated behaviors	-	-	-
-AChE inhibition (EC50)	no	5 - 500	high
Habitat: -prey survival (LC50)	yes	62.4 – 3330	high
-aquatic primary producers (EC50)	yes	1500 - 2800	high
Degradate toxicity	-	-	-
Dimethoate			
Fish:			
-survival (LC50)	yes	1000 – 180,000	high
-growth	yes	840	moderate
-reproduction	-	-	-
-swimming	-	-	-
-olfactory-mediated behaviors	-	-	-
-AChE inhibition (EC50)	yes	273	high
Habitat: -prey survival (LC50)	yes	43 - 15,000	high
Degradate toxicity	yes	22 - 9100	high
Disulfoton			
Fish:			
-survival (LC50)	yes	8.2 – 13,900	high
-growth	yes	420	moderate
-reproduction	yes	2.9 – 32.9	moderate
-swimming	-	-	-
-olfactory-mediated behaviors	-	-	-
-AChE inhibition (EC50)	yes	487	moderate
Habitat: -prey survival (LC50)	yes	5 - 100	high
Degradate toxicity	yes	35.2 – 60,300	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)
Ethoprop			
Fish:			
-survival (LC50)	yes	33 – 13,800	high
-growth	yes	11	moderate
-reproduction	yes	21 - 54	moderate
-swimming	-	-	-
-olfactory-mediated behaviors	-	-	-
-AChE inhibition (EC50)	yes	90.6	high
Habitat: -prey survival (LC50)	yes	93	high
Degradate toxicity	-	-	-
Fenamiphos			
Fish:			
-survival (LC50)	yes	4.5 - 563	high
-growth	yes	7.4	high
-reproduction	-	-	-
-swimming	-	-	-
-olfactory-mediated behaviors	-	-	-
-AChE inhibition (EC50)	no	0.3 – 100	high
Habitat: -prey survival (LC50)	yes	1.3 – 10,000	high
Degradate toxicity:	yes	7.5 - 2653	high
Methamidophos			
Fish:			
-survival (LC50)	yes	5630 – 31,000	high
-growth	-	-	-
-reproduction	-	-	-
-swimming	yes	4500 - 16100	high
-olfactory-mediated behaviors	-	-	-
-AChE inhibition (EC50)	no	15 – 1000	high
Habitat: -prey survival (LC50)	yes	0.042 – 1054	high
Degradate toxicity:	-	-	-
Methidathion			
Fish:			
-survival (LC50)	yes	2.2 – 111.9	high
-growth	yes	12	moderate
-reproduction	-	-	-
-swimming	-	-	-
-olfactory-mediated behaviors	-	-	-
-AChE inhibition (EC50)	yes	1.1	moderate
Habitat: -prey survival (LC50)	yes	3 – 7.2	high
Degradate toxicity	-	-	-
Methyl parathion			
Fish:			
-survival (LC50)	yes	59 – 12,000	high
-growth	yes	10 - 380	high
-reproduction	-	-	-
-swimming	yes	3.5 - 300	high
-olfactory-mediated behaviors	-	-	-
-AChE inhibition (EC50)	yes	28.8	high
Habitat: -prey survival (LC50)	yes	0.14 – 28	high
Degradate toxicity	yes	1.5 - 5000	high

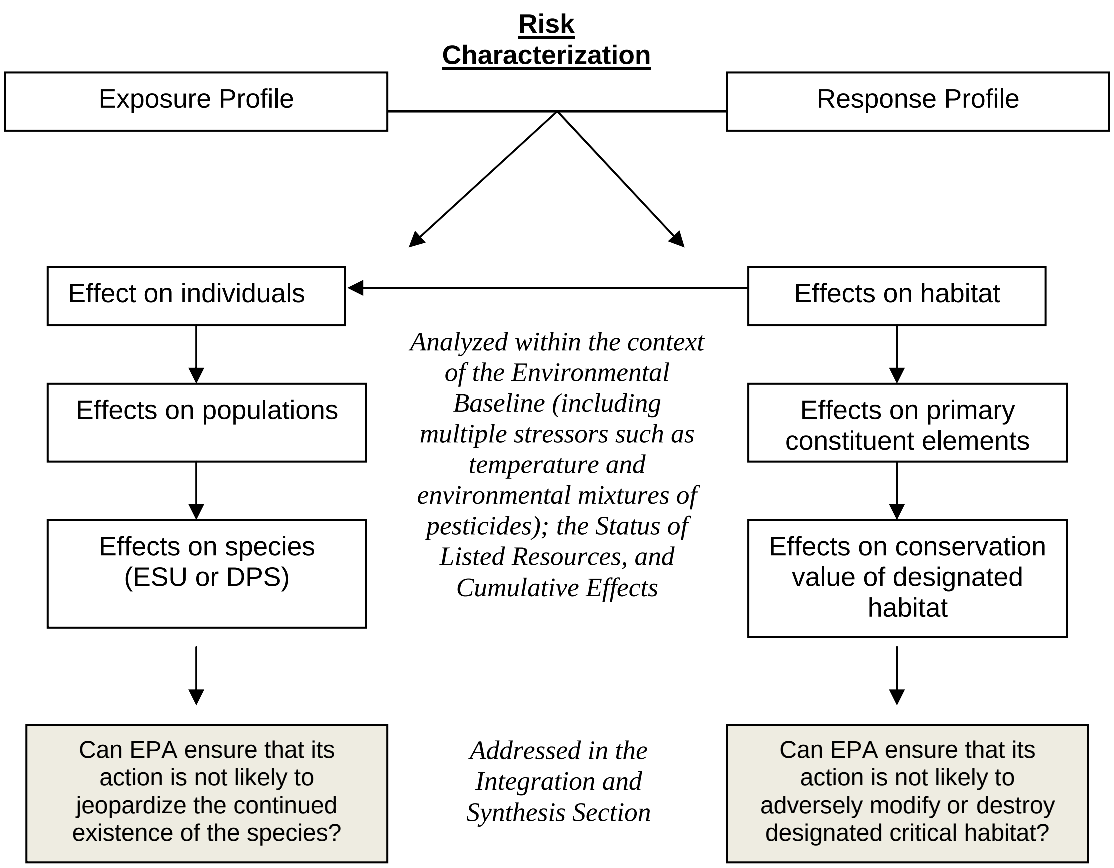
Assessment Endpoint	Evidence of adverse responses (yes/no)	Concentration range of observed effect or concentrations tested showing absence of effect (µg/L)	Degree of confidence in effects (low, moderate, high)
Naled			
Fish:			
-survival (LC50)	yes	92 - 3300	high
-growth	yes	15	high
-reproduction	-	-	-
-swimming	-	-	-
-olfactory-mediated behaviors	-	-	-
-AChE inhibition (EC50)	yes	7.8	high
Habitat: -prey survival (LC50)	yes	0.14 – 230	high
Degradate toxicity	yes	0.066 – 180,000	high
Phorate			
Fish:			
-survival (LC50)	yes	0.36 - 280	high
-growth	yes	4.2 - 190	high
-reproduction	-	-	-
-swimming	-	-	-
-olfactory-mediated behaviors	-	-	-
-AChE inhibition	yes	0.57	high
Habitat: -prey survival (LC50)	yes	0.3 – 65	high
Degradate toxicity	yes	0.4 - 22	high
Phosmet			
Fish:			
-survival (LC50)	yes	32 – 13,000	high
-growth	yes	6.1	high
-reproduction	yes	6.1	high
-swimming	-	-	-
-olfactory-mediated behaviors	-	-	-
-AChE inhibition (EC50)	yes	3.25	high
Habitat: -prey survival (LC50)	yes	1.6 – 3400	high
Degradate toxicity	-	-	-
Other ingredient:			
Nonylphenol (NP)			
Fish:			
-survival (LC50)	yes	130 - >1,000	high
-reproduction	yes	0.15 - 10	high
-smoltification	yes	5 - 100	moderate
-endocrine disruption	yes	5.0 – 100	high
Habitat: -prey survival (LC50)	yes	1- >1,000	high
Additive toxicity of OPs	yes	-	high
Synergistic toxicity of OPs	yes	-	high

Dash (-) indicates no information

Risk Characterization

In this section we integrate our exposure and response analyses to evaluate the likelihood of adverse effects to individuals and populations (Figure 45). 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 45 Schematic of the Risk Characterization Phase



Exposure and Response Integration

In Figure 46 through Figure 57, we show the overlap between exposure estimates for the 12 a.i.s and concentrations that affect assessment endpoints. This portion of the analysis is based on a.i., and does not take into account other stressors of the action that may contribute to toxicity, and/or that other AChE inhibitors may be present, creating additive or synergistic toxicity. The figures show the exposure concentration ranges (minimum – 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 off-channel habitats; and surface water monitoring data from ambient monitoring programs and 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. Some, although not all, of those BEs considered non-crop uses and those estimates have been included in our summary. The effect concentrations are values taken from the toxicity data reviewed in the *Response Analysis Section*. For the survival assessment endpoint, effect concentrations are LC50s, however, death of individuals occurring at concentrations below them are not represented by this metric. Consequently, when LC50 effect concentrations are not exceeded by the exposure estimates, it does not mean there are no incidences of mortality. For those instances where LC50s do not overlap or are not exceeded by an a.i.'s exposure estimates, we discuss the difference in magnitude of the two metrics and apply best professional judgment on whether death of individuals is expected. We also evaluate incident data for each a.i. as a separate line of evidence. We cannot accurately predict at what concentrations death first occurs because dose-response slope information was generally not provided for acute lethality studies. Although we are unable to determine at what concentration an individual salmonid dies, we do incorporate survival endpoints from acute 96 h studies using a default slope (probit slope of 4.5) in mixture analyses and in population modeling exercises discussed later. This slope is recommended by EPA when more relevant information is unavailable (EPA 2004g). Where overlap occurs between exposure concentrations and effect concentrations, NMFS discusses the likelihood of adverse effects. If data suggest exposure exceeds adverse effects thresholds, we discuss the likelihood and expected frequency of effects based on species information and results of the exposure and response analyses.

This is a coarse analysis because it 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 little ecologically relevant sublethal information, 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 the 12 a.i.s. Where significant uncertainty arises, NMFS highlights the information and discusses its influence on our inferences and conclusions. Table 117 compares exposure estimates with effect concentrations for each a.i.

Table 117 Comparison of Exposure Estimates and Effects Concentrations (µg/L)

CONCENTRATION	Azinphos methyl	Bensulide	Dimethoate	Disulfoton	Ethoprop	Fenamiphos	Methamidophos	Methidathion	Methyl parathion	Naled	Phorate	Phosmet
EPA peak PRZM/EXAMS estimates for farm pond	1.9 - 40.6	7.2 - 231	0.1 - 58.3	7.1 - 67	15 - 75	0.3 - 35.4	30 - 65	8.9 - 15.5	1.3 - 67	0.8 - 33	4.6 - 138	3.0-78
NMFS AgDrift estimates for floodplain habitat	0.8 - 11.4	1100 - 2940	46 - 652	16 - 237	6 - 24	No active labels	267 - 490	66 - 1860	134 - 980	16.8-132	NA - only granular	5.0-2,920
Monitoring data[1]	0.001 - 670	0.001-2840	0.001 - 11.6	0.001 - 48.7	0.001-241	0.001-520	0.001-0.13	0.001-15.1	0.001-213	na	0.001-32.3	0.001-0.63
ASSESSMENT ENDPOINTS	**Azinphos methyl**	**Bensulide**	**Dimethoate**	**Disulfoton**	**Ethoprop**	**Fenamiphos**	**Methamidophos**	**methidathion**	**Methyl parathion**	**Naled**	**Phorate**	**Phosmet**
Salmonid survival	1.2 - 27.5	720 - 1100	6200 - 7500	1850 - 13900	1020 - 13,800	68 - 563	25,000 - 51,000	6.6 - 14	1850 - 5300	87 - 345	13 -66	150-1,560
Olfactory-mediated behaviors	na	na	na	na	na	na	na	na	na	na	na	na
Fish reproduction (LOEC)	0.40	na	na	2.9 - 32.9	21 - 54	na	na	na	na	na	na	6.1
Fish growth (LOEC)	0.4 -0.98	na	840.00	420.00	11.00	7.40	na	12.00	10 - 380	15.00	4.2 - 190	6.1
Swimming	0.36 - 4810	na	na	na	na	na	4500 - 16,100	na	3.5 - 300	na	na	na
AChE inhibition (95% CI of EC50)[2]	0.10 - 0.26	na	195.7 -382	112.3 - 2118	69.5 - 118.2	na	na	0.47 - 2.68	21.2 - 39.0	6.5 - 9.5	0.42 - 0.76	2.5-4.2
Prey Survival	0.16 - 56	62.4 - 3330	43 - 15,000	5 - 100	44 - 93	1.3 -10,000	0.042 - 1054	3 - 7.2	0.14 - 28	0.14 - 230	0.3 - 65	1.6-3,400
Primary production	na	1500 - 2800	na	na	na	na	na	na	na	na	na	na

na not available

[1] Monitoring data includes concentrations from water quality monitoring, targeted monitoring, field studies, and incident data.

[2] From NOAA 2009

Measured and modeled concentrations of azinphos methyl exceed toxicity thresholds that kill salmonids and their prey, impact fish reproduction and growth, and inhibit AChE and swimming (Figure 46). For some of the endpoints, the entire range of modeled exposure concentrations is greater than the range of effect concentrations (fish reproduction, fish growth, and AChE inhibition). Both salmonid and prey survival endpoints are within the range of EECs predicted by EPA modeling, and overlap with the range of EECs in floodplain habitats estimated by NMFS. All three ranges of EEC estimates overlap with prey survival and salmonid AChE inhibition endpoints. The maximum concentration values far exceed the lower effect ranges. We note that minimal data exist for the growth and reproduction endpoints (n=2). We found no information regarding azinphos methyl's effect on salmonid olfaction. We also note that temperature is a major factor of azinphos methyl's toxicity to salmonids and their prey. Acute lethality bioassays with azinphos methyl and other OPs showed a distinct, robust relationship of increasing toxicity (measured by 96 h LC50) with increasing temperature (Mayer and Ellersieck 1986). Temperature-enhanced toxicity is discussed in more detail within the risk hypothesis section.

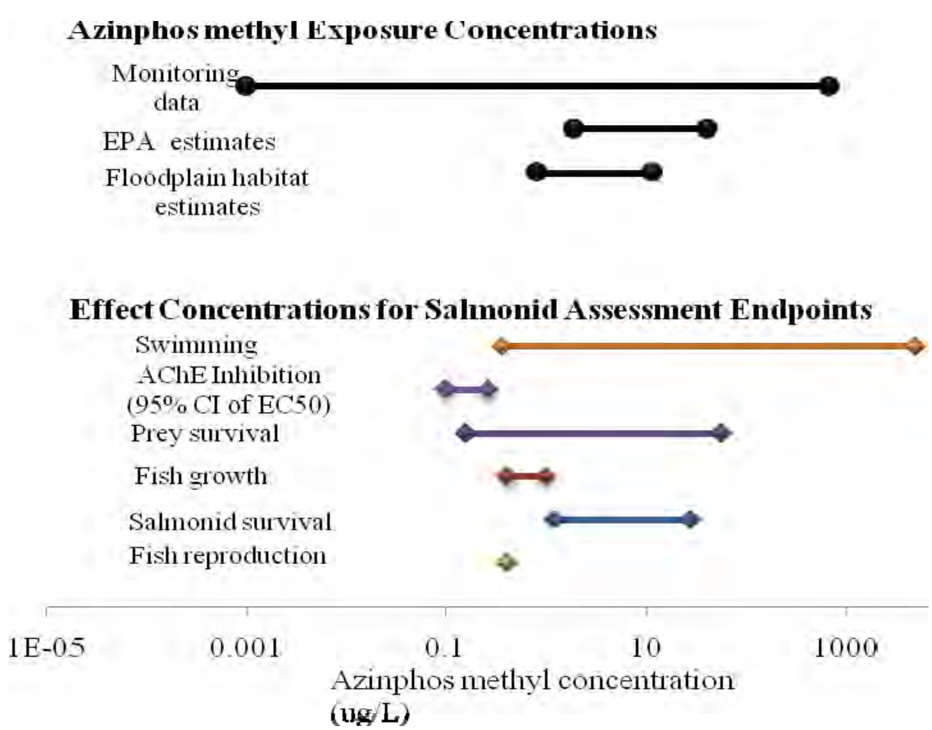


Figure 46 Comparison of exposure concentrations and salmonid assessment endpoints effect concentrations for azinphos methyl

Effects concentration data for bensulide were only available for the assessment endpoints of salmonid survival, prey survival, and primary production. Due to bensulide's herbicidal activity, we also evaluated primary productivity as an endpoint. We note that bensulide showed no consistent dose-response relationship on salmonid AChE when fish were exposed for 96 h to concentrations up to 500 μg/L. However, salmonids were lethargic, excitable, and disoriented at concentrations of 300, 400, and 500 μg/L. These behavioral effects occur within the range of exposure predicted with modeling and measured in surface water. Bensulide is currently registered for food crop uses, and also for uses such as ornamentals, residential lawns, golf courses, and turf grass. Generally, the non-food uses are at higher rates, permit more applications, and produce higher EECs in both EPA and NMFS modeling estimates. Some of the high estimates are corroborated by targeted monitoring studies (EPA 2006e). Salmonid survival and primary production effects concentrations overlap with NMFS EECs in the floodplain habitats and with the maximum values reported in monitoring data (Figure 47). Prey survival effects concentrations overlap with both EPA and NMFS' exposure estimates. We located no toxicity data for fish reproduction, growth, or behavior, resulting in substantial data gaps. We anticipate potential effects on salmonids, prey items, and primary productivity in floodplain habitats, and potential effects on prey items in larger water bodies.

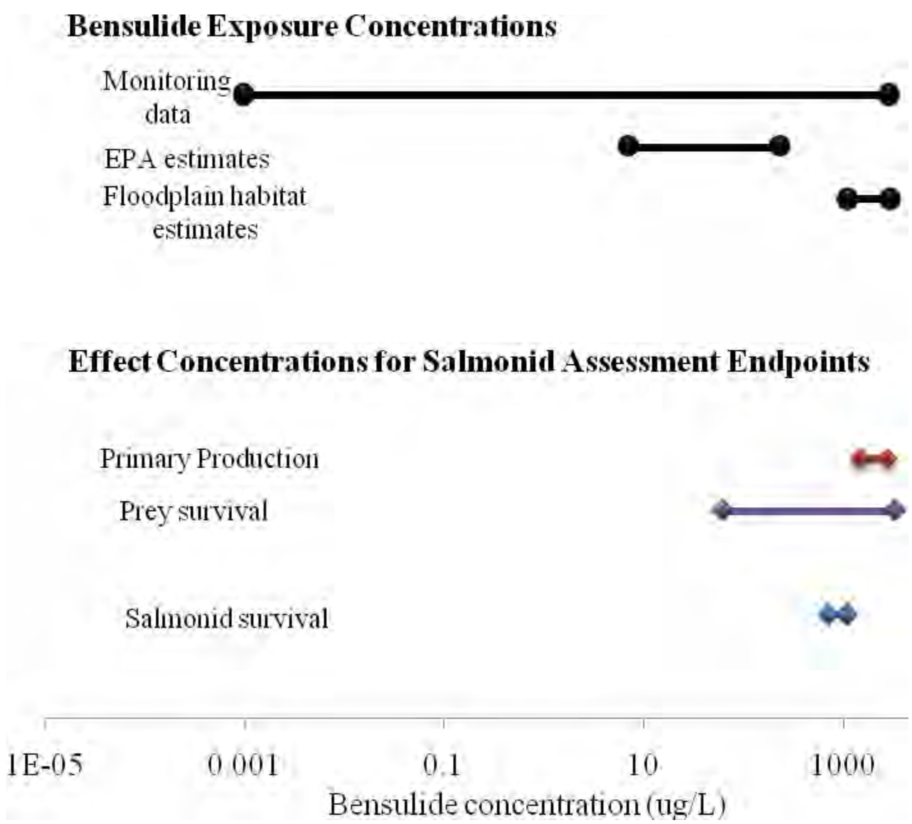


Figure 47 Comparison of exposure concentrations and salmonid assessment endpoints effect concentrations for bensulide

Modeled concentrations of dimethoate did not overlap with salmonid survival endpoints. The highest concentration predicted in floodplain habitats (652 μg/L) was an order of magnitude less than the lowest salmonid LC50 (6,200 μg/L) **(Figure 48).** Modeled concentrations did overlap with prey survival endpoints and AChE inhibition endpoints and were close to effect concentrations available for fish growth. It appears unlikely that concentrations of dimethoate in aquatic habitats would kill salmonids outright, however incidences of fish mortalities from dimethoate-containing formulations have been documented (see *Incidents* section). Concentrations of dimethoate are likely to be sufficient to reduce prey survival, and may affect salmonid growth and behavior. Given the strong correlation between reduced AChE activity and swimming performance, we anticipate impaired swimming ability of individuals exposed to dimethoate. We also expect that at elevated temperatures, dimethoate will be more toxic.

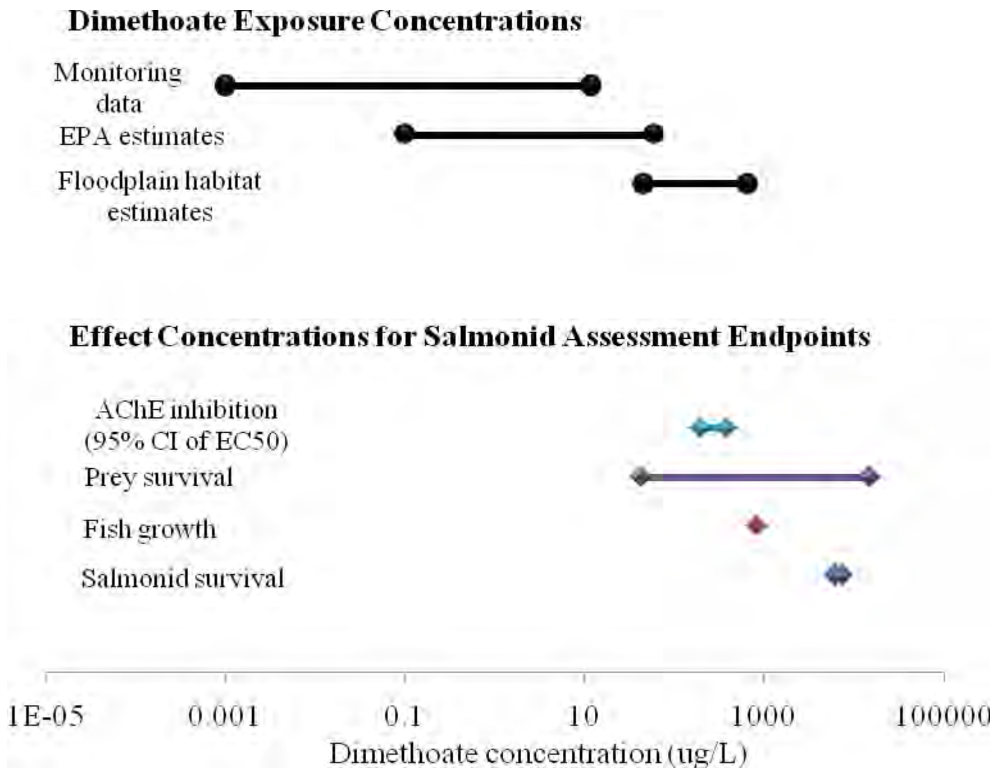


Figure 48 Comparison of exposure concentrations and salmonid assessment endpoints effect concentrations for dimethoate

Disulfoton EECs and monitoring data overlap with and are greater than the range of effects concentrations for fish reproduction endpoints, and also overlap with the EC50s for prey survival (Figure 49). In some cases, EECs account for the degradates disulfoton sulfoxide and sulfone as well as the parent. EPA estimates are approximately half the lower bound of the 95% confidence interval for AChE inhibition, and floodplain habitat estimates are well within it. The maximum floodplain estimate is approximately half the single estimate available for fish growth. All estimates are lower than salmonid LC50 concentrations, with nearly an order of magnitude between the highest EEC and the lowest salmonid LC50. Concentrations of disulfoton in the environment may be sufficient to cause effects on prey survival and fish reproduction, growth, and behavior. However, death of salmonids as a result of exposure to disulfoton is less likely, although in surface waters with elevated temperatures and other AChE inhibitors, death of sensitive individuals may be possible.

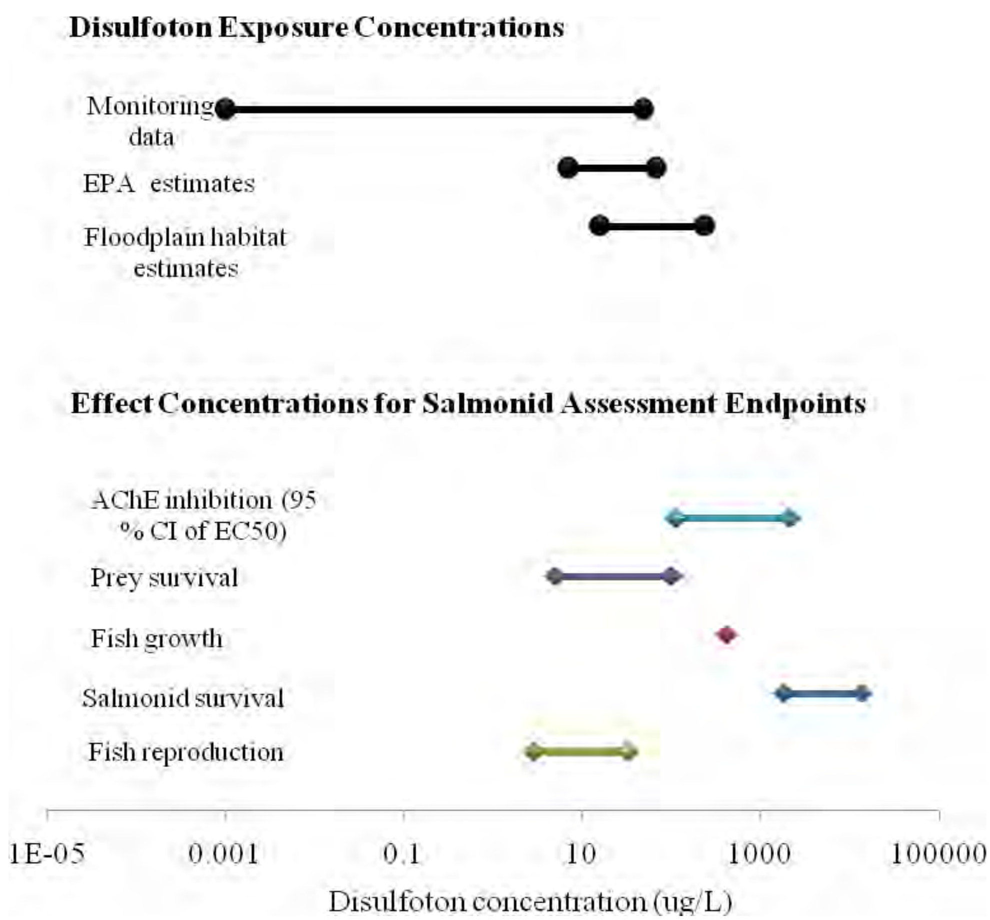


Figure 49 Comparison of exposure concentrations and salmonid assessment endpoints effect concentrations for disulfoton

Modeled ethoprop EECs from EPA and measured concentrations of ethoprop in surface water overlap with effect concentration ranges for all assessment endpoints where data are available, except for salmonid survival (Figure 50). Monitoring data from the databases examined show a maximum concentration of 5.75 µg/L, but there are two reported incidents in which fish deaths occurred at measured concentrations of 3-241 µg/L. Overlap occurs for the endpoints of fish reproduction, fish growth, prey survival, and AChE inhibition. Ethoprop is more persistent in aquatic systems than most OPs, with both hydrolysis and photolysis half-lives entered as “stable” in EPA modeling exercises. Estimated concentrations in floodplain habitats overlapped salmonid growth and reproduction endpoints. Floodplain estimates were less than EPA modeling estimates because they incorporated a 140 ft setback to “inland freshwater habitats” required for spray-applied formulations. However, there are no setback requirements for the granular formulations of ethoprop and the NMFS floodplain estimates do not account for surface water contamination via runoff and leaching. We anticipate adverse effects are likely to occur on fish growth, reproduction, and behavior in habitats exposed to ethoprop. The lack of information on ethoprop’s effect on olfaction and swimming is a noted data gap. We also anticipate adverse effects on prey survival. Death of sensitive individuals from ethoprop remains plausible although salmoid LC50s and EECs do not overlap.

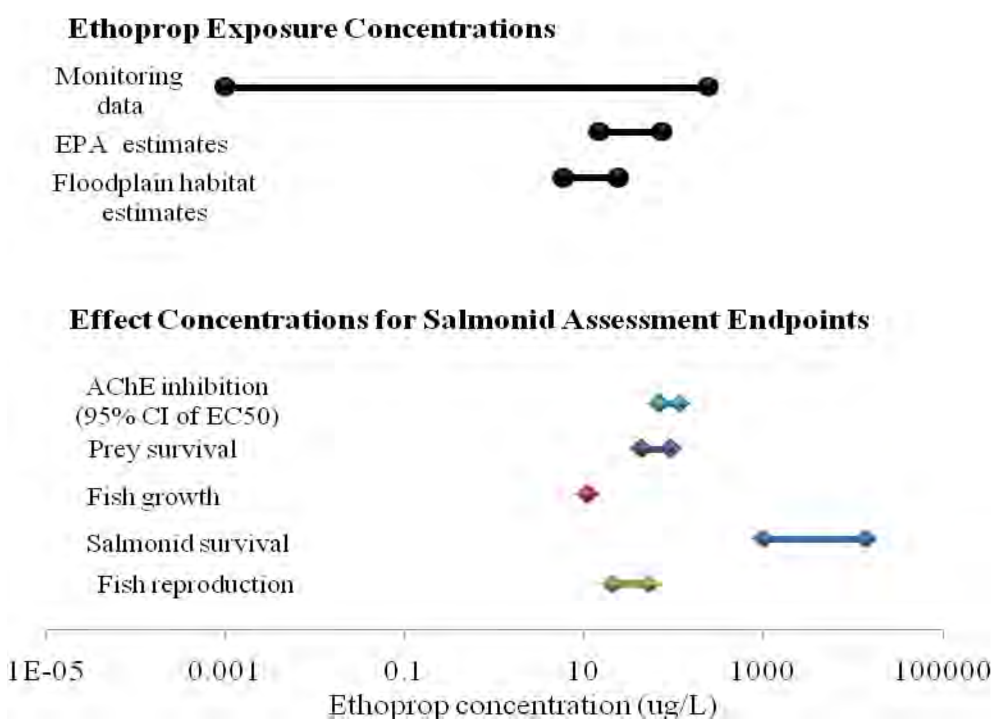


Figure 50 Comparison of exposure concentrations and salmonid assessment endpoints effect concentrations for ethoprop

Effect concentration data on a number of assessment endpoints were not available for fenamiphos (Figure 51). Data were only located for salmonid and prey survival and for fish growth. Thus, we are unable to draw any firm conclusions regarding the potential effects of fenamiphos on behavioral endpoints or fish reproduction. An EC50 for AChE inhibition was not determined at the highest concentration tested, 100 µg/L. No floodplain habitat concentrations were estimated, as there are no active labels for fenamiphos, but EPA estimates and concentrations measured in surface water overlap with effects concentrations for prey survival and fish growth. The maximum EPA estimate of exposure is approximately one-half the lowest salmonid LC50 and the maximum concentration of ethoprop detected in surface water (520 µg/L) overlaps a significant portion of the range of salmonid LC50s (68-563µg/L). We anticipate in situations where existing stocks of fenamiphos are used near salmon-bearing waters, concentrations in those waters may cause adverse effects on prey survival and fish growth, and may cause acute lethality. We cannot discount the possibility of adverse behavioral or reproductive effects also occurring.

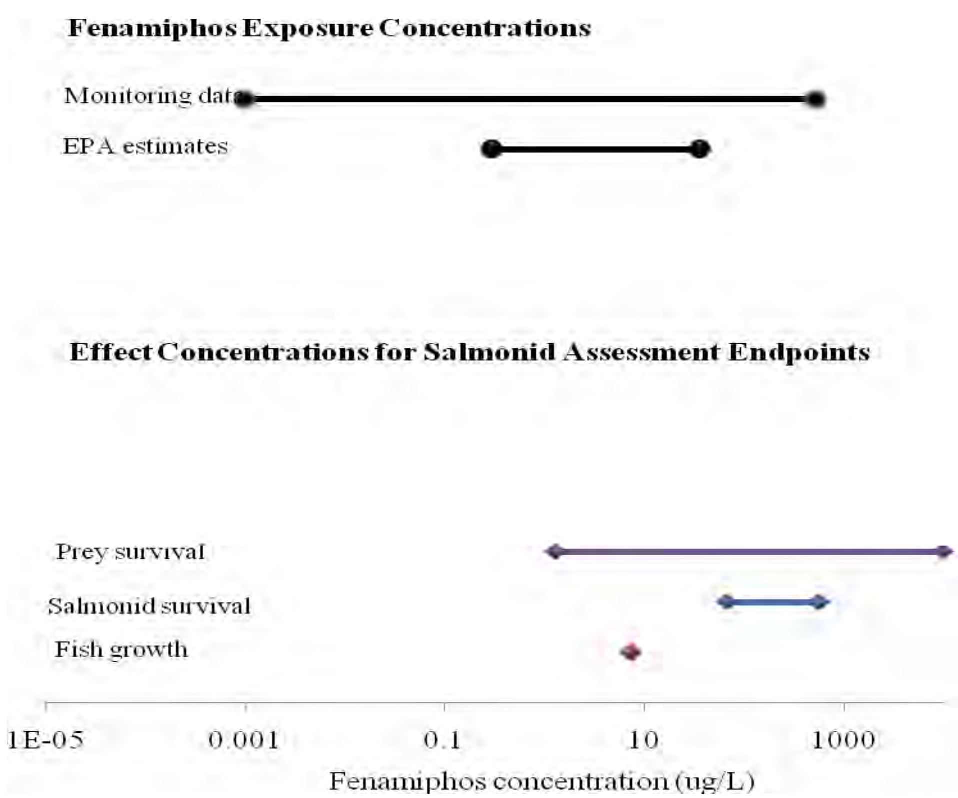


Figure 51 Comparison of exposure concentrations and salmonid assessment endpoints effect concentrations for fenamiphos

For methamidophos, effects concentration data were only available for the endpoints of salmonid survival, prey survival, and swimming behaviors (Figure 52). An EC50 for AChE inhibition was not determined at the highest concentration tested, 500 µg/L. We cannot draw any firm conclusions on the likelihood of adverse effects on fish olfactory mediated behaviors, reproduction, or growth. The estimated modeled concentrations from both EPA and NMFS are substantially higher than the lowest EC50 for prey survival and within the range of monitoring data, thus we do anticipate adverse effects on prey from use of methamidophos.

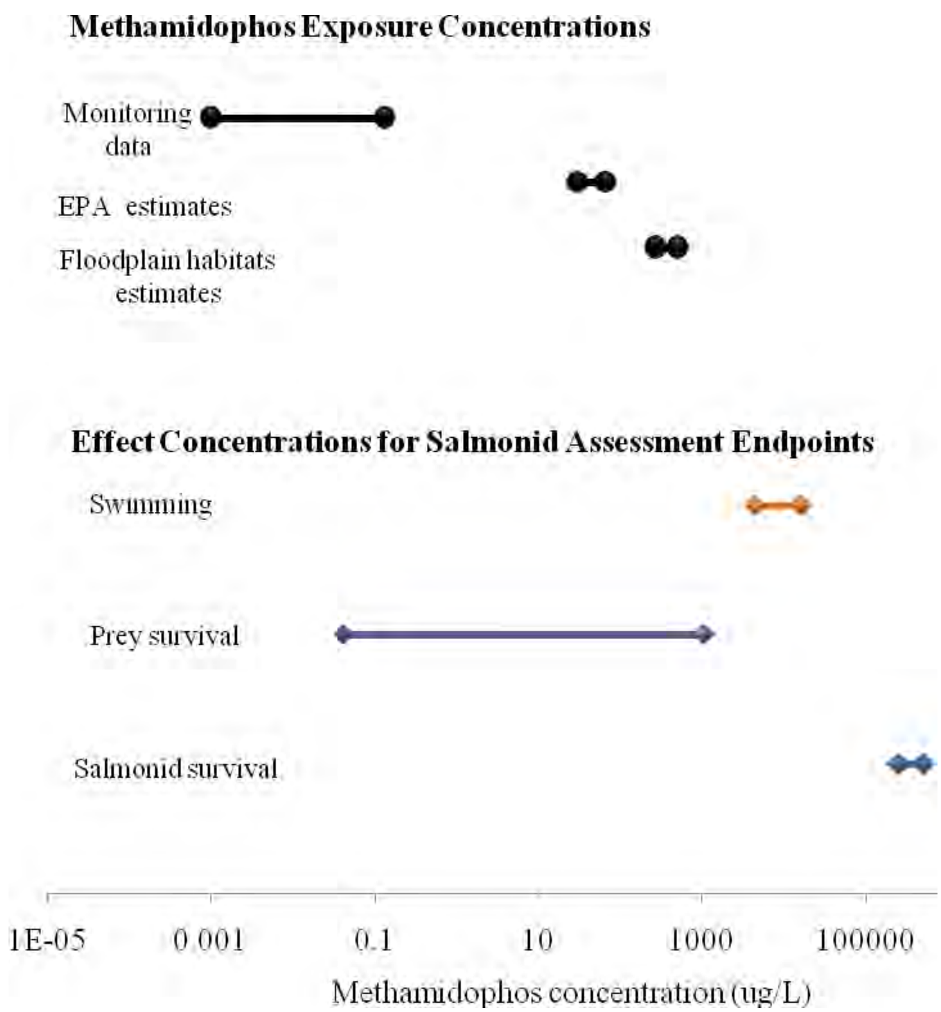


Figure 52. Comparison of exposure concentrations and salmonid assessment endpoints effect concentrations for methamidophos

All exposure estimates for methidathion are in the range of and/or higher than effects concentrations for endpoints where data were available (Figure 53). Additionally, concentrations of methidathion measured in surface water overlapped with the ranges of effect concentrations for all available endpoints (salmonid survival, fish growth, AChE inhibition, and prey survival). Based on the AChE inhibition data, we anticipate impaired swimming of exposed salmonids and potential growth effects due to impaired foraging ability. Thus for methidathion, we anticipate some salmonids exposed to methidathion for sufficient durations will die and show impaired swimming due to AChE inhibition. We also anticipate death of salmonid prey.

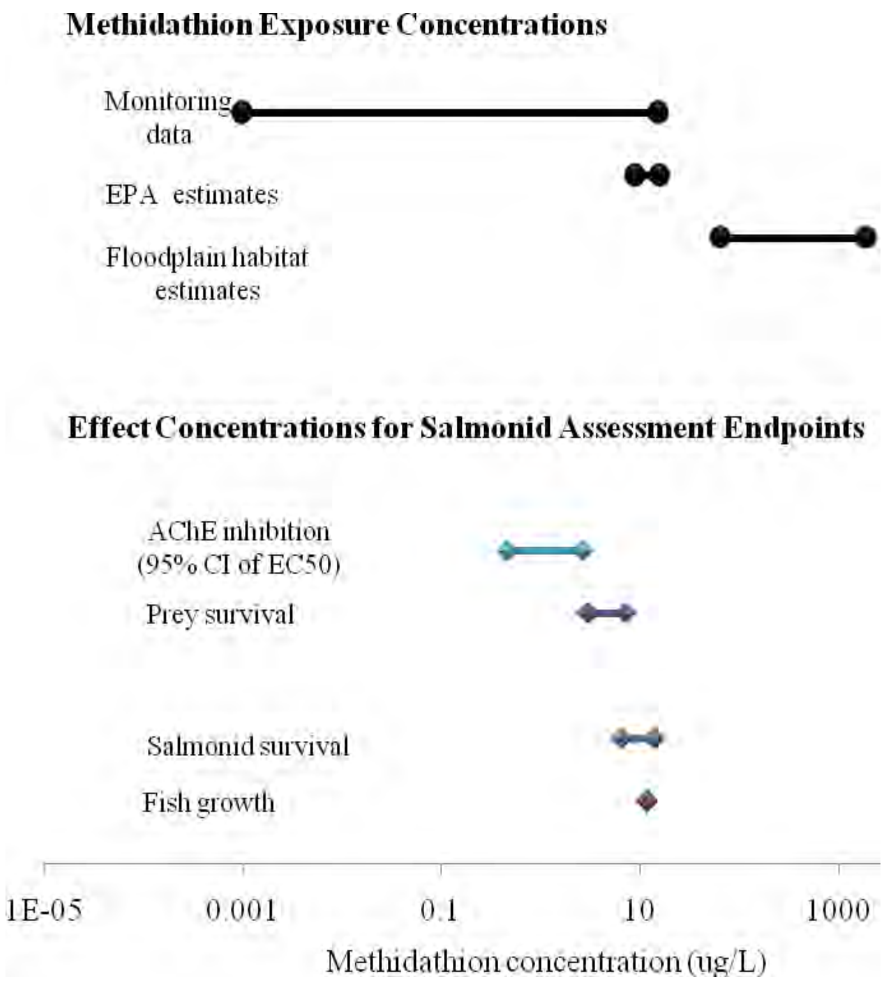


Figure 53. Comparison of exposure concentrations and salmonid assessment endpoints effect concentrations for methidathion

Estimated environmental concentrations for methyl parathion overlap with effects concentrations for all endpoints for which data are available except salmonid survival (Figure 54). The maximum EEC for methyl parathion in floodplain habitats is approximately one-half the lowest salmonid LC50. No data were available regarding effects on olfactory-mediated behaviors or fish reproduction. The lowest prey survival

EC50s occur at concentrations approxiamtely an order of magnitude below the lowest EECs. We expect methyl parathion will cause adverse effects on salmonids due to reduction in prey survival and sublethal effects on growth, reproduction, and behavior. We also anticipate death of sensitive salmonids exposed while rearing in small streams and floodplain habitats.

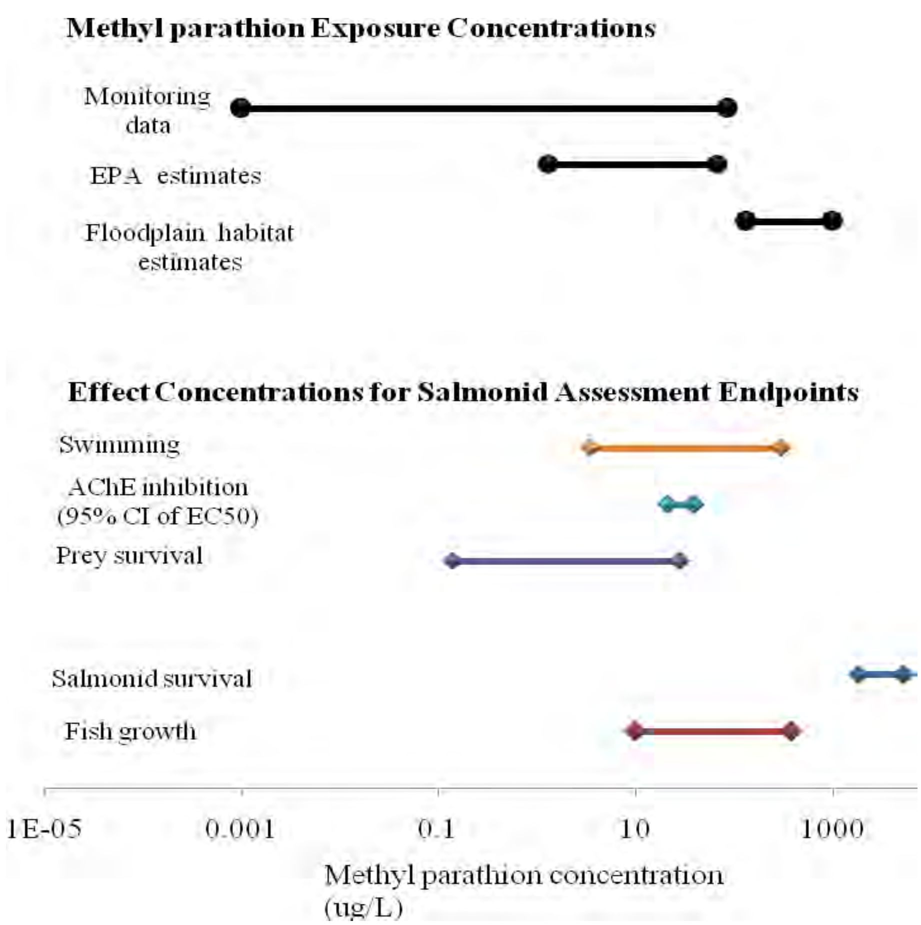


Figure 54. Comparison of exposure concentrations and salmonid assessment endpoints effect concentrations for methyl parathion

EPA EECs range from 0. 8 – 25 μg/L for crop uses of naled. The estimates for crop uses exceed effect thresholds for fish growth, AChE inhibition, and prey survival. NMFS AgDisp estimates for non-crop aerial applications to control flys and mosquitoes ranged from <0.01 – 94 μg/L depending on use rate, release height, and depth of the aquatic habitat. NMFS estimates for floodplain habitats associated with crop (6.8-132μg/L) and noncrop uses overlap with effects concentrations for all available effect endpoints, including salmonid survival (Figure 55).

Based on the modeled EECs for naled applications to crops and for other uses, we anticipate naled is likely to cause adverse effects on prey survival, and could also cause AChE inhibition, reproductive effects, and reduced survival in salmonids.

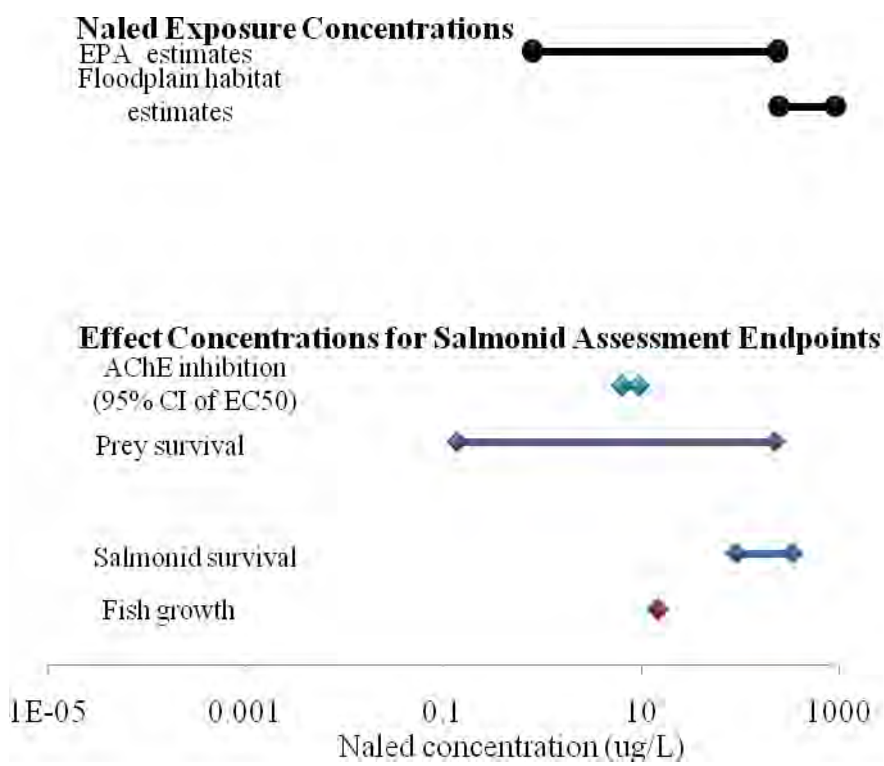


Figure 55 Comparison of exposure concentrations and salmonid assessment endpoints effect concentrations for naled

Phorate is applied only in granular form, thus we did not generate spray drift estimates for floodplain habitats. EPA estimates in the salmonid BE for acute exposure to phorate ranged from 4.6 – 23.1 μg/L for food crop uses. However, these estimates were for application rates between 1 and 1.3 lbs a.i./A, whereas phorate is approved for higher rates in other food crops (*e.g.*, potatoes at 3.54 lbs a.i./A). However, even considering 23.1 μg/L as the maximum EEC, concentrations overlap with effects concentrations for all endpoints where data are available. The EEC for lilies and daffodils in the BE was 115 μg/L at an application rate of 8 lb a.i./A, which is consistent with the current application rate approved for that use under a 24C registration in California. (Table 11). We anticipate adverse effects on salmonid survival, AChE activity, and growth, as well as reductions in prey survival in water bodies receiving runoff from phorate treated crops.

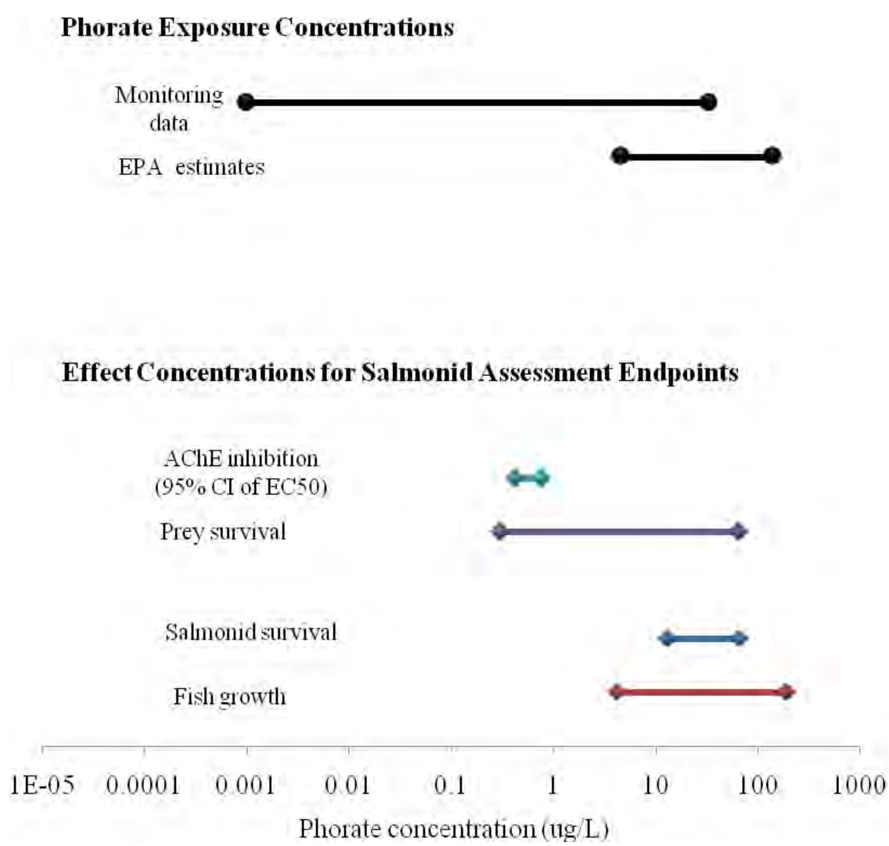


Figure 56 Comparison of exposure concentrations and salmonid assessment endpoints effect concentrations for phorate

EPA acute EECs for phosmet range from 3.0-29.9 µg/L in the salmonid BEs. The highest estimate from the RLF BE for phosmet is 78 µg/L. The acute EEC for forestry is 24 µg/L. NMFS floodplain estimates range from 5.0-2,920 µg/L based on the range of active label use rates. The wide concentration range indicates the variability in active label use rates. EECs from all sets of estimates exceed effects concentrations for fish growth, fish reproduction, and AChE inhibition, as well as prey survival (Figure 57). Some floodplain estimates also exceed the salmonid survival endpoint.

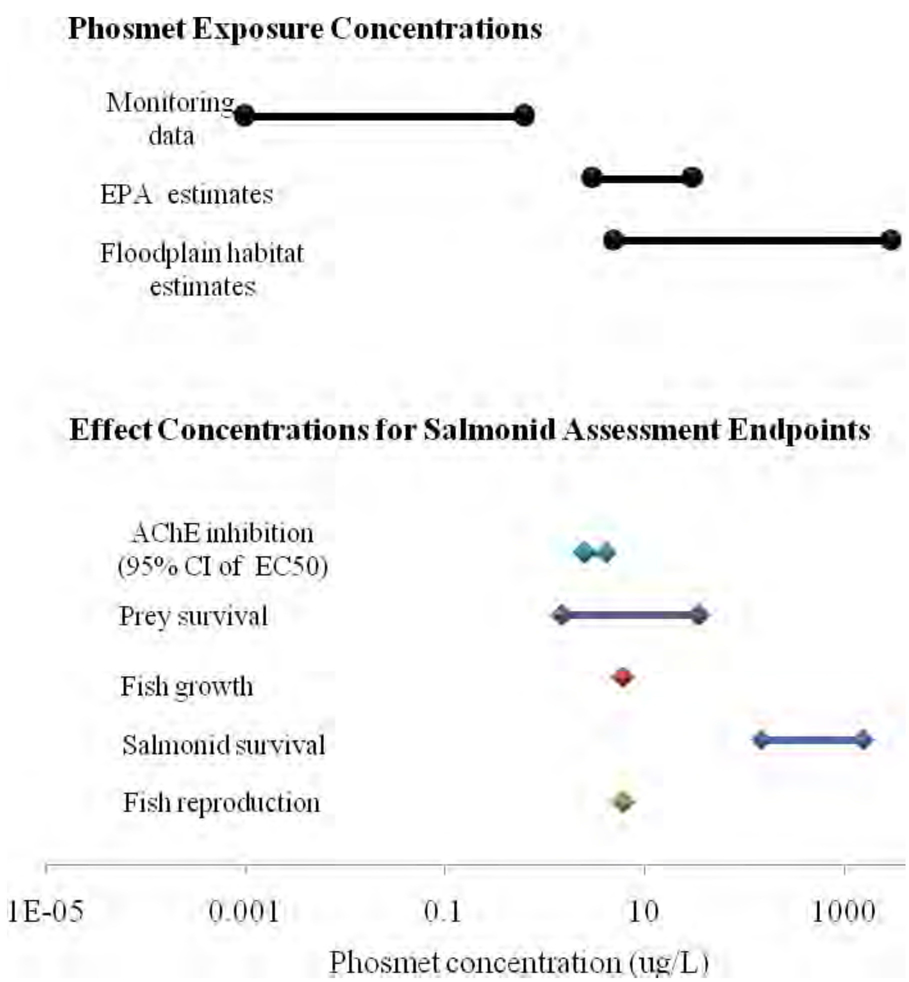


Figure 57 Comparison of exposure concentrations and salmonid assessment endpoints effect concentrations for phosmet

Relationship of pesticide use to effects in the field

Schulz (2004) reviewed 45 field and *in situ* studies published in peer-reviewed journals (from 1982-2003) that evaluated relationships between insecticide contamination and biological effects in freshwater aquatic ecosystems and included invertebrates and fishes. For each study, the author classified the relationship of exposure to effect in one of four categories: no relation, assumed relation, likely relation, or clear relation based on the cited authors' judgment of their own results. A relationship was classified as clear only if the exposure was quantified and the effects were linked to exposure temporally and spatially. The review concluded that “about 15 of the 42 studies revealed a clear relationship between quantified, non-experimental exposure and observed effects *in situ*, on abundance [aquatic invertebrate], drift, community structure, or dynamics” (Schulz 2004).

The three insecticides most frequently detected at levels expected to result in toxicity were chlorpyrifos (OP), azinphos-methyl (OP), and endosulfan (OC). Azinphos methyl (6 studies), and methyl parathion (4 studies) showed clear, likely or assumed relationships to the toxicological effects investigated. Mostly the effects noted were mortality or reduced abundance of invertebrate species, although one azinphos methyl study addressed a die-off of estuarine fish, and another noted reduction in carp (*Cyprinus carpio)* brain cholinesterase activity. In some cases, the water contained multiple pesticides, including mixtures of multiple AChE inhibitors. A number of other AChE inhibitors, both OPs and carbamates, also were judged to have clear or likely effects, including carbaryl, carbofuran, chlorpyrifos, diazinon, ethyl parathion, fenbucarb, malathion, and oxydemeton methyl. For these compounds, effects were generally mortality, reductions in abundance, and changes in community composition for aquatic invertebrates. Two studies also noted drift input into nearby waterbodies (Schulz 2004) . None of the other OPs addressed in this Opinion were included in the studies evaluated for relationships, but based on the mode of action, we assume the environmental effects are similar, although the severity of effects will vary. It should be noted that these studies were not designed to establish effect thresholds and in our assessment, are not sufficient to define thresholds. The review shows robust empirical evidence that OP insecticide applications under field conditions can cause biological and ecological effects to aquatic communities.

Schulz (2004) noted that for all of the studies “that seem to establish a clear link between exposure and effect, the pesticide concentrations measured in the field were not high enough to support an explanation of the observed effects simply based on [laboratory bioassays] acute toxicity.” Some authors have suggested differences in field measured exposure and actual organismal exposure (aquatic, sediment, dietary; environmental variables that affect exposure) as a reason for higher mortalities *in situ* than predicted by laboratory toxicity data. Schulz concluded that on the basis of present knowledge, it cannot be determined whether the measured concentration in the field regularly underestimates the actual exposure or if a general difference between the field and laboratory reactions of aquatic invertebrates is responsible. The review by Schulz shows a body of evidence that natural aquatic ecosystems can be adversely affected by AChE inhibitors. He reviewed multiple studies that showed direct, adverse effects to salmonid prey species following exposures to azinphos methyl and methyl parathion (Table 118). These included not only reduction in individual aquatic species, but also changes in community abundance, richness, and diversity of salmonid prey items. Most of the effects were noted following short exposures, typically a few hours to a few days.

Table 118. Published field studies establishing a relationship between azinphos methyl and methyl parathion contamination of aquatic habitats and toxic responses of fish and aquatic invertebrates (adapted from Table 2 in Schulz 2004).

Source	Concentration µg/L	Endpoint	Species	Relationship of exposure and effect	Reference
			Azinphos methyl		
Leaching (irrigation)	0.2	Brain cholinesterase	Carp (*Cyprinus carpio)*	Likely (mixture)	Gruber and Munn 1998
Runoff	1.42-21	Die-off Abundance	Estuarine fish Shrimp (*Palaemonetes pugio)*	Likely	Finley eta al 1999
Runoff	0.1-7	Mortality	Mummichog (*Fundulus heteroclites)* Shrimp (*Palaemonetes pugio)*	Clear (mixture)	Scott et al 1999
Runoff	0.8	Mortality	Dipteran (*Chironomus sp.)*	Clear (mixture)	Schulz and Peall 2001
Spray drift	0.87	Mortality	Dipteran (*Chironomus sp.)*	Clear	Schulz et al 2001c
Runoff, spray drift	0.82	Community composition	Various invertebrate species	Clear	Schulz et al 2002
			Methyl Parathion		
Runoff	0.4-213	Abundance	Various invertebrate species	Assumed (mixture)	Aufsess et al 1989

Source	Concentration μg/L	Endpoint	Species	Relationship of exposure and effect	Reference
Experimental	1-550	Abundance	Various invertebrate species	Clear	Schulz et al 2003b

One study on cholinesterase inhibition in carp was conducted in the Central Columbia River Plateau. Authors described their results in terms of cholinesterase (ChE) rather than specifically acetylcholinesterase (AChE). It was an evaluation of the effects of agricultural pesticides in a lake receiving irrigation return flow (Gruber and Mann 1998). Organophosphate and carbamate pesticide concentrations were measured in Crab Creek Lateral, which drains into Royal Lake, WA from March 1993-May 1994 (n=29). Streamflow was in the range of 1-2.5 m^3/sec during irrigation season, and nearly 0 m^3/sec from October to February when fields were not being irrigated, indicating the primary water source was irrigation return water. Several of the a.i.s addressed in this Opinion were detected at concentrations ranging from below detection limit to a maximum of 0.2 μg/L, including azinphos methyl (41% of samples), ethoprop (17 % of samples) and disulfoton (7 % of samples, no measurements for sulfoxide or sulfone). They also detected chlorpyrifos (52%), carbaryl (28%), diazinon (7%), and malathion (7%). Two or more a.i.s often appeared in the same samples, especially azinphos methyl and chlorpyrifos. According to authors, azinphos methyl and chlorpyrifos had the "highest application rates, the highest maximum concentrations, and the greatest percentage of samples that exceeded aquatic life criteria[9]." In addition to measuring pesticide concentrations, study authors collected carp (*Cyprinus carpio*, n=20) from Royal Lake and a reference location (Billy Clapp Lake, n=20) and measured brain ChE activity. Fish collected in Crab Creek Lateral exhibited a statistically significant decrease (34.2%) in whole brain ChE activity compared to the fish from the reference lake. Authors did not conduct any behavioral analyses, but ChE inhibition is known to alter feeding, swimming, and other behaviors. This paper provides robust field evidence that AChE

[9] Nowell, L.H., Resek E.A. 1994. Summary of national standards and guidelines for pesticides in water, bed sediment, and aquatic organisms and their application to water-quality assessments. Open-File Report 94-44, U.S. Geological Survey, Sacrament, CA.

inhibitors within irrigation return water can be sufficient to reduce normal AChE activity in wild fish.

Two of the papers evaluated by Schulz (Schulz 2004) were detailed multi-year studies on the effects of runoff in South Carolina estuaries (Finley et al 1999, Fulton et al 1999). As described in (Fulton et al 1999) authors used native grass shrimp (*Palaemonetes pugio*) and mummichogs (*Fundulus heteroclites*) to assess ecological effects of agricultural runoff at three sites. One site was a reference (control) site, one was an agricultural site where best management practices (BMPs) and integrated pest management (IPM) procedures were added and/or modified in the course of the study, and one was an unmodified agricultural site. Effects were evaluated by deployment of caged grass shrimp and mummichogs (n=25 at reference and managed sites, n=12 at the unmanaged site), and monthly counts of indigenous grass shrimp populations at the reference site and the managed agricultural site. Exposure was determined by daily water sample collection during field toxicity testing, water sample collection at predetermined intervals, and water sample collection following "significant rainfall events" (>1.27 cm/24h). Samples were analyzed for azinphos methyl, endosulfan I, endosulfan II, endosulfan cyclic sulfate, ethyl parathion, fenvalerate, and methyl parathion. Analytes were selected based on pesticides commonly used in the area. Rain events resulted in runoff of azinphos methyl, endosulfan, and fenvalerate. Investigators also conducted standard 96 h toxicity tests to determine LC50s for grass shrimp (azinphos methyl 1.05 μg/L (95% CI 0.91-1.21), endosulfan 1.01 μg/L (0.72-1.43), fenvalerate 0.052 μg/L (0.043-0.063)) and mummichogs (azinphos methyl 36.95 μg/L (95% CI 28.30-48.24), endosulfan 1.45 μg/L (1.32-1.59), fenvalerate 2.86 μg/L (2.02-4.06)).

During the course of the study, fish kills (of wild fish) were observed 5 separate times (Fulton et al 1999). On 3 occasions, the kills corresponded with peaks of azinphos methyl (~4-6 μg/L), on one occasion with a peak in endosulfan (~0.25 μg/L) and fenvalerate (~0.05 μg/L), and on one occasion both azinphos methyl (~3.5 μg/L) and endosulfan (~0.9 μg/L). Generally, the mummichogs appeared relatively insensitive to the runoff based on 90-100% survival of caged fish in most cases, although no sublethal endpoints were evaluated. Survival of grass shrimp was affected. At both agricultural sites, there were a number of occasions were survival of grass shrimp was <50%, and in some cases, mortality approached 100%. These mortality events were often correlated with detections of azinphos methyl, but sometimes also with endosulfan, fenvalarate, or some combination of the chemicals. Concentrations of pesticides in the water decreased at the managed agricultural site when a retention pond was constructed and both BMP and IPM practices were enhanced. With the exception of the year when Hurricane Hugo occurred, grass shrimp densities at the agricultural site were always lower than at the reference site. Although there was a great deal of variability in densities from year to year, there was a noticeable increase in densities at the managed agricultural site when the enhanced runoff control measures were instituted. These field results showed grass shrimp were adversely affected by azinphos methyl and other insecticides and fish kills correlated to toxic concentrations of azinphos methyl in estuaries. It also showed when tools were put in place to reduce pesticide loading, ecological effects of the pesticides were reduced.

Based on Schulz's evaluation (2004) and our review of the supporting papers, we conclude that expected concentrations of azinphos methyl in the action area as a result of agricultural practices can cause fish kills, cause sublethal effects in fish, and degrade aquatic invertebrate communities which serve as prey for salmonids. Methyl parathion is implicated in degradation of aquatic invertebrate communities. A majority of these instances are associated with direct runoff or spray drift, but we note that irrigation return water has been linked with effects as well. Equally important, based on the studies in South Carolina, we note that a combination of practices to limit runoff appears to reduce ecological effects associated with application of pesticides.

Field studies in ESA-listed salmonid habitats: Hood River Oregon

A group of field studies evaluated macroinvertebrate community responses in the orchard-dominated Hood River Basin, Oregon and correlated results with azinphos methyl and chlorpyrifos use and detections (Grange 2002, St. Aubine 2004, Vander Linde 2005). Hood River Basin contains several listed anadromous salmonids, including Lower Columbia River steelhead.

The goals of the studies were to determine whether in-stream OPs affected steelhead AChE activity and/or modified the aquatic macroinvertebrate community. An additional objective was to evaluate how changes in macroinvertebrate community structure might affect salmonid growth. Data from these studies, in part, substantiate growth-related population models presented later in the *Risk Characterization section*.

Two sets of field experiments directly investigated juvenile steelhead (hatchery-reared) AChE activity in caged-fish studies {Fulton, 1999 #1937}. Investigators analyzed water samples for chlorpyrifos, azinphos methyl, and malathion before, during, and after orchard spray periods. One of the studies also monitored the aquatic invertebrate community's response in conjunction with the AChE inhibition (St. Aubin 2004). Steelhead from reference sites had statistically significantly greater AChE activity than steelhead from orchard-dominated areas. The reductions in AChE activity corresponded to the application seasons and detections of chlorpyrifos and azinphos methyl insecticides.

The data indicated that OP-insecticides inhibited AChE activity in the caged steelhead. Inhibition was correlated to chlorpyrifos and azinphos methyl detections and, to a lesser degree, with malathion detections (Grange 2002). None of the pesticides were detected at reference sites and both azinphos methyl (0.03- 0.27 μg/L) and chlorpyrifos (0.08- 0.20 μg/L) were frequently detected at orchard stream and river sites (Grange 2002, St. Aubine 2004). AChE activity was inhibited up to 21% in smolts, and 33% in juveniles relative to reference locations. Temperature was a confounding factor, as lower AChE activity occurred at lower temperatures and higher AChE activity occurred at higher temperatures at reference sites. Authors normalized data to temperature to eliminate this variability, and found a greater number of statistically significant reductions in AChE in steelhead. Depending on the percentage of inhibition, these OP-induced AChE activity reductions can manifest into fitness level consequences such as reduced growth (Grange 2002, St. Aubine 2004).

The concurrent macroinvertebrate studies evaluated community assemblages rather than the direct endpoint of AChE inhibition. Aquatic macroinvertebrate communities integrate toxic effects over time, and thus are often a more sensitive indicator than measured water concentrations, which may fail to capture the brief pulses of pesticide inputs that can occur in flowing water. However, the inherent variability associated with community dynamics and site differences can make interpretation of these studies difficult. In the first year of the study, a summer sampling was conducted and significant differences in macroinvertebrate community assemblages were found between upstream reference sites and downstream agricultural sites (St. Aubine 2004). However, no significant differences were found within each individual site. Therefore, a second Hood River study investigated the spring spray events as well as the summer spray events to determine seasonal effects (Van der Linde 2005). Sharp declines in species abundance at the downstream agricultural sites as compared to the upstream reference occurred during the spring spray period. These reductions correlated to chlorpyrifos applications and subsequent aquatic detections (at one site chlorpyrifos was detected at concentrations ranging from 0.032 -0.183 μg/L over an eight day period). Agricultural sites contained more pollution tolerant taxa and less intolerant taxa than reference sites (Van der Linde 2005). Collector–gather species, many of which are salmonid prey items, were less abundant at agricultural sites compared to the reference sites. Illustrating the variability noted previously, the reductions in biodiversity seen in 2001 when agricultural sites were compared to reference sites was not seen in 2002 (Van der Linde 2005).

These field studies conducted in salmonid-bearing waters of the Hood River Basin, Oregon demonstrate two important effects: reduction of salmonids' AChE activity and modification of macroinvertebrate assemblages associated with azinphos methyl and chlorpyrifos applications in orchard-dominated watersheds.

Wide Area Application of Mosquitocides

Naled is commonly used as an adult mosquitocide, and may be sprayed over large areas including aquatic habitats such as estuarine areas and floodplain habitats used by listed salmonids. We were unable to locate any studies detailing the results of such control programs in California or the Pacific Northwest, but did locate a detailed study on the fate and biological effects of naled used in a mosquito control program in South Carolina. We also located a study evaluating aerial drift and tidal transport of naled and dichlorvos in the Florida Keys.

Bolton-Warberg *et. al*, (2007) conducted a targeted study evaluating the effects of naled (Dibrom [commercial product]) spraying by the Charleston County Mosquito Abatement Program (CCMAP) on caged grass shrimp (*Palaemonetes pugio*) and eastern oyster (*Crassostrea virginica*) deployed in a tidal marsh. Investigators deployed caged shrimp subtidally and oysters intertidally at three treatment sites and two reference sites in tidal creeks. Baseline deployments were conducted for the treatment sites, and two separate deployments were done at each of the treatment sites, concurrent with scheduled CCMAP treatments. Investigators measured concentrations of naled's degradate dichlorvos, evaluated effects on AChE inhibition, and recorded mortality events. "Mortality in field-deployed shrimp was <1% (2 out of 638 shrimp deployed)." AChE results were inconclusive, with no apparent pattern when comparing reference sites to treatment sites or baseline measurements to post-spraying inhibition.

4-L water samples for chemical analysis were taken at the treatment sites prior to deployment for the second spray event and upon retrieval of the caged organisms. Water samples were taken upon deployment and retrieval for the baseline at the reference sites. The dichlorvos LOD for water samples was 177.5 μg/L and LOQ was 200 μg/L. Note that both LOD and LOQ are higher than LC50, NOEC, and LOEC values determined in the laboratory toxicity tests, thus there may have been dichlorvos present in the water in toxic concentrations that would not be detectable by the analytical method employed. Authors note in one instance that naled was not detected, but do not provide an LOD or LOQ, or any further discussion. Concentrations of dichlorvos were below LOD for all creeks during the baseline deployment. Concentrations of dichlorvos in water taken upon retrieval was <LOD to 200 μg/L at the treated sites, and <LOD to 177.5 μg/L at the reference sites.

Overall, results from this study are inconclusive, as the actual exposure was not well characterized. The pattern of AChE inhibition did not appear to be correlated specifically with the application of naled and resulting concentrations of dichlorvos in the water. Confounding factors include the possibility of other AChE influencing compounds in the water, variability in the AChE results that precluded distinguishing subtle effects in exposed individuals and/or greater tidal transport of the dichlorvos than anticipated.

Aerial drift and tidal transport of naled and dichlovos resulting from mosquito control applications in Key Largo, Florida was evaluated (Pierce et al 2005). Naled (Dibrom-14) was applied as ultra-low volume (ULV) spray at an application rate of 21.3 g a.i./h w by C-47 aircraft. Two applications, one in June, and one in July, were considered. Details of release height, location, and total amount released were not provided. Authors do discuss windspeed and direction in relation to ground applied permethrin, which occurred approximately 9 hours prior to the naled applications. In both cases, the wind was out of the East-Southeast. Windspeed ranged from 5kts (June) to 12 kts (July). Authors do not indicate if naled was applied over land, or over water and allowed to drift in.

Sampling design included filters to capture airborne drift, samples of the water surface microlayer, and samples of the water column. Samples were collected in a grid pattern on both the Atlantic and Florida Bay sides of Key Largo, with 9 sample sites on each sitde. Samples were collected pre-application, and at 2-4 hours and 10-12 hours following application. QA/QC measures included field blanks, spiked standard recoveries, and surrogate recovery standards. LODs for naled and dichlorvos were 0.05 μg/L in water and 0.1 $\mu g/m^3$ for filter. Sites were sampled prior to application to establish determine background concentrations.

Following the first application, at the 2-4 h post-application sampling, neither naled nor dichlorvos was detected on the filters on the Atlantic side. On the Bay side, naled was detected on one filter (1.6 $\mu g/m^3$) and dichlorvos was detected on another filter (0.16 $\mu g/m^3$). Neither naled nor dichlorvos was detected in the surface microlayer. Naled was detected in one water column sample on the Atlantic side (0.19 μg/L), but dichlorvos was detected in “50% of the water samples” (range 0.08 -0.56 μg/L). At the 10-11 h post application sampling, dichlorvos was detected at 3 of the 9 sampling sites (range 0.05 - 0.33 μg/L. Authors postulate tidal transport from canals on the island to be a source of the dichlorvos.

Following the second application, dichlorvos (0.4 $\mu g/m^3$) was detected on one filter on the Bay side of the island. Naled was not detected on any of the filters at this sampling, which was approximately 5-7 h post application. No naled or dichlorvos was detected in the surface microlayer. Naled was not detected in water column samples. Dichlorvos was detected at 2 sites (range 0.7 -0.09 μg/L), but in lower concentrations than following the first application.

Although data in this paper are limited, it does provide evidence that dichlorvos may be present in the water column following ULV applications of naled, and that concentrations may persist for 10-12 h following application. It also demonstrates there is more variability in deposition from these types of applications than standard crop applications, which typically have a larger droplet size spectrum and are applied by lower-flying aircraft.

Field incidents reported in EPA incident database

NMFS reviewed reported incidents of fish deaths from field observations throughout the U.S. because this information reflects real world scenarios of pesticide applications and corresponding death of freshwater fish. We recognize that much of the information is not described in sufficient detail to attribute an incident to a label-permitted use leading to the death of fish, or to make conclusions regarding the frequency of fish kills that may be associated with the use of pesticides. NMFS uses the information as a component to evaluate a line of evidence- whether or not fish kills have been observed from labeled uses of the 12 a.i.s. EPA categorizes incidents in the database into one of five levels of certainty: highly probable, probable, possible, unlikely, or unrelated. The certainty level indicates the likelihood that a particular pesticide caused the observed effects. EPA uses the following definitions to classify fish kill incidents:

- Highly probable (4): Pesticide was confirmed as the cause through residue analysis or other reliable evidence, or the circumstances of the incident along with knowledge of the pesticides toxicity or history of previous incidents give strong support that this pesticide was the cause.

- Probable (3): Circumstances of the incident and properties of the pesticide indicate that this pesticide was the cause, but confirming evidence is lacking.

- Possible (2): The pesticide possibly could have caused the incident, but there are possible explanations that are at least as plausible. Often used when organisms were exposed to more than one pesticide.

- Unlikely (1): Evidence exists that a stressor other than exposure to this pesticide caused the incident, but that evidence is not conclusive.

- Unrelated (0): Conclusive evidence exists that a stressor other than exposure to the given pesticide caused the incident.

NMFS reviewed incident reports provided by EPA from OPP's incident database. This database is populated with reports received by EPA from registrants that are defined as reportable under FIFRA 6(a)(2) and includes other information received from registrants and other sources. EPA provided a summary of aquatic incidents associated with the 12 a.i.s (Table 119). Incidents classified with a certainty of unlikely are not reported in the table. Incidents associated with product misuses were not provided.

Table 119. EPA summary of aquatic incidents involving the 12 a.i.s

Active ingredient	EPA certainty that incident was caused by the active ingredient		
	Possible (Incident ID)	Probable (Incident ID)	Highly Probable (Incident ID)
Azinphos methyl	B0000-300-50, B0000-500-25, B0000-500-27, B0000-500-37, B0000-500-61, B0000-500-74, B0000-500-79, B0000-500-84, B0000-500-95,	B0000-300-51, B0000-500-17, B0000-500-19, B0000-500-22, B0000-500-23, B0000-500-24, B0000-500-26, B0000-500-28, B0000-500-33,	B0000-500-43, B0000-501-44, B0000-501-45, I000203-001, I000203-002, I012265-001, I017028-001,

Active ingredient	EPA certainty that incident was caused by the active ingredient		
	Possible (Incident ID)	Probable (Incident ID)	Highly Probable (Incident ID)
	I000109-005, I000769-001, I001838-001, I004668-011, I012265-004, I013436-001, I013530-001,	B0000-500-35, B0000-500-36, B0000-500-38, B0000-500-39, B0000-500-40, B0000-500-41, B0000-500-42, B0000-500-45, B0000-500-46, B0000-500-47, B0000-500-48, B0000-500-49, B0000-500-50, B0000-500-51, B0000-500-52, B0000-500-53, B0000-500-54, B0000-500-55, B0000-500-60, B0000-500-62, B0000-500-63, B0000-500-64, B0000-500-65, B0000-500-66, B0000-500-67, B0000-500-68, B0000-500-69, B0000-500-70, B0000-500-71, B0000-500-72, B0000-500-73, B0000-500-76, B0000-500-77, B0000-500-78, B0000-500-80, B0000-500-81, B0000-500-82, B0000-500-85, B0000-500-86, B0000-500-88, B0000-500-89, B0000-500-90,	

Active ingredient	EPA certainty that incident was caused by the active ingredient		
	Possible (Incident ID)	Probable (Incident ID)	Highly Probable (Incident ID)
		B0000-500-91, B0000-500-92, B0000-500-93, B0000-500-94, B0000-501-26, B0000-501-28, B0000-501-29, B0000-501-30, B0000-501-31, I000109-007, I000109-009, I000109-018, I000109-025, I000109-030, I000114-001, I000114-002, I000114-003, I000146-001, I000146-002, I000146-003, I000146-004, I000146-005, I000146-006, I000200-037, I000203-003, I000247-003, I000247-004, I000454-014, I000592-001, I000603-001, I001849-011, I001863-002, I001863-003, I002338-001, I002363-001, I003622-001, I003659-001, I004374-006, I004875-004, I004875-011, I005148-001, I005148-002,	

Active ingredient	EPA certainty that incident was caused by the active ingredient		
	Possible (Incident ID)	Probable (Incident ID)	Highly Probable (Incident ID)
		I005148-003, I012265-002, I012265-003	
Bensulide	NR	NR	NR
Dimethoate	I003826-002	I000403-001	NR
Disulfoton	I003826-002	I001167-001,	NR
Ethoprop	NR	I000221-001, I001712-001, I001849-006	NR
Fenamiphos	I003822-001	I000454-005, I000636-010, I001076-001	I000630-001, I000666-001, I000636-001
Methamidophos	NR	NR	NR
Methidathion	NR	NR	NR
Methyl parathion	B0000-243-01, B0000-255-01, B0000-261-01, B0000-264-01, B0000-501-44	B0000- 244-01, B0000- 252-01, B0000-254-01, B0000-263-01, B0000-271-01, I000109-024, I000383-002, I006861-002	B0000-262-01, I001849-009
Naled	NR	NR	NR
Phorate	B0000-501-37	B0000-300-53, I002814-001, I006718-002	NR
Phosmet	NR	NR	NR

NR none reported

Azinphos methyl incidents

In 2001, EPA indicated their incident database contained more aquatic incidents attributed to azinphos methyl than any other pesticide in the database (EPA 2001). Additionally, the incidents were characterized in the following way:

"Azinphos methyl has 143 incidents reported prior to 2000, only including incidents that are probable or highly probable to have associations with the azinphos methyl and excluding those associated with misuse. This number of incidents is more than twice the number of incidents of the next highest chemical, which is chlorpyrifos with 63 incidents. Azinphos methyl is responsible for over 21% of all aquatic incidents. A large majority of the incidents are associated with the cotton and sugar cane uses. Seventy seven of these incidents are associated with cotton and 37 were associated with sugar cane. In addition, there are 15 incidents that are unclassified or classified as “agricultural”. This accounts for 129 of the 143 incidents. Of the remainder, 1 is associated with apples (MO), 1 with citrus (FL), 3 with potatoes (ME), and 1 with peaches (MO). There are also 7 incidents that unclassified or classified as “orchard” in New York (2), Washington (1), Wisconsin (1), North Carolina (1), Maine (1), and Michigan (1). If all events associated with azinphos methyl are included, which adds misuses, and those with less certainty, there are 256 incidents. These include an almond incident (CA), one more apple (NC), 1 blueberry (ME), 1 forestry (AR), and one “nursery” (GA). The balance are associated with sugar cane and cotton.

“Aside from the number of incidents, the size of the incidents and kinds of species killed for azinphos methyl stand in contrast to other currently registered pesticides. Some of the incidents associated with sugar cane are listed as “6 miles long” and “2 miles long”. Ten others have over 10,000 fish killed. Some of the fish included are those not otherwise found in the incident database including gar, catfish, buffalo, and bowfin, and carp. These “aquatic incidents” also included some otherwise terrestrial or semiaquatic species including turtles, an alligator, a dog, and a pig.”

In our review of the data, we found that a large proportion of the fish kill incidents were associated with boll weevil eradication efforts in the Southeast. More than 50 of those incidents are summarized in Table 120. Although azinphos methyl is no longer permitted for use on cotton, the information provided is pertinent because it indicates measured concentrations of azinphos methyl associated with fish kills. Additionally, it indicates concentrations of pesticides that may end up in aquatic habitats resulting from wide area spray operations. These incidents summarized below resulted from aerial application of 1 pint of Guthion 2L (0.25 lbs azinphos methyl/A). Although current labels restrict azinphos methyl to ground applications, they also allow application at a much higher rate (0.75 – 3 lbs a.i./A) and many uses do not require buffers to aquatic habitats. Ground applications at maximum permitted use rates and with no buffers may produce even higher concentrations than measured in many of these incidents. For example, we estimate average initial concentrations of 0.8-11.4 µg azinphos methyl/L in a modeled floodplain habitat resulting from airblast applications of 0.75-1.5 lb a.i./A (Table 93). The peak concentration detected in the incident investigations was 670 µg/L (B0000-500-23). However, the majority of the incidents were associated with measured concentrations near 1 µg/L, which laboratory studies indicate is lethal to fish. It is likely that measured concentrations associated with these kills did not capture peak exposure concentrations given the expected dissipation of azinphos methyl in the aquatic environment, (hydrolysis half-life of 3.2 days).

Table 120. Monitoring of fish kills associated with aerial spray of azinphos methyl for boll weevil eradication at 0.25 lb a.i./A.

Incident	Distance to surface water (feet)	Number of Fish Killed	Maximum Concentration detected in water µg/L
B0000-500-23	100	2500	670
B0000-500-82	NR	“thousands”	5.48
B0000-501-30	1500	“all”	5.34
B0000-500-40	450	“hundreds”	3.52
B0000-500-33	900	NR	2.93
B0000-500-47	3000	NR	2.38
B0000-500-41	236	“extensive”	2.36
B0000-500-80	150-300	“all”	2.34
B0000-500-78	NR	“severe”	2.3
B0000-500-17	NR	2000	2.24
B0000-501-29	150	NR	1.94
B0000-500-68	200	1500	1.93
B0000-500-43	100	“large number”	1.87
B0000-500-65	50	NR	1.65
B0000-500-42	150	125	1.56
B0000-500-62	75-100	“thousands”	1.53
B0000-500-86	300	NR	1.48
B0000-500-69	100-150	NR	1.47
B0000-500-90	225	NR	1.46
B0000-500-85	100	“all”	1.41
B0000-500-76	50-75	300	1.38
B0000-500-54	150	500	1.38
B0000-500-22	3000	2000	1.34
B0000-500-49	250	3	1.31
B0000-500-63	300	“all”	1.3
B0000-500-45	25-150	“several”	1.3
B0000-500-94	300	NR	1.2
B0000-500-81	40	NR	1.15
B0000-500-39	200	“thousands”	1.1
B0000-500-46	30-40	NR	1.09
B0000-500-88	600	“hundreds”	1.08
B0000-500-24	NR	NR	1.08
B0000-500-91	168	NR	1.04
B0000-500-89	30	“all”	1
B0000-501-31	1300	NR	0.91
B0000-501-26	NR	NR	0.86
B0000-500-64	75-100	“several”	0.73
B0000-500-51	1700	“several”	0.71
B0000-500-26	1050	NR	0.71
B0000-500-77	125-1500	2500	0.67
B0000-500-70	250	4000	0.64
B0000-500-53	400	2000	0.63

Incident	Distance to surface water (feet)	Number of Fish Killed	Maximum Concentration detected in water µg/L
B0000-500-52	NR	"hundreds"	0.61
B0000-500-50	150	"hundreds"	0.59
B0000-500-48	100	55	0.59
B0000-500-92	225-300	"hundreds"	0.54
B0000-500-72	900	"large number"	0.54
B0000-500-38	150-175	"thousands"	0.48
B0000-500-36	40-150	"thousands"	0.47
B0000-500-60	30	"60-70%"	0.45
B0000-500-93	150	NR	0.42
B0000-500-73	150	NR	0.4
B0000-500-55	100	10000	0.39
B0000-501-28	200	"thousands"	0
B0000-500-28	60	NR	0
B0000-500-71	150	10000-12000	NE
B0000-500-67	141	NR	NE
B0000-500-19	50-100	200	NE

NR- Not reported, NE- Not evaluated

EPA considered it highly probable that azinphos methyl was the cause of death in several other incidents. A large fish kill of approximately 10,000 to 20,000 fish covering 2 ½ miles in a Louisiana bayou occurred in 1992 (I000203-001). The kill involved several species and occurred adjacent to cropland which drained directly into the bayou. Several fields were sprayed with azinphos methyl at a rate of 3 pints Guthion/A in the days immediately preceding the fish kill. The pathway of exposure was presumed to be runoff given substantial rainfall (1.6 in) that occurred the day of the fish kill. Azinphos methyl was detected at 17.4 µg/L in a water sample collected at the fish kill site. Another incident reported in Louisiana in 1992 was a fish kill of approximately 1,000 fish of several species (I000203-002). The cause of death was attributed to azinphos methyl and associated with applications of 0.375 "units" Guthion/acre to sugar cane. Lethal concentrations of azinphos methyl were found in water samples (11 µg /L) and high concentrations were also found in all fish tissues examined (>30 µg/kg).

Approximately 400 fish were killed near the border of Arkansas and Louisiana (B0000-501-44, highly probable). Water samples contained detectable concentrations of azinphos methyl, methyl parathion, and atrazine. The fish kill was attributed to azinphos methyl given that several species of fish have LC50 values below the concentration of azinphos methyl measured at the site (5 μg/L). However, we note that 10,000 μg/L of methyl parathion were also detected in the water. These concentrations may also be lethal given laboratory LC50 values for fish as low as 790 μg/L (Table 113). Further, the kill may be a result of mixture toxicity given the presence of the two cholinesterase-inhibiting OPs (azinphos methyl and methyl parathion). The co-occurrence of atrazine with OPs has been shown to cause additive effects in aquatic invertebrates {Van der Linde, 2005 #765}. Atrazine may potentiate the toxicity of some OPs by inducing metabolic enzymes that convert the parent to a more toxic oxygen analog {, 2007 #1766}. However, we have not found studies demonstrating atrazine potentiation in fish.

Another fish kill (B0000-501-45, highly probable) involving approximately 50 fish included measured concentrations of azinphos methyl (7.27μg/L) and atrazine (5.25 μg/L). Again the cause of death was attributed to azinphos methyl given that concentrations were above LC50 values for several species of fish. Information regarding the application (rate, method, etc.) was not provided.

Two fish kill incidents occurred in Prince Edward Island, Canada in 1999 and 2000 (I012265-001 and I017028-001, highly probable). The first incident followed use of azinphos methyl and metiram (a dithiocarbamate fungicide) on potatoes and killed more than 1,200 fish. Trace concentrations of the fungicide were detected in surface waters, as were relatively high concentrations of carbofuran (0.32-18.5 μg/L). Use of carbofuran was not reported. EPA concluded that the fish kill was possibly caused by carbofuran, but that it was highly probable that the kill was caused by azinphos methyl given that surface water concentrations exceeded fish LC50 values (maximum concentrations of 2.31-5.43 μg /L). We recognize that carbofuran and azinphos methyl may have contributed to this kill through additive toxicity since both compounds inhibit cholinesterase. The second fish kill was also associated with azinphos methyl use on potatoes. The total number of dead fish was not reported, although 50 trout were collected for analysis. EPA concluded the deaths were caused by azinphos methyl given concentrations of the pesticide in fish tissues (*e.g.*, up to 0.39 mg/kg in gills).

Dimethoate Incidents

I000403-001 (Probable): EPA listed dimethoate and chlorpyrifos as the probable cause of a large fish kill on the Mackinaw River in Illinois in 1988. Over 9,000 fish were killed two days after applications of dimethoate and chlorpyrifos were made to crops on land near the fish kill site. Chemical analysis was not performed to confirm which chemical was responsible. Exposure to dimethoate and chlorpyrifos is expected to result in additive effects since these compounds both inhibit cholinesterase. The likelihood of exposure to environmental mixtures of dimethoate with other anticholinesterase pesticides is increased when they are co-applied or sequentially applied over a short interval. Dimethoate labels indicate it can be tank mixed with other pesticides that inhibit cholinesterase (EPA Reg No. 19713-231 and 9779-273).

Dimethoate and Disulfoton Incident

I003826-002 (Possible): A fish kill was reported in a 6 acre farm pond and both dimethoate and disulfoton were listed as the possible cause. Applications of several herbicides and insecticides were made to crops within 200 feet of the pond prior to the occurrence of the fish kill. This included separate applications of four cholinesterase inhibiting insecticides (dimethoate, disulfoton, chlorpyrifos, and aldicarb) made during a span of approximately eight weeks before the fish kill. Pesticide runoff was suspected, but not confirmed, as the cause of death. Water samples collected 4 days after the fish kill was observed did not detect the presence of dimethoate, aldicarb, or disulfoton.

Disulfoton Incident

I001167-001 (Probable): A fish kill in a deep, 4.5 acre Colorado "pond" was attributed to the use of disulfoton on wheat. EPA classified this event as having a probable certainty that disulfoton was the primary cause of death, given the measured concentrations in two water samples were 29.5/48.7 µg/L and 0.02/0.21 µg/L for the disulfoton sulfoxide and disulfoton sulfone degradates, respectively. The incident followed two significant rainfall events that caused sheet erosion and an increase in the pond depth from 10 to approximately 20-25 ft. Samples were collected three days after the second rainfall event (or six days after the rainfall first event). The measured concentrations of disulfoton sulfoxide are particularly notable in that they are greater than the screening estimates provided in the salmonid BE for the parent material using PRZM-EXAMS, despite the potential dilution provided by the volume of the large pond.

Ethoprop Incidents

I001712-001 (Probable): Dead fish and eels were found in lagoons on three South Carolina golf courses after applications of ethoprop were followed by heavy rainfall. The ethoprop was applied with a seeder that incorporated the granular formulation below the soil surface, and then further watered the ethoprop into the soil with 0.2 inches of irrigation water. Ten water samples collected from the kill site five days after the incident showed ethoprop concentrations of 3-241 µg /L. A second set of water samples, collected 10 days later, showed significant dissipation and ranged from non detectable to 6 µg/L. Although application of ethoprop to golf courses is not permitted on active labels, the application methods (ground, with soil incorporation) and rate (7.5 lbs a.i./A) are comparable to current agricultural uses. Active labels permit ethoprop to be applied at a maximum application rate of 12 lbs/A in potatoes. The maximum rate for several other crops ranges from 5 – 8 lbs/A (cabbage, corn, mint, tobacco, lilies).

I000221-001 and I001849-006 (Probable): Fish kills were reported on three lakes and one private pond in Louisiana in 1994. Approximately 200 shad were killed in the pond incident (I001849-006). The number and species of fish killed in the three lakes were not reported. Ethoprop was determined to be the probable cause for all of the incidents given measured concentrations in water. The concentrations were not provided for the pond incident we received. Two water samples and one fish were collected for analysis associated with the lake kills. Ethoprop concentrations of 10 and 26 µg/L were detected in the water samples, however, the sampling apparently occurred well after the event given the advanced state of decay of a fish (I000221-001). The source of the ethoprop was not determined for either incident.

Fenamiphos incidents

I000630-001/ I000666-001 (highly probable): A large fish kill occurred on several lakes on a Miami golf course in 1993 following a fenamiphos application and subsequent rainfall. Nemacur 10G was applied to approximately 100 acres on July 6 and 7. A heavy rainfall was reported for July 7 (up to 0.70 inches measured nearby). The fish kill was first noted on July 8. Dead fish were observed in all lakes and concentrations up to 520 µg/L were measured in lake water. Laboratory studies suggest that this concentration would be sufficient to cause lethality in salmonids (LC50s 68 – 560 µg/L). Another series of highly probable incidents that killed several thousand fish were documented on golf courses in Missouri in 1981 (I000636-001). Bluegill, bass, catfish, and other species were killed in several small impounds following applications of fenamiphos and subsequent heavy rains. These observations, and the occurrence of several other probable incidents (I000454-005, I000636-010, and I001076-001) are consistent with environmental fate studies that suggest fenamiphos is mobile in soils and has a high potential to contaminate surface waters through the runoff pathway.

Methidathion incidents

I013170 (unlikely): EPA did not characterize any of the methidathion incidents as possible, probable or highly probable. However, an incident that occurred in 2002 is noteworthy because it documents likely exposure of fish to mixtures of anticholinesterase insecticides. The incident was a fish kill involving over 2,000 fish distributed over 3-4 miles of stream channel were reported in the Salinas River of California. Several species of fish were killed. Applications of methidathion, diazinon, and esfenvalerate to agricultural fields in the vicinity were reported. Surface water samples revealed detectable concentrations of methidathion (0.05-0.24 µg/L) and diazinon (0.10 – 018 µg/L). Diazinon was also detected in fish gill tissue (5.2 - 44 µg/kg). Esfenvalerate was not detected in water, sediment, or fish tissue. Although it was not possible to identify the cause of the fish kill, the data confirm fish exposure to diazinon and suggest likely exposure to methidathion, both anticholinesterase agents that likely cause additive toxicity to fish. The concentrations detected were below expected lethal concentrations. However, the condition of the fish suggested the kill occurred 24 to 48 hours prior to sampling allowing time for significant dissipation of OP contaminants in a riverine environment.

Methyl parathion incidents

B0000-262-01 (highly probable): Methyl parathion was listed as the cause of death in a kill of an estimated 6,400 fish in Missouri in 1973. Methyl parathion was measured in surface water at 25 µg/L. Endrin was also detected at 2.3 µg/L, which may be sufficient to cause lethality in fish (bluegill LC50 0.19-0.73 µg/L, Mayer and Ellersieck 1986). Although the measured concentration of methyl parathion would not be predicted to cause lethality based on laboratory tests with the single active ingredient, dose-response evaluations suggest they would be adequate to inhibit brain cholinesterase by approximately 47% (NOAA 2009). Another highly probable incident involving methyl parathion resulted in the death of approximately 2,400 hundred fish. The fish were killed in three lakes following the application of methyl parathion and profenofos, another organophosphate pesticide that inhibits cholinesterase (I001849-009). The application of these compounds was reportedly to a "large acreage of cotton" that drained into the lakes following a heavy rainfall. It was not indicated if these products were applied as tank mixtures or separate applications. However, the cause of death was attributed to both pesticides.

Methyl parathion was reported as the probable cause of death in seven incidents. The majority of these incidents were attributed to exposure to methyl parathion and another neurotoxic pesticide (B0000- 244-01, B0000- 252-01, B0000-263-01, B0000-271-01, I000109-024). These were primarily older incidents that occurred in the 1970s. The other insecticides implicated were organochlorine pesticides (lindane, toxaphene, endrin, endosulfan). Use of organochlorine pesticides has been largely replaced by other pesticides although exposure to these persistent compounds still occurs. Few details regarding the application of methyl parathion associated with these incidents was provided and it is uncertain how representative they may be of current use patterns. The reported magnitude of the incidents ranged in size from a few hundred to 20-30 million fish.

Phorate incidents

B0000-300-53 (probable): Three separate farms reported fish kills in farm ponds shortly after heavy rainfall in Illinois. All had used phorate in combination with herbicides. The three ponds had phorate concentrations of 6.8 to 32.3 µg /L. Phorate concentrations in this range could result in acute lethality to 8 – 96% of the exposed population (assuming LC50 of 13 µg/L and sigmoid slope of 3.63). The phorate application rates associated with these incidents were 5-7 lbs a.i./A, higher than the rate permitted for most crops on active labels (1.3-3.5 lbs a.i./A). However, a SLN registration in California allows for phorate application at a greater rate than those associated with these incidents (8 lbs a.i./A in lilies and daffodils) suggesting fish kills may occur when significant rainfall follows phorate applications to these crops. It is not clear to what extent the herbicides, which were not identified, may have contributed to these incidents.

I002814-001/ I006718-002 (probable): Five separate fish kill events where phorate was classified as the probable cause of death were reported by the American Cyanamid Company in 1995. These events occurred during the 1992 – 1995 growing season. The registrant indicated that "each of these events occurred under similar conditions; heavy rainfalls shortly after planting in heavy soils which drain toward small, mostly shallow man-made farm ponds." Heavy rainfall following spring planting can occur in some of the more arid regions of California, Idaho, Oregon, and Washington and is likely in many agricultural areas in these states, including areas west of the Cascade Range. These events suggest phorate runoff to smaller bodies, such as floodplain habitats used by salmon, can result in concentrations sufficient to cause direct lethality.

Overall, there were large discrepancies in the occurrence of incident reports associated with the 12 a.i. For example, disproportionately large numbers of incidents in the database were attributed to azinphos methyl, while any incidents associated with bensulide, methamidophos, methidathion, naled, or phosmet were considered unrelated or an unlikely cause of the event. This may be partially explained by differential toxicity or environmental fate characteristics among the compounds (*e.g.* azinphos methyl versus bensulide). However, several other contributing factors are likely involved. For example, different levels of investigation appear to be associated with certain different uses (*e.g.,* the boll weevil eradication program versus other agricultural applications). Incidents associated with some uses are also more likely to be discovered and reported than others (*e.g.*, public golf course versus private agricultural lands,(Vyas 1999)). There are several other factors that influence what proportion of kills are observed, reported, investigated, and confirmed. The number of incidents confirmed is believed to be only a small subset of actual mortality associated with pesticide exposure. Uncertainty regarding response to multiple stressors and the toxicity of mixtures further complicate the confirmation of cause of death (Vyas 1999).

Mixture Analysis of the 12 a.i.s

As noted earlier, pesticides most often occur in the aquatic environment as mixtures. In our review and synthesis of the available exposure information, we find that mixtures including combinations of two or more of the a.i.s are expected to co-occur in salmonid habitats. These pesticides share the same mechanism of toxic action (AChE inhibition). Therefore, we employ a simple mixture analysis derived from empirical data with Pacific salmonids to predict potential effects to individual salmonid's AChE activity and their survival from short-term exposures. The analysis is predicated on the toxic potencies of the active ingredients added together to predict the resulting cumulative effect to AChE activity and mortality.

Mixture toxicity is typically described by three general responses: antagonistic, additive, or synergistic. Antagonism and synergism are where the toxic response is not predicted by the individual potencies of the pesticides found in the mixture. Antagonistic effects of a mixture lead to less than expected toxicity on the organismal endpoint. Mechanistically, the pesticides are likely interacting with one another to reduce the toxic potency of individual pesticides. Synergistic effects of a mixture lead to a greater than expected effect on the organismal endpoint and the pesticides within the mixture enhance the toxicity of one another. The third general type of mixture toxicity and the one most frequently reported is additivity (known also as dose-addition or concentration-addition). This type of response is defined by adding the individual potencies of pesticides together to predict the effect on the biological endpoint. Additivity has been demonstrated for many pesticide classes as well as other organic compounds such as PAHs, PCBs, and dioxins.

Additive toxicity of chemicals that share a mode or mechanism of toxic action is well established in the scientific literature, and as a result, has been informing regulatory decisions for more than a decade. In 1996, the National Academy of Sciences recommended a dose-additive approach to assessing risks to human infants and children from pesticide exposure. EPA currently assesses human risk of pesticide mixtures for pesticides that share a common mechanism of toxic action (*e.g.*, *N*-methyl carbamates, organophosphorus insecticides, chloroacetanilide and triazine herbicides), as mandated by Food Quality Protection Act. The analysis EPA conducts is predicated on additive toxicity and applies dose-addition to set tolerance limits of pesticide residues on food. For example, the toxic potencies of the OPs are added together to determine pesticide tolerance limits for edible crops. Although additive toxicity is evaluated when determining risk to humans, EPA OPP has yet to apply a similar approach to address cumulative toxicity of pesticides that share a common mode or mechanism of action in the evaluation of terrestrial and aquatic species. That said, the use of dose-addition for mixtures containing acetylcholinesterase-inhibiting pesticides is well established and has been extended to protection of aquatic life (Belden et al 2007).

Dose-addition assumes the cumulative toxicity of the mixture can be predicted from the sum of the individual toxic potencies of each component of the mixture. Within the past decade, government regulatory bodies started to use dose-addition models to predict toxicity for those chemicals that share a common mode of action. In California, the Central Valley Regional Water Quality Board (CVRWQB) used dose-addition models (based on the toxic-unit approach) to develop TMDLs for the OP insecticides diazinon and malathion. NMFS Biological Opinions have also recognized the environmental reality of co-occurring pesticides in species' aquatic habitats and applied additive toxicity models to predict potential responses of salmonids (NMFS 2004, NMFS 2005d, NMFS 2005e, NMFS 2005f, NMFS 2008c, NMFS 2009b),

In salmon, dose-additive inhibition of brain AChE activity by mixtures of OPs and carbamates was demonstrated *in vitro* (Scholz et al 2006). More recently, it has been found that salmonid responses to OP and carbamate mixtures vary *in vivo*; some interactions were synergistic, rather than just additive (Laetz et al 2009).

We used the dose-addition method to predict responses by applying the modeling exposure estimates for OPs presented in the *Exposure Section*. Figure 22 provides an example of the predicted effects on A) AChE inhibition, and (B) survival from exposure to methidathion, phorate, and a binary mixture of the two a.i.s. Based on additivity, the mixture is expected to be more toxic than the individual OPs for both endpoints. Due to the steep slopes of the two dose-response curves, and especially the mortality slope, small changes in concentrations along the linear portion of the curve elicit large changes in observed toxicity. The exposure values represent concentrations from EPA PRZM-EXAMS 60 d average modeling estimates for surface waters (methidathion: one ground application at 1 lb a.i./acre in wheat; phorate: one soil incorporated T-banded application at 1.3 lbs a.i./acre in sweet corn). These estimates assume relatively low application rates compared to current labels; assuming higher rates provides a poor example of mixture toxicity as the individual compounds alone lead to substantial AChE inhibition and mortality. Methidathion is spray-applied by ground and air in a variety of crops at rates of 1 – 10 lb a.i./A. Phorate is applied by ground application and soil incorporated in several crops at rates of 1.3 – 8.0 lbs a.i./acre. We recognize that this approach is likely to under-predict toxicity for mixtures that produce synergistic rather than additive responses.

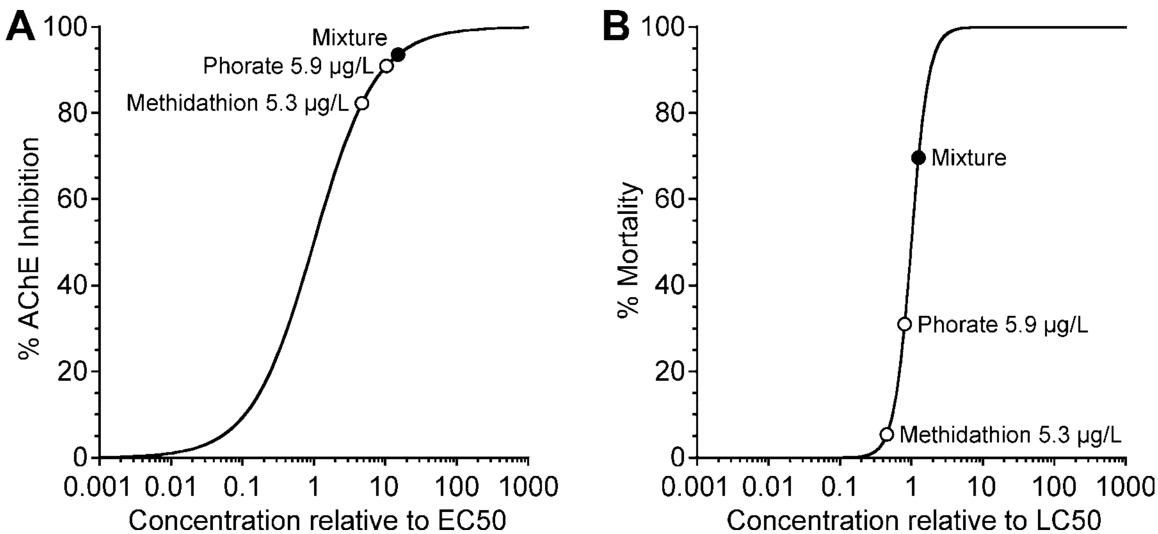


Figure 58 Percent AChE inhibition (A.) and percent mortality (B.) for salmonids expected from exposure to methidathion and phorate as separate constituents and as mixtures (5.3 µg/L and 5.9 µg/L)[10].

We used a variety of exposure estimates to evaluate responses to different mixtures of OPs (Table 121). The predicted additive responses from these mixtures ranged from 93-99% inhibition of AChE and 70-100% mortality. The predicted additive response to AChE inhibition is likely to result in increased AChE-mediated behavioral consequences to salmonids. What is not captured in these responses is the likelihood of exposure to the various mixture concentrations. The PRZM-EXAMS values were estimates selected from EPA simulations of western crops. The scenarios were representative of use rates and numbers of applications on current product labels. Additionally, we used 60 d, time-weighted averages of exposure rather than predicted peak concentrations as exposure to multiple pesticides would be expected to occur more frequently over chronic durations. This may underestimate effects as responses are measured following 96 h exposures. Site specific considerations will also have an influence on the frequency and duration of exposure.

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[10] EPA's default pesticide slope was used for acute mortality (3.63 or probit slope of 4.5) [EPA 2004]. The slope used for AChE inhibition was based on the mean of thirteen cholinesterase-inhibiting insecticides (slope = 0.99); including azinphos methyl, dimethoate, phorate, methidathion, naled, methyl parathion, phosmet, ethoprop, carbofuran, carbaryl, chlorpyrifos, diazinon, and malathion consistent with methods of {Majewski, 2006 #1935;Aston, 1997 #1760}

Table 121. Predicted AChE inhibition and mortality from estimated mixtures of OP pesticides.

a.i.	Concentration (µg/L)	% AChE Inhibition	% Mortality
Modeling: PRZM-EXAMS 60-day averages (from Table 87)			
Methidathion	5.3	82.29	31.08
Phorate	5.9	91	5.38
Additive response		**93.62**	**69.63**
Modeling: GENEEC 90-day averages (from Table 89)			
Potatoes			
ethoprop	120	56.91	0.04
methyl-parathion	38	56.86	0.00
phorate	44	98.67	98.82
Additive response		**98.71**	**98.98**
Oranges			
methidathion	122	99.04	100.00
naled	15	65.51	0.17
phosmet	1.6	33.15	0.00
Additive response		**99.06**	**100.00**
Cherries			
azinphos methyl	17	99.00	99.99
methidathion	63	98.18	99.97
phosmet	0.4	11.17	0.00
Additive response		**99.35**	**100.00**

The GENEEC estimates are 90 d, time-weighted averages that were based on labeled uses of OPs in potatoes, oranges, and cherries. We found no restrictions that would prevent co-application or sequential applications of these compounds or combinations of other AChE-inhibitors. The application rates assumed were consistent with current labels and frequently representative of use rates authorized for many crop and non-crop uses.

Evaluation of Risk Hypotheses:

In this phase of our analysis we examine the weight of evidence from the scientific and commercial data to determine whether it supports or refutes a given risk hypothesis. This is not a statistical analysis, but rather a qualitative weighing of the available lines of evidence. 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 in the analysis. If the evidence supports the hypothesis we determine whether it warrants an assessment at the population level. Although 12 a.i.s are addressed in this Opinion, we recognize the toxicities of these compounds vary widely, and have considered them separately through the analysis. In some cases, a group of compounds may be discussed together in this section if toxicities are in a similar range, or the toxicity/exposure profiles are similar. Although we discuss the risk of fenamiphos to salmonids and their habitat, we do not carry forward a population-level analysis with fenamiphos because its registration has been cancelled, although existing stocks may still be used. EPA estimates there are less than 25,000 lbs of existing stocks of fenamiphos available for future application nationwide.

The available information to characterize pesticide exposure included surface water monitoring data and estimates from pesticide transport models. We combine this information with the distribution and life-history characteristics of listed Pacific salmonids. 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 listed Pacific salmonid habitats. Consequently, models were used to supplement monitoring data and together the information was used to describe the potential range of pesticide concentrations in salmonid habitats. The NMFS AgDrift model runs provided estimates for concentrations resulting from drift to a shallow and narrow body of water, such as those found in floodplain habitats used by listed Pacific salmonids. Small streams and many floodplain habitats 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 habitats used by salmonids, and exposure durations will be reduced in flowing water systems where higher velocities occur. There is uncertainty as to what the magnitude of response of fish and salmonid prey will be under different environmental dissipation patterns. Standardized toxicity tests for pesticide registration are poor predictors of real world aquatic ecosystems as fish and other test organisms are exposed to relatively constant pesticide concentrations for arbitrary durations (*e.g.* acute, 96 h and chronic, 21 d) that may poorly reflect field exposures, which tend to be repeated pulses. The response of fish and their prey to different durations of exposure, and exposure mimicking different environmentally relevant dissipation patterns of the 12 a.i.s is a prominent data gap. For example, within 2 hours of exposure to a carbamate insecticide, juvenile salmonids showed maximum inhibition of brain AChE (Labenia et al 2007). This result illustrates that exposure durations to elicit toxicological responses are chemical specific, depend on toxicokinetic factors, and can occur at durations much shorter than standard toxicity tests. Furthermore, OPs inhibition of AChE can last several weeks prolonging adverse effects well beyond four days due to the irreversible binding of OPs to AChE (Habig andDi Giulio 1991). We therefore did not average exposure concentrations over time, so called time-weighted averages, because adverse responses to short term OP exposures such as pulses would likely be masked.

Large spatial and temporal variability exists in the use of aquatic habitats by listed Pacific salmonids. These differences occur at multiple scales of biological organization (*i.e.*, individual, population, and species). Both an individual's lifestage 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 the stressors of the action for any one individual let alone for a population or species.

Consequently, we used general life history information to evaluate potential exposure in the myriad aquatic habitats. For example, all listed Pacific salmon and steelhead occupy habitats that could contain high concentrations of these pesticides at one or more life stages. That said, populations show temporal variation in use of those habitats. Most species use shallow floodplain habitats and/or small streams during their freshwater and estuarine rearing period. These periods of development and growth can differ significantly between species and populations. Coho, steelhead, and stream-type Chinook spend much longer in freshwater systems prior to migrating to the ocean, while ocean-type Chinook and chum spend less time rearing in freshwater. Ocean-type Chinook migrate from their natal stream within 2-6 months of hatching and spend several months rearing in floodplain, estuary, nearshore habitats before continuing on to the open ocean. Chum spawn in side channels, tributary streams, and mainstem rivers. The egg and alevin life stages reside at these sites until they approach or reach the fry stage. Swim-up fry immediately migrate downstream to estuarine areas, where they typically reside near the shoreline for one or more weeks. Thus, a chum fry's freshwater residency period is only a few days, compared with more than a year for other species such as steelhead.

To account for the temporal and spatial variation of aquatic habitats across individuals, populations, and species, 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. We divided salmonid habitats into two basic groups.

The first group is composed of spawning and rearing habitats. These freshwater aquatic habitats range from first order streams to large mainstem rivers as well as lakes. They are essential to successful reproduction and to the development and growth of young fish.

The second habitat group is composed of migratory corridors, estuaries, and nearshore marine areas. Salmonids use these habitats to migrate and rear (feed, develop, shelter), prior to moving into open ocean areas. In general, pesticide exposure will likely be less intense in these areas compared to freshwater systems given their size, flow, and use by salmonids. Exceptions include estuaries and nearshore marine environments where juveniles are rearing for extended periods (weeks-months) proximate to high pesticide use areas such as agricultural operations in tidal areas and dense urban centers with stormwater runoff during salmonid rearing.

Although we recognize this as an oversimplification of the diversity in life histories as well as aquatic habitats used by listed Pacific salmonids, the framework allows us to evaluate risk hypotheses based on differences in habitats and their use by salmonids. We explicitly address species differences in the *Integration and Synthesis* section by evaluating the potential for the stressors of the action to jeopardize the continued existence of the species. Ultimately, for each of the risk hypotheses we make a determination of whether fitness of individuals is compromised to warrant an analysis at the population level.

Risk Hypotheses

Here we evaluate the available evidence to determine whether each risk hypothesis is supported. If the available information supports a hypothesis, we analyze the effects at the population scale. If the available information does not support a hypothesis, we do not conduct population level analyses.

1. Exposure to azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, methamidophos, methidathion, methyl parathion, naled, phorate, and phosmet is sufficient to:

A. Kill salmonids from direct, acute exposure

Species' life history information indicates that listed salmonids are at the greatest risk of exposure to acutely toxic concentrations of the 12 a.i.s during freshwater occupancy. Salmonids that rear in small streams and floodplain habitats are particularly vulnerable to the highest expected concentrations. We found no survival data comparing the various salmonid lifestage (*i.e.*, eggs, freshwater juveniles, smolting juveniles, returning jacks, and returning adults) for any of the 12 a.i.s. We also located no survival data for estuarine or marine salmonid life stages. The vast majority of lethality data is based on standard toxicity laboratory tests conducted with juvenile salmonids (predominantly rainbow trout) that determine the LC50. These data show the 12 a.i.s have a wide range of LC50s, and the salmonid species tended to be among the most sensitive of the freshwater fish species tested. We relied on these data as well as incident information to evaluate whether expected concentrations of the 12 a.i.s are sufficient to kill individual salmonids.

Of the chemicals assessed, azinphos methyl, methidathion, and phorate are the most toxic based on salmonid survival data, with LC50 ranges of 1.2-27.5 μg/L, 6.6-14 μg/L, and

13-66 μg/L, respectively. We expect concentrations of azinphos methyl, methidathion, and phorate will reach lethal levels based on the range of toxicity and exposure values derived from monitoring data, EPA's modeling estimates, and NMFS modeling estimates. Additional support for acute lethality to fish is found in field incidents where many cases of fish mortality were attributed to azinphos methyl and phorate. The evidence supports evaluating population level consequences from reductions in salmonid survival for azinphos methyl, methidathion, and phorate.

Fenamiphos (LC50 68-563 μg/L), naled (LC50 87-345 μg/L), and phosmet (LC50 150-1,560 μg/L) are also highly toxic[11] to salmonids. Although we did not specifically estimate floodplain concentrations for fenamiphos, as there are no active labels, the application rates described in EPA's BE (EPA 2003d) are sufficiently high to result in concentrations which would cause lethal effects from fenamiphos in floodplain habitats. Additionally, lethal concentrations of fenamiphos have been measured in surface water (up to 520 μg/L) and EPA has characterized fenamiphos as the probable or highly probable cause of death in several fish kill incidents. However, given the lack of currently registered labels we do not evaluate population level effects resulting from reduced survival of individuals.

We also expect concentrations of naled and phosmet to kill juvenile and adult salmon in floodplain habitats and small streams, based on NMFS modeling. We therefore evaluate the effects to populations from exposure to naled and phosmet based on reduced survival.

Bensulide is also categorized as highly toxic[12] to fish, but appears to be less toxic than fenamiphos, naled, and phosmet. However, approved application rates for all uses are

[11] EPA uses a descriptive scale for acute aquatic effects: very highly toxic (LC50 <100 μg/L), highly toxic (LC50 100-1,000 μg/L), moderately toxic (LC50 >1,000-10,000 μg/L), slightly toxic (LC50 >10,000-100,000 μg/L), and practically non-toxic (LC50 >100,000 μg/L), as published in Kamrin 1997.

[12] EPA uses a descriptive scale for acute aquatic effects: very highly toxic (LC50 <100 μg/L), highly toxic (LC50 100-1,000 μg/L), moderately toxic (LC50 >1,000-10,000 μg/L), slightly toxic (LC50 >10,000-100,000 μg/L), and practically non-toxic (LC50 >100,000 μg/L), as published in Kamrin 1997.

high, ranging from 6 to16 lbs a.i./A. The highest application rates, 12-16 lbs a.i./A, are for turf and lawn uses. Turf uses also allow multiple applications, unlike other crop uses. Although bensulide is limited to ground applications, AgDrift estimates indicate lethal concentrations are expected for some aquatic habitats. Runoff is also a likely pathway given the persistence, solubility, and lack of buffer requirements to aquatic habitats. We expect bensulide may kill salmonids in some situations. We therefore evaluate potential population level consequences from reduced survival of salmonids following exposure to bensulide.

Methyl parathion (LC50 1,850-5,300 μg/L), ethoprop (LC50 1,020-13,900 μg/L), dimethoate (LC50 6,200-7,500 μg/L), disulfoton (LC50 1,850-13,900 μg/L), and methamidophos (LC50 25,000-51,000 μg/L) are classified as moderately toxic compounds by EPA, based on lethality tests with salmonids.

We expect concentrations of methyl parathion in salmonid floodplain habitats will reach lethal levels based on NMFS modeling estimates for currently registered uses. Additional support for acute lethality to fish is found in field incidence data where EPA has determined methyl parathion to be the "probable" cause of death in several incidents. We therefore evaluate potential population level consequences from reduced survival of salmonids following exposure to methyl parathion.

We also expect concentrations of ethoprop will reach lethal levels in some salmonid habitats. This conclusion is supported by the occurrence of several fish kill events attributed to ethoprop, including events associated with application methods comparable to current registered uses. We therefore evaluate potential population level consequences from reduced survival of salmonids following exposure to ethoprop.

There is less evidence suggesting dimethoate, disulfoton, or methamidophos concentrations will reach lethal concentrations based on measured and estimated exposure to the single active ingredients. Methamidophos was not reported as the probable cause of death in any of the fish kills in EPA's incident database, and dimethoate and disulfoton are each implicated in a single “probable” fish kill event. The reports for both incidents suggest other chemicals may have been involved. However, it is not known to what degree these compounds contributed to the observed response. We therefore do not evaluate population level responses from reduced survival of salmonids for dimethoate, disulfoton, or methamidophos.

We expect concentrations of some of the OPs in salmonid floodplain habitats will reach lethal levels based on exposure concentrations derived from monitoring data, EPA’s modeling estimates, and NMFS modeling estimates (See *Exposure Analysis*). The youngest swimming salmonids appear to be the most likely to die from short-term, acutely toxic exposures in these habitats. It is less likely that adults would be killed by acute concentrations in most freshwater aquatic habitats compared to juveniles. However, if adults are present in smaller floodplain habitats during spray applications or severe runoff events death is possible, particularly from azinphos methyl, methidathion, and phorate. The available monitoring data, if representative of salmonid habitats, indicated that concentrations rarely achieve LC50 values for most of compounds in freshwaters. However, it is unlikely that peak concentrations are reflected in the monitoring data and none of the sampling targeted real time applications. Given the acutely toxic nature of OPs, a brief exposure would be sufficient to cause effects. As described in the *Exposure Analysis*, monitoring data are limited when compared to the range of habitats used by salmonids. Few data were found that targeted applications and subsequent concentrations in edge of field habitats which typically show much higher concentrations than weekly, monthly, or seasonal monitoring efforts. Although we found no information on egg survival following acute exposures, we do not expect death of eggs from these insecticides as entry into the eggs via the water column is unlikely.

In conclusion, the available information on measured and expected concentrations of the a.i.s and field incident data supports this risk hypothesis for azinphos methyl, bensulide, ethoprop, fenamiphos, methidathion, naled, phorate, phosmet, and methyl parathion. We translate the fitness level consequences of reduced survival from mortality of juvenile salmonids to potential population-level consequences using population models (see

Population modeling) for these a.i.s, except fenamiphos. Fenamiphos was not carried forward for population modeling because of its registration status (no active labels).

B. Reduce salmonid survival through impacts to growth.

Fish growth can be affected by OP chemicals in two ways: by a reduction in somatic processes (inhibition of AChE and other enzymes) and by behavior modifications that reduce foraging (primarily via AChE inhibition). Salmonids are at the greatest risk of reduced growth from pesticide exposure during their juvenile life stage where rapid growth is needed in order to survive. The longer salmonids remain in freshwater the greater the probability for pesticide exposure. Juveniles rearing in estuaries and nearshore environments are also susceptible to growth impacts. For most of the listed salmonid species, but especially stream-type Chinook and coho, extended periods of growth occur in shallow, low-flow habitats, including floodplain habitats and small streams. Time to first feeding is a critical lifestage transition period for all salmonids. Following the adsorption of the yolk sac, fry need adequate prey upon which to feed and the ability to capture them before the onset of starvation. Given effects from OP-inhibited AChE can last from days to weeks, starvation and impaired development is anticipated following exposure, even for fry that migrate to the ocean in a matter of days if they have been exposed during freshwater residency.

We did not identify any studies conducted with the 12 a.i.s that provided a quantitative relationship between growth and fish survival in the lab or field. However, there is abundant ecological literature showing smaller salmonids have reduced first year survival (discussed in *Appendix 1*). Additionally, exposure to sublethal concentrations of other AChE inhibitors (chlorpyrifos and diazinon) for acute durations has been shown to cause reduced feeding success, which in turn reduces growth (Sandahl et al 2005, Scholz et al 2000).

Exposure concentrations will likely vary temporally and spatially for salmonids depending on life history, pesticide use, and environmental conditions. We expect that juvenile fish exposed to azinphos methyl, ethoprop, fenamiphos, methidathion, methyl parathion, naled, phorate, and phosmet during their freshwater residency will feed less successfully, resulting in lower growth rates and reduced size. Available information support likely reductions in growth when salmonids are exposed to more than 0.4 µg/L, 11 µg/L, 7.4 µg/L 12 µg/L, 10 µg/L, 15 µg/L, 4.2 µg/L, and 6.1 µg/L of these chemicals, respectively. We are unable to draw a specific conclusion regarding this risk hypothesis for bensulide or methamidophos, due to lack of data on this endpoint. Inhibition of AChE by methamidophos was not affected at or below 1,000 µg/L, suggesting foraging ability would remain intact for individuals at these concentrations. Given the highest concentrations estimated in floodplain habitats is 490 ug/L, direct effects to growth are not anticipated. Some modeled and measured concentrations of bensulide are high enough to cause lethality, thus we assume that concentrations are also likely to be high enough to affect growth, which typically is affected at lower concentrations than survival for OPs. Dimethoate and disulfoton appear unlikely to cause reduced growth based on data from laboratory assays, but may affect behavior based on inhibition of AChE, leading to reduced foraging ability and ultimately growth.

The weight of evidence supports the conclusion that fitness level consequences from reduced size are likely to occur in rearing salmonids exposed to azinphos methyl, ethoprop, fenamiphos, methidathion, methyl parathion, naled, phorate and phosmet. Fitness level consequences from reduced size may also occur in individual salmonids exposed to bensulide, dimethoate, and disulfoton. Therefore, we address the potential for population-level repercussions due to anticipated reductions in growth. Fenamiphos was not carried forward for population modeling because of its registration status (no active labels).

C. Reduce salmonid growth through impacts on the availability and quantity of salmonid prey

This hypothesis focuses on rearing juveniles and the amount of prey available to ensure adequate growth and ultimately, size. As mentioned previously, habitats most vulnerable to pesticide contamination are shallow, low flow habitats where salmonids congregate to feed on a variety of terrestrial and aquatic invertebrates. Other aquatic habitats used by rearing salmonids are also vulnerable to reductions in prey, including channel edges along larger streams, rivers, estuaries, and nearshore marine areas.

We address several lines of evidence to determine the likelihood of reduced salmonid growth from impacts to aquatic invertebrate prey. The first line of evidence we evaluated is whether salmonid prey items are sensitive to acute and chronic exposures from expected concentrations of the 12 a.i.s. This primarily involved evaluating laboratory experimental results reporting on acute toxicity of the pesticides to aquatic invertebrates. Multiple survival estimates were available for all chemicals, and ≥10 data points were available for 7 of the 12 a.i.s evaluated (azinphos methyl, dimethoate, disulfoton, fenamiphos, methyl parathion, naled, and phosmet). Bensulide, ethoprop, methamidophos, methidathion, and phorate have smaller datasets, with acute data sets of 4, 2, 7, 5, and 9, respectively. Data for longer tests evaluating reproduction and growth were also available for all chemicals. Aquatic invertebrates were sensitive to all of the pesticides. Five a.i.s had EC50s of < 1 μg/L for one or more species (azinphos methyl, methamidophos, methyl parathion, naled, and phorate). For 4 others (disulfoton, fenamiphos, methidathion, and phosmet) the most sensitive invertebrate EC50s were in the 1-10 μg/L range. Bensulide, dimethoate, and ethoprop were slightly less toxic, with the most sensitive invertebrate EC50s in the 40-65 μg/L range. All a.i.s are classified by EPA as very highly toxic to aquatic invertebrates (EC50 <100 μg/L). In some cases, there was a large range of sensitivity for various organisms, but high value salmonid prey (mayflies, stoneflies, cladocerans) were generally at the more sensitive end of the distribution. For all cases, there were overlaps and/or exceedances in measured and estimated concentrations for the a.i.s and the assessment endpoints. We expect death and a variety of sublethal effects to salmonid prey items to occur when exposed to any of the OPs singly. Co-occurences of the a.i.s (mixtures) are anticipated to result in greater toxicity due to dose addition. Sequential exposures to the same a.i. applied multiple times, or different a.i.s applied within the same time frame may also cause a prolonged reduction in prey availability. We expect death as well as a variety of sublethal effects to salmonid prey items from use of all a.i.s evaluated in this Opinion.

The second line of evidence evaluated is whether field-level reductions in aquatic invertebrates correlate to use of insecticides addressed in this Opinion. Although we located few microcosm or community studies for the pesticides evaluated in this Opinion, data from other pesticides with the same mode of action (OP and carbamate AChE inhibitors) show marked reductions in prey abundance and changes in community composition in waterbodies receiving runoff or spray drift from OP applications. Schulz (2004) found that azinphos methyl and methyl parathion caused "clear" effects on aquatic invertebrates in receiving waters in a review of field studies. He also noted these types of effects for other OP chemicals. Effects of OPs generally included reductions in abundance and AChE activity, direct mortality, increased drift, and reduced community diversity. Insecticides, particularly OPs and carbamates, can trigger catastrophic drift of salmonid prey items (Courtemanch and Gibbs 1980, Davies and Cook 1993, Hianes 1981, Hatakeyama et al 1990, Schulz 2004). Available literature from field experiments indicates that populations of aquatic insects and crustaceans are likely the first aquatic organisms impacted by exposures to OPs and other AChE-inhibiting insecticides. Shifts in benthic communities - from sensitive mayfly, stonefly and caddisfly taxa, the preferred prey of salmonids, to worms and midges, occur in areas with degraded water quality (Cuffney et al 1997, Hall et al 2006). Reduced salmonid prey availability correlated to OP use in salmonid bearing watersheds (Hall et al 2006). Recovery of salmonid prey communities following acute and chronic exposures to AChE-inhibiting compounds depends on the organisms' sensitivity, life stage, and length of life cycle, among other characteristics. Univoltine species will take longer than multivoltine species to recover (Liess and Schulz 1999). Recovery of high quality salmonid prey items such as caddisflies, stoneflies, and mayflies will be slow, given they have long life cycles and infrequent reproduction.

The third line of evidence we evaluated was whether salmonids showed reduced growth in areas of low prey availability. An evaluation of this line is complicated by multiple factors affecting habitat quality (*e.g.*, water quantity, quality, temperature, riparian zone condition, *etc*.), which in turn affects prey items and salmonids. We were unable to locate information attributing reduced growth in salmonids to prey reduction caused by specific insecticide exposure, as most studies focused on measuring direct effects on salmonids or direct effects on invertebrates (see review by Schulz 2004). However, there are multiple field experiments and studies that demonstrate reduced fish growth resulting from reduced prey availability (Baxter et al 2007, Barzner and kline 1990, Metcalfe et al 1999) or document fish growth rates below maximal potential growth rates when prey are limited (Dineen et al 2007).

One study, in particular, tested the hypothesis that single applications of the OP insecticide chlorpyrifos (0.5, 5, 20 μg/L) to outdoor ponds (littoral enclosures) would reduce the abundance of invertebrates and cause diet changes resulting in reduced growth rates of juvenile fish (Brazner and Kline 1990). The results are direct, empirical evidence supporting this risk hypothesis. Growth rates of fathead minnow larvae were reduced significantly in all chlorpyrifos-containing treatments due to reduction in prey abundance. At 15 d post-treatment, the reductions in growth rate compared to control fish were the most pronounced and coincided with the greatest reductions in invertebrates. Stomach contents of minnows were identified throughout the experiment. By day 7 mean numbers of protozoans, chironomids, rotifers, cladocerans, mean total number of prey being eaten per fish, and mean species richness were greater in unexposed treatments than in those exposed to chlorpyrifos. On day 15, most of the differences were more pronounced. The results strongly support the conclusion that foraging opportunities were better in untreated enclosures and unexposed larvae grew significantly more compared to chlorpyrifos-treated enclosures. Furthermore, the reductions in prey items in diets mirrored the reduction in prey items in the enclosures. This study supports the hypothesis that reduction in prey abundances translates to reductions in subsequent ration as well as individual growth. The authors concluded that "low levels of contaminants that induce slower growth in young-of-the-year fish through food chain effects or other means may eventually reduce the survival and recruitment of these fish." (Brazner and Kline 1990).

Collectively, the lines of evidence strongly support the overall hypothesis for all a.i.s. We conducted population modeling exercises based on reduced abundances of salmonid prey, presented in the next section (*Effects to Salmonid Populations from the Proposed*

Action). Fenamiphos was not carried forward for population modeling because of its registration status (no active labels).

D. Impair swimming which leads to reduced growth (via reductions in feeding), delayed and interrupted migration patterns, survival (via reduced predator avoidance), and reproduction (reduced spawning success).

Swimming is a critical function for anadromous salmonids. They rely upon swimming to avoid predators, capture prey, migrate, and reproduce, all of which are essential to individual and species survival. When the ability to swim effectively is compromised, direct individual fitness-level consequences result. We evaluate several lines of evidence to determine the potential for the 12 a.i.s to affect swimming of individual salmonids.

The first (and most direct) line of evidence is measured impairment of swimming behaviors following exposure to the 12 a.i.s at concentrations estimated to occur in salmonid habitats. We located no studies that measured swimming behaviors in any fish species following exposures to azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, naled, phorate or phosmet. Methyl parathion, methamidophos, and methidathion showed disrupted fish swimming behaviors. Methyl parathion and methidathion are expected to adversely affect swimming as expected concentrations in salmonid habitats exceed salmonid effect thresholds. Methamidophos caused swimming effects to rainbow trout at concentrations (>16 mg/L), well above those expected in salmonid habitats. Swimming experiments with methidathion indicated that carp behavior was affected at concentrations of 2-6 mg/L (suggesting that carp are fairly tolerant to methidathion). Unfortunately, no experimental data were located on swimming responses of salmonids to methidathion; a recognized data gap. The lowest methidathion LC50 located is 10 µg/L for juvenile rainbow trout and we anticipate effects on swimming behaviors will occur well below the LC50. Swimming-related behaviors are frequently impaired at 0.3 – 5.0% of reported fish LC50s (Little and Finger 1990).

The second line of evidence is whether available information supports that AChE inhibition is expected following exposure to the 12 a.i.s. The rationale for this line of evidence is that inhibition of AChE is significantly correlated to impaired swimming behaviors (Little et al 1990, Sandahl et al 2005, Brewer et al 2001). We found compelling evidence that OPs impair salmonid swimming behaviors while simultaneously showing inhibited AChE activity. The evidence showed measured activity of AChE is reduced in juvenile salmonids following exposures to azinphos methyl, dimethoate, ethoprop, methidathion, methyl parathion, naled, phorate, and phosmet at concentrations expected in salmonid habitats. The pesticides showed a wide range of potency in AChE inhibition. EC50s ranged from 0.57 µg/L for phorate up to 486 µg/L for disulfoton. Although AChE is reduced in juvenile salmonids exposed to disulfoton, the EC50 for AChE inhibition is greater than the highest concentration of disulfoton modeled for floodplain habitats (237 µg/L). Bensulide, fenamiphos, and methamidophos showed no inhibition of AChE in salmonids at the concentrations tested. However, clear signs of neurotoxicity (*e.g.*, swimming erratically, loss of orientation, *etc*.) were evident in fish exposed to fenamiphos and bensulide at concentrations expected in salmonid habitats. No indication of neurotoxicity was observed in fish exposed to methamidophos (up to 1,000 µg/L). We do not expect concentrations of methamidophos to exceed 1,000 µg/L in salmonid habitats given that the highest concentration expected in a floodplain habitat is 490 µg/L.

A third line of evidence we reviewed were studies showing swimming behavior modification following exposure to other AChE inhibitors, including OPs and carbamates, as these pesticide groups share a mode of action. We found compelling evidence that other OPs impair salmonid swimming behaviors at concentrations expected in salmonid habitats. The most sensitive swimming endpoints were those associated with swimming activity rather than measurements of swimming capacity (Little and Finger 1990, Little et al 1990). Irrespective, there are robust data that showed reductions in swimming speed, distance swam, acceleration, predator avoidance, schooling, feeding behaviors, social behaviors, as well as other swimming activities following exposures to malathion, diazinon, chlorpyrifos, carbaryl, and carbofuran. These studies were reviewed and described in detail in NMFS' 2008 and 2009 pesticide Opinions (NMFS 2008c, NMFS 2009b). These results suggest similar responses may occur with exposure to other cholinesterase inhibitors.

The ecological consequences to salmonids from impairment of swimming are myriad. Juvenile salmonids may experience impaired feeding that results in reduced growth and size. Size of individual salmonids during migration to the ocean is a key determinant for survival and ultimately for successful lifecycle completion. Additionally, impaired swimming behavior can lead to increased mortality from predation (Labenia and others 2007). Although we were unable to locate results from field or laboratory experiments for the other remaining endpoints of this hypothesis, we conclude that swimming behaviors will likely be affected by 10 of the OPs (azinphos methyl, bensulide, dimethoate, ethoprop, fenamiphos, methidathion, methyl parathion, naled, phorate, and phosmet), where the magnitude of impaired swimming is dependent on the concentration and exposure duration. When exposure is sufficient, evidence indicates that adverse effects to swimming behaviors are directly attributed to neurotoxicity (and for most of the pesticides via inhibition of AChE), leading to potential reductions in an individual's fitness (i.e., growth, migration, survival, and reproduction). We therefore translate impaired swimming to potential impacts on salmonid populations when concentrations are expected to substantially inhibit AChE activity.

Based on the three lines of evidence evaluated, we conclude that individual fitness of salmonids via swimming-related effects will be compromised from exposure to azinphos methyl, bensulide, dimethoate, ethoprop, fenamiphos, methidathion, methyl parathion, naled, phorate, and phosmet. Therefore, we evaluate whether population level responses are anticipated based on impaired swimming for these a.i.s. In contrast, we do not expect fitness level consequences to salmonids from expected concentrations of disulfoton and methamidophos.

E. Reduce olfactory-mediated behaviors resulting in consequences to survival, migration, and reproduction.

Information reviewed for the Opinion on chlorpyrifos, diazinon, and malathion (NMFS 2008c) showed impaired salmonid olfactory-mediated responses following exposure to diazinon and chlorpyrifos, indicating that as a class of compounds, OPs can impair this salmonid sensory system. However, we located no studies directly measuring olfactory-mediated behavioral responses of fish following exposures to the 12 a.i.s addressed in this Opinion. This recognized data gap introduces uncertainty as to whether the 12 a.i.s impair olfaction and, if so, at what concentrations effects might occur. We did locate one study evaluating olfactory responses from exposure to a mixture of pesticides which included dimethoate and methamidophos. This study demonstrated environmentally realistic concentrations of a mixture of methamidophos, dimethoate, ethyl parathion, chlorpyrifos, diazinon, and malathion compromised juvenile steelhead's ability to detect changes in odorant concentrations (Tierney et al 2008a). Without properly functioning olfaction, behaviors that rely on smell, such as homing and migration may be impaired. Uncertainty remains on the contribution of OPs to the observed toxicity and whether the reduction in olfactory detection results in affected behaviors. Study results from experiments with other OPs (*e.g.*, diazinon) do demonstrate impairment of essential behaviors including homing, predator avoidance, reproductive priming, and milt production (Scholz et al 2000). Considering that data are unavailable for these 12 a.i.s while data on other OPs support this hypothesis, and giving the benefit of the doubt to the species, NMFS assumes that these effects may occur with some of the 12 a.i.s at expected environmental concentrations. Therefore, we evaluate qualitatively how populations may respond when individuals show impaired olfactory–mediated behaviors for all a.i.s. Fenamiphos was not carried forward for to the population analysis because of its registration status (no active labels).

2. Exposure to mixtures of the 12 a.i.s can act in combination to increase adverse effects to salmonids and salmonid habitat.

The exposure and toxicity information we compiled, reviewed, and analyzed support the risk hypothesis, although less mixture data were available for the OPs addressed in this Opinion compared to the OPs addressed in the previous Opinion (NMFS 2008c). Evidence of additive and synergistic effects on survival and AChE inhibition in salmonids were identified. Multiple independent study results supported additive toxicity based on measured AChE inhibition. We therefore conducted an analysis of potential mixtures on the levels of AChE inhibition and the potential for a greater reduction in survival predicated on simple additivity. The analysis showed that both survival and AChE inhibition of individuals is likely affected to a greater degree from exposure to a mixture than from exposure to a single chemical. We also expect assessment endpoints influenced by AChE inhibition are likely affected to a greater degree when in the presence of more than one of the insecticides. Considerable uncertainty arises as to the level of impairment caused by mixtures for some endpoints, as dose responses have not been characterized for some pesticide combinations. We conclude that this hypothesis is supported by the available information and we assess the potential for population-level consequences below.

3. Exposure to degradates of azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, methamidophos, methidathion, methyl parathion, naled, phorate and phosmet, cause adverse effects to salmonids and their habitat.

Ethoprop and methamidophos have no degradates of concern based on the available information. Of the 12 a.i.s addressed in this Opinion, three (disulfoton, fenamiphos, and phorate) form sulfoxide and sulfone degradates, which are more persistent in the environment than the parent, but of a similar acute toxicity based on LC50 data. EPA's EECs for disulfoton and phorate factor in the environmental fate characteristics of the sulfoxide and sulfone metabolites and thus represent the estimated sum concentration of the parent, sulfoxide, and sulfone degradates (TTR). For both of these pesticides, the parent, sulfoxide, and sulfone also form oxons, so there are potentially six toxic compounds in the environment from each application of the parent a.i. EPA did not report data regarding anticipated concentrations of parent, sulfoxide or sulfone oxon forms for disulfoton or phorate, and they were not included in the TTR. Six of the other a.i.s (azinphos methyl, bensulide, dimethoate, methidathion, methyl parathion, and phosmet) also form oxons in the environment. EPA reported that <10% of applied azinphos methyl, methidathion, methyl parathion, and phosmet form oxons in soils, while >10% of the applied bensulide is converted to an oxon degradate. The percentage of dimethoate converted to the oxon was not reported.

Oxons are the metabolically activated form of the OPs, and once inside the organism, are more toxic than the parent. Oxon toxicity is estimated to be 10 to 100 times the parent, based on mammalian data (EPA 2006b). Data for only two a.i.s (dimethoate and methyl parathion) were available to evaluate whether oxons are also of greater toxicity when aquatic organisms are exposed to them in the water column. For dimethoate and the dimethoate oxon, available data included LC50s for rainbow trout 6,200-7,500 μg/L (dimethoate, n=2) versus 9,100 μg/L (oxon, n=1), EC50s for *D. magna* 3,320-5,040 μg/L (dimethoate, n=2) versus 22 μg/L (oxon, n=1), and survival and growth rate NOAEC/LOEAC values for *D. magna* of 40/100 μg/L (dimethoate, n=1) versus 42/140 μg/L (oxon, n=1). The methyl paraoxon EC50 for *D. magna* (n=1) was 2.3 μg/L, compared to the methyl parathion EC50s of 0.14-2.6 μg/L (n=3) for *D. magna* and *C. dubia*. Although the difference in the *D. magna* EC50s for dimethoate and the oxon was extreme (>2 orders of magnitude), other data indicate similar ranges of toxicity for the parent and the oxon. Few data were available regarding toxicity via this route of exposure for other OPs not evaluated in this Opinion.

Based on data we reviewed, persistence of oxons in the water column is not well-understood, although measurable quantities have been detected for some OPs. Oxons may also form during atmospheric transport, and be found in waters distant from the application site. In absence of better data and to give benefit of doubt to the species, we assume the mammalian toxicity relationships apply. Roughly, this means that some percentage of the EEC for oxon-forming compounds is 1-2 orders of magnitude more toxic than the parent. The oxons of azinphos methyl, bensulide, dimethoate, disulfoton, methidathion, methyl parathion, phorate, and phosmet may affect individual fitness of the salmon either by effects directly on the fish, or indirectly by prey reduction. We carry this forward from the individual-level analysis to the population-level analysis.

Potential effects associated with the disulfoton and phorate sulfoxides and sulfones have been considered by EPA and NMFS via their inclusion in the Total Toxic Residues (TTR). Methyl parathion degrades to 4-nitrophenol, but based on parent EECs and available toxicity data showing toxic effects occur at concentrations approximately 3 orders of magnitude greater than the parent, it does not appear sufficient quantities would be formed to cause an effect. None of these degradates appear likely to cause effect on individual fitness not accounted for in the analysis of the parent compound, and are not carried forward to the population-level analysis.

Parent fenamiphos, and the sulfoxide and sulfone degradates do not form oxons. However, fenamiphos EECs do not include the sulfoxide and sulfone degradates, so EECs of fenamiphos and its degradates are likely underestimated. Based on toxicity data for other sulfoxides and sulfones, and the fact degradation to sulfoxide and sulfone does not affect the toxic moiety (the phosphorothione), we assume the fenamiphos sulfoxide and sulfone will have a toxicity similar to the parent. Although no fate data were provided, we also assume fenamiphos sulfoxide and sulfone will be more persistent in the environment, based on fate properties of the disulfoton and phorate sulfoxides and sulfones. We believe the sulfoxides and sulfones may affect individual fitness. Available data indicate phenolic derivatives of fenamiphos, fenaminphos sulfoxide, and fenamiphose sulfone are 2-4 orders of magnitude less toxic than parent, sulfoxide, and sulfone compounds (Caceres et al 2007, Caceres et al 2008). Fenamiphos was not carried forward for population-level analyses because of its registration status (no active labels).

Naled degrades to form dichlorvos, another registered pesticide, and dichloroacetic acid (DCAA). Approximately 20% of naled degrades to dichlorvos (EPA 2008g) and while dichlorvos was not included in EECs in the salmonid BEs, it was included in the EECs for the California red-legged frog, which NMFS has considered in this Opinion. Dichlorvos is more likely to occur in aquatic systems than naled, as it is more resistant to hydrolysis and photolysis than parent naled. Based on available LC50 data, dichlorvos is similar in toxicity to salmonids, but approximately an order of magnitude more toxic to aquatic invertebrates such as *Daphnia* (Table 43). We carry this uncertainty forward from the individual-level analysis to the population-level analysis. Based on the information we located on DCAA, effects were noted in zebrafish embryos at ~4,000 mg/L and in aquatic macrophytes at 1-10 mg/L (Table 44). NMFS did not locate information regarding percentage of the parent that degrades to DCAA, but parent EECs are all <1 mg/L. We anticipate DCAA will be formed at concentrations well below toxic levels, therefore we do not carry it forward to the population-level analysis.

4. Exposure to other stressors of the action including additional active ingredients, and inert/other ingredients in pesticide products and tank mixes cause adverse effects to salmonids and their habitat.

In addition to exposure to the a.i.s, 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, as well as adjuvants, such as 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 of the various compounds will cause them to move through the environment at different rates and partition into different compartments. We expect some percentage of these other stressors will be present in 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 response than laboratory animals exposed only to one a.i, thus available toxicity data generally underestimate the response in a field-applied pesticide mixture.

4.a. Additional Active Ingredients

Three of the active labels reviewed for this Opinion contained multiple a.i.s. While the a.i.s will move through the environment at different rates, it is reasonable to believe that these a.i.s and the OP in the product will co-occur in receiving waters, especially from drift deposition and in the first runoff from the field following application. Some of these a.i.s may be as acutely toxic as the a.i.s specifically considered in this Opinion[13], although they may have a different mode of action. Others may not be as acutely toxic, but may cause reproductive effects, bioaccumulate, or otherwise adversely affect the salmon or their environment in some way.

[13] EPA uses a descriptive scale for acute aquatic effects: very highly toxic (LC50 <100 μg/L), highly toxic (LC50 100-1,000 μg/L), moderately toxic (LC50 >1,000-10,000 μg/L), slightly toxic (LC50 >10,000-100,000 μg/L), and practically non-toxic (LC50 >100,000 μg/L), as published in Kamrin 1997.

Two bensulide products also contain oxadiazon (an oxidiazolone herbicide) which is highly toxic to fish and moderately toxic to aquatic invertebrates (EPA 2003j). One disulfoton product contained both pentachloronitrobenzene (an organochlorine fungicide, highly toxic to fish and aquatic invertebrates, strong potential to bioaccumulate) (EPA 2006o) and etridiazole (a thiazole fungicide, moderately toxic to fish and aquatic invertebrates) (EPA 2000). Azinphos methyl, bensulide, dimethoate, disulfoton, methyl parathion and phosmet labels all have recommendations for one or more tank mixtures that contain additional a.i.s (Table 109). While it is less certain these a.i.s will co-occur in the water than it is for multiple a.i. products, it is reasonable to assume it will occur upon occasion. Specific interactions between additional a.i.s in products and tank mixes and the OPs addressed in this Opinion are unknown, but it is reasonable to assume toxicity of the OPs may be enhanced. In general, exposure to other active ingredients in pesticide products and tank mixes is expected to increase adverse effects to salmonids and their habitat.

4. b. Inert/other ingredients

Labels for bensulide, dimethoate, disulfoton, ethoprop, methidathion, methyl parathion, naled, and phosmet list aromatic solvents, xylene range solvents, and petroleum distillates as other ingredients (Table 107). PAHs (in some aromatic solvents) are frequent aquatic contaminants, and some have been linked to carcinogenic and immunogenic effects. Other aromatic solvents may have a narcotic effect. "Mixed xylenes and xylene isomers are moderately to highly toxic to aquatic species" (EPA 2005). EPA's assessment of aliphatic solvents (which would include some petroleum distillates) indicated that they are generally not acutely toxic to fish, but may adversely affect aquatic invertebrates (EPA 2007c). Toxic effects vary dependent on the specific chemical. As with tank mixes, 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 following applications.

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 for addition 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 the nonylphenol polyethoxylates, which have been linked to endocrine disruption and were addressed at length in previous Opinions on EPA pesticide registrations (NMFS 2008c, NMFS 2009b). There are however, myriad others, some of which may increase the toxicity of the a.i.s. The majority of a pesticide formulation is often composed of inert ingredients. For example, bensulide formulations considered in this Opinion are >90% inert ingredients. Consequently, salmonid exposure to these ingredients may be greater than exposure to the assessed active ingredient. 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 in a qualitative sense. 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 the additional a.i.s and inert/other ingredients in pesticide formulations as well as tank mixes. These stressors of the action are an important consideration when assessing potential effects on listed salmonids and their habitat. Thus, to provide the benefit of the doubt to the species, we therefore 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 the 12 a.i.s to increase effects to salmonids and their habitat.

The available toxicity and exposure data support the hypothesis. Other OPs and carbamates found in the action area likely result in additive or synergistic effects to exposed salmonids and aquatic invertebrates. In particular, methamidophos and naled are associated with other OP pesticides currently registered in the U.S. but not included in this action. Methamidophos is a degradate of acephate, and naled degrades to dichlorvos. Thus, exposure to acephate and diclorvos are expected to cause additive effects. The magnitude of effects will depend on the duration and concentrations of exposure. Effects of exposure to other pesticides is 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 OPs in general and in particular for the 12 a.i.s assessed in this opinion. Multiple experimental results from separate studies indicated that increases in temperature resulted in lower LC50s for fish, including salmonids (Mayer and Ellersieck, 1988). The phosmet BE reported a positive correlation between temperature and phosmet-induced mortalities. Acute lethality bioassays with OPs showed a distinct, robust relationship between toxicity (measured by 96 h LC50s) and temperature (Mayer and Ellersieck, 1988). The experiments were conducted with several species of fish and OPs including bluegill sunfish (phosmet, parathion, malathion, trichlorfon), rainbow trout (phosmet, chlorpyrifos, trichlorfon), yellow perch (azinphos methyl), Atlantic salmon (trichlorfon), and brook trout (trichlorfon). We also reviewed studies showing increases in toxicity to aquatic invertebrates as temperature rises. In aggregate, these data support the hypothesis and we therefore carry forward temperature effects when we discuss effects to salmonid populations.

A summary of effects on individual fitness for each of the a.i.s is presented in Table 122. Fenamiphos is not included because of its registration status (no active labels).

Table 122 Summary of individual-based risk hypotheses.

Risk Hypotheses	Is individual fitness of exposed salmonids compromised?										
	Azinphos methyl	Bensulide	Dimethoate	Disulfoton	Ethoprop	Methamidophos	Methidathion	Methyl parathion	Naled	Phorate	Phosmet
1.A. Kill salmonids from direct, acute exposure	Yes	Yes	No	No	Yes	No	Yes	Yes	Yes	Yes	Yes
1. B. Reduce salmonid survival through impacts to growth	Yes	Yes	Yes	Yes	Yes	No	Yes	Yes	Yes	Yes	Yes
1. C. Reduce salmonid growth through impacts on the availability and quantity of salmonid prey	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
1. D. Impair swimming which leads to reduced growth (reductions in feeding), delayed and interrupted migration patterns, survival (reduced predator avoidance), and reproduction (reduced spawning success)	Yes	Yes	Yes	No	Yes	No	Yes	Yes	Yes	Yes	Yes
1. E. Reduce olfactory-mediated behaviors resulting in consequences to survival, migration, and reproduction.	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
2. Exposure to mixtures of the 12 a.i.s can act in combination to increase adverse effects to assessment endpoints	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
3. Exposure degradates cause adverse effects to salmonids and their habitat.	Yes (oxon)	Yes (oxon)	Yes (oxon)	Yes (oxon)	No	No	Yes (oxon)	Yes (oxon)	Yes (DDVP)	Yes (oxon)	Yes (oxon)
4. Exposure to other stressors of the action including additional a.i.s and inert/other ingredients in formulations and tank mixes cause adverse effects to salmonids and their habitat.	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
5. Exposure to other pesticides present in the action area can act in combination with the 12 a.i.s to increase effects to salmonids and their habitat.	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
6. Exposure to elevated temperatures can enhance the toxicity of the stressors of the action	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes

Salmonid Population Models

We selected four life-history strategies to model (*Appendix 1*). We ran life-history matrix models for ocean-type and stream-type Chinook salmon (*O. tshawytscha*), coho salmon (*O. kisutch*), and sockeye salmon (*O. nerka*). We did not construct a steelhead (*O. mykiss*) life-history model due to the lack of demographic information. Stream-type Chinook salmon were used a a surrogate for steelhead. Chum salmon (*O. keta*) were omitted from the growth model exercise because they migrate to marine systems soon after emerging from the gravel and the model assesses early life stage growth effects over a minimum of 140 d in freshwater systems. If we anticipated chum would be exposed to the a.i.s in their estuarine rearing environment, we considered model output for the other species. The basic salmonid life history we modeled consisted of hatching and rearing in freshwater, smoltification in estuaries, migration to the ocean, maturation at sea, and returning to the natal freshwater stream for spawning followed shortly by death. For specific information on the construction and parameterization of the models see Appendix 1.

Effects to salmonid populations from death of sub-yearling juveniles

An acute toxicity model was constructed that estimated the population-level impacts of sub-yearling juvenile (referred to as juveniles within this section) mortality resulting from exposure to concentrations of the single active ingredients azinphos methyl, bensulide, ethoprop, methidathion, methyl parathion, naled, phorate, and phosmet. Dimethoate, disulfoton, and methamidaphos were not run because we do not expect death of juveniles from short term exposures to EECs. Fenamiphos exposures were not used in the acute toxicity models because there are no current active labels. The acute toxicity models excluded sublethal and indirect effects of the pesticide exposures and focused on the population-level outcomes resulting from an annual 96 hr exposure of all juveniles in the population to single exposures of the active ingredients. This duration was chosen because it is consistent with exposure required in standardized fish LC50 toxicity tests. Fish survival response to other exposure durations is largely uncertain. Death of juveniles was implemented as a change in first-year survival rate for each of the salmon life-history strategies modeled. We also evaluated potential population responses resulting from mixture toxicity utilizing EPA exposure estimates and assuming a single exposure to a binary combination of phorate and methidathion. Finally, we evaluated population level responses resulting from varying the proportion of the population exposed to a single event equivalent to the 96 hr LC50.

Acute exposure to a single active ingredient. The percent changes in the intrinsic population growth rates (lambdas) increased as concentrations of the OPs increased. Increases in direct mortality during the first year of life produced large impacts on the population growth rates for all four life-history strategies. Model results for stream-type Chinook salmon showed significant impacts at lower concentrations than the other modeled populations. This result is primarily due to the lower size of the standard deviation of the data used to parameterize the unexposed population for stream-type Chinook compared to data available for the other life-history strategies. Percent changes in lambda were deemed significant if they were outside of one standard deviation from the unexposed population. The relative sensitivity of the life-history models producing the greatest to the least changes in population growth rate for equivalent impact on survival rates was coho salmon, ocean-type Chinook salmon, stream-type Chinook salmon, and sockeye salmon. We note that the choice of LC50 and related slope values are important drivers for these results. Therefore, an LC50 above or below the ones used here will result in a different dose-response. We selected the lowest reported salmonid LC50 from the available information to ensure that risk is not underestimated. However, if the actual environmental 96 hr LC50 is lower, then the model will under predict mortality. If the actual environmental 96 hr LC50 is higher, then the model will over-predict mortality.

Table 123 Modeled output for ocean-type Chinook salmon exposed to an OP for 96 hrs. Impacted factors including survival (as percent dead), lambda and standard deviation, and percent change in lambda compared to an unexposed population are reported in the bottom 3 rows. The estimated threshold concentration for population level effects is reported in the far right column.

a.i.	Model input concentrations (μg/L)								Threshold for significant change in lambda[b] (μg/L)
Azinphos methyl	0	0.12	0.6	0.96	1.2	1.44	1.8	2.4	0.92
Methidathion	0	0.66	3.3	5.28	6.6	7.92	9.9	13.2	5.06
Phorate	0	1.3	6.5	10.4	13	15.6	19.5	26	10.0
Naled	0	8.7	43.5	69.6	87	104.4	130.5	174	66.7
Phosmet	0	15	75	120	150	180	225	300	115
Bensulide	0	72	360	576	720	864	1080	1440	552
Ethoprop	0	102	510	816	1,020	1,224	1,530	2,040	782
Methyl parathion	0	185	925	1,480	1,850	2,220	2,775	3,700	1,418
% dead	0	0	7	31	50[a]	66	81	93	
Lambda (STD)	1.09 (0.1)	1.09 (0.1)	1.07 (0.1)	0.98 (0.09)	0.89 (0.08)	0.80 (0.07)	0.68 (0.06)	0.53 (0.05)	
% change in lambda[b]	na	na	ns (-2)	-10	-18	-27	-38	-52	

na denotes non applicable, unexposed population; ns denotes values less than one standard deviation of lambda expressed as the percent of lambda. (Calculated value, omitted when less than or equal to one); [a] Model input concentrations in this column represent the lowest 96 hr LC50 for salmonids and the corresponding a.i.; [b]A reduction in lambda of -9.1% is statistically significant.

Table 124 Modeled output for stream-type Chinook salmon exposed to an OP for 96 hrs. Impacted factors including survival (as percent dead), lambda and standard deviation, and percent change in lambda compared to an unexposed population are reported in the bottom 3 rows. The estimated threshold concentration for population level effects is reported in the far right column.

a.i.	Model input concentrations (μg/L)								Threshold for significant change in lambda[b] (μg/L)
Azinphos methyl	0	0.12	0.6	0.96	1.2	1.44	1.8	2.4	0.66
Methidathion	0	0.66	3.3	5.28	6.6	7.92	9.9	13.2	3.61
Phorate	0	1.3	6.5	10.4	13	15.6	19.5	26	7.1
Naled	0	8.7	43.5	69.6	87	104.4	130.5	174	47.6
Phosmet	0	15	75	120	150	180	225	300	82
Bensulide	0	72	360	576	720	864	1,080	1,440	394
Ethoprop	0	102	510	816	1,020	1,224	1,530	2,040	558
Methyl parathion	0	185	925	1,480	1,850	2,220	2,775	3,700	1,012
% dead	0	0	7	31	50 [a]	66	81	93	
Lambda (STD)	1.0 (0.03)	1.0 (0.03)	0.98 (0.03)	0.91 (0.03)	0.84 (0.03)	0.77 (0.02)	0.66 (0.02)	0.53 (0.02)	
% change in lambda[b]	na	ns	ns (-2)	-9	-16	-23	-34	-47	

na denotes non applicable, unexposed population; ns denotes values less than one standard deviation of lambda expressed as the percent of lambda. (Calculated value, omitted when less than or equal to one); [a] Model input concentrations in this column represent the lowest 96 hr LC50 for salmonids and the corresponding a.i.; [b]A reduction in lambda of -3.1% is statistically significant.

Table 125 Modeled output for coho salmon exposed to an OP for 96 hrs. Impacted factors including survival (as percent dead), lambda and standard deviation, and percent change in lambda compared to an unexposed population are reported in the bottom 3 rows. The estimated threshold concentration for population level effects is reported in the far right column.

a.i.	Model input concentrations (μg/L)								Threshold for significant change in lambda[b] (μg/L)
Azinphos methyl	0	0.12	0.6	0.96	1.2	1.44	1.8	2.4	0.69
Methidathion	0	0.66	3.3	5.28	6.6	7.92	9.9	13.2	3.81
Phorate	0	1.3	6.5	10.4	13	15.6	19.5	26	7.6
Naled	0	8.7	43.5	69.6	87	104.4	130.5	174	52.7
Phosmet	0	15	75	120	150	180	225	300	90
Bensulide	0	72	360	576	720	864	1,080	1,440	415
Ethoprop	0	102	510	816	1,020	1,224	1,530	2,040	611
Methyl parathion	0	185	925	1,480	1,850	2,220	2,775	3,700	1,067
% dead	3	0	7	31	50[a]	66	81	93	
Lambda (STD)	1.03 (0.05)	1.03 (0.05)	1.00 (0.05)	0.91 (0.05)	0.82 (0.04)	0.72 (0.04)	0.59 (0.03)	0.43 (0.02)	
% change in lambda[b]	na	na	ns (-3)	-12	-21	-30	-43	-58	

na denotes non applicable, unexposed population; ns denotes values less than one standard deviation of lambda expressed as the percent of lambda. (Calculated value, omitted when less than or equal to one); [a] Model input concentrations in this column represent the lowest 96 hr LC50 for salmonids and the corresponding a.i.; [b]A reduction in lambda of -5.3% is statistically significant.

Table 126 Modeled output for sockeye salmon exposed to an OP for 96 hrs. Impacted factors including survival (as percent dead), lambda and standard deviation, and percent change in lambda compared to an unexposed population are reported in the bottom 3 rows. The estimated threshold concentration for population level effects is reported in the far right column.

a.i.	Model input concentrations (μg/L)								Threshold for significant change in lambda[b] (μg/L)
Azinphos methyl	0	0.12	0.6	0.96	1.2	1.44	1.8	2.4	0.82
Methidathion	0	0.66	3.3	5.28	6.6	7.92	9.9	13.2	4.52
Phorate	0	1.3	6.5	10.4	13	15.6	19.5	26	8.9
Naled	0	8.7	43.5	69.6	87	104.4	130.5	174	59.6
Phosmet	0	15	75	120	150	180	225	300	103
Bensulide	0	72	360	576	720	864	1,080	1,440	493
Ethoprop	0	102	510	816	1,020	1,224	1,530	2,040	699
Methyl parathion	0	185	925	1,480	1,850	2,220	2,775	3,700	1,267
% dead	0	0	7	31	50 [a]	66	81	93	
Lambda (STD)	1.01 (0.06)	1.01 (0.06)	0.99 (0.06)	0.93 (0.05)	0.86 (0.05)	0.78 (0.04)	0.68 (0.04)	0.55 (0.03)	
% change in lambda[b]	na	ns	ns (-2)	-8	-15	-23	-33	-46	

na denotes non applicable, unexposed population; ns denotes values less than one standard deviation of lambda expressed as the percent of lambda. (Calculated value, omitted when less than or equal to one); [a] Model input concentrations in this column represent the lowest 96 hr LC50 for salmonids and the corresponding a.i.; [b]A reduction in lambda of -5.7% is statistically significant.

These results indicate that exposure of salmonid populations to azinphos methyl, bensulide, ethoprop, methidathion, methyl parathion, naled, phorate, and phosmet for 96 hrs at their respective LC50s would result in severe consequences to a population's growth rate. If exposure occurred every year for each new cohort, population growth rate would be reduced and recovery efforts would be slowed. For those natural populations with low abundance or current population growth rates (lambdas) of less than one (decreasing), the risk of extinction would increase substantially, especially if several successive generations were exposed. For each of the combinations of species and OP, we denoted the relative concentration at which the percent change in lambda is deemed significantly different from an unexposed population, (Table 124, Table 125, and Table 126). These population effect thresholds assume exposure to all the juveniles in the population.

These results can be compared to expected concentrations shown in Table 117. Thresholds for azinphos methyl, bensulide, methidathion, naled, phorate, and phosmet were exceeded by one or more of the expected environmental concentrations. Exceedances most frequently occurred with modeled estimates for floodplain habitats. Azinphos methyl, methidathion, and phorate concentration thresholds were exceeded by EPA PRZM/EXAMS estimates, NMFS floodplain estimates, and monitoring data. Model estimates for naled and its toxic degradates also exceed population thresholds some uses (*e.g.*, maximum labeled rated for mosquitos and some crops). Methyl parathion's maximum concentration estimated for floodplain habitats, 980 μg/L, was slightly lower than the concentration at which a significant percent change in lambda is expected (1,012-1,418 μg/L). Expected concentrations for ethoprop showed no exceedances of thresholds, thus for this compound it appears highly unlikely that a single 96 hr exposure would lead to population level consequences based on acute lethality. Other individual fitness consequences will be addressed including the potential for mixtures and other ingredients to result in lethality following acute exposures.

When we compare the population threshold concentrations to concentrations expected in salmonid habitats described in the exposure section, it is likely that some individuals within a population will be exposed during their freshwater juvenile life stage, particularly while using floodplain habitats. The likelihood of population effects from death of juveniles increases for those populations that spend longer periods in freshwaters such as steelhead, stream-type Chinook, and coho salmon. Additionally, individuals that experience elevated temperatures and/or exposures to additional AChE inhibitors are expected to show higher levels of mortality. Reductions in lambda from death of juveniles lead to reduced abundance and productivity, two key parameters in assessing population viability. Consequently, attainment of recovery goals would take longer to achieve for populations with reduced lambdas. Many of the populations that are categorized as core populations, or are important to individual strata, have lambdas just above one, (increasing) and are essential to survival and recovery goals. Slight changes in lambda, even as small as 3-4%, would result in reduced abundances and/or increased time to meet population recovery goals. We apply the results of these modeling exercises to populations comprising ESUs/DPS within the integration and synthesis section.

Acute exposure to pesticide mixtures. The population exercises discussed thus far focused on the effects from exposures to a single OP from one application of a pesticide; however, we know that pesticides are frequently applied multiple times per season and pesticide ingredients often occur together in environmental mixtures. To address the potential population-level effects to environmental mixtures of the OPs, we used estimates of acute mortality from pesticide mixtures generated with the dose-addition model in the previous *Mixtures* section (Table 121). Exposure to the binary mixture of methidathion (5.3 µg/L) and phorate (5.9 µg/L) predict a cumulative mortality of 70% in the exposed population. Based on modeling, these EECs were derived assuming a single ground application at the lower end of the labeled use rates (approximately 1 lb a.i./A), thus they are relevant to a large number of crops and application methods (Table 87). These estimates represent the average concentrations expected to persist for 60 days under the modeled conditions (a static farm pond). We expect dissipation to be more rapid in many habitats used by salmonids and assumed a single acute exposure with a duration of 96 hrs for our simulation given that the rate of dissipation would be greater in many habitats used by juvenile salmonids. Table 127 shows the predicted population response of stream-type Chinook from this level of mortality. Exposure to this scenario showed a 26% reduction in lambda, a severe reduction in the population growth rate compared to the modeled control population. We did not run population simulations for the other mixture combinations assessed in the *Risk Characterization* section as the dose-addition calculations with these mixtures predicted even greater levels of juvenile mortality, indicating greater reductions in population growth rate would be expected.

Table 127 Modeled output for stream-type Chinook exposed to an environmental mixture of phorate and methidathion for an acute duration (96 h). The table denotes the impacted factors of survival as percent dead, lambda and standard deviation, and percent change in lambda compared to an unexposed population.

Methidathion concentration	5.3 µg/L
Phorate concentration	5.9 µg/L
% dead	70
Lambda (SD)	0.74 (0.02)
% change in lambda	-26

Although pesticide mixtures are common in the environment, the likelihood of the modeled scenario occurring is difficult to predict due to the lack of detailed information on watershed characteristics, salmonid presence, the numbers of salmonids exposed, the duration of exposure, and the climatic variables leading to runoff and drift events. The occurrence of the specific modeled scenario may represent an infrequent event, but if a substantial part of a population of listed salmonids is exposed to these mixtures or other comparable mixtures of cholinesterase-inhibiting pesticides, a severe reduction in a population's abundance is expected.

Acute exposure to a segment of the population. So far, we have presented population responses based on every individual of the population being exposed to a given concentration of a pesticide or mixture of pesticides. However, exposure among individuals of a population will vary depending on the spatial and temporal distribution of both individuals and pesticide applications, and differences in the various site- and application-specific conditions that contribute to the likelihood of exposure (*e.g.*, application method and rate, meteorological conditions, soil-type, etc). To address this issue, we evaluated population response by varying the percent of the population exposed to an acute exposure event. Table 128 shows the expected change in lambda for different percentages of a hypothetical stream-type Chinook population exposed to a pesticide at a concentration equivalent to the LC50.

Table 128 Modeled output for different percentages of stream-type Chinook exposed to the LC50 concentration of a pesticide for an acute duration (96 h).

% of stream-type Chinook population exposed	Change in lambda (%)	Significant reduction in lambda (yes or no)
0	0	No
17	-3	Approximate threshold
25	-4	Yes
50	-8	Yes
75	-12	Yes
100	-16	Yes

These results suggest that significant population level effects may occur when roughly 17% or more of a population incurs an acute exposure equivalent to the LC50. In applying these outputs to real world populations caution is needed. Response will vary depending on specific characteristics of the population (*e.g.* survival, reproductive contribution, numbers of individuals within a population, *etc.*). Additionally, individuals of the population will most likely be exposed to a range of concentrations. The likelihood of exposure to concentrations greater than, or less than the LC50 is chemical specific and will vary depending on environmental conditions, application rates and methods, and environmental fate characteristics.

Effects to salmonid populations from reduced size of juveniles due to impaired feeding and reduced abundance of aquatic prey

We developed a second model to evaluate the potential for adverse effects to juvenile growth resulting from exposure to the active ingredients (Appendix 1). The model links AChE inhibition, feeding behavior, prey availability, and somatic growth of individual salmon to the productivity of salmon populations expressed as a percent change in lambda (a population's intrinsic rate of growth). The model scenarios assume annual exposure of sub-yearling juveniles and their prey to the pesticide. Similar to the acute toxicity model, we developed the growth model for four species of salmonids: ocean-type Chinook, stream-type Chinook, sockeye, and coho salmon. The four populations were used to assess the response to a single annual exposure to the active ingredients. We also evaluated population-level effects from repeated pulsed-exposures, and exposure to varying portions of the population with the stream-type Chinook model. Although the available information suggests that impacts to juvenile growth are possible with all 12 a.i.s, the model was not run for bensulide or fenamiphos. Bensulide showed no

AChE inhibition up to 500 µg/L and the Prey Abundance EC50 value (580 µg/L) was close enough to the salmonid LC50 (720 µg/L) that significant acute mortality in salmon would be predicted before effects on the population could occur due to prey loss. Fenamiphos was not evaluated using the population model because there are currently no active labels containing fenamiphos.

We integrated two avenues of effect to juvenile salmonids' growth from exposure to the 10 a.i.s (Appendix 1). The first avenue is a result of AChE inhibition on the feeding success and subsequent effects to growth of juvenile salmonids. Study results with juvenile salmonids show that feeding success is reduced following exposures to AChE inhibitors (Sandahl et al 2005). The second avenue the model addresses is the potential for reductions in juvenile growth due to reduction in available prey. Salmon are often found to be food limited in freshwater aquatic habitats, suggesting that a reduction in prey due to insecticide exposure may further stress salmon and lead to reduced growth rates. Field mesocosm data support this assertion, showing reduced growth of juvenile fish following exposure to the AChE inhibitor, chlorpyrifos (Brazner and Kline 1990). Based on our review of the sensitivities of aquatic invertebrates to the 10 a.i.s, we expect reductions in densities and altered composition of the salmonid prey communities.

Reductions in aquatic prey are included in the model because of the high relative toxicity of pesticides to salmonid prey and the extended duration of effects on prey communities. Juvenile salmonids are largely opportunistic, feeding on a diverse community of aquatic and terrestrial invertebrate taxa entrained in the water column or on the surface (Higgs et al 1995). As a group, these invertebrates are among the more sensitive taxa for which there is toxicity data, but within this group, there is a wide range of sensitivities (Table 113 and Table 114). The 10 a.i.s are toxic to aquatic macroinvertebrates, and concentrations that are not expected to kill salmonids are often lethal for their invertebrate prey (*e.g.*, for methyl parathion, range of LC50s for salmonids = 1,850-5,300 µg/L, vs. the range of EC50s for freshwater, estuarine, and marine invertebrates = 0.14-28 µg/L). In particular, prey items that are preferred by small juvenile salmonids (including midge larvae, water fleas, mayflies, caddisflies, and stoneflies) are among the most sensitive aquatic macroinvertebrates. Effects on the prey community can persist for extended periods of time (weeks, months, years), resulting in effects on fish feeding and growth long after an exposure has ended (Colville et al 2008, Liess and Schulz 1999, Van der Brink et al 1996, Ward et al 1995).

Selection of aquatic invertebrate toxicity values to represent salmonid prey items

The model requires an EC50 for each pesticide (defined as a 50% reduction in the biomass of salmonid prey items) and a corresponding slope (Appendix 1). The term "EC50" will be used in this section to describe short-term survival data for aquatic invertebrates (death and immobility). To determine what levels of the OPs reduce aquatic invertebrate numbers, we reviewed the available field and laboratory studies. We found a wide spectrum of available data for the 10 a.i.s. We did not locate a field study that measured aquatic community response to a range of concentrations of these pesticides. Therefore, we did not select concentration data from field experiments as we did in NMFS' 2008 Opinion on the registration of chlorpyrifos, diazinon, and malathion (NMFS 2008c). Due to the scarcity of data for many of the a.i.s, we did not develop probability plots. Instead, we selected the lowest available survival EC50 for *D. magna* for each a.i. to represent the salmonid prey community EC50 because *D. magna* data were available for all a.i.s.

Table 129 48 h survival EC50s of *Daphnia magna*

Organophosphate	*Daphnia magna* 48 hr EC50 (µg/L) (95% CI)	Data Source
Azinphos methyl	1.13	MRID 00068678
Dimethoate	3320 (1730-4120)	Song, M.Y., J.D. Stark, J.J. Brown. 1997. Comparative Toxicity of Four Insecticides, Including Imidacloprid and Tebufenozide, to Four Aquatic. Environ.Toxicol.Chem. 16(12):2494-2500
Disulfoton	13	MRID 00143401
Ethoprop	44	MRID 00068325
Methamidophos	26 (20-34)	MRID 00041311
Methidathion	3	MRID 42081704
Methyl parathion	0.14 (0.09-0.2)	MRID 40094602
Naled	0.3	MRID BA0NAL02
Phorate	37 (30-44)	MRID 0161825
Phosmet	5.6	MRID 00063194

Data from EPA documents, denoted by MRID unless other wise specified.

A growing number of studies on a variety of insecticides have reported that concentrations well below LC50s can cause delayed mortality or sublethal effects that may scale up to affect aquatic invertebrate populations, especially in scenarios with multiple exposures and/or other stressors. Evidence for ecologically significant sublethal or delayed effects to aquatic invertebrates includes reduced growth rates Forbes and Cold 2005, Schulz and Liess 2001a), altered behavior (Johnson et al 2008) reduced emergence (Johnson et al 2008, Schulz and Liess 2001a), reduced reproduction (Cold and Forbes 2004, Forbes and Cold 2005) and reduced predator defenses (Johnson et al 2008, Sakamoto et al 2006).

Additionally, the available toxicity data – and therefore the data included for these analyses– are from studies using taxa, such as *D. magna*, hearty enough to survive laboratory conditions. Studies specifically examining salmonid prey, which are more difficult to rear in the laboratory, have documented relatively low survival EC50 values when exposed to current use insecticides (Johnson et al 2008).

Modeling availability of unaffected prey

Reductions in benthic invertebrate densities can lead to long-term reductions in prey availability and reductions in fish growth (Davies and Cook 1993). That said, prey densities are not usually reduced to zero (Wallace et al 1989). Therefore, it is assumed that regardless of the exposure scenario, prey abundance would not drop below a specific "floor" of prey availability. This floor is included in the model to reflect an assumption that a minimal yet constant terrestrial subsidy of prey and/or an aquatic community with tolerant individuals would be available as prey, regardless of pesticide exposure and in addition to the constant recovery rate (see below).

Therefore, even in extreme exposure scenarios, some prey will be available, as determined by the value assigned to the floor. In some highly degraded systems this may or may not be the case. No studies have quantified this floor for the purpose of estimating prey availability, but several studies have documented reductions in overall aquatic benthic insect densities of 75-98% (Anderson et al 2003a, Anderson et al 2006a, Wallace et al 1989). Because benthic densities are typically correlated with drift densities (Hildebrand 1974, Waters and Hokenstrom 1980), these reductions likely result in similar reductions of prey. Therefore, assuming there is also some constant rate of terrestrial invertebrate subsidy in addition to a residual aquatic community, a floor of 0.20, or 20% of fish ration, is reasonable. The model does not include any additional impacts to fish via dietary exposure from contaminated prey, or any potential synergistic or additive effects to the aquatic invertebrates that may be result from multiple stressors (Schulz and Liess 2001b).

Modeling spikes in invertebrate drift following insecticide exposure

"Catastrophic drift" of invertebrates, due to acute mortality and/or emigration of benthic prey into the water column is frequently observed following exposure to insecticides (Davies and Cook 1993, Schulz 2004, Schulz and Liess 2001a). Drift rates within hours of exposure can be more than 10,000 times the natural background drift (Cuffney 1984) and fish have been found to exploit this by feeding beyond satiation (Davies and Cook 1993, Haines 1981). The duration and magnitude of the spike in drift of prey is dependent in part on the physical properties and dose of the pesticide; however, the spike is generally ephemeral and returns to natural, background levels within hours to days (Haines 1981, Kreutzweiser and Sibley 1991). Likewise, the magnitude of the spike is dependent in part on the benthic density of prey; the spike in drift from communities that have been reduced by previous exposures is smaller than the spike from previously undisturbed communities (Cuffney 1984, Wallace et al 1991). To reflect this temporary increase in prey availability, the model includes a one-day prey spike for the day following an exposure (Appendix 1). The model also accounts for this short-term increase in prey availability by allowing fish to feed at a maximum rate of 1.5 times their normal, optimal ration.

Modeling recovery of salmonid prey

We selected a 1% recovery in prey biomass per day. Reports of recovery of invertebrate prey populations, once pesticide exposure has ended, range from within days to more than a year (Colville et al 2008, Cuffney 1984, Kreutzweiser and Sibley 1991, Liess and Schulz 1999, Pusey et al 1994, Van den Brink et al 1996, Ware et al 1995). The dynamics of recovery are complicated by several factors, including the details of the pesticide exposure(s) as well as habitat and landscape conditions (Liess and Schulz 1999, Van den Brink et al 2007). In watersheds with undisturbed upstream habitats, recovery can be rapid due to a healthy source of invertebrates that can immigrate via drift and/or aerial colonization (for adult insects) (Heckmann and Freiberg 2005). However, in watersheds dominated by agricultural or urban land uses, healthy upstream or nearby habitats may be limited and consequently, recolonization by salmonid prey is likely reduced Liess and Von der Ohe 2005, Schriever et al 2007). Additionally, many large, high-quality prey take a year or more to develop (Merritt and Cummins 1995) indicating that recovery of biomass (as compared to prey density) is likely a limiting factor (Cuffney 1984). Recovery to pre-disturbance levels is unlikely in aquatic habitats where invertebrate abundances are repeatedly reduced by stressors. We consider a 1% (control prey abundance per day) recovery rate as ecologically realistic to represent recolonization by invertebrates in salmonid habitats (Colville et al 2008, Van der Brink et al 1996, Ward et al 1995).

Growth model results

Exposure to single insecticides for 4-, and 21 day exposure durations

Population model outputs for the four salmon populations are summarized as dose-response curves in Figure 59 - Figure 68. As expected, greater reductions in population growth resulted from longer exposures to the pesticides. For several pesticides, the primary factor driving the magnitude of change in lambda was the Prey Abundance parameter (*i.e.*, the EC50 value for *D. magna)*. The AChE parameter was a secondary factor compared to Prey Abundance for disulfoton, ethoprop, methamidophos, methyl parathion, and naled. This is largely because the salmonid EC50s for AChE were much higher, typically by an order of magnitude, than the prey survival EC50s. However, the salmonid AChE EC50 was a more sensitive parameter for other chemicals; azinphos methyl, dimethoate, methidathion, phorate, and phosmet all showed significant reductions in population growth rates well below their respective Prey Abundance EC50s, due at least partially to reduced feeding activity associated with predicted levels of AChE-inhibition.

Similar trends in effects were seen for each pesticide across all four life-history strategies modeled. This is apparent by the similar shape of the dose-response curves across species. For compounds driven by prey abundance, the curves plateau when there is no more reduction possible in the aquatic community (*i.e.*, when the 20% biomass of the aquatic invertebrate community is reached). Once that plateau is achieved, further reductions in lambda are minimal with increasing concentrations. In contrast, pesticides such as azinphos methyl, whose response was driven by predicted behavioral impacts associated with AChE-inhibition appeared to exhibit more of a linear response. Although these curves would eventually plateau, higher concentrations were not tested because those concentrations would result in significant mortality, and consequently would be more appropriately modeled using the acute toxicity model. The most toxic of the pesticides affected salmon populations at concentrations in the low μg/L range (azinphos methyl, methidathion, methyl parathion, naled, and phorate) while significant decreases in the populations' growth rates occurred at much higher concentrations for less toxic compounds (dimethoate, disulfoton, ethoprop, and methamidophos).

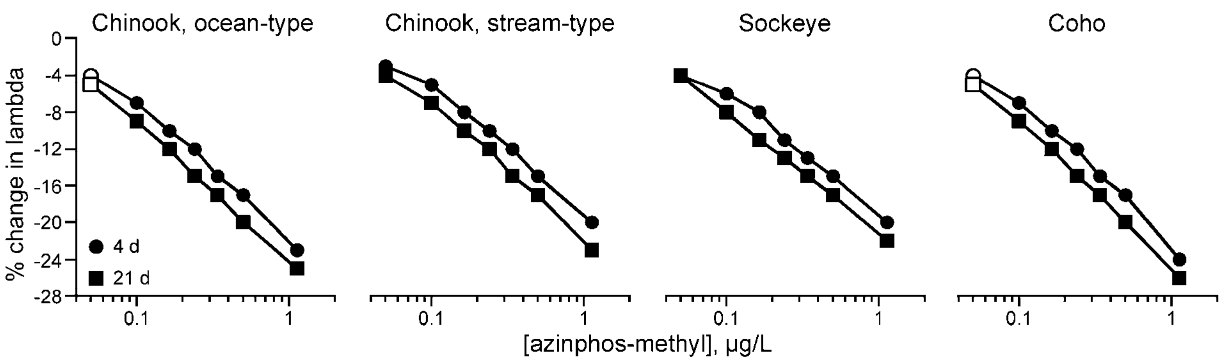

Figure 59 Percent change in lambda for modeled species following 4 d and 21 d exposures to azinphos methyl. Open symbols denote a percent change in lambda of less than one standard deviation from control population. Closed symbols represent a percent change in lambda of more than one standard deviation from control population.

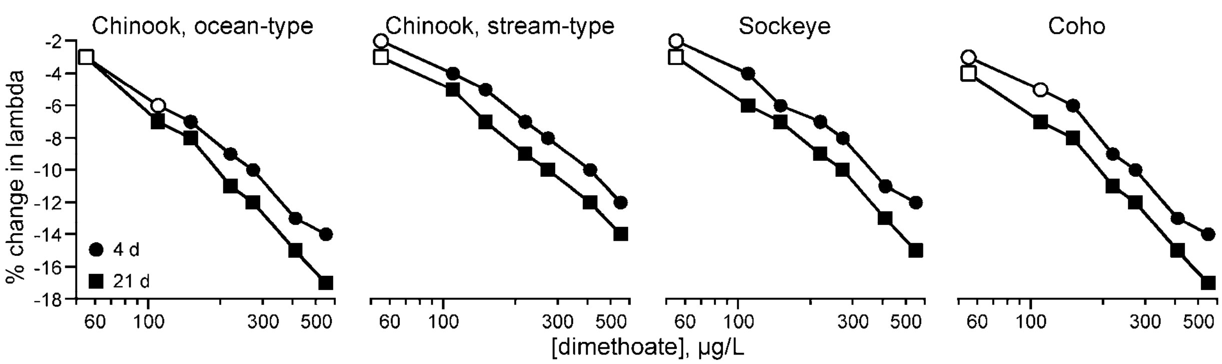

Figure 60 Percent change in lambda for modeled species following 4 d and 21 d exposures to dimethoate. Open symbols denote a percent change in lambda of less than one standard deviation from control population. Closed symbols represent a percent change in lambda of more than one standard deviation from control population.

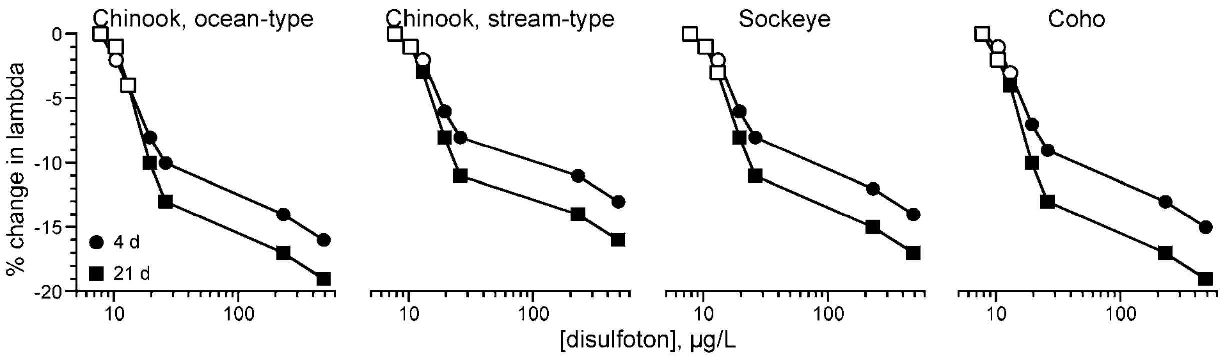

Figure 61 Percent change in lambda for modeled species following 4 d and 21 d exposures to disulfoton. Open symbols denote a percent change in lambda of less than one standard deviation from control population. Closed symbols represent a percent change in lambda of more than one standard deviation from control population.

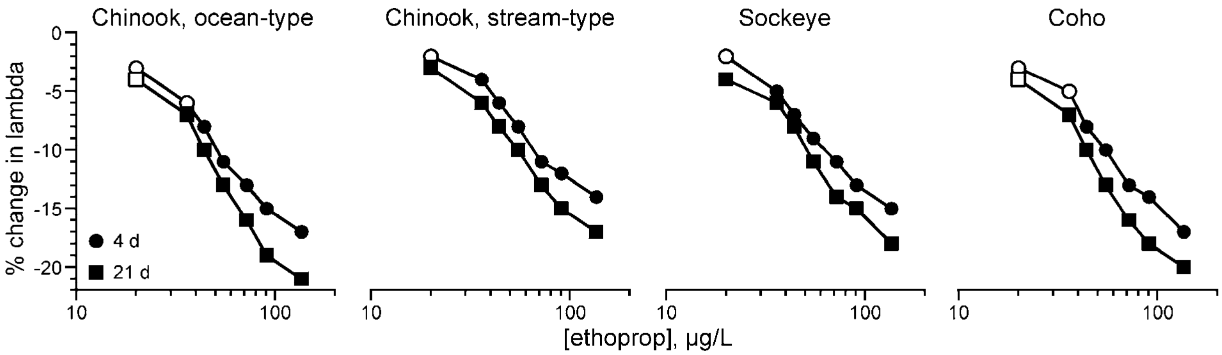

Figure 62 Percent change in lambda for modeled species following 4 d and 21 d exposures to ethoprop. Open symbols denote a percent change in lambda of less than one standard deviation from control population. Closed symbols represent a percent change in lambda of more than one standard deviation from control population.

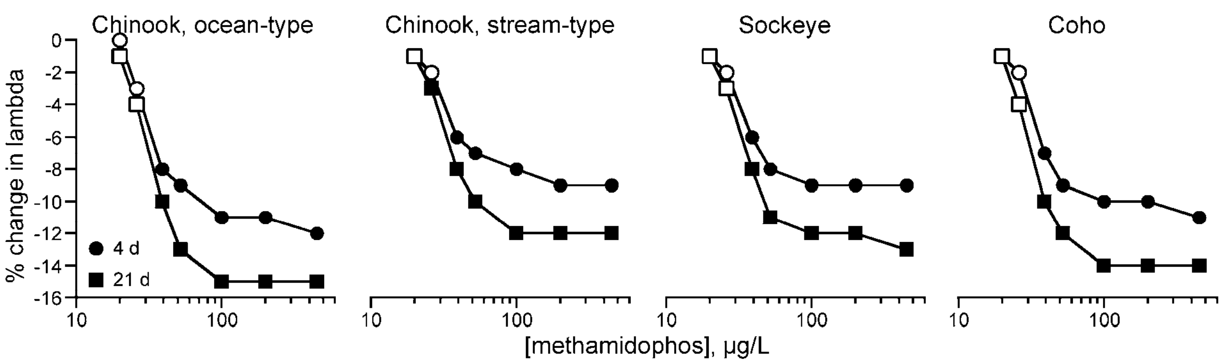

Figure 63 Percent change in lambda for modeled species following 4 d and 21 d exposures to methamidophos. Open symbols denote a percent change in lambda of less than one standard deviation from control population. Closed symbols represent a percent change in lambda of more than one standard deviation from control population.

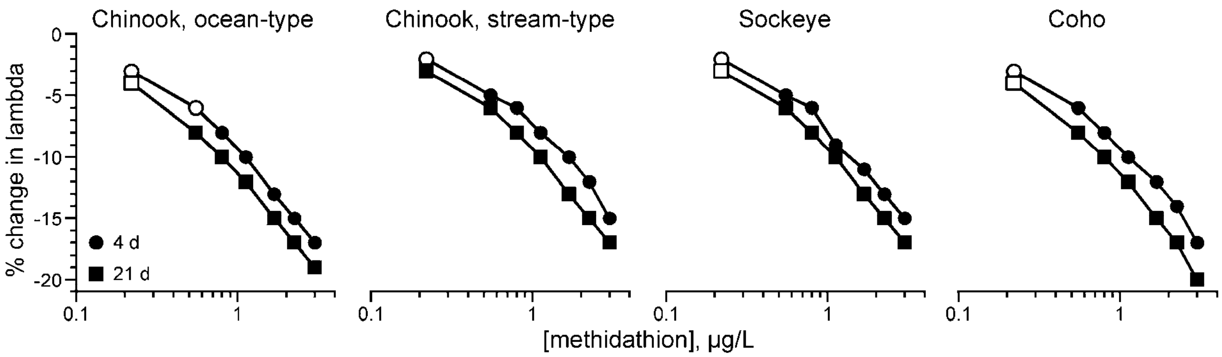

Figure 64 Percent change in lambda for modeled species following 4 d and 21 d exposures to methidathion. Open symbols denote a percent change in lambda of less than one standard deviation from control population. Closed symbols represent a percent change in lambda of more than one standard deviation from control population.

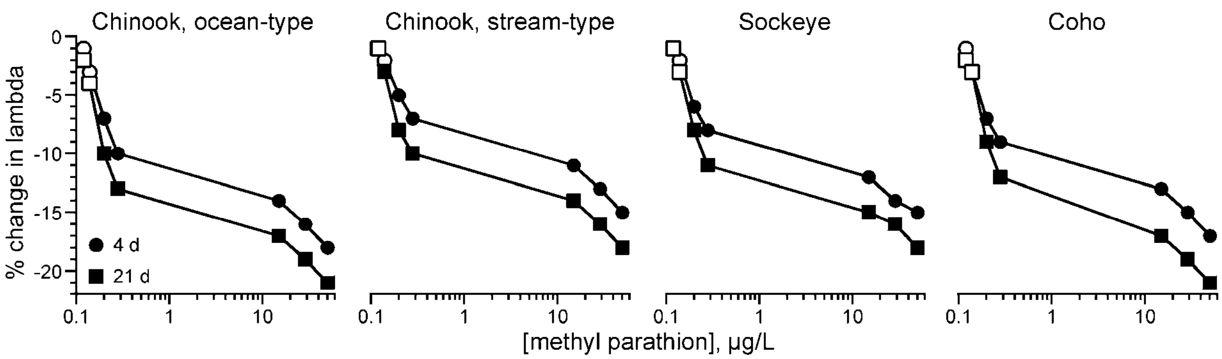

Figure 65 Percent change in lambda for modeled species following 4 d and 21 d exposures to methyl parathion. Open symbols denote a percent change in lambda of less than one standard deviation from control population. Closed symbols represent a percent change in lambda of more than one standard deviation from control population.

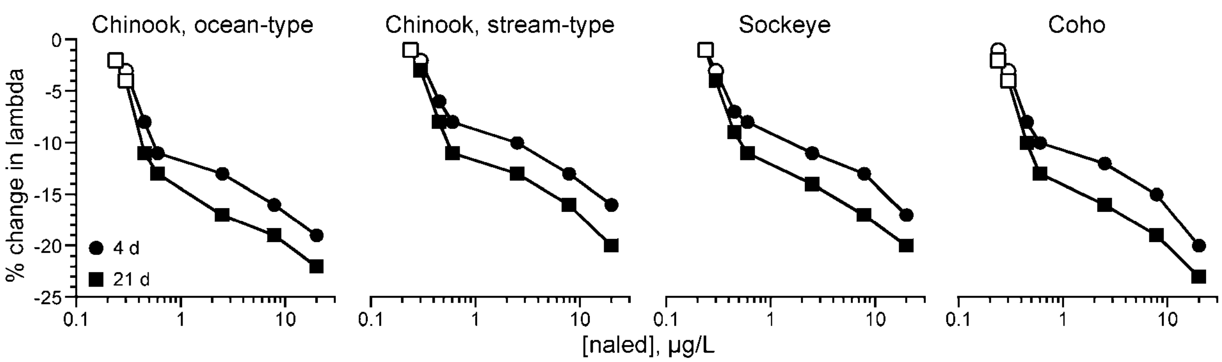

Figure 66 Percent change in lambda for modeled species following 4 d and 21 d exposures to naled. Open symbols denote a percent change in lambda of less than one standard deviation from control population. Closed symbols represent a percent change in lambda of more than one standard deviation from control population.

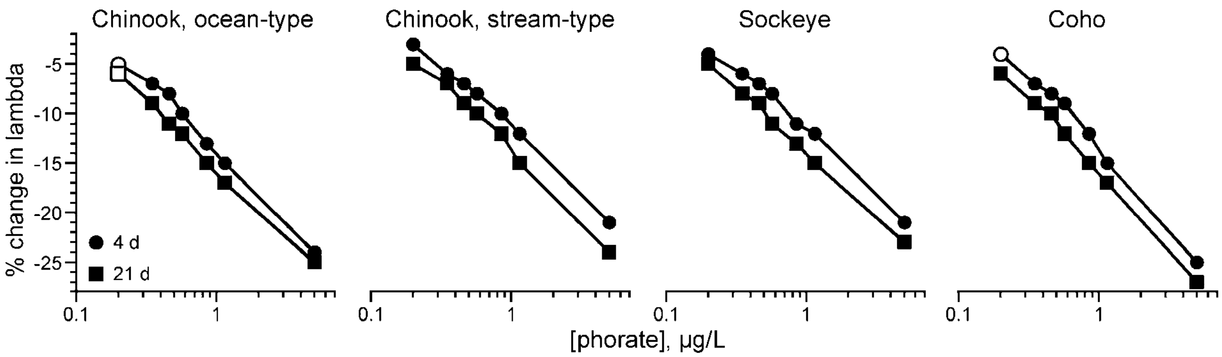

Figure 67 Percent change in lambda for modeled species following 4 d and 21 d exposures to phorate. Open symbols denote a percent change in lambda of less than one standard deviation from control population. Closed symbols represent a percent change in lambda of more than one standard deviation from control population.

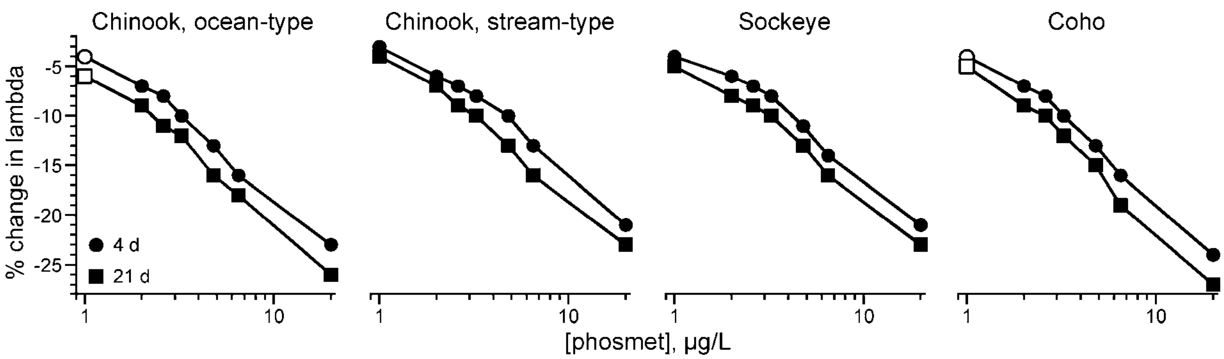

Figure 68 Percent change in lambda for modeled species following 4 d and 21 d exposures to phosmet. Open symbols denote a percent change in lambda of less than one standard deviation from control population. Closed symbols represent a percent change in lambda of more than one standard deviation from control population.

By applying some of these changes in lambda to known threatened and endangered populations' lambdas from Appendix 2, significant reductions in population viabilities are anticipated. For example, if the Puget Sound Chinook salmon Green River population with a lambda of 0.67 is exposed to methyl parathion at 0.28 µg/L for 4 d, a concentration attainable in many salmonid habitats based on monitoring and modeling, we would expect a reduction in lambda by 10% or 7% (Figure 65) depending whether the individuals exhibit ocean-type or stream-type life histories. These reductions would be severe for a population already in decline and are primarily a result of reductions in juvenile growth based on reductions in salmonid prey. Even for those lambdas that are well above one such as Central Valley Chinook salmon Spring Runs' Butte Creek population (lambda = 1.3), reductions of 10% would have major consequences to a population's viability from reduced growth of juveniles and could inhibit the species recovery.

The repercussions to these populations' viabilities are increased with increasing concentrations, durations, multiple applications, and when mixtures are incorporated.

Exposure to multiple applications

So far, we have presented population responses based on a single, annual exposure event. However, some of the OPs addressed in this Opinion (azinphos methyl, dimethoate, methamidophos, methidathion, methyl parathion, naled, and phosmet) are approved for multiple applications per year. Additionally, crops frequently receive applications of several different pesticides during the course of a single growing season, increasing the likelihood of the occurrence of multiple exposures. To evaluate the potential population effects from multiple applications of OPs, we constructed a scenario based on the labeled use of methyl parathion in alfalfa, a crop common in California, Idaho, Oregon, and Washington. Six applications of methyl parathion were assumed at 14-day application intervals. We assumed 6 pulsed acute exposures (96 h) of 1.7 µg/L, the peak concentration reported in CDPR's monitoring database (Table 97 and Table 98). A single exposure to methyl parathion resulted in a significant decrease in lambda (-9%,Table 130). Each successive exposure decreased lambda, with the 6th exposure resulting in a severe decline in the population growth rate (-20%). As discussed with other simulations, the likelihood of this scenario depends on a number of environmental and chemical specific factors. We assumed a relatively low concentration for these simulations to ensure their relevance (*e.g.* EPA PRZM-EXAMS estimates for methyl parathion ranged from 1.3-67 µg/L, NMFS floodplain habitat estimates ranged from 134-980 µg/L).

Table 130 Predicted percent change in lambda for stream-type Chinook exposed to 6 acute pulses of methyl parathion (1.7 µg/L).

Number of Applications	% Change in Lambda	Mean Lambda (sd)
1	-9	0.91 (0.03)
2	-12	0.88 (0.03)
3	-14	0.86 (0.03)
4	-16	0.84 (0.03)
5	-18	0.82 (0.03)
6	-20	0.80 (0.03)

Exposure to a segment of the population

To evaluate the potential for adverse effects to juvenile growth under variable exposure conditions, we evaluated population responses by adjusting the percent of the population exposed to single acute (96h) episode annually. For these simulations we assumed naled concentrations of either 3.4 µg/L or 239 µg/L, representing the range in expected environmental concentrations estimated by EPA for mosquito control applications. These results suggest that exposure of 10 -25% of the population to a single exposure event would cause a significant reduction in the population growth rate of stream-type Chinook (Table 131). As previously discussed, the likelihood of exposure is dependent on several factors. NMFS does not expect frequent exposure at or near 239 µg/L. This estimate was derived assuming direct application of naled to water and would constitute a misuse according to label specifications. However, NMFS estimates suggest that naled concentrations can exceed 90 µg/L in salmon habitat when applied for mosquito control at the maximum labeled rate (1.25 lbs a.i./A) and can exceed 100 µg/L for aerial applications in crops. Naled is approved for use at a large number of sites, increasing the likelihood of exposure (e.g. tidal wetlands, woodlands, swamps, corrals, holding pens, feedlots, pastureland, rangeland, around food processing plants, loading docks, cull piles, refuse areas, in greenhouses and on outdoor-grown ornamentals, fruits, nuts, and field crops). Additionally, naled is approved for repeated applications within most use sites, further increasing the likelihood of exposure to a large segment of the population. For adult mosquito control, some use sites contain no limit on the number of times naled can be reapplied. Others have a seasonal maximum of over 10 lbs a.i./acre (*i.e.* more than 40 applications are allowable on a single site if applied at the recommended rate of 0.25 lbs/Acre). Naled can be applied up to five times or more in most agricultural crops.

Table 131 Growth Model output for different percentages of stream-type Chinook exposed to naled.

% of Stream-type Chinook population exposed	3.4 µg/L (AgDrift Model)		239 µg/L (RICE Model)	
	(%) Change in lambda	Significant reduction in lambda (yes or no)	(%) Change in lambda	Significant reduction in lambda (yes or no)
0	0	no	0	no
10	-1	no	-3	Approximate threshold
25	-3	Approximate threshold	-6	yes
50	-6	yes	-14	yes
75	-8	yes	-21	yes
100	-11	yes	-26	yes

Both the acute toxicity and juvenile growth models produced similar shaped dose-responses among the four life-history strategies given the use of the same input values (*e.g.* exposure, fish LC50, prey EC50, *etc.*). For example, the stream-type Chinook salmon and sockeye salmon models produced very similar results as measured as the percent change in population growth rate. The ocean-type Chinook salmon model produced similar output to the coho salmon model and showed the greatest changes in lambda resulting from the pesticide exposures. When looking for similarities in parameters to explain the ranking, no single life-history parameter or characteristic, such as lifespan, reproductive ages, age distribution, lambda and standard deviation, or first-year survival show a pattern that matches this consistent output **(**Appendix 1**)**. Combining these factors into the transition matrix for each life history and conducting the sensitivity and elasticity analyses revealed that changes in first-year survival produced the greatest changes in lambda. While some life-history characteristics may cause a population to be more vulnerable to a specific effect, the combination of age structure, survival and reproductive rates as a whole strongly influences the population-level response. We discuss the applicability of the population models results for each ESU/DPS within the *Integration and Synthesis* section below.

Population-level consequences from other affected salmonid assessment endpoints and other stressors of the action

In this section we present the population-level consequences from individual effects not amenable to population modeling. In most cases we lack the empirical data to conduct population modeling for these endpoints. Thus, we qualitatively infer population-level responses. We focus on population abundance and productivity, metrics used by NMFS to assess a population's viability. Both can be compromised by the stressors of the action assessed in this Opinion. Individual fitness consequences that reduce survival, growth, reproduction, or migration can lead to reduced salmonid population viability if sufficient numbers of individuals comprising a population are affected, and are more pronounced when individuals are affected over multiple generations. If the reductions in fitness result in reducing a population's survival or recovery potential, then we consider whether specific ESUs or DPSs are affected (See *Integration and Synthesis* section).

With these proposed actions it is difficult to place an exact number on the percentage of a population affected or frequency of effect on the population because of the uncertainty associated with the spatial and temporal uses of the currently registered formulations of azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, methamidophos, methidathion, methyl parathion, naled, phorate and phosmet, compounded by the imperfect data regarding salmonid location at any given time. However, NMFS has sufficient information to make inferences from the available uses, exposure, and response data, on the likelihood of population-level consequences. Below we address whether the fitness level consequences for individuals identified from the risk hypotheses affect the viability of salmonid populations. As mentioned earlier, we focus on the potential for reduced population abundance and productivity.

Impaired swimming and olfactory–mediated behaviors

All life stages of salmonids rely on their inherent ability to swim and to navigate through a variety of habitats over their life span in order to ultimately spawn successfully in natal waters and complete their life cycle. OPs and other AChE inhibitors (carbamates) have been shown to affect swimming and other behaviors at concentrations below those necessary to cause lethal effects (Little and Finger 1990, Little et al 1990). Our previous analysis indicated all of the a.i.s, except disulfoton and methamidophos, may impact swimming of individuals at expected environmental concentrations. Very few data regarding these types of effects were available for the 12 a.i.s addressed in this Opinion. Based on the data available, knowledge of how these classes of compounds affect behavior, and laboratory data linking exposure to OPs with AChE inhibition and impaired feeding (Sandahl 2005) we expect swimming will be impaired by azinphos methyl, bensulide, dimethoate, ethoprop, methidathion, methyl parathion, naled, phorate, and phosmet at concentrations expected to occur in salmon habitats. Specifically, we expect that salmonids with impaired swimming behaviors from AChE inhibition will show reduced feeding, delayed or interrupted migration, reduced survival, and reduced reproductive success. We conclude that anticipated exposures are likely to reduce a population's abundance and productivity as a result of impaired swimming.

No information on olfaction was available for the 12 a.i.s addressed in this Opinion, other than one mixture study that included dimethoate and methamidophos (Tierney et al 2008b). The contribution of these two a.i.s to the effects was not evaluated by the authors, nor were we able to estimate it based on the information presented. However, information reviewed within the first OP Opinion (NMFS 2008c) showed olfactory impairment of salmon by other OPs, such as diazinon and malathion, as well as by carbamates. The specific mode of impairment to olfactory neurons by OPs has not been elucidated and it does not appear to be linked with AChE inhibition, thus we are unable to determine which OPs in this Opinion will impair olfaction. We anticipate some of the a.i.s will cause olfactory impairment, and consequently modify olfactory-mediated behaviors. This is a noted data gap that introduces substantial uncertainty with potential impairment of olfactory-mediated behaviors.

Because olfaction plays an important role in a suite of ecologically relevant behaviors that are affected when an individual salmonid's olfaction is impaired, we include this endpoint in our analysis for azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, methamidophos, methidathion, methyl parathion, naled, phorate, and phosmet. Lack of predator avoidance behaviors by juvenile and adult salmonids reduces the probability of surviving predation events. Juvenile salmonids with impaired olfaction may fail to properly imprint on their natal waters, which later in life leads to adult straying (*i.e.*, migrating into and spawning in streams other than their natal stream). Adults that do not return to natal waters are a functional loss to recruitment of a population. Adult male salmonids that do find their way back to their natal stream or river reaches and are subsequently exposed to the insecticides may lose some or all of their olfactory capacity, even from a short-term exposure. Female salmonids release odorants to trigger male priming hormones and to alert males of a female's spawning condition. Male fish with reduced olfactory capacity may not detect these cues, as demonstrated in a study on carbofuran(Bretaud et al 2002). Thus, spawning synchronization could be compromised and recently laid eggs may go unfertilized. Unfertilized eggs may result in reduced productivity and abundance for a population if sufficient numbers of spawning events are missed. Again, we find it difficult to accurately predict when these impairments and missed spawning opportunities occur, primarily as a result of lack of olfactory toxicity data on the a.i.s, incomplete pesticide use information, difficulty in conducting field experiments with adult salmonids, and uncertainties surrounding the extent of effects and concentrations which may trigger them. Because imprinting, avoiding predators, homing, and spawning are likely affected when exposed to OPs, we conclude these additional effects cannot be dismissed. Therefore, we expect populations exposed to OPs that affect olfaction may show reduced reproductive rates, reduced return rates, and reduced intrinsic rates of growth when sufficient numbers of individuals are affected.

Starvation during a critical life stage transition

In the *Population Modeling* section above we discussed population level impacts from reduced growth associated with reduced prey availability, however the models do not address starvation occurring from lack of prey at a critical life stage transition. Limitations in prey can cause starvation which can further limit abundance and productivity. Salmonids emerge from redds (nests) with a yolk-sac as their initial food source (yolk-sac fry). Once the yolk-sac has been absorbed, they must begin exogenous feeding. Fry have limited energy reserves, and if they are unable to swim properly or detect and capture prey, the onset of starvation occurs rapidly. Because juvenile salmon are limited by gape width, prey for this life stage is limited to very small aquatic and terrestrial invertebrates. The stressors of the action likely affect this critical life stage transition in several ways, leading to increased early life stage mortality. Impaired swimming and olfaction affects the fry's ability to detect and capture prey. Prey may be killed outright by the stressors of the action, leading to reduced prey availability or the complete absence of prey, although this is rare. Floodplain habitats where fry seek shelter and food are highly susceptible to the highest concentrations of the a.i.s, as these habitats are often low-flow, and/or shallow. Therefore, we expect that death of yolk-sac fry from exposure to the stressors of the action may reduce population abundance for populations with small numbers of individuals. All salmonid ESUs share this common life stage transition and therefore are at risk.

Death of returning adults

We discussed and analyzed with models the importance of juveniles to population viability. However, we did not address possible implications of returning adults dying from exposure to the stressors of the action before they successfully spawn. Pre-spawn adults have used up most of their accumulated fat stores, converting it into gamete production and typically die within hours to days after spawning. We anticipate that returning adults in this condition are likely less tolerant of chemical stressors. An adult returning from the ocean to natal freshwaters is important to a population's survival and recovery for many reasons. Notably, less than one percent of salmon generally survive to complete their life cycle. For populations with low abundance, every adult is crucial to a population's viability. We expect some sensitive adults will die from short-term exposures before they spawn, particularly those that spawn in or migrate through intensive agricultural watersheds and urban/suburban environments where elevated temperatures, other AChE-inhibiting insecticides, and other toxics may be present in addition to the a.i.s addressed in this Opinion. We are particularly concerned about azinphos methyl, methidathion, and phorate, which are the most acutely toxic of the a.i.s, with salmonid LC50s in the 1-20 μg/L range. EECs from all methods of estimation are in this range, as are monitoring data. Bensulide, naled, and phosmet are also expected to kill some returning adults, based on overlaps between the EECs and salmonid LC50s. Bensulide and phosmet are also expected to kill some returning adults, based on overlaps between the EECs and salmonid LC50s. We believe risk of death to returning adults from applications of dimethoate, disulfoton, ethoprop, methamidophos, methyl parathion, and naled is lower than the other two groups, given EECs and salmonid LC50s for these a.i.s. However, we expect sensitive or highly stressed individuals exposed to concentrations below the LC50 will die. The length of time the adults are exposed may vary widely for these a.i.s, depending on the persistence of the specific a.i.s and the hydrological regime of the exposed habitat, but we anticipate a greater likelihood of toxic exposure in shallow, small first and second order streams.

Another important consideration for returning adults is the fact a large number may be migrating together, and a fish kill of any magnitude may effectively eliminate a portion of the population bound for a specific natal stream, contributing to extirpation of that sub-population. This is particularly a concern for many coho salmon populations, which reproduce in distinct yearly cohorts, with virtually no year group overlap. Elimination of a cohort would result in approximately a one-third reduction of that sub-population as they reproduce in 3-year cycles. The missing cohort would result in depressed productions for many generations and may not be replaced.

We cannot quantify the number of adults lost to a given population in a given year. For those few populations where each adult salmonid is important to viability, we expect reductions in both productivity and abundance. In cases where a large fish kill occurs, it may also affect distribution via extirpation of sub-populations.

Synergistic toxicity

With certain combinations and concentrations of various OPs and carbamates, synergism occurs, resulting in increased inhibition of AChE and in some cases death (Laeta et al 2009). We currently have no predictive model for synergistic toxicity, nor any data showing synergism on the 12 a.i.s addressed in this Opinion. While synergism may occur, we cannot predict the intensity or effective concentrations at which it might occur for any of these compounds. Environmental situations where we anticipate synergism are likely to occur are in aquatic areas where two or more of the AChE inhibitors are present at concentrations sufficient to inhibit AChE as single compounds, but cumulatively result in enhanced toxicity. Swimming-related behaviors are particularly at risk from synergistic toxicity, as swimming is strongly correlated to the degree of AChE inhibition. Furthermore, where temperatures are elevated we expect a greater probability of synergism occurring within exposed salmonids. Based on the registered uses for the 12 a.i.s, these co-occurrences of a.i.s are most likely in agricultural dominated areas where applications of multiple a.i.s may overlap spatially and temporally.

If synergistic effects occur, we anticipate increased mortality than predicted based on additive toxicity. Based on the types of habitats potentially subject to concentrations which may cause synergistic effects, both juveniles and returning adults may be affected. Whether or not death occurs is dependent on exposure duration and concentrations of the insecticides. Typically, adult fish are less sensitive than early lifestages based on LC50 comparisons, however pre-spawn salmonids (*i.e.*, returning adults) are already highly stressed physiologically (due to the severe, rapid reduction in body fat), and little is known regarding how sensitive they are to toxics compared to the juvenile life stages or from healthy adult fish. Returning adults could be equally or more sensitive than juveniles. We anticipate synergism may affect productivity and abundance in exposed populations.

Toxicity from other stressors of the action

As described in the individual-level risk hypotheses, we expect toxic degradates of nine of the a.i.s addressed in this Opinion to contribute to the toxicity of the parent a.i, although based on existing data, we could not quantify the extent of this effect. Specifically, we expect the oxon degradates of azinphos methyl, bensulide, dimethoate, disulfoton, methidathion, methyl parathion, phosmet, and phorate, and also dichlorvos, a degradate of naled, to affect salmonids and their prey. Additional active ingredients contained in pesticide formulations and tank mixes likely increase the toxicity associated with the use of these products. Specific interactions between additional a.i.s in products and tank mixes and the a.i.s addressed in this Opinion are unknown, but it is reasonable to assume toxicity of the OPs may be enhanced by these ingredients. We discussed toxic properties of other/inert ingredients identified in the products we evaluated. However, thousands of other compounds are approved by EPA for addition to pesticide products without any specific requirement for the compound identity or amount to be listed on the labels. Many of these are known to be toxic to fish and other aquatic species. There is substantial uncertainty regarding the ingredients that occur in pesticide products containing the 12 a.i.s. Additionally, there are data gaps regarding the expected concentrations of these chemicals in salmonid habitats and the toxicity of these ingrediets. Exposed populations are at increased risk of reduced abundance and productivity from these chemicals. However, NMFS is unable to accurately describe the level of risk.

Conclusions on population level effects

We conclude that many of the populations of threatened and endangered salmonids covered by this consultation will likely show reductions in viability, particularly those that are comprised of juvenile life histories that rear for weeks to years in freshwater habitats found in intensive agricultural and residential/urban areas (Table 132). Juvenile coho salmon, steelhead, and ocean- and stream-type Chinook salmon use these types of rearing areas for extended periods which overlap with pesticide applications. Of greatest concern are those independent populations for each ESU or DPS distributed in high use areas of the pesticides.

Effects to abundance and productivity are anticipated from exposure to all 12 a.i.s, except fenamiphos, where the geographic ranges of listed population overlaps with intensive cropping patterns and residential/urban areas. Fenamiphos is expected to have individual fitness consequences to listed salmonids and negative impacts to salmonid habitat. However, the incidents of exposure are expected to be insufficient to cause population level effects given all uses of fenamiphos have been canceled and there is limited availability of existing stocks. The terms and conditions for cancelation of use of azinphos methyl, disulfoton, and methamidophos will also reduce the incidents of exposure to listed salmonids and their designated critical habitat. However, sale and use of pesticide products containing these active ingredients before cancellation may result in sufficient exposure to reduce the abundance and productivity of some populations of listed salmonids. Cancellation of these products is considered in the *Integration and Synthesis Section*. Predicted exposure of juvenile salmonids to azinphos methyl, methidathion, naled, and phorate can cause severe population declines through direct acute lethality. Additionally, significant population effects due to prey reductions are expected for some populations due to predicted exposure to azinphos methyl, disulfoton, methamidophos, methidathion, methyl parathion, naled, phosmet, and phorate. Population-level effects from exposure to single a.i.s through acute lethality and/or prey reductions are less likely for bensulide, dimethoate, and ethoprop. However, population modeling indicates reduction in salmonid abundance and productivity may occur with these a.i.s based on predicted and measured concentrations. We also anticipate potential reductions to population viability from death of returning adults exposed to the stressors of the action. Reductions in prey that occur when yolk sac fry are transitioning to exogenous feeding may result in starvation and consequently affect population viability.

Additionally, several factors increase the likelihood of population-level effects for the active ingredients: repeated exposures to AChE-inhibiting pesticides due to repeat applications of the a.i.s and applications of other OPs and carbamates; exposure to environmental mixtures of AChE-inhibiting pesticides that cause additive or synergistic effects; sublethal effects including impaired swimming and olfactory-mediated behaviors that have consequences for survival, migration, and reproduction; exposure to toxic degradates of the active ingredients; exposure to other stressors of the action such as other toxic a.i.s and inert ingredients present in the pesticide formulations and tank mixtures; and exposure to elevated temperatures that enhance the toxicity of the stressors of the action.

Table 132 Summary of Population-Level Analyses. Anticipated denotes that where exposure is expected, population-level consequences may occur. In contrast, Not anticipated denotes that where exposure is expected, population-level consequences are not expected.

Effects to populations	Azinphos methyl	Bensulide	Dimethoate	Disulfoton	Ethoprop	Methamidophos	Methidathion	Methyl parathion	Naled	Phorate	Phosmet
Death of sub yearling juveniles causes reductions in lambda	Anticipated	Anticipated	Not anticipated	Not anticipated	Not anticipated	Not anticipated	Anticipated	Anticipated	Anticipated	Anticipated	Anticipated
Reduced growth of sub yearlings results in reduced first year survival causing reductions in lambda	Anticipated	Not anticipated	Anticipated	Anticipated	Anticipated	Anticipated	Anticipated	Anticipated	Anticipated	Anticipated	Anticipated
Impaired swimming and olfactory-mediated behavior	Anticipated	Anticipated	Anticipated	Anticipated	Anticipated	Anticipated	Anticipated	Anticipated	Anticipated	Anticipated	Anticipated
Starvation during critical life stage transition	Anticipated	Anticipated	Anticipated	Anticipated	Anticipated	Anticipated	Anticipated	Anticipated	Anticipated	Anticipated	Anticipated
Death of returning adults	Anticipated	Anticipated	Anticipated	Anticipated	Anticipated	Anticipated	Anticipated	Anticipated	Not Anticipated	Anticipated	Anticipated
Synergistic toxicity	Anticipated	Anticipated	Anticipated	Anticipated	Anticipated	Anticipated	Anticipated	Anticipated	Anticipated	Anticipated	Anticipated
Toxicity from degradates in combination with the parent compounds	Anticipated	Anticipated	Anticipated	Anticipated	Not anticipated	Not anticipated	Anticipated	Anticipated	Anticipated	Anticipated	Anticipated
Toxicity from other stressors of the action: Other actives, inert/other ingredients, and chemicals added to tank mixtures	Anticipated	Anticipated	Anticipated	Anticipated	Anticipated	Anticipated	Anticipated	Anticipated	Anticipated	Anticipated	Anticipated

Fenamiphos not carried forward for population-level analysis.

Conclusions Regarding Risk Associated with Specific a.i.s

Below we describe the risk posed by each of the a.i.s to a generic population of salmonids, if those salmonids are present in one or more waterbodies expected to receive input of that a.i. from registered uses. In the ESU/DPS specific summaries, we determine co-occurrence of the a.i. and the fish based primarily on land use overlap with salmon-bearing waters. In some cases, use sites may not occur directly adjacent to occupied waterbodies but if there are sufficient use sites upstream to reasonably expect exposure concentrations downstream are of concern, we consider these use sites as well. Due to the physico-chemical properties of the a.i.s addressed in this Opinion, we have focused on direct runoff and spray drift from application sites as the primary drivers in exposure. Although the majority of the a.i.s considered have relatively short half- lives in the environment, and are not considered either persistent or bioaccumulative, we do note there is some atmospheric transport from the application sites to more distant environments. Unlike the three OPs considered in NMFS 2008 Opinion (chlorpyrifos, diazinon, and malathion (NMFS 2008c)), all of which were very toxic to both fish and invertebrates, the a.i.s considered in this Opinion vary widely in toxicity, especially to fish.

NMFS does not use the deterministic RQ/LOC approach used by EPA in evaluating risk to salmonids, nor do we consider only the a.i. Although each a.i. has been evaluated separately in the *Effects Analysis*, we consider all stressors of the action, in addition to other stressors such as elevated temperature and toxic chemicals that may already exist in the environment. For the evaluation of the a.i., we consider the full range of toxicity endpoints including sublethal effects, and the full range of EECs

Chemicals which EPA is currently in the process of canceling.

During the process of consultation on these a.i.s, EPA has proceeded with cancellation for some or all of the uses of several a.i.s, for reasons unrelated to the consultation itself As applicable, terms of cancellation are described in the chemical-specific summaries below.

Fenamiphos

Fenamiphos is in the final stages of cancellation. Distribution by the registrant ceased in 2007 and other distributers were required to halt sales on March 31, 2009. There is only one active label, a SLN use for iris and narcissus bulbs in Washington State, which expires on December 31, 2010. The terms of the cancellation order allow use of existing stocks of fenamiphos products until they are fully depleted (73 FR 21942). Predictably, fenamiphos use has decreased dramatically in response to the phases of cancellation.

In the event that salmon are exposed to fenamiphos, we expect there may be adverse effects to individuals, ranging from mortality to sublethal effects on behavior and/or growth. The assessment provided by EPA states that labeled uses of some fenamiphos products are expected to adversely affect aquatic communities, including salmon and invertebrates. Juvenile coho salmon exhibit signs of neurotoxicity following exposure to fenamiphos, though the mechanism is currently unknown. However, due to the minimal use of these products, we believe that salmon exposure to fenamiphos is extremely unlikely. We expect that fenamiphos will be applied to some agricultural crops over the next few years before tapering off to zero usage.

As use will be minimal, we expect that salmon will have little to no exposure to fenamiphos. Thus, exposure is not expected to rise to the level of affecting a population. The expected use of fenamiphos poses very low risk to the survival and recovery of all 28 ESUs/DPSs.

Azinphos methyl

Azinphos methyl is also well along in the process of cancellation, with current use sites restricted to a small number of crops, and all use prohibited after 2012. Current registered uses are all orchard crops, with the exception of two Section 24 (c) registrations for alkali bee beds. Some of these orchards are located in watersheds containing listed salmonids, and potentially toxic concentrations of azinphos methyl have been detected in these waters. Thus, we did conduct a full analysis of the potential effects of this a.i. on Pacific salmonids, including a population-level analysis.

Azinphos methyl is one of the most toxic a.i.s addressed in this Opinion, with LC50s for salmonids in 1 - 30 μg/L range, and EC50s for aquatic invertebrates in the 0.2 - 60 μg/L range. Sublethal effects in fish, including AChE inhibition, swimming behavior changes, and impaired growth occur in the 0.1 - 1.0 μg/L range. Azinphos methyl is mobile and fairly persistent in the environment, and has been detected in air samples and rainfall. In aquatic systems, it degrades within days to weeks. Current registrations allow for 2 applications, with a 7 - 14 day interval. EPA EECs and NMFS floodplain EECs are both higher than assessment endpoints. The oxon is not included in the EECs. Evidence is unclear as to how toxic oxons are compared to parent compounds, but EPA assumes oxons are 10 - 100 times as toxic, and NMFS has also made this assumption. Based on information supplied by EPA, it appears approximately 1 - 10% of applied a.i. may be converted to the oxon in the environment.

Based on our analysis, EECs of azinphos methyl may cause direct lethality to individuals, and/or impair growth, swimming, or olfaction. EECs are also sufficient to decrease prey abundance and/or diversity. Based on models, EECs may be sufficient to cause a significant decline in exposed populations due to lethality and reductions in growth. Field studies conducted in orchard areas where azinphos methyl was applied correlated reductions in stream macroinvertebrate assemblages (Grange 2002, St. Aubin 2004, Van der Linde 2005) and adverse responses in caged juvenile steelhead (Grange 2002, St. Aubin 2004) with concentrations of azinphos methyl and other chemicals in the water.

We anticipate azinphos methyl will cause sublethal, lethal, and population-level effects when it is applied near listed species habitat. However, although it poses high risk, this risk is counterbalanced somewhat by the fact that it is only legal for use for approximately two years following issuance of this Opinion.

Disulfoton

In July of 2009, registrants requested a voluntary cancellation for disulfoton. Some products were canceled as of December 31, 2009. Registrants can sell many products through December 31, 2010, and others through June 30, 2011. Persons other than registrants may sell and distribute exisiting stocks of these products until they are exhausted.

Based on the labels provided by EPA for this consultation, disulfoton is registered for a wide range of uses, including food and non-food crops, and urban/residential uses such as home gardens and ornamentals. Aerial applications are prohibited, with the exception of asparagus. Disulfoton is often limited to a single application per year, although not all labels specify number of applications. When application intervals are given, they are generally 42 days.

Disulfoton is less toxic to fish than some of the other a.i.s addressed in this Opinion, with LC50s for salmonids in 1,850 - 13,900 μg/L range. It is much more toxic to aquatic invertebrates, with EC50s for aquatic invertebrates in the 5 - 100 μg/L range. Available data show effects on fish reproduction at concentrations as low as 2.9 μg/L, and the EC50 for AChE inhibition is in the 112 - 2,118 μg/L range. Disulfoton degrades to a sulfoxide and a sulfone, both of which are more persistent in the environment than parent disulfoton. Based on the more stable degradates, half-life in water is in the neighborhood of a year (323 - 385 days). All three forms can convert to oxons. As with azinphos methyl, NMFS assumes the oxon is 10 - 100 times more toxic. EPA EECs for disufoton ranged from 7.1 - 67 μg/L, and NMFS floodplain estimates were 16 - 237 μg/L. The sulfoxide and sulfone are included in the EECs. No information was provided regarding what portion of the overall residues might be oxons.

Based on overlap between the EECs and assessment endpoints, NMFS does not anticipate direct lethality for salmonids, but does expect direct effects on sublethal endpoints such as behavior and growth. Available data indicates reproduction could be affected as well. We expect a reduction in prey availability, and subsequent effects on growth. The population-level analysis indicated a significant decline in lambda due to decreased growth.

Overall, as currently registered, disulfoton poses a moderate risk to salmon where there are co-occurrences of use sites and salmon habitat. Given persistence in water, this a.i. could also pose a downstream risk.

Methamidophos

In July of 2009, registrants requested a voluntary cancellation for methamidophos. The cancellation order will permit sale by the registrants until December 31 of 2010, and stocks belonging to "other than the registrants" can be sold until exhausted. NMFS did not have information regarding existing stocks while preparing this Opinion. Thus we cannot predict when actual use might cease. However, we do assume use will cease within a few years following the issuance of the Opinion. Currently, methamidophos is only registered for use on four crops: cotton, alfalfa grown for seed (CA only), tomatoes (CA only), and potatoes. Cotton is not grown in any of the Northwest states, and use of existing stocks on cotton must cease in September of 2010. Thus, following issuance of this Opinion, methamidophos may only be used on three crops in California, and one crop in the Northwest for a limited time.

Methamidophos is one of a.i.s addressed in this Opinion that is least toxic to fish, with LC50s for salmonids in 25,000 - 51,000 μg/L range. It is, however, toxic to aquatic invertebrates, with EC50s in the 0.042 - 1,054 μg/L range. We did locate information indicating it affects swimming at concentrations of 4,500 - 16,100 μg/L, less than 20% of a lethal concentration. Laboratory tests did not establish an EC50 for AChE inhibition. Little toxicity data other than standard survival tests were available for this a.i. methamidophos is very mobile in the environment. Half-life in water is dependent on pH, with breakdown occurring more quickly at higher pHs. Within the physiological tolerance range for salmonids, aquatic half-life is probably on the order of weeks to months. Methamidophos does not break down into an oxon or other toxic degradate. EPA EECs ranged from 30 - 65 μg/L, and NMFS floodplain estimates were 267 - 490 μg/L.

Based on overlap between the EECs and assessment endpoints, NMFS does not anticipate direct lethality nor sublethal effects on salmonids. However, in locations where uses co-occur with salmonid habitats, decreases in aquatic invertebrate populations may be sufficient to affect growth, especially given that methamidophos may be applied multiple times only 7 - 10 days apart. Population models indicated a significant decrease in lambda based on reductions in growth.

Given the minimal number of crops methamidophos is currently registered for, and current plans for use to end within several years following issuance of the Opinion, we believe the risk to listed salmonids is low. We note that risk still exists, as data indicate prey populations may be decreased enough to reduce growth.

Methidathion

A final cancellation notice for methidathion was published June 2, 2010. After December 31, 2012, registrants are prohibited from selling or distribution existing stocks of methidathion. After December 31, 2014, persons other than registrants are prohibited from selling or distributing existing stocks of products containing methidathion. After December 31, 2014, existing stocks of products containing methidathion already in the hands of users can be used legally until they are exhausted. Given the terms of cancellation, NMFS assumes methidathion may continue to be applied to crops for several years following the end of sales, thus use may continue until 2015, or 2016.

Based on the labels provided by EPA for this consultation, methidathion is currently registered for a range of agricultural uses, including row crops, orchard crops, and pasture/rangeland uses such as alfalfa. Non-agricultural uses are limited to nursery stock. Usage data located while preparing this Opinion do not provide a clear picture of the crops on which it is used, and usage may have shifted. It does appear to have been commonly used on orchard crops.

Methidathion is one of the most toxic a.i.s addressed in this Opinion, with LC50s for salmonids in 6.6 - 14 μg/L range, and EC50s for aquatic invertebrates in the 3.0 - 7.2 μg/L range. The EC50 for AChE inhibition is in the 0.47 - 2.7 μg/L range, and fish growth was affected at 12 μg/L. We located no specific data on swimming, olfaction, or other sublethal effects specific to this a.i. but based on acute toxicity information, believe it is reasonable to assume sublethal effects may occur at concentrations <1 μg/L. EPA EECs ranged from 8.9 - 15.5 μg/L, and NMFS floodplain estimates were 66 - 1,860 μg/L. In all cases, EECs are above assessment endpoints.

Methidathion is moderately mobile in the environment. It breaks down to an oxon form, and both the parent and oxon have been shown to transport atmospherically (Aston and Seiber 1997, Majweksi et al 2006). Available data indicate it will persist in aquatic systems for days to weeks. Current registrations allow multiple applications of methidathion for some crops, with intervals typically in the range of 7 - 14 days. Studies conducted in California were able to track pulses of methidathion applied as a dormant orchard spray from the San Joaquin and Sacramento Rivers all the way to San Francisco Bay (Kuivila and Foe 1995).

Based on our analysis, we anticipate current uses of methidathion to cause acute lethality to salmonids, to cause sublethal effects such as reduced growth, impaired swimming, and impaired olfaction, and to reduce available prey. Population modeling exercises show a significant decline in lambda due to both lethality and reduced growth. EECs are high enough methidathion may cause the death of returning adults.

While termination of sales in 2014 will sharply decrease risk associated with use of methidathion, the terms and conditions associated with the cancellation are not well defined as NMFS prepares this Opinion. Currently, the toxicity and wide range of uses of methidathion pose a high risk to salmonids in ESUs where use sites and salmonid habitat co-occur. Like disulfoton, extensive use of methidathion upstream of the habitat also poses a risk.

Methyl parathion

Methyl parathion is currently registered for a number of agricultural uses, and for rangeland and pasture. In the April 28, 2010 *Federal Register*, EPA published a notice regarding voluntary cancellation of all product registrations for methyl parathion. The final cancellation order was published on July 16, 2010. Under the cancellation order, registrations would be terminated effective December 31, 2012. End-use products cannot be sold after August 31, 2013, and cannot be legally used after December 31, 2013.

Methyl parathion comes in a microencapsulated formulation as well as the emulsifiable concentrate. Most fate and toxicity data are for the technical a.i., and it is unknown specifically how the microencapsulation affects fate and toxicity properties. Generally, methyl parathion appears to degrade quickly in aquatic systems. It does degrade to an oxon form, and both parent and oxon have been detected in water samples. It is the only one of the a.i.s registered for use on rice. Norberg-King, *et al., (1991)* identified methyl parathion as a likely toxicant in Colusa Basin Drain water using a TIE procedure Detectable quantities of methyl parathion were found in Colusa Basin Drain samples in 1995-1998, but not in samples from 1999-2002, indicating mandatory water holding times may be reducing pesticide loading into irrigation return systems. Aerial, ground, and chemigation application methods are allowed for methyl parathion. It may be applied 2 - 6 times a year. Application intervals are often not specified, but those that are range from 4 - 14 days.

For aquatic invertebrates, methyl parathion is one of the more toxic a.i.s addressed in this Opinion, and it is in the middle of the range in terms of toxicity to fish. EC50s for aquatic invertebrates are in the 0.14 - 28 μg/L range and LC50s for salmonids are in 1,850 - 5,300 μg/L range. EC50 for AChE inhibition was 21.2 - 39.0 μg/L. Decreased fish growth was noted at 10 - 380 μg/L, and effects on swimming occurred at 3.5 - 300 μg/L. EPA EECs ranged from 1.3 - 18.2 μg/L, and NMFS floodplain estimates were 134 - 980 μg/L. In one case, a measured concentration of 213 μg/L was reported as a result of spray drift from aerial application (Schulz 2004).

In some cases, we believe direct lethality will occur, and we believe sublethal effects on growth, swimming, and olfaction are likely in smaller waterbodies. Reduction in prey base is anticipated in all waterbodies receiving spray drift or runoff, given the toxicity, application rates, and potential frequency of re-application. For this chemical, we do anticipate population-level effects, as modeling showed a probable decline in lambda due to lethality and a significant decline in lambda due to growth. Overall, risk to salmonids from methyl parathion is moderate, depending on the spatial relationship of use sites and habitat. However, the risk posed by methyl parathion is decreased by the cessation of legal use in 2013, three years following the release of this Opinion, thus risk to the species is low.

Chemicals for which registrations appear to be static

Of the 12 a.i.s initially to be addressed in this Opinion, 6 appear to have registrations that remain static. NMFS presumes the uses analyzed will be permitted for the 15-year duration of the registration review timeline. Should EPA commence cancellation proceedings for any of these chemicals prior to issuance of the final Opinion, those changes will be considered. The a.i.s which currently fall into this category include bensulide, dimethoate, ethoprop, naled, phorate, and phosmet. Of the a.i.s in this group, some are registered for many use sites. A wider range of uses makes it both more difficult to predict where it might be used, and increases the chance that it could be used on multiple, unrelated use sites in a watershed. Dimethoate, naled, and phosmet have many use sites, whereas bensulide, ethoprop, and phorate are more limited.

Chemicals with Many Use Sites

Dimethoate

Dimethoate is one of the less toxic a.i.s addressed in this Opinion. It is currently registered for agricultural uses and also for forestry uses on douglas fir, cottonwood, and poplar in Washington and Oregon. It can be applied aerially, on the ground, or via chemigation. Maximum number of applications are generally not specified, and reapplication intervals are short (3 - 14 days). There are buffers for application near aquatic habitats.

LC50s for salmonids are in 6,200 - 7,500 μg/L range, and EC50s for aquatic invertebrates in the 43 - 15,000 μg/L range. EC50 for AChE inhibition was 196 - 382 μg/L. EPA

EECs ranged from 0.1 - 58 μg/L, and NMFS floodplain estimates were 46 - 652 μg/L. It is mobile in the environment (K_d 0.06 - 0.66) and very soluble in water. It forms an oxon, omethoate, which is a registered pesticide in some locations, but not the U.S. Available data show omethoate is more toxic than dimethoate to aquatic invertebrates on an acute basis, and similar in toxicity to parent for fish. The specific amount of omethoate formed is not known, but EPA estimates it to be <10%. The primary route of degradation in aquatic systems appears to be microbial, with the anaerobic aquatic metabolism half-life estimated at 16 days. Photolysis and hydrolysis half - lives are longer (68 days @pH7, and 353 days, respectively).

Based on overlap between assessment endpoints and EECs, we believe it is possible but unlikely that fish will be killed by dimethoate, but sublethal effects on growth and behavior could occur. We do anticipate reductions in the prey base in areas where dimethoate applications co-occur with salmon habitat. Population modeling showed a possible decline in lambda due to growth reductions. Based on the range of use patterns, frequency of application, and length of time it remains in the water column, dimethoate is particularly of concern in areas where agriculture or forestry co-occur with floodplain habitats. Overall, it poses a low to moderate risk to listed salmonids, depending on the spatial arrangement of the use sites in association with the habitat, and the life history of the species. Risk is higher for species that rear in floodplain habitats, or forage in areas likely to receive drift or runoff.

Naled

Naled is unique in this group of a.i.s, because in addition to agricultural uses, it is also registered as a vector control. Based on overlap of EECs and assessment endpoints, we expect naled to cause direct sublethal and lethal effects to salmonids and to decrease salmon prey populations. Population models showed a significant decline in lambda due to both lethality and effects on growth. Agricultural uses appear likely to cause higher water concentrations than noncrop uses based on model estimates. However, some naled labels allow for mosquito adulticide applications at rates comparable to crop uses (*e.g.* 1.25 lbs a.i./A). Additionally, naled may be applied over vast areas of freshwater habitats occupied by listed salmonids and the frequency of reapplication for the vector control measures are an important concern as reapplications may prevent recovery of salmonid prey for extended durations. Overall, we believe naled poses a high risk to all ESUs/DPSs.

Naled is registered for a range of agricultural uses, including a number of row crops, orchard crops, and some applications like forest and shade trees and ornamental plants. Application rates range from 0.63 - 2.12 lb a.i./A, and in most cases it can be applied multiple times (up to 7) at intervals of 7 - 14 days. Some labels currently include risk reduction measures, but others do not.

In the environment, naled degrades quickly to dichlorvos, which is also a registered pesticide. Dichlorvos is more water soluble, and more persistent in water than naled, with an aquatic half-life of 5 - 10 days as compared to naled's 1 - 5 days. Dichlorvos is also more toxic to aquatic invertebrates by about an order of magnitude. Based on available data, toxicity to salmonids appears similar.

EC50s for aquatic invertebrates are in the 0.14 - 230 μg/L range and LC50s for salmonids are in 87 - 345 μg/L range. EC50 for AChE inhibition was 6.5 - 9.5 μg/L. Decreased fish growth occurred at 15 μg/L. EPA EECs ranged from 0.8 - 33 μg/L, and NMFS floodplain estimates were 251 - 921 μg/L. The higher EECs were generally from high application rates (1.88 lb a.i./A), which are mostly crop uses, and drift from aerial applications.

Based on overlap of EECs and assessment endpoints, we expect naled to cause direct lethality to salmonids in some waterbodies, and to decrease prey populations and cause sublethal effects due to AChE inhibition in all waterbodies. Population models showed a significant decline in lambda due to both lethality and effects on growth. Based on application rates alone, agricultural uses appear likely to cause higher water concentrations. However, the inability to determine specific areas of use, and frequency of reapplication for the vector control measures are an important concern. Overall, we believe naled poses a high risk to all ESUs/DPSs.

Phosmet

Phosmet is registered for a wide variety of uses. Agricultural uses include row crops, orchard crops, and alfalfa (pasture/rangeland). It is also registered for use on cranberries, which are grown in bogs. It has forestry uses, on conifers and deciduous trees, urban/residential uses on ornamentals, and also may be used as a livestock spray. On current labels, multiple applications are permitted. Number of applications and application intervals are not specified for some crops. Some crops permit up to 5 applications, and when intervals are specified, they are typically around 10 days. It may be applied aerially, from the ground, or via chemigation.

Phosmet is stable to photolysis but has an extremely short hydrolysis half-life (0.4 days). Aquatic metabolism data were not available. Given the range of organic carbon partitioning coefficients (K_{oc}s 716 - 10,400) it may bind to sediment. Phosmet does form an oxon, although available fate data indicated it was a small amount (<0.5% of applied). LC50s for salmonids are in the 150 - 1,560 μg/L range, and EC50s for aquatic invertebrates in the 1.6 - 3,400 μg/L range. EC50 for AChE inhibition was 2.5 - 4.2 μg/L, and effects on fish reproduction and growth occurred at 6.1 μg/L. EPA EECs ranged from 3.0 - 29.9 μg/L and NMFS floodplain EECs were 5.0 - 2,920 μg/L.

We expect in some cases, concentrations of phosmet could be high enough to kill fish, due in part to drift estimates in floodplain habitats, but also due to the fact there are so many potential uses, some of which may occur concurrently in a particular watershed. Residues from applications on single crops are sufficiently high to cause reductions in the prey base, and cause sublethal effect due to AChE inhibition. Population models showed a probable decline in lambda due to lethality, and a significant decline in lambda due to growth. Overall, phosmet poses a high risk to all ESUs/DPSs.

Chemicals with More Limited Use Sites

Bensulide

Bensulide is unusual for an OP in that it is registered for use as an herbicide rather than an insecticide. The mode of action against plants is via inhibition of cellular division in roots and shoots. It is toxic to fish and invertebrates, presumably via activity on the central nervous system in the same fashion as other OPs. Laboratory tests were unable to establish an EC50 for AChE inhibition at test concentrations of up to 500 μg/L. Bensulide is registered for agricultural uses, primarily on row crops. It is also registered for urban/residential uses such as turf grass, golf courses, and residential lawns. Turf, grass, and lawn uses are permitted at roughly double the rate of food crop uses, and applications are allowed two or more times a year. Food crop uses allow only a single application per year. Aerial applications are prohibited and it must be soil incorporated or watered-in.

Of the OPs considered in this Opinion, bensulide is in the middle of the range in terms of toxicity to fish, and one of the least toxic to aquatic invertebrates. LC50s for salmonids are in 720 - 1,100 μg/L range, and EC50s for aquatic invertebrates in the 62.4 - 3,300 μg/L range. Because of bensulide's herbicidal properties, we also considered primary productivity. EC50s for freshwater plants ranged from 1,500 - 2,800 μg/L. EPA EECs ranged from 7.2 - 231 μg/L, and NMFS floodplain estimates were 1,100 - 2,640 μg/L. In general, the higher estimates were for the turf uses. In one case, a fish kill on a golf course was attributed to bensulide, with measured concentration in the water of 2,840 μg/L.

Bensulide is relatively persistent in the environment for an OP, with expected half-lives in water of 200 - 220 days based on photolysis and hydrolysis rates. It has a Koc of 1,400 - 4,350, and may also be more likely to partition to sediment than other OPs. It does form an oxon, which is more mobile in the environment than the parent. We did not locate information comparing the toxicity or persistence of the oxon compared to the parent.

Based on overlap of EECs and assessment endpoints, we anticipate effects on the prey community, and in some cases, direct lethality to the salmonids, especially in floodplain habitats. No data were available for this a.i. regarding sublethal effects such as growth, swimming, or olfaction. In order to give the benefit of the doubt to the species, we assume some such effects will occur. Based on persistence of bensulide, EECs, and assessment endpoints for primary productivity, it is possible that primary productivity will be affected in some locations. Modeling exercises indicate a possible decline in lambda due to lethality.

Overall, we believe that risk from use of bensulide is low to moderate in locations where agricultural use sites co-occur with listed salmonid habitats, but that it is moderate to high where turf/grass use sites co-occur, due to the much higher application rates and shorter application intervals.

Ethoprop

Ethoprop is registered primarily for agricultural uses, although in California, Oregon, and Washington, it may also be used on ornamentals. For crops located within the range of listed salmonids, it can only be applied once a year. Application methods are limited to ground and it must be soil incorporated or watered-in. Ethoprop comes in an emulsifiable concentrate, and a granular formulation. The liquid formulation has a buffer of 140 ft for inland freshwater habitats. The granular form does not.

For an OP, ethoprop is persistent in the environment, and EPA models the aquatic parameters as stable. LC50s for salmonids are in 1,020 - 13,800 μg/L range, and EC50s for aquatic invertebrates in the 44 - 93 μg/L range. EC50 for AChE inhibition was 196 - 382 μg/L, and effects on fish growth and reproduction occurred at 11 - 54 μg/L. EPA EECs ranged from 15 - 75 μg/L, and NMFS floodplain estimates were 6 - 24 μg/L. In one case, incident data reported a fish kill at a golf course at a much higher concentration (241 μg/L).

At the individual level, we expect ethoprop may sometimes cause direct lethality, and anticipate more often it will cause reduced growth due to effects on AChE inhibition and reduction of prey. It may also affect behavioral endpoints such as swimming and olfaction. Population-level modeling indicated a significant decline in lambda due to growth effects. Overall, we believe ethoprop poses a low to moderate risk to listed salmonids in locations where habitat co-occurs with use sites. Due to the length of time it may be in the water column, floodplain habitats where water is slow-moving are of the greatest concern, and habitat downstream of major use sites may also be at risk.

Phorate

Phorate is one of the most toxic a.i.s addressed in this Opinion, and the only one of the most toxic a.i.s not currently scheduled for cancellation. The other two most toxic a.i.s are azinphos methyl and methidathion. Phorate is currently only available in granular form, and use is limited to agricultural crops. It may only be applied on row crops, with the exception of a California 24(c) label for lilies and daffodils. It must be soil incorporated, and may only be applied once a year. Active labels specify use of BMPs, including vegetated buffer strips in certain situations. It breaks down quickly in aquatic systems (photolysis and hydrolysis half lives of 1.1 and 3.2 days, respectively). However, it does form a sulfoxide and sulfone, both of which are more mobile and persistent than the parent. The parent, sulfoxide, and sulfone all form oxons. No toxicity data was available for the oxons, thus we assumed they are 10 - 100 times more toxic than the parent. EECs include the sufoxide and sulfone.

LC50s for salmonids are in the 13 - 66 μg/L range, and EC50s for aquatic invertebrates in the 0.3 - 65 μg/L range. EC50 for AChE inhibition was 0.42 - 0.76 μg/L, and effects on fish growth occurred at 4.2 - 190 μg/L. EPA EECs ranged from 8 - 27 μg/L for crop uses around 1.3 lb a.i./A. The EEC for lilies and daffodils, which are labeled for 8.0 lb a.i./A, was 138 μg/L. For several crops, the maximum single use rate is between 2 and 4 lb a.i./A (beans, cotton, potatoes, soybeans, and radishes), so we anticipate EECs will be correspondingly higher than those estimated for crops with rates of 1.3 lb a.i./A. Given there are only granular uses, NMFS made no drift estimates for floodplain habitats. Presumably, EECs developed for the salmonid BEs include the risk reduction measures, although we note that crop EECs in more recent assessment conducted by EPA for the California red-legged frog are lower (0.3 - 16 μg/L) than those in the salmonid BEs.

NMFS expects use of phorate is likely to kill salmonids where use sites co-occur with salmon habitat. We also anticipate a reduction in the prey base, and sublethal effect on behavioral endpoints and growth. Population models showed a significant decline in lambda due to lethality and reduced growth. Phorate poses a high risk to salmon in ESUs/DPSs where use sites overlap with habitat.

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. 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 all 28 ESUs/DPSs. Changes in the near-term (five-years; 2014) are more likely to occur than longer-term projects (10-years; 2019). 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.

The states of the west coast region, which contribute water to major river systems, are projected to have 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). Overall, the west coast region has a projected population of 72.2 million people in 2010. The U.S. Census Bureau predicts this figure will grow to 76.8 million in 2015 and 81.6 million in 2020.

Although general population growth stems from development of metropolitan areas, growth in the western states is projected from the enlargement of smaller cities rather than from major metropolitan areas. Of the 46 western state metropolitan areas that experienced a 10% growth or greater between 2000 and 2008, only seven have populations greater than one million people. Of these major cities, one and two cities are from Oregon and California, respectively. They include Portland-Vancouver-Beaverton, OR (1.81% per year), Riverside-San Bernadino-Ontario, CA (3.31% per year), and Sacramento-Arden-Arcade-Roseville, CA (2.18% per year) (USCB 2009).

As these cities border coastal or riverine systems, diffuse and extensive growth will increase overall volume of contaminant loading from wastewater treatment plants and sediments from sprawling urban and suburban development into riverine, estuarine, and marine habitats. Urban runoff from impervious surfaces and roadways may also 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.

Mining has historically been a major component of western state economies. 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 many western streams that feed into or provide spawning habitat for threatened and endangered salmonid populations will be exacerbated.

As the western states have large tracts of irrigated agriculture, a 2.2% rise in agricultural output is anticipated (Figueroa and Woods 2007). Impacts from heightened agricultural production will likely result in two negative impacts on listed Pacific salmonids. The first impact is the greater use and application of pesticide, fertilizers, and herbicides and their increased concentrations and entry into freshwater systems. Carbaryl, carbofuran, and methomyl, and other pollutants from agricultural runoff may further degrade existing salmonid habitats. Second, 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.

The western states are widely known for scenic and natural beauty, and 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. Hiking, camping, and recreational fishing in these natural areas is unlikely to have any extensive effects on water quality.

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 28 ESUs. 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).

Coupled with EPA's registration of carbaryl, carbofuran, and methomyl, the effects from anthropogenic growth on the natural environment will continue to affect and influence the overall distribution, survival, and recovery of Pacific salmonids in California, Idaho, Oregon, and Washington.

Integration and Synthesis for Threatened and Endangered Pacific Salmonids

The *Integration and Synthesis* section describes NMFS' assessment of the potential for EPA's registration of azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, methamidophos, methidathion, methyl parathion, naled, phorate, and phosmet to reduce the reproduction, numbers or distribution of listed Pacific salmonids, taking into account status of the species, the environmental baseline, and cumulative effects.

We start with *Conclusions Regarding Specific a.i.s*, based on the analyses presented in the *Effects of the Proposed Action to Threatened and Endangered Pacific Salmonids* chapter. Inherent in the modeling used to determine some of the EECs 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, or resulting from more distant uses in the watershed or airshed. Modeling EECs and monitoring data are not ESU/DPS specific.

For the *Integration and Synthesis,* to evaluate areal extent of application sites near salmon-bearing waters, NMFS used a GIS overlay containing landuse classifications and salmon distributions to determine overlap. Because cropping patterns and registered use sites may change over time, landuse classifications (agricultural, forestry, urban/developed) are used rather than specific crops. Details of the GIS analysis and the maps are provided in *Appendix 7*. Occurrence of land uses where specific a.i.s could be applied near salmon-bearing waters for each ESU/DPS is shown in Tables 132-159.

Based on the risk presented in *Conclusions Regarding Specific a.i.s*, the co-occurrence of land uses where that a.i. may be applied, the status of the species, the environmental baseline, and the cumulative effects, we determine the potential for use of that a.i. as registered to reduce the reproduction, numbers, or distribution of populations within each ESU/DPS. This is expressed qualitatively as low, medium, or high (Tables 132-159).

Salmon exist as discrete population(s) within each ESU/DPS. These populations support the survival and recovery of the species, but may not all be equally affected by the use of an a.i.(s). Taking into account both the unevenness of use, and the importance of various populations to the ESU/DPS, we then determine the potential for appreciable reduction in the reproduction, numbers, or distribution of the species. This is expressed qualitatively as low, medium, or high (Tables 132-159).

In the *Conclusion* section, we present jeopardy and no jeopardy determinations (Table 194 and Table 195). For species listed as “threatened”, a high potential for reduction in the reproduction, numbers, or distribution of the species was determined to jeopardize the ESU/DPS. For species listed as “endangered”, which are more vulnerable, to extinction, a medium or high potential for reduction in the reproduction, numbers, or distribution was determined to jeopardize the ESU/DPS.

ESU/DPS Specific Evaluations

Below, we summarize the current status of each species, including baseline stressors. VSP parameters (abundance, growth rate, genetic variability, and spatial structure) are presented as a measure of the ESU/DPS’s relative health. As exposure to a.i.s during the juvenile life stage is of particular concern, we highlight the length of time juveniles are found in shallow, more vulnerable habitats. The number of extant populations that co-occur with agricultural and urban areas is also given.

The tables below list a.i.s addressed in this Opinion in three separate categories: those under cancellation proceedings, those that are currently registered for a wide range of uses, and those that are registered for a more limited range of uses. Within each subgroup, toxicity, fate properties, and use patterns differ. For two of the cancellation chemicals (azinphos methyl and methamidophos), uses are sufficiently restricted that crop specific analyses were done. Azinphos methyl is restricted to orchard uses, and methamidophos can only be used on potatoes in Washington, Oregon, and Idaho, although it is permitted for some additional uses in California. The length of time the cancellation chemicals are allowed to be used following publication of the Opinion was an important consideration.

Puget Sound Chinook Salmon (Threatened Species)

The Puget Sound ESU is comprised of 22 extant populations. Eleven of these populations have declining productivity; the remaining populations are at replacement value. Current spawner abundance is significantly lower than historical estimates. The spatial structure for this species is compromised by extinct and weak populations that are disproportionately distributed in the mid- to southern Puget Sound and the Strait of Juan de Fuca. The genetic diversity of this ESU has been reduced due to a disproportionate loss of populations exhibiting the early-run life history.

The Puget Sound Chinook salmon are faced with many challenges to recovery, including lost and degraded habitat, loss of in-river large wood, poor water quality from land use practices, water diversions, and elevated temperatures. Pesticide use and detections in the ESU's watershed are well documented. NAWQA sampling conducted in 2006 in the Puget Sound basin detected numerous pesticides and other synthetic organic chemicals in streams and rivers.

Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 133. More than 50 percent of the ESU is composed of evergreen, deciduous, or mixed forests. Other pesticide use areas include urban/residential development (15%) and agricultural uses (4%). Cultivated crops (1%) and hay crops and pastures (3%) are primarily distributed on the floodplain and other lowland habitats. The majority of urban/residential land use also occurs within river and stream valleys in lowland areas, and much of the nearshore marine area also consists of urban/residential. Our GIS analysis indicates 22 populations in this ESU are exposed to pesticides applied in agriculture and urban areas. These areas serve as spawning, rearing, and migration habitat for Puget Sound Chinook. Juveniles generally have long freshwater residences of one or more years before migrating to the ocean. Given their long residency period and use of freshwater, estuarine, and nearshore areas, juveniles and migrating adults have a high probability of exposure to pesticides that are applied near their habitats.

Table 133. Puget Sound Chinook Salmon

a.i.	Co-occurrence			Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Populations	Species
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Orchard crops only	NA	NA	High	Low
Disulfoton	Yes	Yes	Yes	High	Low
Methamidophos[1]	Potatoes only	NA	NA	Medium	Low
Methidathion	Yes	Yes	NA	High	High
Methyl parathion	Yes	NA	NA	High	Low
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Medium	Medium
Naled	Yes	Yes	Yes	High	High
Phosmet	Yes	Yes	Yes	High	High
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Medium	Medium
Ethoprop	Yes	NA	NA	Medium	Low
Phorate	Yes	NA	NA	High	Medium

1: Crop-specific analysis was conducted
NA: Not applicable

Lower Columbia River (LCR) Chinook Salmon (Threatened Species)

The LCR Chinook salmon ESU includes 20 fall- and 2 late-fall runs and 9 spring-run populations. The majority of spring-run LCR Chinook salmon populations are nearly extirpated. Total returns for all runs are substantially depressed, and only one population is considered self-sustaining. The spatial structure for this ESU is relatively intact despite a 35% reduction in habitat. The genetic diversity of all populations (except the late fall-runs) has been eroded by large hatchery influences and low effective population size.

Obstacles to the recovery of LCR Chinook salmon include hydropower development, reduced access to habitat, loss of habitat, harvest, elevated water temperature, and sedimentation. NAWQA sampling detected more than 50 pesticides in streams within this ESU's range, ten of which also exceeded EPA's chronic toxicity aquatic life criteria (Wentz et al 1998).

Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 134. The percentage of agriculture lands that overlap with LCR Chinook salmon ESU is about 6 %, with 2% as cultivated crop crops and 4% as hay/pasture. More than 76% of the ESU is composed of evergreen, deciduous forest, and mixed forests. Urban/residential development (13 %) is a fairly substantial portion of this ESU. Most of the highly developed land and agricultural areas in this ESU's range are adjacent to salmonid habitat. Our GIS analysis indicates that all populations may be exposed to pesticides applied in agriculture and urban areas. Given their long juvenile residency period, use of river mainstem and upstream tributaries for spawning, juveniles and migrating adults have a high probability of exposure to pesticides that are applied near their habitats.

Table 134. Lower Columbia River Chinook Salmon

a.i.	Co-occurrence			Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Populations	Species
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Orchard crops only	NA	NA	Low	Low
Disulfoton	Yes	Yes	Yes	Low	Low
Methamidophos[1]	Potatoes only	NA	NA	Medium	Low
Methidathion	Yes	Yes	NA	High	Medium
Methyl parathion	Yes	NA	NA	Medium	Low
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Medium	Medium
Naled	Yes	Yes	Yes	High	High

a.i.	Co-occurrence			Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Populations	Species
Phosmet	Yes	Yes	Yes	High	High
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Medium	Medium
Ethoprop	Yes	NA	NA	Medium	Low
Phorate	Yes	NA	NA	High	High

1: Crop-specific analysis was conducted
NA: Not applicable

Upper Columbia River (UCR) Spring run Chinook Salmon (Endangered Species)

The UCR Spring-run Chinook salmon ESU is comprised of three extant populations. These populations are affected by low abundances and failing recruitment. The long-term trend for abundance and lambda for all three populations indicate a decline. The ESU's genetic integrity is compromised by periods of low effective population size and a low proportion of natural-origin fish. Spatial structure of this ESU is fairly intact.

Recovery of the UCR Chinook salmon is hindered by altered channel and floodplain morphology, habitat degradation, loss of in-river wood, reduced flow, impaired fish passage and fish mortality from dams, harvest impacts, impaired water quality, and elevated temperature. Concentrations of azinphos methyl, triallate, chlorpyrifos, diazinon, lindane, and parathion have been detected in surface water samples and all exceeded EPA freshwater chronic criteria for the protection of aquatic life (Williamson et al. 1998).

Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 135. The percentage of agricultural and developed lands that overlap with UCR Chinook salmon habitat is about 5.4% and 4.7%, respectively. Forested lands make up about 45% of the ESU. Our GIS analysis indicates that all three populations are exposed to pesticides applied in agriculture and urban areas. Fish spawn and rear in the major tributaries leading to the Columbia River between Rock Island and Chief Joseph dams. Given their residency period and use of freshwater tributaries and floodplain areas, juveniles have a high probability of exposure to pesticides that are applied near salmonid aquatic habitats within the range of this ESU.

Table 135. Upper Columbia Spring-Run Chinook Salmon

a.i.	Co-occurrence			Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Populations	Species
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Orchards only	NA	NA	Low	Low
Disulfoton	Yes	Yes	Yes	Medium	Low
Methamidophos[1]	Potatoes only	NA	NA	Low	Low
Methidathion	Yes	Yes	NA	High	High
Methyl parathion	Yes	NA	NA	High	Low
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Medium	Medium
Naled	Yes	Yes	Yes	High	High
Phosmet	Yes	Yes	Yes	High	High
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Low	Low
Ethoprop	Yes	NA	NA	Low	Low
Phorate	Yes	NA	NA	High	High

1: Crop-specific analysis was conducted
NA: Not applicable

Snake River (SR) Fall-run Chinook Salmon (Threatened Species)

The SR Fall-run Chinook salmon ESU consists of one population that spawns in the lower mainstem Snake River. Its spatial distribution has been reduced to 10 to 15% of the historical range. The annual population growth rate for the population is just over replacement, and the ESU remains highly vulnerable due to low abundance. Genetic diversity has been reduced with the loss of additional populations and influx of hatchery raised spawners.

The major threats to this ESU include spawning habitat loss and degradation, impaired stream flows, barriers to fish passage, mortality from hydropower systems, poor water quality, and elevated water temperature.

Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 136. Pesticide use areas for the 12 a.i.s within this ESU's and above the Columbia River migratory corridor include evergreen forests (49%), cultivated crops (15%), pastures (1%), and developed lands (1%).

Historically, SR Fall-run Chinook salmon exhibited a largely ocean-type life history. However, as a consequence of dam construction, this ESU now resides in water that is cooler than the historic spawning areas, and alteration of the Lower Snake River by hydroelectric dams has created a series of low-velocity pools in the Snake River. Thus, Fall-run Chinook salmon in the Snake River Basin now exhibit one of two life histories: ocean-type and reservoir-type (Conner et al 2005, Tiffen et al 2001). The reservoir-type life history is one where juveniles overwinter in the reservoirs created by the dams, prior to migrating out of the Snake River. SR Fall-run Chinook salmon spend one to four years in the Pacific Ocean before beginning their spawning return migration. Given the freshwater residency period and migration distance traveled along the edges/margins of rivers, juveniles and migrating adults have a high probability of exposure to pesticides that are applied near their habitats.

Table 136. Snake River Fall-run Chinook Salmon

a.i.	Co-occurrence			Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Populations	Species
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Orchard crops only	NA	NA	Low	Low
Disulfoton	Yes	Yes	Yes	NLAA[3]	NLAA[3]
Methamidophos[1]	Potatoes only	NA	NA	NLAA[3]	NLAA[3]
Methidathion	Yes	No	No	Medium	Medium

a.i.	Co-occurrence			Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Populations	Species
Methyl parathion	Yes	NA	NA	Medium	Low
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Low	Low
Naled	Yes	Yes	Yes	Medium	Medium
Phosmet	Yes	Yes	Yes	Medium	Medium
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Low	Low
Ethoprop	Yes	NA	NA	Low	Low
Phorate	Yes	NA	NA	Low	Low

1: Crop-specific analysis was conducted
NA: Not applicable

Snake River (SR) Spring/Summer-run Chinook Salmon (Threatened Species)

This ESU includes 31 historical populations. Productivity trends are approaching replacement levels, though most populations are far below their respective interim recovery targets. Many individual populations have highly variable abundance and no positive long-term growth. The genetic diversity and spatial distribution of this ESU are intact.

The major obstacles to the recovery of this ESU include altered channel and floodplain morphology, excessive sediment, reduced stream flow, degraded water quality from land use activities, hydroelectric dams, water diversions, and elevated water temperature.

The percentage of cultivated croplands and developed lands that overlap with SR Spring/Summer-run Chinook salmon habitat are 6.6% and 1.7%, respectively. Our GIS analysis indicates 20 populations in this ESU are exposed to pesticides applied in agriculture and urban areas. Juvenile fish mature in fresh water for one year and may migrate from natal reaches into alternative summer-rearing or overwintering areas.

Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 137. This ESU spawns and rears primarily in the smaller tributaries, many of which are located on U.S. Forest Service lands. Agricultural and urban areas are not common in the watersheds comprising the ESU, and those that are present are clustered mostly around the mainstem Snake and Columbia Rivers. The Snake River is a high-volume, high-flow system, and the salmon use it primarily as a migratory corridor.

Table 137. Snake River Spring/Summer-run Chinook Salmon

a.i.	Co-occurrence			Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Populations	Species
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphosmethyl[1]	Orchard crops only	NA	NA	Low	Low
Disulfoton	Yes	Yes	Yes	NLAA[3]	NLAA
Methamidophos[1]	Potatoes only	NA	NA	NLAA[3]	NLAA
Methidathion	Yes	No	NO	Medium	Low
Methyl parathion	Yes	NA	NA	Medium	Low
Many Use Sites					
Dimethoate	Yes	Yes	Not in ID	Low	Low
Naled	Yes	Yes	Yes	Medium	Medium
Phosmet	Yes	Yes	Yes	Medium	Medium
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Low	Low
Ethoprop	Yes	NA	NA	Low	Low
Phorate	Yes	NA	NA	Low	Low

1: Crop-specific analysis was conducted
NA: Not applicable

Upper Willamette River (UWR) Chinook Salmon (Threatened Species)

The UWR Chinook salmon ESU is composed of seven populations. Of these, only the McKenzie population is producing naturally. Abundance is low for all populations, and growth rates are negative. The spatial distribution of this ESU has been dramatically reduced, with 30 to 40% of the total historic habitat blocked by dams. The genetic diversity of this ESU has been compromised by hatchery stocks and mixing between populations.

The obstacles to recovery for this ESU include loss/degraded floodplain connectivity and stream habitat, reduced stream flow, reduced access to spawning/rearing habitat, degraded water quality, and elevated water temperature. Fifty pesticides were detected in streams that drain both agricultural and urban areas. Ten of these pesticides exceeded EPA criteria for the protection of freshwater aquatic life from chronic toxicity

The percentage of cultivated and developed lands that overlap with UWR Chinook salmon habitat are 10.5% and 9%, respectively. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 138. Our GIS analysis indicates all populations in this ESU may be exposed to pesticides applied in agriculture and urban areas. Juveniles rear in the mainstem Willamette River and floodplain wetlands during the inundation period. Residence periods range from 6 months to over a year, with three distinct emigration runs. Given their residency period and habitat preference, juveniles and migrating adults have a high probability of exposure to pesticides that are applied near their habitat.

Table 138. Upper Willamette River Chinook Salmon

a.i.	Co-occurrence			Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Populations	Species
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Yes	NA	NA	Low	Low
Disulfoton	Yes	Yes	Yes	Medium	Low
Methamidophos[1]	Potatoes only	NA	NA	Low	Low
Methidathion	Yes	No	No	High	Medium
Methyl parathion	Yes	NA	NA	High	Low
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Medium	Medium

a.i.	Co-occurrence			Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Populations	Species
Naled	Yes	Yes	Yes	High	High
Phosmet	Yes	Yes	Yes	High	High
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Medium to High	Medium
Ethoprop	Yes	NA	NA	Medium	Medium
Phorate	Yes	NA	NA	High	High

1: Crop-specific analysis was conducted
NA: Not applicable

California Coastal (CC) Chinook Salmon (Threatened Species)

The CC Chinook salmon ESU's spatial structure has been drastically altered through the loss of several historic populations. Genetic diversity has been significantly reduced by the loss of the spring-run and coastal populations. Current population structure is uncertain, though fish are concentrated in 15 geographic locations. Populations in the Eel River and Russian River are larger than some of the others, and are important to the ESU. Overall ESU productivity is low and all populations have low abundance.

The major threats to this ESU's recovery include fisheries, timber harvest, vineyards and other agriculture, introduced fish species, migration barriers, habitat degradation, increased predation, and elevated water temperatures. Pesticides may be used within these watersheds, though very little monitoring has occurred.

Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 139. The percentage of cultivated croplands and developed lands that overlap with CC Chinook salmon habitat are 1% and 5.4%, respectively. Our GIS analysis indicates 15 populations in this ESU are exposed to pesticides applied in agriculture and urban areas. The most abundant populations are in the Eel River and tributaries, and in the Russian River watershed. While there is little overlap of use sites with the habitat of the Eel River populations, there is substantial overlap in the Russian River watershed. Due to the importance of this population to the ESU, likelihood of negative effects ratings were based primarily on the overlap in this watershed. Juveniles rear in freshwater streams for a few months, and may reside in the estuary for an extended period before entering the ocean. Given their residency period and use of estuaries, juveniles and migrating adults have a high probability of exposure to pesticides that are applied near their habitat.

Table 139. California Coastal Chinook Salmon

a.i.	Co-occurrence			Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Populations	Species
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Yes	NA	NA	Low	Low
Disulfoton	Yes	Yes	Yes	Medium	Low
Methamidophos[1]		NA	NA	Low	Low
Methidathion	Yes			High	Medium
Methyl parathion	Yes	NA	NA	High	Low
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Medium	Low
Naled	Yes	Yes	Yes	High	High
Phosmet	Yes	Yes	Yes	High	High
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Medium	Low
Ethoprop	Yes	NA	NA	Medium	Medium
Phorate	Yes	NA	NA	High	High

1: Crop-specific analysis was conducted
NA: Not applicable

Central Valley (CV) Spring-run Chinook Salmon (Threatened Species)

The CV Spring - run Chinook salmon ESU includes four populations in the upper Sacramento River and three of its tributaries. 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 all severely depressed from historic estimates, though time series data show that all three tributary populations have growth rates just above replacement.

Juvenile emigration in the Sacramento River is highly variable; individuals may migrate as fry or as yearlings. Floodplain habitats are particularly important for CV Spring - run Chinook salmon juveniles during rearing and migration (Sommer at al 2001, Sommer et al 2005). Given the residency period and use of non - natal tributaries, intermittent streams, and floodplain habitats for rearing and migration, juveniles and adults have a high probability of exposure to pesticides that are applied near their habitat.

The major threats to the recovery of this ESU include impaired or loss of habitat, predation, water management (dams, levees, reservoirs), and impaired water quality. Pesticides detected in the Sacramento River include thiobencarb, carbofuran, molinate, simazine, metolachlor, and dacthal, chlorpyrifos, carbaryl, and diazinon. Heavy use of agricultural pesticides and the high probability of mixtures increase likelihood of negative effects for this species.

The percentage of cultivated croplands and developed lands that overlap with CV Chinook salmon habitat are 21.3% and 10.8%, respectively. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 140. Our GIS analysis indicates all four populations in this ESU are exposed to pesticides applied in agriculture and urban areas. Fish must also migrate through the San Francisco-San Pablo-Suisan Bay estuarine complex, which is heavily influenced by input from California's Central Valley.

Table 140. Central Valley Spring-run Chinook Salmon

a.i.	Co-occurrence			Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Populations	Species
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Yes	NA	NA	Low	Low

a.i.	Co-occurrence			Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Populations	Species
Disulfoton	Yes	Yes	Yes	Medium	Low
Methamidophos[1]		NA	NA	Low	Low
Methidathion	Yes	No	No	High	High
Methyl parathion	Yes	NA	NA	High	Low
Many Use Sites					
Dimethoate	Yes	Yes	Yes	High	High
Naled	Yes	Yes	Yes	High	High
Phosmet	Yes	Yes	Yes	High	High
More Limited Use Sites					
Bensulide	Yes	Yes	Yes	High	High
Ethoprop	Yes	NA	NA	Medium	Medium
Phorate	Yes	NA	NA	High	High

1: Crop-specific analysis was conducted
NA: Not applicable

Sacramento River Winter-run Chinook Salmon (Endangered Species)

The Sacramento River Winter-run Chinook salmon ESU is now comprised of a single population. This population rears in the mainstem 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 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.

The obstacles to the recovery of this ESU are impaired or loss of habitat, predation, water management (dams, levees, and reservoirs), and increased water temperatures. Today, the ESU depends on reservoir storage and releases for access to cold water. Pesticides frequently detected in the Sacramento River include thiobencarb, carbofuran, molinate, simazine, metolachlor, dacthal, chlorpyrifos, carbaryl, and diazinon. Heavy use of agricultural pesticides and the high probability of mixtures increase likelihood of negative effects for this species. Juveniles rearing in the river system and floodplains may encounter high concentrations of pollutants at the onset of the rainy season.

Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 141. The percentage of cultivated croplands and developed lands that overlap with Sacramento River Winter-run Chinook salmon are 25% and 10%, respectively. Our GIS analysis indicates the sole winter - run population in this ESU is exposed to pesticides applied in agriculture and urban areas. Juvenile winter-run fish are found in the Delta primarily from November through early May, though some spend up to 10 months in the river system. Given their residency period and use of the Sacramento River and Delta for rearing, juveniles and migrating adults have a high probability of exposure to pesticides that are applied near their habitat.

Table 141. Sacramento Winter-run Chinoook Salmon

a.i.	Co-occurrence			Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Populations	Species
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Yes	NA	NA	Low	Low
Disulfoton	Yes	Yes	Yes	Medium	Low
Methamidophos[1]		NA	NA	Low	Low
Methidathion	Yes	No	No	High	High
Methyl parathion	Yes	NA	NA	High	Low
Many Use Sites					
Dimethoate	Yes	Yes	Yes	High	High
Naled	Yes	Yes	Yes	High	High
Phosmet	Yes	Yes	Yes	High	High
More Limited Use Sites					
Bensulide	Yes	Yes	NA	High	High
Ethoprop	Yes	NA	NA	Medium	Medium
Phorate	Yes	NA	NA	High	High

1: Crop-specific analysis was conducted
NA: Not applicable

Hood Canal Summer-run Chum Salmon (Threatened Species)

This ESU has two remaining independent populations. Much of the historical spatial structure has been lost; all populations on the eastern side of the canal are extirpated. The genetic diversity of the ESU has also declined. The two populations have long-term trends above replacement, though abundance is very low. The life history of this ESU strongly influences the potential for exposure. Following emergence, fish typically migrate quickly to nearshore marine areas in Puget Sound to rear and grow. Average rearing time for juveniles in Hood Canal is around 23 days before emigration.

The major threats to the survival and recovery of this ESU include habitat (floodplain, estuarine, and riparian) degradation, reduced stream flow, sedimentation, and hatcheries. The widespread loss of estuary and lower floodplain habitat has impacted the ESU's spatial structure and connectivity. NAWQA detections in surface waters in the Puget Sound Basin reported 26 of 47 analyzed pesticides.

Land use within the ESU is predominantly forested (73%), open water (17%), urban/residential (9%), and agriculture (2%). The percentage of cultivated croplands and developed lands that overlap with HC Summer-run chum salmon habitat is about 0.04% and 8.9%, respectively. Most of the agriculture and urban/residential occurs within river and stream valleys in lowland areas. Nearshore marine areas are frequently adjacent to urban/residential areas. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 142 Our GIS analysis indicates that both populations of HC Summer-run chum salmon may be exposed to pesticides applied in agriculture and urban areas

Table 142. Hood Canal Summer-run Chum Salmon

a.i.	Co-occurrence			Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Populations	Species
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Yes	NA	NA	Low	Low

a.i.	Co-occurrence			Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Populations	Species
Disulfoton	Yes	Yes	Yes	Low	Low
Methamidophos[1]	Yes	NA	NA	Low	Low
Methidathion	Yes	Yes	NA	Low	Low
Methyl parathion	Yes	NA	NA	Low	Low
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Low	Low
Naled	Yes	Yes	Yes	High	High
Phosmet	Yes	Yes	Yes	Low	Low
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Low	Low
Ethoprop	Yes	NA	NA	Low	Low
Phorate	Yes	NA	NA	Low	Low

1: Crop-specific analysis was conducted
NA: Not applicable

Columbia River (CR) Chum Salmon (Threatened Species)

This ESU has been reduced to two populations: the Lower Gorge tributaries and Grays River. The population abundances for the Grays River and Lower Gorge are significantly depressed. Short- and long-term productivity trends for these populations are at or below replacement. Much of the genetic diversity of this population has been lost due to the extirpation of 15 populations.

The major threats to this ESU include overharvests, hatcheries, hydromodification, habitat loss, elevated temperatures, and poor water quality. Of the salmonids, chum salmon are most averse to negotiating obstacles in their migratory pathway. Thus, they are more highly impacted by the Columbia River hydropower system – specifically the Bonneville Dam (Johnson et al 1997b).

The percentage of cultivated croplands, hay/pasture, and developed lands that overlap with CR chum salmon habitat is about 2%, 5%, and 15%, respectively. More than 50%

of the ESU is covered by deciduous, evergreen, or mixed forests. Within the ESU, agriculture and development are predominantly distributed in the low-lying areas near the Columbia River and its tributaries. Our GIS analysis indicates 14 populations of CR chum salmon are exposed to pesticides applied in agriculture, developed, and forested areas. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 143.

Chum salmon spawning migration in the Columbia River occurs in the late fall, from mid-October to December. They primarily spawn along the edges of the mainstem or in tributaries or side channels. The fry emerge between March and May and emigrate shortly thereafter to nearshore estuarine environments (Salo 1991). Juveniles spend around 24 days feeding in the estuary. The Columbia River estuary is extremely large with tidal influence extending from its mouth at the Pacific Ocean to the Bonneville Dam, 235 km upstream.

Table 143. Columbia River chum Salmon

a.i.	Co-occurrence			Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Populations	Species
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Orchard crops only	NA	NA	Low	Low
Disulfoton	Yes	Yes	Yes	Low	Low
Methamidophos[1]	Potatoes only	NA	NA	Low	Low
Methidathion	Yes	Yes	NA	Medium	Low
Methyl parathion	Yes	NA	NA	Low	Low
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Low	Low
Naled	Yes	Yes	Yes	High	High
Phosmet	Yes	Yes	Yes	Medium	Low
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Low	Low

a.i.	Co-occurrence			Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Populations	Species
Ethoprop	Yes	NA	NA	Low	Low
Phorate	Yes	NA	NA	Low	Low

1: Crop-specific analysis was conducted
NA: Not applicable

Lower Columbia River Coho Salmon (Threatened Species)

The LCR coho salmon ESU now consists of two populations found in the Sandy and Clackamas Rivers. Both populations have low levels of abundance. The diversity of populations has been eroded by large hatchery influences and low effective population sizes. The spatial structure for this ESU has also been drastically reduced compared to historical levels.

The major obstacles to LCR coho salmon's survival and recovery are reduced water flow from irrigation diversions and hydroelectric dams, degraded water quality, and elevated temperature. NAWQA sampling in surface waters within the ESU range detected more than 50 pesticides in streams. Ten pesticides exceeded EPA's criteria for the protection of aquatic life from chronic toxicity, including azinphos methyl.

The percentage of cultivated crop lands overlap with LCR coho ESU is about 6 %, 4% as hay/pasture land and 2% as cultivated crop land. More than 76% of the ESU is composed of evergreen, deciduous forest, and mixed forests. Urban/residential development lands (12%) were a fairly substantial portion of this ESU. The percentage of cultivated croplands and developed lands that overlap with LCR chum salmon habitat are 2% and 11.7%, respectively. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 144. Our GIS analysis indicates both populations of LCR coho salmon may be exposed to pesticides applied in agriculture and urban areas. Juveniles rear in fresh water for more than a year. During the day, they show a preference for near-shore habitats and use floodplain habitats (Johnson 1991).

Table 144 Lower Columbia River coho Salmon

a.i.	Co-occurrence			Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Populations	Species
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Orchards only	NA	NA	Low	Low
Disulfoton	Yes	Yes	Yes	Low	Low
Methamidophos[1]	Potatoes only	NA	NA	Low	Low
Methidathion	Yes	No	No	Low	Low
Methyl parathion	Yes	NA	NA	Medium	Low
Many Use Sites					
Dimethoate	Yes	Yes	Yes	High	Medium
Naled	Yes	Yes	Yes	High	High
Phosmet	Yes	Yes	Yes	High	High
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Medium	Medium
Ethoprop	Yes	NA	NA	Medium	Low
Phorate	Yes	NA	NA	High	High

1: Crop-specific analysis was conducted
NA: Not applicable

Oregon Coast (OC) Coho Salmon (Threatened Species)

The OC coho salmon ESU includes 13 functionally independent populations. Current abundance levels are less than 10% of historic populations. Long-term trends in ESU productivity remain strongly (Good 2005). Spatial distribution is relatively intact. Populations within the ESU experience recruitment failure and long-term negative growth. As with other coho, there is a 3 year brood cycle, and depletion of a specific brood year may reduce the resiliency of the ESU.

The major threats to this ESU include reduced habitat complexity, loss of overwintering habitat, excessive sediment, habitat degradation, elevated temperature, water diversions, and poor water quality.

The percentage of cultivated croplands and developed lands that overlap with OC coho salmon habitat are 0.23% and 6.6%, respectively. Most of the cropland is hay/pasture, and is primarily located in the Umpqua watersheds. While this is an important population for this ESU, there are a number of other functionally imdependent populations in other watersheds with less overlap. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 145. Our GIS analysis indicates all 13 populations of OC coho salmon may be exposed to pesticides applied in agriculture and urban areas. Juvenile coho salmon are often found in small streams less than five ft wide and rear in fresh water for 18 months.

Table 145. Oregon Coast Coho Salmon

a.i.	Co-occurrence			Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Populations	Species
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Orchard crops only	NA	NA	Low	Low
Disulfoton	Yes	Yes	Yes	Low	Low
Methamidophos[1]	Potatoes only	NA	NA	Low	Low
Methidathion	Yes			Low	Low
Methyl parathion	Yes	NA	NA	Low	Low
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Low	Low
Naled	Yes	Yes	Yes	Medium	Medium
Phosmet	Yes	Yes	Yes	Medium	Medium
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Low	Low
Ethoprop	Yes	NA	NA	Low	Low
Phorate	Yes	NA	NA	Low	Low

1: Crop-specific analysis was conducted
NA: Not applicable

Southern Oregon/Northern California Coast (SONCC) Coho Salmon (Threatened Species)

The SONCC coho salmon ESU includes coho salmon in streams between Cape Blanco, Oregon, and Punta Gorda, California. The disproportionate loss of southern populations has decreased the genetic diversity of this ESU. Coho distribution within individual watersheds has been reduced as well. There is very limited information on population growth rates for this ESU. Available data indicates that the Eel River and southern populations have critically low abundances. Coho have a 3 year brood cycle, and depletion of a specific brood year may reduce the resiliency of the ESU.

The major obstacles to the survival and recovery of this ESU include habitat loss and degradation, reduced stream flow, migratory barriers, timber harvest, agricultural activities, water management, and elevated temperatures.

The percentage of cultivated croplands and developed lands that overlap with SONCC coho salmon habitat are 2.5% and 4.3%, respectively. Our GIS analysis indicates that fish may be exposed to pesticides applied in agriculture and urban areas in all watersheds. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 146. As little population data were available for this ESU, we were not able to determine if agricultural and developed areas, which cluster in certain watersheds, co-occur with important populations. Areas with more cropland include the Scott and Shasta watersheds in the Klamath basin, and the Upper and Middle Rough River[14] watersheds. Of the development in this ESU, much is in the Rough River basin, with most of the rest distributed along the coastline and estuaries. The fry rear in backwater, side channels, and shallow channel edges for up to 18 months.

[14] The Rough River is also be referred to as the Rouge or Rouge River in other publications, maps, or websites

Table 146. Oregon/Northern California Coast Coho Salmon

a.i.	Co-occurrence			Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Populations[A]	Species[B]
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Orchard only	NA	NA	Low	Low
Disulfoton	Yes	Yes	Yes	Medium	Low
Methamidophos[1]	Yes	NA	NA	NLAA	NLAA
Methidathion	Yes	No	NO	High	Medium
Methyl parathion	Yes	NA	NA	High	Low
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Medium	Medium
Naled	Yes	Yes	Yes	High	High
Phosmet	Yes	Yes	Yes	High	High
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Medium	Medium
Ethoprop	Yes	NA	NA	Medium	Medium
Phorate	Yes	NA	NA	Medium	Medium

1: Crop-specific analysis was conducted
NA: Not applicable

Central California Coast (CCC) Coho Salmon (Endangered Species)

The CCC coho salmon ESU includes 11 independent populations. The spatial structure for CCC coho salmon has been substantially modified due to lack of viable source populations and loss of dependent populations. Long-term population trends are unknown, though all populations have very low abundances. This year's low return suggests that all three year classes are faring poorly across the species' range. Loss of a specific year class may decrease the overall resiliency of the population. Juveniles rear for 18 months, spending two winters in fresh water.

The major threats to the survival and recovery the ESU include loss of riparian cover, elevated water temperature, alteration of channel morphology, loss of winter habitat, and siltation. Highly contaminated runoff into the Russian River, San Francisco Bay, and into rivers south of the Golden Gate Bridge is expected during the first fall storms.

The percentage of cultivated croplands and developed lands that overlap with CCC coho salmon habitat are 2.3% and 9.4%, respectively. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 147. Much of the development is centered around San Francisco Bay, and there are also developed areas and agriculture in the Russian River watershed. Coho is the San Francisco Bay are considered effectively extirpated, and the Russian River, which was once a source population for this ESU, is in serious decline (Spence 2008). Our GIS analysis indicates that all 11 populations may be exposed to pesticides applied in agriculture and urban areas. The majority of the salmon remaining is in the northern, undeveloped watersheds around the Navarro and Big Rivers.

Table 147. Central California Coast Coho Salmon

a.i.	Co-occurrence			Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Populations	Species
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Yes	NA	NA	Low	Low
Disulfoton	Yes	Yes	Yes	Medium	Low
Methamidophos[1]		NA	NA	Low	Low
Methidathion	Yes			High	High
Methyl parathion	Yes	NA	NA	High	Low
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Medium	Low
Naled	Yes	Yes	Yes	High	High
Phosmet	Yes	Yes	Yes	High	High
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Medium	Low
Ethoprop	Yes	NA	NA	Medium	Medium
Phorate	Yes	NA	NA	High	High

1: Crop-specific analysis was conducted
NA: Not applicable

Ozette Lake Sockeye Salmon (Threatened Species)

The Ozette Lake sockeye salmon ESU consists of a single population made up of five spawning aggregations. Uncertainty remains on the growth rate and productivity of the natural component of the ESU. Genetic differences occur between age cohorts and different age groups do not spawn with each other. Genetic diversity within the ESU, however, is low. Spatial structure of the population has been altered, as only two beaches are used for spawning. Overall abundance is also significantly depressed.

Major threats to this population include degraded habitat, loss of in-river large wood, and siltation of spawning habitat. Roughly 77% of the land in Ozette Basin is managed for timber production (Jacobs 1996).

Ozette Lake is in a sparsely populated area, with less than 1% of land developed within the range of this ESU. Similarly, there is no cultivated cropland. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 148. Our GIS analysis indicates that Ozette Lake sockeye salmon have minimal risk of exposure to pesticides applied for agricultural or urban uses. They may be at risk of exposure from forestry related uses. Fry rear in the limnetic zone of Ozette Lake for a full year.

The life histories of this ESU strongly influence the potential for exposure to the 12 a.i.s. Adult spawners enter Ozette River from April to early August and may remain in Ozette Lake for extended periods before spawning (October- February). Spawning occurs along the lakeshore and historically in some of the lakes' tributaries. Fry migrate immediately to the lake where they rear for a year or so before entering the ocean. Land use of this ESU is primarily forest with private, state, and federal ownership (86% forested, 13% open water, 1% developed land, 0% agriculture). No crops were indentified within the NLCD data for this ESU. The entire circumference of the lake is within Olympic National Park. The predominant pesticide use sites (*i.e.*, urban/residential and forestry uses) overlap with the Lake's freshwater tributaries. As such, the greatest risk of exposure is to those sockeye that utilize freshwater tributary habitats. Based on juvenile sockeye's lake rearing, we do not anticipate reductions in prey within the tributaries to affect juvenile growth. Direct effects to fish remain a concern within tributaries. No effect determinations were made for those a.i.s that showed little to no overlap of sockeye with labeled uses. Although no cropland occurred within the 2.5 km area analyzed, private residences along tributaries may have small, non-commercial crops where applications of the a.i.s could occur. We assumed it is unlikely that restricted use pesticides would be applied in these situations. Not likely to adversely affect determinations were made for those a.i.s with very low probability of exposure and expect effects to be discountable or insignificant.

Table 148. Ozette Lake sockeye Salmon

a.i.	Co-occurrence			Potential for adverse effects	
	Agriculture	Urban/ Residential	Forest	Populations	Species
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	No	NA	NA	No Effect[2]	NA
Disulfoton	No	Yes	Yes	NLAA[3]	NA
Methamidophos[1]	No	NA	NA	No Effect[2]	NA
Methidathion	No	Yes	NA	NLAA[3]	NA
Methyl parathion	No	NA	NA	No Effect[2]	NA
Many Use Sites					
Dimethoate	No	Yes	Yes	NLAA[3]	NA
Naled	No	Yes	Yes	Low	Low
Phosmet	No	Yes	Yes	Low	Low
More Limited Use Sites					
Bensulide	No	Yes	NA	NLAA[3]	NA
Ethoprop	No	NA	NA	NLAA[3]	NA
Phorate	No	NA	NA	No Effect[2]	NA

1: Crop-specific analysis was conducted
2: At the scale of the individual, no exposure anticipated
3: At the scale of the individual, effects are anticipated to be discountable
NA: Not applicable

Snake River Sockeye Salmon (Endangered Species)

The SR sockeye salmon ESU is comprised of one remaining population in Redfish Lake, Idaho. Abundance and productivity are highly variable; around 30 fish of hatchery origin return to spawn each year (FCRPS 2008). However, this figure has increased to adults numbering in the hundreds over the last two years. The ESU's genetic diversity has been reduced based on low population abundance and a high proportion of hatchery-origin fish.

The major threats to the survival and recovery of this ESU include altered channel morphology, impaired tributary and stream flow and passage, migration barriers, degraded water quality, hydromodification of the Columbia and Snake Rivers, and fish mortality from hydropower systems.

About 1% of the land surrounding Redfish Lake has been developed, and another 1% is used for agriculture, primarily hay and pasture. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 149. Juvenile sockeye remain in the lake for one to three years before migrating through the Snake and Columbia Rivers for several hundred miles to the ocean. Given the distance traveled between Redfish Lake and the ocean, juveniles and returning adults have a high probability of exposure to pesticides that are applied near salmonid habitats during migration. More than 50% of the ESUs is in evergreen forests. Consequently, forestry uses are the major source of pest exposure during spawning and rearing activities.

Table 149. Snake River Sockeye Salmon

a.i.	Co-occurrence			Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Populations	Species
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Orchard crops only	NA	NA	Low	Low
Disulfoton	Yes	Yes	Yes	NLAA[2]	NLAA

a.i.	Co-occurrence			Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Populations	Species
Methamidophos[1]	Potatoes only	NA	NA	NLAA[2]	NLAA
Methidathion	Yes	No	NO	Low	Low
Methyl parathion	Yes	NA	NA	Low	Low
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Low	Low
Naled	Yes	Yes	Yes	Medium	Medium
Phosmet	Yes	Yes	Yes	Medium	Low
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Low	Low
Ethoprop	Yes	NA	NA	Low	Low
Phorate	Yes	NA	NA	Low	Low

1: Crop-specific analysis was conducted
2: At the scale of the individual, anticipated effects are expected to be discountable
NA: Not applicable

Puget Sound Steelhead (Threatened Species)

The Puget Sound steelhead is comprised of 53 populations (37 winter-run and 16 summer-run). Summer-run populations are concentrated in northern Puget Sound and Hood Canal. The WDFW 2002 stock assessment categorized 5 populations as healthy, 19 as depressed, 1 as critical, and 27 of unknown status. Median population growth rates indicate declining population growth for nearly all populations in the DPS (NMFS 2005). Overall, the DPS experiences declining abundance, reduced genetic diversity, and abbreviated spatial complexity.

The major threats to the survival and recovery of this DPS include habitat degradation, water diversions, poor water quality, hatchery domestication, and elevated temperature. Over two million people inhabit the area, with most development occurring along rivers and coastline. NAWQA sampling conducted in 2006 within the Puget Sound basin detected 26 pesticides and 74 other synthetic organic chemicals in streams and rivers.

More than 50 percent of the ESU is composed of evergreen, deciduous, or mixed forests. Other pesticide use areas include urban/residential development (15%) and agricultural uses (4%). Cultivated crops (1%) and hay crops and pastures (3%) are primarily distributed on the floodplain and other lowland habitats. The majority of urban/residential also occurs within river and stream valleys in lowland areas, and much of the nearshore marine area also consists of urban/residential development. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 150. Our GIS analysis indicates all populations in this DPS are exposed to pesticides applied in agriculture and urban areas. Fry usually inhabit shallow water along banks of stream or aquatic habitats on stream margins. Juveniles rear in a wide variety of freshwater habitats, generally for two years with a minority migrating to the ocean as one or three-year olds.

Table 150. Puget Sound steelhead

a.i.	Co-occurrence			Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Populations	Species
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Orchard crops only	NA	NA	High	Low
Disulfoton	Yes	Yes	Yes	High	Low
Methamidophos[1]	Potatoes only	NA	NA	Medium	Low
Methidathion	Yes	Yes	NA	High	High
Methyl parathion	Yes	NA	NA	High	Low
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Medium	Medium
Naled	Yes	Yes	Yes	High	High
Phosmet	Yes	Yes	Yes	High	High
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Medium	Medium
Ethoprop	Yes	NA	NA	Medium	Low
Phorate	Yes	NA	NA	High	Medium

1: Crop-specific analysis was conducted
NA: Not applicable

Lower Columbia River Steelhead (Threatened Species)

The LCR steelhead DPS includes 23 extant populations. Spatial structure within the DPS, especially in Washington, has been substantially reduced by the loss of access to the upper portions of some basins from tributary hydropower development. Many of the populations in this DPS are small, and the long- and short-term trends in abundance of all individual populations are negative. The genetic diversity of this DPS has also been substantially reduced.

The major threats to this DPS include dams, water diversions, destruction/ degradation of habitat, altered channel morphology, reduced floodplain connectivity, sedimentation, reduced stream flow, land use practices, poor water quality, and elevated water temperature. NAWQA sampling detected more than 50 pesticides. Ten pesticides exceeded EPA's criteria for the protection of aquatic life from chronic toxicity.

The percentage of cultivated crop lands overlap with LCR Steelhead DPS is about 7%, 4.5 % as hay/pasture land and 2.5% as cultivated crop land. More than 61% of the DPS is composed of evergreen, deciduous forest, and mixed forests. Urban/residential development lands (12%) were a fairly substantial portion of this DPS. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 151. Our GIS analysis indicates all populations are exposed to pesticides applied in agriculture and urban areas. Juveniles typically rear in floodplain habitats associated with their natal rivers and streams for more than a year, and remain in fresh water systems for at least two years.

Table 151. Lower Columbia River steelhead

a.i.	Co-occurrence			Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Populations	Species
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA

a.i.	Co-occurrence			Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Populations	Species
Azinphos methyl[1]	Orchards only	NA	NA	Low	Low
Disulfoton	Yes	Yes	Yes	Low	Low
Methamidophos[1]	Potatoes only	NA	NA	Low	Low
Methidathion	Yes	Yes	Yes	Low	Low
Methyl parathion	Yes	NA	NA	Medium	Low
Many Use Sites					
Dimethoate	Yes	Yes	Yes	High	Medium
Naled	Yes	Yes	Yes	High	High
Phosmet	Yes	Yes	Yes	High	High
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Medium	Medium
Ethoprop	Yes	NA	NA	Medium	Low
Phorate	Yes	NA	NA	High	High

1: Crop-specific analysis was conducted
NA: Not applicable

Upper Willamette River Steelhead (Threatened Species)

The UWR steelhead DPS is comprised of four extant populations that occupy tributaries draining the east side of the UWR basin. Populations within this DPS have been declining and have exhibited large fluctuations in abundance. Abundance is moderately depressed for the entire DPS. The DPS's spatial distribution and genetic diversity are moderately intact.

The major threats to the survival and recovery of this DPS include habitat loss due to blockages, lost or degraded floodplain connectivity, and degraded water quality within the Willamette mainstem and the lower reaches of its tributaries. Fifty pesticides were detected in streams that drain both agricultural and urban areas. Forty-nine pesticides were detected in streams draining agricultural land, while 25 pesticides were detected in streams draining urban areas. Ten of these pesticides, including azinphos methyl, exceeded EPA criteria for the protection of freshwater aquatic life.

The percentage of cultivated crop lands and developed lands overlapping with this DPS are 14.5% and 10%, respectively. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 152. Our GIS analysis indicates all four populations in this DPS are exposed to pesticides applied in agriculture and urban areas. After emergence, steelhead fry typically rear in floodplain habitats associated with their natal rivers and streams for two years. Given their residency period and habitat preference, juveniles and migrating adults have a high probability of exposure to pesticides that are applied near their habitat.

Table 152. Upper Willamette Steelhead

a.i.	Co-occurrence			Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Populations	Species
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Yes	NA	NA	Low	Low
Disulfoton	Yes	Yes	Yes	Medium	Low
Methamidophos[1]	Potatoes only	NA	NA	Low	Low
Methidathion	Yes	NA	NA	High	Medium
Methyl parathion	Yes	NA	NA	High	Low
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Medium	Medium
Naled	Yes	Yes	Yes	High	High
Phosmet	Yes	Yes	Yes	High	High
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Medium to high	Medium
Ethoprop	Yes	NA	NA	Medium	Medium
Phorate	Yes	NA	NA	High	High

1: Crop-specific analysis was conducted
NA: Not applicable

Middle Columbia River Steelhead (Threatened Status)

The MCR steelhead DPS includes 16 extant populations in Oregon and Washington. The spatial structure of this population is relatively intact. The genetic diversity has been compromised by interbreeding with resident and hatchery fish. Population growth rates are near replacement, though abundances are depressed in relation to historic levels.

The major threats to this DPS include altered floodplain and channel morphology, sedimentation, reduced stream flow, migratory barriers, hydroelectric system mortalities, agricultural practices, poor water quality, and elevated water temperature. Seventy-six pesticide compounds were detected within the Yakima River Basin.

The percentage of cultivated crop lands and developed lands within the range of this DPS are 17% and 3%, respectively. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 153. Our GIS analysis indicates 16 populations are exposed to pesticides applied in agricultural and urban areas. Swim–up fry usually inhabit shallow water along banks of streams or aquatic habitats on stream margins. Juveniles rear in a variety of freshwater habitat for two years . Essentially there is a continuous run of steelhead throughout the year thus adults and rearing juveniles are in freshwater habitats throughout the year.

Table 153. Middle Columbia River steelhead

a.i.	Co-occurrence			Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Populations	Species
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Orchards only	NA	NA	Low	Low
Disulfoton	Yes	Yes	Yes	Medium	Low
Methamidophos[1]	Potatoes only	NA	NA	Low	Low
Methidathion	Yes	Yes	NA	High	High
Methyl parathion	Yes	NA	NA	High	Low
Many Use Sites					

a.i.	Co-occurrence			Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Populations	Species
Dimethoate	Yes	Yes	Yes	High	Medium
Naled	Yes	Yes	Yes	High	High
Phosmet	Yes	Yes	Yes	High	High
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Medium	Medium
Ethoprop	Yes	NA	NA	Medium	Medium
Phorate	Yes	NA	NA	High	High

1: Crop-specific analysis was conducted
NA: Not applicable

Upper Columbia River Steelhead (Endangered Species)

The UCR steelhead DPS consists of four extant populations in Washington State. Abundance data indicate that these populations are below the minimum threshold for recovery and have negative growth rates. Adult returns are dominated by hatchery fish and experience reduced genetic diversity from homogenization of populations. The spatial structure of this DPS has been severely altered, with 50% of its habitat cutoff by the Grand Coulee Dam.

The major obstacles to the survival and recovery of the UCR steelhead include hatcheries, dams that block fish migration, altered floodplain and channel morphology, water diversions, loss of LWD, destruction of riparian habitat, harvest, hydroelectric system mortality, land use practices, poor water quality, and elevated water temperature. Pesticides have been detected in UCR steelhead freshwater habitats. Concentrations of six pesticides exceeded the guidelines for aquatic life.

The percentage of cultivated crop lands and developed lands within the range of the ESU are 13% and 4%, respectively. Our GIS analysis indicates all 4 populations in this DPS are exposed to pesticides applied in agriculture and urban areas. Newly emerged fry move about considerably and seek suitable rearing habitat, such as stream margins or cascades. The majority of juveniles smolt as two-year olds, though some individuals may rear for as long as seven years in these fresh water systems.

Co-occurrence of agriculture, forestry and urban areas with salmonid habitat is shown in Table 154. There is some agriculture in the spawning and rearing areas in the Wenatchee, Methow, and Okenogan watersheds. In the Entiat, there is intense agriculture outside the buffer, in the Upper Columbia Irrigation District. The water is heavily used and re-used in irrigation. We expect that the fish will also be exposed to a number of the a.i.s on their migratory pathway along the Columbia River, where the valley is heavily agricultural. A portion of the waters the salmonids use are 303 (d) listed for high temperature, and we expect this will excerabate effects of the a.i.s.

Table 154. Upper Columbia River steelhead

a.i.	Co-occurrence			Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Populations	Species
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Orchard crops only	NA	NA	Low	Low
Disulfoton	Yes	Yes	Yes	Medium	Low
Methamidophos[1]	Potatoes only	NA	NA	Low	Low
Methidathion	Yes			High	High
Methyl parathion	Yes	NA	NA	High	Low
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Medium	Medium
Naled	Yes	Yes	Yes	High	High
Phosmet	Yes	Yes	Yes	High	High
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Low	Low
Ethoprop	Yes	NA	NA	Low	Low
Phorate	Yes	NA	NA	High	High

1: Crop-specific analysis was conducted
NA: Not applicable

Snake River Basin Steelhead (Threatened Species)

The SR basin steelhead DPS includes 23 populations that are spatially distributed in each of the six major geographic areas in the Snake River basin (Good 2005). The historic spatial structure is relatively unaltered. While population growth rates show mixed long- and short-term trends in productivity, overall abundances remain well below their interim recovery criteria. Genetic diversity has been reduced, particularly for the B-run steelhead, those whose life history pattern includes spending two or more years in freshwater, and two or more years in the ocean before their upriver migration. A-run steelhead are smaller, have a shorter freshwater and ocean residence.

The major threats to the survival and recovery of this DPS include hatcheries, harvest impacts, altered floodplain and channel morphology, hydrosystem mortality, water diversions, sedimentation, degraded water quality, and elevated temperature. Pesticides have been detected in SR basin steelhead freshwater habitats, including eptam, atrazine, desethylatrazine, metolachlor, and alachlor.

SR basin steelhead are generally classified as summer-run fish. They enter the Columbia River from late June to October. After remaining in the river through the winter, SR basin steelhead spawn the following spring (March to May). Juveniles typically rear in floodplain habitats associated with their natal rivers and streams for more than a year. SR basin steelhead typically smolt after two or three years. During their freshwater residence they may be exposed to pesticides used for a variety of uses. Potential exposure from use within the DPS includes use on evergreen forests (52%), agricultural lands including use on cultivated crops (8%) and hay/pasture (1%), and use in urban/residential or other developed areas (2%). Our GIS analysis indicates 15 populations of this DPS could be exposed to pesticides applied for these uses during their freshwater rearing period. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 155.

Table 155. Snake River steelhead

a.i.	Co-occurrence			Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Populations	Species
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Orchard crops only	NA	NA	Low	Low
Disulfoton	Yes	Yes	Yes	Low	Low
Methamidophos[1]	Potatoes only	NA	NA	Low	Low
Methidathion	Yes	NA	NA	Low	Low
Methyl parathion	Yes	NA	NA	Low	Low
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Medium	Low
Naled	Yes	Yes	Yes	Medium	Medium
Phosmet	Yes	Yes	Yes	High	High
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Medium	Low
Ethoprop	Yes	NA	NA	Medium	Low
Phorate	Yes	NA	NA	High	Medium

1: Crop-specific analysis was conducted
NA: Not applicable

Northern California Steelhead (Threatened Species)

The NC steelhead DPS includes 15 historically independent populations of winter steelhead and 4 extant populations of summer steelhead. The loss of summer-run steelhead populations has significantly reduced the genetic diversity. Most populations are in decline and have low abundances and production. Although the DPS spatial structure is relatively intact, the distribution within most watersheds has been restricted by physical and temperature barriers. Juvenile steelhead remain in fresh water for two or more years, rearing in streams and lagoons.

The major threats to the survival and recovery of this DPS include land use practices, migratory barriers, timber harvest, loss of large woody debris, reduced riparian vegetation, elevated water temperature, increased predation, and barriers that limit access to tributaries.

The percentage of cultivated crop lands and developed lands overlapping with NC steelhead habitat are also less than 1% and 19%, respectively. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 156. Our GIS analysis indicates 19 populations of NC steelhead are exposed to pesticides applied in urban areas. Of these, 15 are also exposed to pesticides applied in agriculture areas. However, there are few areas of concentrated agriculture. Most appears to hay/pasture, concentrated in the Lower Eel watershed and some of the other coastal valleys. Development is concentrated primarily near Eureka, on the coast in the Mad River and Redwood Creek watersheds. Much of the land area in this DPS is heavily forested, and there are a number of state and national parks.

Table 156. Northern California steelhead

a.i.	Co-occurrence			Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Populations	Species
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Yes	NA	NA	Low	Low
Disulfoton	Yes	Yes	Yes	Low	Low
Methamidophos[1]	Yes	NA	NA	NLAA	NLAA
Methidathion	Yes			Low to medium	Low
Methyl parathion	Yes	NA	NA	Low	Low
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Low	Low
Naled	Yes	Yes	Yes	Medium	Medium
Phosmet	Yes	Yes	Yes	Medium	Medium
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Low	Low
Ethoprop	Yes	NA	NA	Low	Low
Phorate	Yes	NA	NA	Low	Low

1: Crop-specific analysis was conducted
NA: Not applicable

Central California Coast (CCC) Steelhead (Threatened Species)

The CCC steelhead DPS includes nine historic independent populations, all of which are nearly extirpated. Data on abundance and population growth rates are scarce, but available information strongly suggests that no population is viable. The loss of spatial structure and hatchery influences have likely reduced the genetic diversity for this DPS. Juvenile steelhead remain in fresh water for one or more years rearing in small tributaries and floodplain habitats. Age to smoltificaton for this DPS is typically 1 to 4 years. Steelhead have a more adaptive life history than some of the other salmon species, including overlapping generations and iteropary.

The major threats to this DPS include dams and other migration barriers, urbanization and channel modification, agricultural activities, predators, hatcheries, and water diversions. Throughout the species' range, habitat conditions and quality have been degraded by a lack of channel complexity, eroded banks, turbid and contaminated water, low summer flow and high water temperatures, and restricted access to cooler head waters from migration barriers.

High densities of crop farming occur throughout the San Joaquin Basin, the Delta, and along the lower Sacramento River. There is also agriculture in the Russian River valley. The Russian River population is one of the largest runs. Southern portions of DPS include the heavily developed areas around San Francisco Bay. The percentage of cultivated croplands and developed lands that overlap with CCV steelhead habitat are 27% and 10%, respectively. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 157. Most of the watersheds in this DPS are heavily developed, and/or have intensive agriculture in the river valley. A number of the populations must migrate through the San Francisco-San Pablo-Suisan Bay estuarine complex, which is heavily influenced by input from California's Central Valley.

Table 157. Central California Coast steelhead

a.i.	Co-occurrence			Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Populations	Species
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Yes	NA	NA	Low	Low
Disulfoton	Yes	Yes	Yes	Medium	Low
Methamidophos[1]		NA	NA	Low	Low
Methidathion	Yes			High	High
Methyl parathion	Yes	NA	NA	High	Low
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Medium	Medium
Naled	Yes	Yes	Yes	High	High
Phosmet	Yes	Yes	Yes	High	High
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Medium to high	Medium
Ethoprop	Yes	NA	NA	Medium	Medium
Phorate	Yes	NA	NA	High	High

1: Crop-specific analysis was conducted
NA: Not applicable

California Central Valley (CCV) Steelhead (Threatened Species)

The CCV steelhead DPS consisted of 81 historical and independent populations. The spatial structure of the CCV steelhead 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 for this DPS (NMFS 2009). Population losses and reduction in abundance have reduced the genetic diversity that existed within the DPS.

The major threats to the survival and recovery of this DPS include dams and other migration barriers, urbanization and channel modification, agricultural activities, non-native predators, hatcheries, large scale water management and diversions, habitat degradation, increased water temperature, and decreased water quality from contaminants including pesticides. Numerous NAWQA, CDPR, and other assessments found high concentration of contaminants in both the San Joaquin and Sacramento Rivers and their tributaries. Monitoring in the San Joaquin basin found seven pesticides exceeded criteria for the protection of aquatic life which included OPs.

The percentage of agriculture, developed, and forested lands that overlap with CCV steelhead habitat are 32%, 10%, and 58%, respectively. Heavy use of agricultural pesticides and the high probability of mixtures increase likelihood of negative effects for this species. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 158. Our GIS analysis indicates that CCV steelhead are exposed to pesticides applied in urban areas and agriculture areas. Juveniles feed and rear in a variety of habitats, including the Sacramento River, the Delta, non-natal intermittent tributaries, tidal marshes, non-tidal freshwater marshes, and other shallow areas in the Delta as rearing areas for short periods during out-migration to the sea. Essentially there is a continuous run of steelhead throughout the year thus adults and rearing juveniles are in freshwater habitats throughout the year. Juveniles typically rear for multiple years in freshwaters where they rely upon a variety of prey dependent on their age and size.

Table 158. California Central Valley steelhead

a.i.	Co-occurrence			Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Populations	Species
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Yes	NA	NA	Low	Low
Disulfoton	Yes	Yes	Yes	Medium	Low
Methamidophos[1]	Yes	NA	NA	Medium	Low
Methidathion	Yes	Yes	NA	High	High
Methyl parathion	Yes	NA	NA	High	Low
Many Use Sites					

a.i.	Co-occurrence			Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Populations	Species
Dimethoate	Yes	Yes	Yes	High	High
Naled	Yes	Yes	Yes	High	High
Phosmet	Yes	Yes	Yes	High	High
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Medium	Medium
Ethoprop	Yes	NA	NA	Medium	Medium
Phorate	Yes	NA	NA	High	High

1: Crop-specific analysis was conducted
NA: Not applicable

South-Central California Coast (S-CCC) Steelhead (Threatened Species)

The S-CCC steelhead DPS includes all naturally spawned steelhead in streams from the Pajaro River to the Santa Maria River. Population growth rates are unknown, though abundances are very depressed. Generally, juvenile steelhead remain in fresh water for one or more years before migrating downstream to smolt. Steelhead have a more adaptive life history than some of the other species, including overlapping generations, and iteropary. Following emergence, fry rear in smaller tributaries and floodplain habitats

Little information is available on the spatial structure or genetic diversity of this DPS. Because of the lack of information as to which populations are more important to the DPS, we have given the benefit of doubt to the species, and assumed that the populations in the mainstem of the Salinas and Pajaro Rivers, both of which have areas of intensive agriculture and development, are important.

The major obstacles to the survival and recovery of this DPS include dams and other migration barriers, urbanization and channel modification, agricultural activities, wildfires, eroded banks, increased water temperature, and decreased water quality from contaminants.

The percentage of cultivated crop lands and developed lands that overlap with this DPS' range are 7% and 10%, respectively. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 159. Agriculture is the dominant land use in the Salinas River valley, and there are areas of intense agriculture in the Pajaro watershed as well. Areas higher in the Salinas and Pajaro watersheds and along some of the coastal areas are less affected. Crops for which phorate may be used appeared unlikely to be planted in this area.

Table 159. South-Central California Coast steelhead

a.i.	Co-occurrence			Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Populations	Species
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Yes	NA	NA	Low	Low
Disulfoton	Yes	Yes	Yes	Medium	Low
Methamidophos[1]		NA	NA	Low	Low
Methidathion	Yes	No	No	High	High
Methyl parathion	Yes	NA	NA	High	Low
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Medium	Medium
Naled	Yes	Yes	Yes	High	High
Phosmet	Yes	Yes	Yes	High	High
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Medium	Medium
Ethoprop	Yes	NA	NA	Medium	Medium
Phorate	Yes	NA	NA	Medium	Medium

1: Crop-specific analysis was conducted
NA: Not applicable

Southern California (SC) Steelhead (Endangered Species)

The SC steelhead DPS includes populations in five major and several small coastal river basins in California from the Santa Maria River to the U.S.–Mexican border. Long-term estimates and population trends are lacking for the streams within the DPS. The DPS experiences reduced and fragmented distribution, and large variations in annual spawner runs. Abundance is extremely low. Genetic variability in this DPS is of particular interest, as SC steelhead can withstand higher water temperatures. SC steelhead juveniles may rear in fresh water or at the upper end of coastal lagoons for the first or second summer before migrating downstream to smolt.

The major threats to this DPS include dams and other migration barriers, urbanization and channel modification, agricultural activities, wildfires, and compromised water quality. The NAWQA analysis detected more than 58 pesticides in ground and surface waters within the heavily populated Santa Ana basin, including multiple AChE inhibitors.

The percentage of cultivated crop lands and developed lands within SC steelhead habitat are about 5% and 34%, respectively. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 160. Three counties within this DPS are included within the fire ant use area for phosmet. Although presumably treated areas are small, the application rate for this use is high (listed in sq. ft, but equivalent to 379 lb a.i./A when scaled up). All of the rivers are affected by anthropogenic inputs.

Table 160. Southern California steelhead

a.i.	Co-occurrence			Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Populations	Species
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Yes	NA	NA	Low	Low
Disulfoton	Yes	Yes	Yes	Medium	Low
Methamidophos[1]		NA	NA	Medium	Low
Methidathion	Yes	No	No	High	High
Methyl parathion	Yes	NA	NA	High	Low
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Medium	Medium
Naled	Yes	Yes	Yes	High	High
Phosmet	Yes	Yes	Yes	High	High

a.i.	Co-occurrence			Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Populations	Species
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Medium	Medium
Ethoprop	Yes	NA	NA	Medium	Medium
Phorate	Yes	NA	NA	Medium	Medium

1: Crop-specific analysis was conducted
NA: Not applicable

Table 161. Species Calls for a.i.s undergoing cancellation

Species	ESU	Undergoing Cancellation					
		Fenamiphos	Azinphos methyl	Disulfoton	Methamidophos	Methidathion	Methyl parathion
Chinook	Puget Sound		Low	Low	Low	High	Low
	Lower Columbia River		Low	Low	Low	Medium	Low
	Upper Columbia River Spring - Run		Low	Low	Low	High	Low
	Snake River Fall - Run		Low	NLAA	NLAA	Medium	Low
	Snake River Spring/Summer - Run		Low	NLAA	NLAA	Low	Low
	Upper Willamette River		Low	Low	Low	Medium	Low
	California Coastal		Low	Low	Low	Medium	Low
	Central Valley Spring - Run		Low	Low	Low	High	Low
	Sacramento River Winter - Run		Low	Low	Low	High	Low
Chum	Hood Canal Summer - Run		Low	Low	Low	Low	Low
	Columbia River		Low	Low	Low	Low	Low
Coho	Lower Columbia River		Low	Low	Low	Low	Low
	Oregon Coast		Low	Low	Low	Low	Low
	Southern Oregon and Northern California Coast		Low	Low	NLAA	Medium	Low
	Central California Coast		Low	Low	Low	High	Low
Sockeye	Ozette Lake		No Effect	NLAA	No Effect	NLAA	No Effect
	Snake River		Low	NLAA	NLAA	Low	Low
Steelhead	Puget Sound		Low	Low	Low	High	Low
	Lower Columbia River		Low	Low	Low	Low	Low
	Upper Willamette River		Low	Low	Low	Medium	Low
	Middle Columbia River		Low	Low	Low	High	Low
	Upper Columbia River		Low	Low	Low	High	Low
	Snake River		Low	Low	Low	Low	Low
	Northern California		Low	Low	NLAA	Low	Low
	Central California Coast		Low	Low	Low	High	Low
	California Central Valley		Low	Low	Low	High	Low
	South-Central California Coast		Low	Low	Low	High	Low
	Southern California		Low	Low	Low	High	Low

Table 162. Species calls for static a.i.s

Species	ESU	Many Uses			Limited Uses		
		Dimethoate	Naled	Phosmet	Bensulide	Ethoprop	Phorate
Chinook	Puget Sound	Medium	High	High	Medium	Low	Medium
	Lower Columbia River	Medium	High	High	Medium	Low	High
	Upper Columbia River Spring - Run	Medium	High	High	Low	Low	High
	Snake River Fall - Run	Low	Medium	Medium	Low	Low	Low
	Snake River Spring/Summer - Run	Low	Medium	Medium	Low	Low	Low
	Upper Willamette River	Medium	High	High	Medium	Medium	High
	California Coastal	Low	High	High	Low	Medium	High
	Central Valley Spring - Run	High	High	High	High	Medium	High
	Sacramento River Winter - Run	High	High	High	High	Medium	High
Chum	Hood Canal Summer - Run	Low	High	Low	Low	Low	Low
	Columbia River	Low	High	Low	Low	Low	Low
Coho	Lower Columbia River	Medium	High	High	Medium	Low	High
	Oregon Coast	Low	Medium	Medium	Low	Low	Low
	Southern Oregon and Northern California Coast	Medium	High	High	Medium	Medium	Medium
	Central California Coast	Low	High	High	Low	Medium	High
Sockeye	Ozette Lake	NLAA	Low	Low	NLAA	NLAA	No Effect
	Snake River	Low	Medium	Low	Low	Low	Low
Steelhead	Puget Sound	Medium	High	High	Medium	Low	Medium
	Lower Columbia River	Medium	High	High	Medium	Low	High
	Upper Willamette River	Medium	High	High	Medium	Medium	High
	Middle Columbia River	Medium	High	High	Medium	Medium	High
	Upper Columbia River	Medium	High	High	Low	Low	High
	Snake River	Low	Medium	High	Low	Low	Medium
	Northern California	Low	Medium	Medium	Low	Low	Low
	Central California Coast	Medium	High	High	Medium	Medium	High
	California Central Valley	High	High	High	Medium	Medium	High
	South-Central California Coast	Medium	High	High	Medium	Medium	Medium
	Southern California	Medium	High	High	Medium	Medium	Medium

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. We apply the analysis framework discussed in the Approach to the Assessment section. Figure 1 shows the pathway of analysis conducted in this section. 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 (Figure 69; and Figure 1). 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 that comprise critical habitat. Water quality and prey availability 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 chemicals rather than affecting other parameters, we use a concentration of a.i. likely to adversely affect fish as our measure of reduction. 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. We then determine whether the response of PCEs in designated critical habitat is sufficient to reduce the conservation value of designated critical habitat for an ESU/DPS in the action area.

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 (quantity, quality, and/or distribution) 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. For watersheds of medium or low conservation value we examined the relative number, spatial distribution, and magnitude of effects to the exposed watersheds to determine whether the conservation values. We assess these watersheds within the *Integration and Synthesis for Designated Critical Habitat* section.

NMFS has designated critical habitat for 26 of 28 listed Pacific salmonids. The two species lacking final critical habitat designations are under development and include the LCR coho salmon and Puget Sound steelhead. The action area for this Opinion encompasses all designated critical habitat for listed Pacific salmon and steelhead. The PCEs for each listed species, where they have been designated, 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[15]; and
6. offshore marine areas.

Water quality and prey availability (forage) 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 the 12 a.i.s, we do not expect offshore marine areas to be affected. Therefore, a risk hypothesis was not developed for offshore marine areas and further evaluation of this PCE is unwarranted. For nearshore marine areas, only Puget Sound nearshore marine areas are expected to substantially overlap with the stressors of the action resulting in degraded water quality and reduced prey availability. We do not expect other ESU/DPS nearshore marine areas listed as critical habitat to receive sufficient loading to impair water quality and prey availability. The large volume of ocean water combined with environmental fate profiles of the stressors suggest limited effects in the nearshore marine areas outside of Puget Sound.

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. In addition to the exposure and response data, we planned on highlighting instances of water bodies not meeting local, state, or federal water quality standards and criteria for the 12 pesticides. As none of the 12 a.i.s have water

[15] 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).

quality criteria currently available, we do not use 303(d) lists as an additional line of evidence for demonstrating the potential degradation of water quality.

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. 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 (EC50/LC50), prey growth, prey drift, prey reproduction, prey abundance, health condition of invertebrate aquatic communities (using indices of biological integrity), and recovery of aquatic communities following pesticide exposure to determine whether expected concentrations are sufficient to affect PCEs for salmonid critical habitats. Given the environmental baseline conditions of many of the aquatic systems, the existing invertebrate community may already be depauperate and may require longer periods for populations of prey to rebound to pre-exposed levels following each pesticide application event. We therefore evaluate available information on the biological integrity of aquatic invertebrate communities (where available). These data were evaluated within the *Effects of the Proposed Action* section.

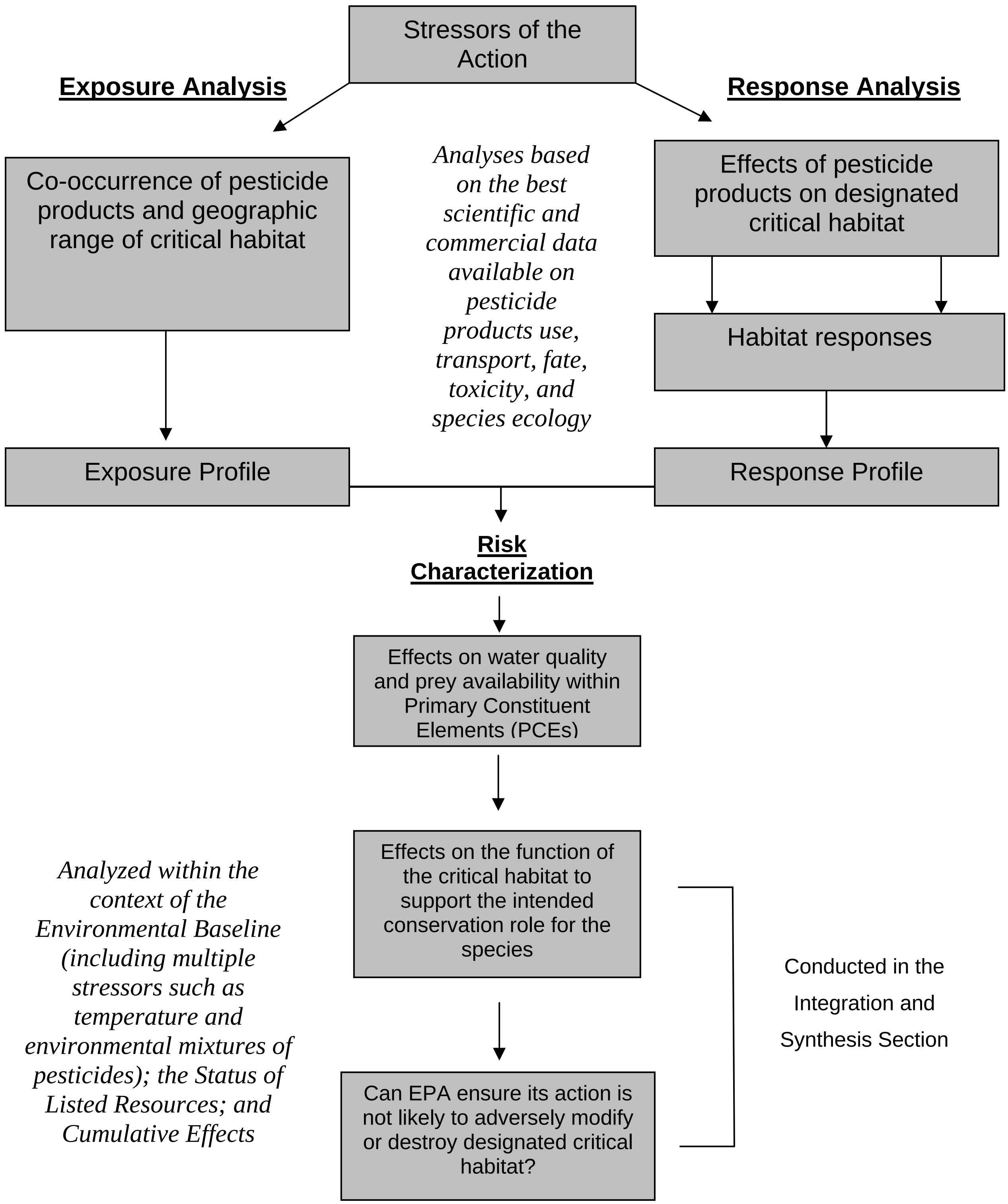

Figure 69. Assessment Framework for Designated Critical Habitat

Exposure of designated critical habitats to the stressors of the action:

All designated critical habitat is located within the action area and overlaps with the allowable uses of the authorized pesticide formulations. The stressors of the action contaminate salmonid habitats via drift, runoff (including from irrigation returns), and atmospheric deposition (see *Exposure* section). Once in aquatic systems, the 12 a.i.s show a wide range in degradation rates *i.e.*, persistence, depending on their physical properties as well as the chemical, biological, and physical environment of the contaminated aquatic habitats. In some circumstances, a.i.s degrade into chemicals that remain toxic and are more resistant to degradation than the parent (*e.g.*, naled degrades to dichlorvos). In other cases, an a.i. may degrade rapidly within aquatic systems (*e.g.*, phosmet). Expected concentrations of other/inert ingredients and adjuvants added to formulations prior to application remain unknown - a substantial data gap. Table 163 shows expected concentrations of the 12 a.i.s and their 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, estuarine, and nearshore marine PCEs. The vast majority of exposure information applies more readily to freshwater habitats compared to estuarine and nearshore marine habitats where much less information is available.

Table 163. Expected levels of the 12 active ingredients in aquatic ecosystems

CONCENTRATION	Azinphos methyl	Bensulide	Dimethoate	Disulfoton	Ethoprop	Fenamiphos	Methamidophos	Methidathion	Methyl parathion	Naled	Phorate	Phosmet
EPA peak PRZM/EXAMS estimates for farm pond	1.9 - 40.6	7.2 - 231	0.1 - 58.3	7.1 - 67	15 – 75	0.3 - 35.4	30 - 65	8.9 - 15.5	1.3 - 18.2	0.8 - 33	4.6 - 138	3.0 - 29.9
NMFS AgDrift estimates for floodplain habitat	0.8 - 11.4	1,100 – 2,940	46 – 652	16 - 237	6 – 24	No active labels	267 - 490	66 – 1,860	134 - 980	251 - 921	NA - only granular	5.0 - 2,920
Monitoring data	0.001 - 670	0.001- 2,840	0.001 - 11.6	0.001 - 48.7	0.001- 241	0.001 - 520	0.001 - 0.13	0.001 - 15.1	213	na	0.001- 32.3	0.001- 0.63

Responses of salmonid habitats to the stressors of the action:

If PCEs are exposed to the stressors, we evaluate the level at which the a.i.s, any toxic degradates, and inert/other ingredients adversely affect water quality and prey availability. 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 the 12 a.i.s can result in concentrations that reduce salmonid survival, growth, reproduction, and essential behaviors, all which independently translate to a degradation of water quality (Table 117 from *Response Analysis*). We have also demonstrated that many of the a.i.s, through mortality, can reduce growth rates (lambdas) of salmonid populations (see *Salmonid Population Model* section). These types of individual and population-level effects demonstrate a severe degradation of water quality in affected habitats. Even more pronounced is the effect of the stressors of the action on the prey community (forage attribute of multiple PCEs) that supports salmonid growth and development (purposes for which the habitat was designated), and ultimately the successful completion of their life cycle (required for species conservation). We modeled the effects of reduced prey availability to foraging juveniles and found that juvenile growth was retarded - resulting in reduced survival. Reductions in juvenile survivorship resulted in reductions in an affected population's growth rate (Table 123 through Table 127 in *Population Modeling* section).

We summarized the available toxicity information in the *Response Analysis* and also present prey toxicity information below by a.i. in Table 164. It is important to note that the toxicity of the a.i.s 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. The most toxic of the 12 a.i.s based on acute lethality and AChE inhibition in salmonids are azinphos methyl, fenamiphos, methidathion, naled, phorate, and phosmet. Dimethoate, disulfoton, ethoprop, methamidophos are less toxic based on acute lethality to salmonids. All of the a.i.s are considered highly toxic to invertebrates with azinphos methyl, methidathion, naled, phorate, and phosmet expected to be of greatest toxicity compared to the other a.i.s.

Table 164. Effect concentrations of the twelve active ingredients in aquatic ecosystems

Assessment Endpoints	Azinphos methyl	Bensulide	Dimethoate	Disulfoton	Ethoprop	Fenamiphos	Methamidophos	Methidathion	Methyl parathion	Naled	Phorate	Phosmet
Salmonid survival (LC50)	1.2 - 27.5	720 – 1,100	6,200 – 7,500	1,850 - 13,900	1,020 - 13,800	68 - 563	25,000 - 51,000	6.6 - 14	1,850 – 5,300	87 - 345	13 - 66	150- 1,560
Olfactory-mediated behaviors	na	na	na	Na	Na	na	Na	na	na	na	na	na
Fish reproduction (LOEC)	0.40	na	na	2.9 - 32.9	21 – 54	na	Na	na	na	na	na	6.1
Fish growth (LOEC)	0.4 - 0.98	na	840.00	420.00	11.00	7.40	Na	12.00	10 - 380	15.00	4.2 - 190	6.1
Swimming	0.36 - 4,810	na	na	Na	na	na	4,500 - 16,100	na	3.5 - 300	na	na	na
AChE inhibition (95% CI of EC50)[1]	0.10 - 0.26	na	195.7 - 382	112.3 – 2,118	69.5 - 118.2	na	Na	0.47 - 2.68	21.2 - 39.0	6.5 - 9.5	0.42 - 0.76	2.5 - 4.2
Prey Survival	0.16 - 56	62.4 – 3,330	43 - 15,000	5 – 100	44 - 93	1.3 -10,000	0.042 – 1,054	3 - 7.2	0.14 - 28	0.14 - 230	0.3 - 65	1.6 - 3,400
Primary production	na	1,500 – 2,800	Na	Na	na	na	Na	na	na	na	na	

Risk Characterization: Evaluation of Critical Habitat Risk Hypotheses

We use the toxicity information presented earlier in the *Effects of the Proposed Action* section to evaluate the scientific lines of evidence that support or refute risk hypotheses developed for critical habitats. We determined that freshwater spawning and rearing sites, migration corridors, estuarine areas, and nearshore marine areas within designated critical habitats are likely to be exposed to the stressors of the action over the 15-year registration duration. We do not evaluate fenamiphos in this analysis because there are no active labels due to the cancellation of all fenamiphos products. We will discuss, and when possible, 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 and prey availability within freshwater spawning habitat, freshwater rearing habitat, migration corridors, estuarine areas, and nearshore marine areas will rise to the level expected to reduce the intended conservation role of designated critical habitats - evaluated within the *Integration and Synthesis for Designated Critical Habitat* section. The final conclusion of whether EPA's proposed actions 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 in freshwater spawning sites.

Freshwater spawning sites require water quality conditions that support spawning, incubation, and larval development. The degradation of water quality by exposure to the stressors of the action is indicated via the toxic responses in a variety of aquatic organisms including listed salmonids. Based on allowable application timings of the pesticide products, we expect episodes of water quality degradation to coincide with spawning events within spawning habitats. The levels of contamination expected are highly variable resulting from the diversity of species spawning habitats (small, shallow first and second order streams to mainstem rivers with variable flow patterns) and year-to-year variation in climate and pesticide applications. All 11 a.i.s are expected to attain concentrations that degrade water quality within spawning PCEs at some point during the 15-year registration period. The most severe effects to water quality within spawning sites will be those sites that experience multiple applications of the a.i.s, are shallow, low flow systems with elevated water temperatures, and are located in high pesticide use areas such as intensive agricultural or urbanized watersheds.

Contamination of spawning sites by the stressors of the action can degrade water quality in several ways. Exposure to the stressors of the action can impair spawning rituals as well as kill spawning adults which diminishes the purposes for which spawning sites were identified as critical habitat. Expected concentrations of many of the OPs subject to this Opinion are sufficient to kill a percentage of spawning adults. Other spawners may experience impaired olfaction which leads to reduced ability to detect spawning events. For those salmonids with impaired nervous systems, due to the irreversible binding of OPs to the enzyme acetylcholinesterase, reduced swimming ability can reduce spawning success. Rigorous swimming is necessary to complete the intensive act of spawning. We expect the concentrations of many of the OPs to impair swimming based on the degree to which AChE is inhibited, effectively reducing the ability of the salmonids to spawn. Based on the use of the 11 OPs, mixtures containing two or more OPs within spawning areas are likely which would intensify the toxic responses of exposed aquatic habitats. Other ingredients and degradates of the 11 OPs may also result in degraded water quality. If the spawning areas are in locations that experience elevated water temperatures, we expect exposure to the OPs would result in greater toxicity.

Collectively, the overlap of spawning sites with application areas combined with expected concentrations and toxicity effect thresholds to aquatic organisms indicates that degradation of the water quality attribute of the spawning PCE is likely. We evaluate whether the degradation of this PCE, sites for spawning, in combination with other affected PCEs reduce the conservation value of the 26 designated critical habitats within the *Integration and Synthesis for Designated Critical Habitat* section.

Risk hypothesis 2. Exposure to the stressors of the action is sufficient to degrade water quality and /or reduce prey availability in freshwater rearing sites.

Freshwater rearing sites need to provide good water quality and abundant forage to support juvenile development. Reductions in either, can limit the existing and potential carrying capacity of rearing sites and subsequently reduce their conservation value. Recovery of listed salmonid populations 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. Chum salmon utilize fresh water for the shortest periods (generally a few days). Chinook, coho, steelhead, and sockeye salmon spend much longer periods rearing in freshwater systems with steelhead trout spending up to several years before ocean migration. Freshwater rearing area are diverse, extensive, complex sites that can range from small, shallow, intermittent floodplain habitats to channel edges of large river systems. As such, expected concentrations range from some of the highest estimates (via spray drift into floodplain habitats; Table 163) to some of the lowest estimates (monitoring results from large rivers, Table 163).

Many freshwater salmonid rearing sites are located in floodplains where shallow, low flow habitats are at high risk of pesticide drift and runoff. These habitats provide some of the most important foraging areas for developing juveniles. Expected floodplain concentrations of 10 of the OPs are shown in Table 163. At these levels, water quality and many types of salmonid prey would be severely affected if the lower end of the survival range in Table 164 is representative of salmonid prey communities. We located few data on the response of real world prey communities to the 10 a.i.s; reduced prey communities in the field were correlated to reduced prey availability in steelhead habitat. We assume that many of salmonid prey items are either as sensitive as or more sensitive than the standard toxicity test organisms for which data exist. This assumption is supported, in part, by comparative data showing that caddisflies, stoneflies, and mayflies are typically more susceptible to pesticide toxicity than laboratory-reared freshwater invertebrates (Peterson et al. 2001; Johnson et al. 2008). Reductions and removal of prey biomass in rearing habitats may substantially reduce this PCE’s role in recovering salmonid populations. Concentrations in other freshwater habitats that support rearing are also expected to reach levels that reduce both water quality and prey abundance (Table 163).

The IBI and other metrics of aquatic community health were reviewed. In areas of intensive agriculture, where we expect use of the stressors of the action, biological integrity is often significantly reduced (Cuffney et al 1997). Many of the preferred salmonid prey items are present only in low numbers or absent altogether in these areas. We see similar depauperate communities in urban areas. We recognize many other limiting factors contribute to the poor condition of these aquatic communities. However, these insecticides and their formulations may be responsible for a portion of these reductions. In fact, several studies have shown toxicity to salmonid prey items from field collected waters and sediment due to OP insecticides (Sommer et al 2001, Sommer et al 2005).

Based on the use of the 11 OPs, mixtures containing two or more OPs within rearing areas are likely and would intensify the toxic responses of exposed aquatic habitats. Other ingredients and degradates of the 11 OP- containing formulations may result in aquatic toxicity, further degrading water quality. If the rearing areas are in locations that experience elevated water temperatures, we expect exposure to the 11 OPs would result in greater toxicity to this PCE. Many areas designated as critical habitats are listed under EPA's 303(d) program for elevated water temperatures (Table 56 of *Environmental Baseline* section).

Collectively, substantial data indicated that expected concentrations of the stressors of the action are sufficient to adversely affect both water quality and salmonid prey (forage) of freshwater rearing 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).

Risk hypothesis 3. Exposure to the stressors of the action is sufficient to degrade water quality and/or reduce prey availability in freshwater migration corridors.

Freshwater migration corridors require good water quality and sufficient prey abundance to support juvenile and adult mobility and survival. Contaminating these sites with the stressors of the action degrades water quality and further impedes the mobility and survival of juveniles and adults. Expected contaminant concentrations may limit prey availability in migratory sites where juveniles pause to rest and feed during their migration to the ocean. Frequently rest areas such as undercut banks, side channels, submerged and overhanging large wood, log jams, and beaver dams are far and few between in many West Coast salmonid producing streams and rivers. Salmonid recovery plans call for restoration of these sites to improve juvenile survival and overall fitness. Lack of adequate prey resources due to the degradation of water quality at these rest areas may cause migrating juveniles to continue downstream thus avoiding needed rest and food, ultimately affecting their health and ability to successfully transition to saltwater environments. Many of these rest areas are located in places where water flow is reduced compared to the main channels. Stressors of the action may persist longer in these areas due to reduced flow. Additionally, channel-edge habitats that are proximate to applications of the stressors of the action are at risk, increasing the probability of exposure to high concentrations from drift and runoff following application events. Many migratory sites overlap with some of the highest use areas for the stressors of the action such as intensive agricultural valleys. Based on the size, flow rate, and proximity to application sites, exposure durations and concentrations within migratory habitats are expected to be highly variable. That said, we expect uses of the 11 OPs will at times lead to concentrations that will degrade water quality and kill salmonid prey. The degradation of water quality within migratory sites may affect the mobility of juveniles and adults by impairing their olfaction and their swimming ability as well as their survival when concentrations reach lethal concentrations. The 11 a.i.s are expected to degrade water quality. In migratory sites where elevated water temperatures co-occur with the OPs the degradation of water quality is intensified as OPs become more toxic at elevated temperatures. Additionally, when the 11 OPs occur in various combinations, additive and synergistic toxicity is expected - a further degradation of water quality.

Collectively, the available data indicated that expected concentrations of the stressors of the action are sufficient to adversely affect water quality and salmonid prey (forage) of migratory 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).

Risk hypothesis 4. Exposure to the stressors of the action is sufficient to degrade water quality 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 prey resources sufficient to support growth and maturation. Prey resources for Pacific salmonids within estuaries include a diverse group of organisms - from aquatic invertebrates to small fishes depending on the size of the salmonid. The allowable uses of the stressors of the action overlap with estuaries designated as critical habitat.

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). We located no estuarine monitoring data specific to the stressors of the action. This is a large data gap as the available exposure data derived for freshwater habitats (EPA modeling estimates, NMFS modeling estimates and monitoring data) are not representative of estuarine habitats. Pacific estuaries are incredibly variable to one another; size, tidal volume, exchange rate, freshwater input, salinity, watershed land uses, trophic structures, bathometry, etc., influence and shape estuarine ecosystems (Salo 1991). Estuaries remain dynamic, complex systems that are not completely understood. As such, predictive models are not available to estimate concentrations of pesticides within estuaries. Therefore, we evaluate whether applications of the stressors of the action are allowed within estuarine-containing watersheds and if so, we assume they may contaminate estuarine habitats. Dissipation rates are another source of uncertainty that remains an active area of research. All of the a.i.s except for fenamiphos (no currently registered labels) are allowed for use in estuarine-containing watersheds.

Naled is registered for mosquito and fly control within swamps and tidal marshes as well as on a variety of other use sites (See *Description of the Proposed Action* section). For these uses, either ground or aerial applications are allowed with a maximum application rate of 0.1 lbs a.i./acre. There are no apparent restrictions within applications to estuaries which may be considered tidal marshes. We expect direct loading to estuarine habitats via this use.

The available toxicity information for estuarine and marine organisms for the a.i.s is presented inTable 165. The majority of aquatic toxicity data are from survival assays for the sheepshead minnow (fish) and mysid (estuarine invertebrate). The 11 a.i.s are all toxic to aquatic invertebrates as evidenced by ppb toxicity values. It is unclear how representative mysids are for other salmonid prey items.

Table 165. Assessment endpoint toxicity values (µg/L) for saltwater aquatic organisms presented in salmonids BEs, CRLF BEs, REDs, IREDs, and EFED Science Chapters. Abbreviations as follows: NR = Not Reported; T = Technical grade; Formulation = formulated product; a96 h test; b48 h test.

Azinphos methyl			
Assessment endpoint	**Assessment measure**	**> 95% a.i. (% a.i.)**	**< 95% a.i. (% a.i.)**
Survival	estuarine, and marine fish LC_{50} (96 h)	Striped mullet (*Mugil cephalus*) (96%; T) = 3.2[b] Spot (*Leiostomus xanthurus*) (96%; T) = 28[b]	Sheepshead minnow (*Cyprinodon variegatus*) (89%; T) = 2.7 Sheepshead minnow (*C. variegatus*) (% a.i. NR) = 2 Threespine stickleback (*Gasterosteus arculeatus*) (Formulation, % a.i. NR) = 4.8; 12.1
Reproduction or larval survival	NOEC/LOEC		Sheepshead minnow (*C. variegatus*) (92.5%) = 0.2/ 0.4
Habitat-salmonid prey	invertebrate survival (48 h EC/LC50)	Brown shrimp (*Penaeus aztecus*) (96%) T= 2.4[b]	Mysid (*Mysidopsis bahia*) (89%) = 0.21[b] Mysid (*M. bahia*) (22% Guthion 2L) = 0.26[b]
Bensulide			
Assessment endpoint	**Assessment measure**	**> 95% a.i. (% a.i.)**	**< 95% a.i. (% a.i.)**
Survival	Estuarine, and marine fish LC_{50} (96 h)		Sheepshead minnow (*Cyprinodon variegatus*) (92%; T) = 560 Spot (*Leistomus xanthurus*) (95%; T) = 320
Habitat-salmonid prey	Invertebrate survival (48 h EC/LC50)		Mysid (*Americamysis bahia*) (92%; T) = 62.4[a]
Dimethoate			

Assessment endpoint	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)
Survival	Estuarine, and marine fish LC_{50} (96 h)	Longnose killifish (*Fundlus similis*) (99.3%; T) = >1,000[b] Sheepshead minnow (*Cyprinodon variegatus*) (99.1%; T) = >111,000	
Habitat-salmonid prey	Invertebrate survival (48 h EC/LC50)	Salt marsh mosquito (*Aedes taeniorhychus*) (> 95%; T) = 31[b] Mysid shrimp (*Mysidopsis bahia*) (99.1%; T) = 15,000[a] Brown shrimp (*M. azteccus*) (99.3%; T) = >1,000[b] Brine shrimp (*Artemia* sp.) (>95%) = 15,730[b]	
Disulfoton			
Assessment endpoint	**Assessment measure**	**> 95% a.i. (% a.i.)**	**< 95% a.i. (% a.i.)**
Survival	Estuarine, and marine fish LC_{50} (96 h)	Sheepshead minnow (*Cyprinodon variegatus*) (sw) (95.5%) = 520 Sheepshead minnow (*C. variegatus*) (sw) (97.8%) = >1,000	
Reproduction or larval survival	NOEC/LOEC	Sheepshead minnow (*C. variegatus*) (sw) (98%) = 0.96/2.9	Sheepshead minnow (*C. variegatus*) (sw) (94.7%) = 16.2/32.9
Habitat-salmonid prey	Invertebrate survival (48 h EC/LC50)	Mysid (*Americamysis bahia*) (97.8%) = 100 Brown shrimp (*Penaeus aztecus*) (95.5%) = 15	
	Invertebrate growth NOEC/LOEC	Mysid (*Americamysis bahia*) (98.5%) = 2.35/8.26	
Ethoprop			
Assessment endpoint	**Assessment measure**	**> 95% a.i. (% a.i.)**	**< 95% a.i. (% a.i.)**

Survival	Freshwater, estuarine, and marine fish LC_{50} (96 h)	Sheepshead minnow (*Cyprinodon variegatus*) (96.8%;T) = 180; 958 Pinfish (*Logodon rhomboides*) (95%; T) = 6.3 Spot (*Leiostomus xanthurus*) (95%; T) = 33	
Reproduction or larval survival	NOEC/LOEC	Sheepshead minnow (*C. variegatus*) (95%;T) = 12/21	
Fish growth	NOEC/LOEC	Sheepshead minnow (*Cyprinodon variegatus*) (96.8%; T) = 5.9/11	
Habitat-salmonid prey	Invertebrate survival and growth NOEC/LOEC	Mysid (Americamysis bahia) (95%; T) = 360/ 620 Mysid (*Americamysis bahia*) (96.8%; T) = 1,400/ 2,700	
Methamidophos			
Assessment endpoint	**Assessment measure**	**> 95% a.i. (% a.i.)**	**< 95% a.i. (% a.i.)**
Survival	Estuarine, and marine fish LC_{50} (96 h)		Sheephead minnow (*Cyprinodon variegatus*) (70.1%) = 5,630 (95% CI 4,130-6,890)
Habitat-Salmon prey	Invertebrate survival (48 h EC/LC50)		Mysid (*Americamysis bahia*) (%a.i. NR; T) = 1,054 Blue shrimp (*Penaeus stylirostris*) (%a.i. NR) = 0.16 Eastern Oyster (*Crassostrea viginica*) (72.9%) T = 36,000 (30,000-47,000)
Methidathion			
Assessment endpoint	**Assessment measure**	**> 95% a.i. (% a.i.)**	**< 95% a.i. (% a.i.)**
Survival	Estuarine, and marine fish LC_{50} (96 h)		Sheepshead minnow (*Cyprinodon variegatus*) (25.2%) = 7.8 - 111.9 n=3

Habitat-Salmonid prey	Invertebrate survival (48 h EC/LC50)	Mysid (*Americamysis bahia*) (97.2%)= 0.7 (95 % CI 0.44-0.99)	Mysid (*A. bahia*) (25.2%)2.34; 0.59
Habitat-Salmonid prey	Invertebrate reproduction and growth NOEC/LOEC 21 d		Mysid (*A. bahia*) (%a.i. NR) = 0.02/0.06
		Methyl parathion	
Assessment endpoint	**Assessment measure**	**> 95% a.i. (% a.i.)**	**< 95% a.i. (% a.i.)**
Survival	Estuarine, and marine fish LC_{50} (96 h)	Spot (*Leiostmous xanthrus*) (99%; T) =59 (45 – 74)	Sheephead minnow (*Cyprinodon variegatus*) (43.2%) = 3,400 (2,800-4,100); (90%: T) =12,000 Striped Bass (*Morone saxatilis*) (80%) = 790 (170-1,400)
Habitat-Salmon prey	Invertebrate survival (48 h EC/LC50)	Copepod (*Acartia tonsa*) (99%; T)=28.0 Mysid *(Americamysis bahia)* (%a.i. NR; T) =0.77 (95 % CI 0.64-0.98) Mysid *(A. bahia)* = (99%; T) = 0.78 (95 % CI 0.58-1.1)	Mosquito 4th instar (*Culex tarsalis*)= formulation (%a.i. NR)= 3.6 Mysid (*A. bahia*) (43.2%)= 0.35 (95% CI 0.31-0.39)
Habitiat-Primary productivity	Aquatic plant EC50 (96 h)	Marine diatom (*Skeletonema costatum*) (99%; T) = 5,300 (4,300 – 5,700)	
Habitat-Salmonid prey	Invertebrate reproduction and growth NOEC/LOEC		Mysid (*A. bahia*) (%a.i. NR) = 0.11/0.37
		Naled	
Assessment endpoint	**Assessment measure**	**> 95% a.i. (% a.i.)**	**< 95% a.i. (% a.i.)**
Survival	Estuarine, and marine fish LC_{50} (96 h)		Sheepshead minnow (*Cyprinodon variegatus)* (T, 90% a.i.) = 1,200 Sheepshead minnow (*C.variegatus)* (Formulation, 59.5%) = 1,200
Habitat-salmonid prey	Invertebrate survival (48 h LC/EC50)		Mysid (*Mysidopsis bahia)* (Formulation, 59.6%) = 8.8[a]

	Invertebrate NOEC/LOEC (length and weight)		Mysid *Mysidopsis bahia* (sw) (89.2%; T) = NOEC < 0.2 LOEC = 0.2 (31 d)
Dichlorvos (degradate of Naled)			
Assessment endpoint	**Assessment measure**	**> 95% a.i. (% a.i.)**	**< 95% a.i. (% a.i.)**
Survival	Estuarine, and marine fish LC_{50} (96 h)	Sheepshead Minnow *(Cyprinodon variegatus) =* (98% dichlorvos) = 7350	Sheepshead Minnow *(Cyprinodon variegatus) =* (42.4% dichlorvos) = 14,400
Fish growth	NOEC/LOEC (Length and weight)	Sheepshead minnow *Cyprinodon variegatus* (98% dichlorvos) NOEC = 960 LOEC = 1840 (34 days)	
Habitat-salmonid prey	Invertebrate survival (48 h LC/EC50)	Mysid (*Mysidopsis bahia)* = (98% dichlorvos) = 19;	Mysid (*Mysidopsis bahia)* = (42.4% dichlorvos) = 44
	Invertebrate NOEC/LOEC (length and weight)	Mysid (*Mysidopsis bahia)* (98% dichlorvos) NOEC = 1.48 LOEC = 3.25 length, growth: (28 d)	
Phorate			
Assessment endpoint	**Assessment measure**	**> 95% a.i. (% a.i.)**	**< 95% a.i. (% a.i.)**
Survival	Estuarine, and marine fish LC_{50} (96 h)		Sheepshead minnow (*Cyprinodon variegatus*) (90%) = 1.3 Sheepshead minnow (*C. variegatus)* (89.5%) = 4 (95% CI 3.5 – 4.5) Sheepshead minnow (*C. variegatus*) (20%G, % a.i. NR) = 8.2 (95% CI 5.5 – 10) Longnose Killifish (*Fundulus similis*) (90%) = 0.36[b] Spot (*Leistomus xanthurus*) (89.5%) = 5 (95% CI 4.2 – 5.6) Spot (*L. xanthurus*) (90%) = 3.9
Fish growth	NOEC/LOEC	Sheepshead minnow (*Cyprinodon variegatus*) (99%) = 96/190	

Habitat-salmonid prey	Invertebrate survival (48 h EC/LC50)		Mysid (*Americamysis bahia*) (89% a.i.) = 1.9 (95% CI 1.0 – 3.2) Mysid (*A. bahia*) (90%) = 0.31 Mysid (*A. bahia*) (20G%, % a.i. NR) = 0.3; 1.4
	Invertebrate survival and growth rate NOEC/LOEC	Mysid (*A. bahia*) (99%; T) =9/21	
Phosmet			
Assessment endpoint	**Assessment measure**	**> 95% a.i. (% a.i.)**	**< 95% a.i. (% a.i.)**
Survival	Estuarine, and marine fish LC_{50} (96 h)	Longnose killifish (*Fundulus similis)* (95% a.i.; T) = 32[b] Striped mullet (*Mugil cephalus)* (95%; T) = 32[b]	Sheepshead minnow(*Cyprinodon variegatus*) (94%;T)= 170
Habitat-salmonid prey	Invertebrate survival (48 h EC/LC50)	Mysid (*Mysidopsis bahia)* (94.3% T) = 1.6 Brown shrimp (*Penaeus aztecus)* (95%; T)= 2.5	
	Invertebrate reproduction and growth NOEC/LOEC	Mysid *Mysidopsis bahia* (95.5%; T) = NOEC 0.37 LOEC 0.69	

Collectively, the exposure and toxicity information supports that degradation of water quality is expected by the 11 a.i.s based on their expected contamination of estuarine habitats. Prey resources for juveniles may be reduced from pulses of the stressors of the action in high risk areas such as tidal mudflats and channels draining diked agricultural areas where the pesticide products are applied. Adult salmonid forage (small fishes) may be reduced by those a.i.s that are highly toxic. It is difficult to determine at what levels forage is affected given the paucity of exposure and response information. The highest risk to forage fishes are in areas where pesticides persist. We discuss the potential for these stressors to reduce the conservation value of estuarine habitats within the *Integration and Synthesis for Designated Critical Habitat* section.

Risk hypothesis 5. Exposure to the stressors of the action is sufficient to degrade water quality and /or reduce prey availability in nearshore marine areas.

Nearshore marine areas require water quality conditions and forage, including aquatic invertebrates and fishes to support growth and maturation. Similar to estuarine sites, nearshore marine sites have very few data on the behavior and dissipation rates of the stressors of the action. More information, yet still insufficient to make a definitive conclusion, is available on the toxicity to a few saltwater fauna. The available toxicity information shows that the stressors of the action can kill and reduce growth of the tested organisms. The representativeness of these standard test species for salmonid prey is unknown, yet the concentrations of many of the OPs are in the low µg/L range suggesting high acute toxicity to sensitive taxa.

There is no doubt that the stressors of the action contaminate nearshore environments based on allowable uses. However, significant uncertainty arises to the persistence and rate of degradation of the stressors. Fundamental environmental fate data are lacking, not to mention experimental results from environmental realistic exposure scenarios for key salmonid prey taxa including small, forage fish. We found no information on the environmental fate of these materials in nearshore marine habitats. The available toxicity data suggest that the materials are indeed toxic to estuarine and marine organisms, sometimes at ng/L concentrations. Whether and how frequently the stressors of the action attain toxic levels for sufficient durations within nearshore marine environments remains unknown. We anticipate some level of degradation of water quality of these habitats, particularly for those species with nearshore marine areas of designated critical habitat within Puget Sound. In Puget Sound we expect the greatest deposition and loading from allowable applications as compared to other nearshore marine areas along the California, Oregon, and Washington coasts because of the longer residence time of the water. For this reason, we discuss effects to Puget Sound nearshore marine areas within the *Integration and Synthesis for Designated Critical Habitat* section by evaluating land uses proximate to these habitats.

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 each of the five risk hypotheses. We expect water quality and forage to be reduced in spawning, rearing, migratory, estuarine, and nearshore marine habitats. 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.

Integration and Synthesis for Designated Critical Habitat

The *Integration and Synthesis of Effects to Designated Critical Habitat* section describes NMFS' assessment of the likelihood that EPA's registration of azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, methamidophos, methidathion, methyl parathion, naled, phorate, and phosmet will destroy or adversely modify designated critical habitat for 26 of 28 ESUs/DPSs covered in this Opinion. Critical habitat has not been designated for the LCR coho salmon and Puget Sound steelhead.

All species addressed in this Opinion have similar PCEs, as described in the *Effect to Designated Critical Habitat*. 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[16]; and
5. offshore marine areas,

which 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 and a lack of predators. We expect stressors of the proposed action to primarily affect water quality and abundance of prey. Destruction or adverse modification of designated critical habitat is evaluated in this Opinion based on whether the stressors of the action are expected to cause appreciable reductions in water quality or prey abundance.

As noted in many of the recovery plans, during all freshwater life stages, 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

[16] 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).

maintain growth which subsequently reduces freshwater predation mortality, increases overwintering success, initiates smoltification, and improves their survival at sea.

The stressors of the action include the a.i.s, degradates, inert/other ingredients in formulations, surfactants, and tank mixtures; and their individual and collective interactions when applied in agricultural, urban, and residential landscapes throughout the action area. Data are not available for some of these other stressors, thus they are evaluated in a qualitative fashion. Most of our quantitative analysis is based on exposure and response data for only the a.i.s, although there may be substantial toxicity from some of the other stressors of the action.

We start with *Conclusions Regarding Specific a.i.s*, based on the analyses presented in the *Effects of the Proposed Action to Threatened and Endangered Pacific Salmonids* chapter. Inherent in the modeling used to determine some of the EECs is the assumption that the pesticide is applied in a location next to or draining into designated critical habitat. Monitoring data may reflect pesticide applications proximate to the waterbody, or resulting from more distant uses in the watershed or airshed. Modeling EECs and monitoring data are not ESU/DPS specific.

For the *Integration and Synthesis for Designated Critical habitat*, NMFS used a GIS overlay containing landuse classifications and salmon distributions to determine overlap of application sites and designated critical habitat. Because cropping patterns and registered use sites may change over time, landuse classifications (agricultural, forestry, urban/developed) are used rather than specific crops. Details of the GIS analysis and the maps are provided in *Appendix 7*. Occurrence of land uses where specific a.i.s could be applied near salmon-bearing waters for each ESU/DPS is shown in Tables 165-190.

Based on the risk presented in *Conclusions Regarding Specific a.i.s*, the co-occurrence of land uses where that a.i. may be applied, and the conservation values of designated critical habitat we determine the potential for use of that a.i. to adversely affect the PCEs of water quality and prey availability. This is expressed qualitatively as low, medium, or high (Tables 165-190). Taking into account both the unevenness of use, and the conservation value of the various watersheds we then determine the potential for appreciable reduction in that conservation value. We consider the conservation value appreciably reduced if effects on water quality and prey availability degrade the habitat to the point it no longer supports the species. This is expressed qualitatively as low, medium, or high (Tables 165-190). In the *Conclusion* section, we present adverse modification and no adverse modification determinations (Table 196 and Table 197).

Designated Critical Habitat Specific Evaluations for Each a.i.

Below, we summarize the current status of high and medium conservation value watersheds for each species, including baseline stressors. As exposure to a.i.s in salmonid spawning, rearing, and migration habitat is of concern, we highlight exposure from the a.i.s in shallow, more vulnerable habitats. The number of exposed watersheds that co-occur with agricultural and urban areas is also given. Using both chemical and species habitat information, we determine whether each a.i. will co-occur and have negative effects on PCEs and if those effects will cause an appreciable decline in the conservation value of that habitat. Determinations are presented at the end of each species summary, stating whether there is a likelihood of each a.i. causing a negative effect to either water quality or prey abundance. We then determine whether they could result in an appreciable reduction in the value of critical habitat for that species. As part of that determination, we considered the extent of overlap between the ESU/DPS, land use categories for the specific chemicals, cancellation information (where applicable), and the number of watersheds exposed. In some cases, there is considerable overlap across a substantial portion of a species' range. In other cases, there may be overlap only within a portion of a species' range for spawning and rearing, or primarily along migratory corridors.

We present co-occurrence of salmonid PCEs with land use and the 11 a.i.s in a table format for each species. Each table lists the a.i.s addressed in this Opinion in three separate categories: those under cancellation proceedings, those currently registered for a wide range of uses, and those registered for a limited range of uses. Within each subgroup, toxicity, fate properties, and use patterns differ. For two of the cancellation chemicals (azinphos methyl and methamidophos), uses are sufficiently restricted and allowed for crop specific analysis. Azinphos methyl is largely restricted to orchard uses and methamidophos is only authorized for potatoes in Idaho, Oregon, and Washington. The length of time the cancellation chemicals are allowed for use following issuance of the Opinion was an important consideration in the decision on the likelihood of effects to PCEs and critical habitat.

Puget Sound Chinook Salmon

Of 61 assessed watersheds (HUC 5), 40 and 9 are of high and medium conservation value, respectively. Nineteen nearshore marine areas are also of high conservation value. Of the high value conservation watersheds, 32 and 40 are exposed to pesticides from agriculture and urban land uses, respectively. Among the medium value watersheds, six and nine are exposed to pesticides from agriculture and urban land uses, respectively. These areas serve as spawning, rearing, and migration habitat for Puget Sound Chinook salmon.

Migration, spawning, and rearing PCEs in upper watersheds of most river systems in the lower alluvial valleys of mid- to southern Puget Sound and the Strait of Juan de Fuca have been heavily altered by forestry, agriculture, and urban land uses. These activities have resulted in the loss of floodplain habitat, reduced substrate conditions for spawning and incubation, and degraded water quality. Estuary PCEs in the northwest Puget Sound are also degraded from impaired water quality (*e.g.*, contaminants), altered salinity conditions, lack of natural cover, and modification of and lack of access to tidal marshes and their channels.

Spawning, rearing, and migration PCEs in these exposed watersheds likely experience a reduction in water quality and prey abundance in freshwater, estuarine, and nearshore areas, especially during allowable pesticide applications adjacent to Puget Sound Chinook salmon habitat. Based on concentrations of agriculture and developed areas in the Lower Skagit Valley, we expect reductions in spawning, rearing, and migration PCEs in watersheds for this valley. As elevated water temperature prevents this ESU from inhabiting about 374 km of streams within its range, suitable PCE conditions in remaining available species habitat become important for ensuring long-term species conservation.

Co-occurrence of agriculture, urban/residential, and forestry uses is indicated in Table 166. More than 50% of the ESU is composed of evergreen, deciduous, or mixed forests. Other pesticide use areas include urban/residential development (15%) and agricultural uses (4%).

Cultivated crops (1%) and hay crops and pastures (3%) are primarily distributed on the floodplain and other lowland habitats. The majority of urban/residential land use also occurs within river and stream valleys in lowland areas, much of the nearshore marine area also consists of urban/residential.

Table 166. Co-occurrence of land use types and likelihood of a reduction in PCEs for Puget Sound Chinook salmon

a.i.	Co-occurrence			Reduction in PCEs	
	Agriculture	Urban/Residential	Forest	PCEs Affected	Appreciable
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Orchard crops only	NA	NA	Yes	No
Disulfoton	Yes	Yes	Yes	Yes	No
Methamidophos[1]	Potatoes only	NA	NA	Yes	No
Methidathion	Yes	Yes	NA	Yes	Yes
Methyl parathion	Yes	NA	NA	Yes	No
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Yes	No
Naled	Yes	Yes	Yes	Yes	Yes
Phosmet	Yes	Yes	Yes	Yes	Yes
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Yes	No
Ethoprop	Yes	NA	NA	Yes	No
Phorate	Yes	NA	NA	Yes	No

1: Crop-specific analysis was conducted

Lower Columbia River (LCR) Chinook Salmon

Thirty-one and 13 watersheds are of high and medium conservation value, respectively. Four additional unoccupied watersheds received a "possibly high" rating for species conservation as well. Our GIS analysis indicates 26 of 31 high conservation value watersheds are exposed to pesticide applications from agriculture and urban land uses, respectively. All 13 medium conservation watersheds are also exposed to pesticide applications from both land uses.

Spawning and rearing PCEs for LCR Chinook salmon have been degraded by timber harvests, agriculture, and urbanization. These land uses have reduced floodplain connectivity and water quality, and removed natural cover in several rivers. Hydropower development projects have also reduced the timing and magnitude of water flows, thereby altering required water quantity to form and maintain physical habitat conditions for juvenile fish growth and mobility. Migration PCEs are also affected by several dams along the migration route used by adult and juvenile fish. The survival of yearlings in the ocean is also affected by habitat conditions in the estuary, such as changes in food availability and the presence of contaminants.

Co-occurrence of agriculture, urban/residential, and forestry uses is indicated on Table 167. The percentage of agriculture lands that overlap with LCR Chinook salmon is about 6% and 2% as cultivated crops and 4% as hay/pasture. More than 76% of the ESU is composed of evergreen, deciduous forest, and mixed forests. Urban/residential development (13%) is a fairly substantial portion of this ESU. Most of the highly developed land and agricultural areas in this ESU's range are adjacent to salmonid habitat.

Spawning and migration PCEs in these exposed watersheds, as well as the river mainstem, and upstream tributaries likely experience reductions in water quality and prey abundance during allowable pesticide applications adjacent to these systems. As elevated water temperature prevents LCR Chinook salmon from inhabiting about 275 km of streams within its range, suitable PCE conditions in available species habitat are important for ensuring long-term species conservation.

Table 167 Co-occurrence of land use types and likelihood of a reduction in PCEs for Lower Columbia River Chinook salmon

a.i.	Co-occurrence			Reduction in PCEs	
	Agriculture	Urban/Residential	Forest	PCEs Affected	Appreciable
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Orchard crops	NA	NA	Yes	No
Disulfoton	Yes	Yes	Yes	Yes	No
Methamidophos[1]	Potatoes only	NA	NA	Yes	No

a.i.	Co-occurrence			Reduction in PCEs	
	Agriculture	Urban/Residential	Forest	PCEs Affected	Appreciable
Methidathion	Yes	Yes	NA	Yes	No
Methyl parathion	Yes	NA	NA	Yes	No
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Yes	No
Naled	Yes	Yes	Yes	Yes	Yes
Phosmet	Yes	Yes	Yes	Yes	Yes
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Yes	No
Ethoprop	Yes	NA	NA	Yes	No
Phorate	Yes	NA	NA	Yes	Yes

1: Crop-specific analysis was conducted.

Upper Columbia River (UCR) Spring-run Chinook Salmon

Twenty-six and five watersheds are of high and medium conservation value, respectively. Our GIS analysis indicates 23 and 26 high conservation watersheds are exposed to pesticide applications from agriculture and urban land uses, respectively. All medium conservation value watersheds are also exposed to pesticides from both land uses.

Urbanization in lower reaches, irrigation and diversion in the major upper drainages, and grazing in the middle reaches have degraded spawning and rearing PCEs in tributary systems. Migration PCEs for adult and juvenile fish are heavily degraded by Columbia River federal dam projects and a number of mid-Columbia River Public Utility District dam projects.

Co-occurrence of agricultural, urban/residential, and forestry land uses is indicated on Table 168. The percentage of agricultural and developed lands that overlap with UCR Chinook salmon habitat is about 5.4% and 4.7%, respectively. Forested lands compose about 45% of the ESU. Fish spawn and rear in the major tributaries leading to the Columbia River between Rock Island and Chief Joseph dams. Spawning, rearing, and migration PCEs likely experience a reduction in water quality and prey abundance especially in freshwater tributaries and shallow low flow floodplain habitats during allowable applications adjacent to UCR Spring-run Chinook salmon habitat.

Table 168. Co-occurrence of land use types and likelihood of a reduction in PCEs for Upper Columbia River Chinook Salmon

a.i.	Co-occurrence			Reduction in PCEs	
	Agriculture	Urban/Residential	Forest	PCEs Affected	Appreciable
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Orchard crops only	NA	NA	Yes	No
Disulfoton	Yes	Yes	Yes	Yes	No
Methamidophos[1]	Potatoes only	NA	NA	Yes	No
Methidathion	Yes	Yes	NA	Yes	Yes
Methyl parathion	Yes	NA	NA	Yes	No
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Yes	Yes
Naled	Yes	Yes	Yes	Yes	Yes
Phosmet	Yes	Yes	Yes	Yes	Yes
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Yes	No
Ethoprop	Yes	NA	NA	Yes	No
Phorate	Yes	NA	NA	Yes	Yes

1: Crops specific analysis conducted

Snake River (SR) Fall-run Chinook Salmon

Individual watersheds within the range of SR Fall-run Chinook salmon have not been evaluated by the CHART team for their conservation value. However, the Lower Columbia River corridor is of high conservation value as it connects several populations with the ocean and is used by rearing/migrating juveniles and migrating adults. The Columbia River estuary is also a unique and essential area for juveniles and adults making the physiological transition between life in freshwater and marine habitats.

Baseline conditions for this ESU include reduced spawning habitat and impaired stream flows and barriers to fish passage in tributaries from hydroelectric dams. Stream water quality and biological communities in the downstream portion of the upper Snake River basin are also degraded. We note that elevated water temperature currently prevents SR Fall-run Chinook salmon from inhabiting 2,401 km of streams within its range.

In lieu of CHART data on the conservation value ratings of salmonid watersheds, we recognize that all watersheds within the range of SR Fall-run Chinook salmon are of high conservation value. We used GIS data to assess the overlap between spawning and migration PCEs and use sites and their exposure in the Columbia River estuary and migratory corridor.

Co-occurrence of agriculture, forestry, and urban areas with salmonid habitats is shown in Table 169. Pesticide use areas for the 11 a.i.s within the range for this ESU and above the Columbia River migratory corridor include evergreen forests (49%), cultivated crops (15%), pastures (1%), and developed lands (1%).

Table 169. Co-occurrence of land use types and likelihood of a reduction in PCEs for Snake River Fall-run Chinook Salmon

a.i.	Co-occurrence			Reduction in PCEs	
	Agriculture	Urban/Residential	Forest	PCEs Affected	Critical Habitat
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Orchard crops only	NA	NA	No	No
Disulfoton	yes	Yes	Yes	Yes	No
Methamidophos[1]	Potatoes only	NA	NA	No	No
Methidathion	Yes	No	No	Yes	No
Methyl parathion	Yes	NA	NA	Yes	No
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Yes	No
Naled	Yes	Yes	Yes	Yes	No
Phosmet	Yes	Yes	Yes	Yes	Yes
More Limited Use Sites					
Bensulide	Yes	Yes	NA	No	No
Ethoprop	Yes	NA	NA	No	No
Phorate	Yes	NA	NA	Yes	No

1: Crop-specific analysis was conducted.

Snake River (SR) Spring/Summer-run Chinook Salmon

Watersheds within the range of SR Spring/Summer-run Chinook salmon were not evaluated by the CHART team for their conservation value. However, the Lower Columbia River is of high conservation value as it connects every population with the ocean and is used by rearing/migrating juveniles and migrating adults. Juveniles of this ESU rely on adequate fresh water quality and prey abundance for migrating and rearing in freshwater habitats including migratory routes from natal reaches leading to alternative summer-rearing or overwintering areas.

Spawning and juvenile PCEs are regionally degraded by changes in flow quantity, water quality, and loss of cover. Juvenile and adult migrations are obstructed by reduced access stemming from altered flow regimes from hydroelectric dams. As elevated water temperature prevents SR Spring/Summer-run Chinook salmon from inhabiting 1,596.3 km of streams within its range, suitable PCE conditions in remaining species habitat become important for ensuring the long-term conservation for this species.

Co-occurrence of agriculture, urban/residential, and forestry land use is shown in Table 170. This ESU spawns and rears primarily in the smaller tributaries, many of which are located on U.S. Forest Service lands. Agricultural and urban areas are not common in the watersheds comprising the ESU, and those that are present are clustered mostly around the mainstem Snake and Columbia Rivers. The Snake River is a high-volume, high-flow system, and salmon use it primarily as a migratory corridor.

Table 170. Co-occurrence of land use types and likelihood of a reduction in PCEs for Snake River Spring/Summer-run Chinook Salmon

a.i.	Co-occurrence			Reduction in PCEs	
	Agriculture	Urban/Residential	Forest	PCEs Affected	Appreciable
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	Na
Azinphos methyl[1]	Orchard crops only	NA	NA	No	No
Disulfoton	Yes	Yes	Yes	Yes	No
Methamidophos[1]	Potatoes	NA	NA	No	No

a.i.	Co-occurrence			Reduction in PCEs	
	Agriculture	Urban/Residential	Forest	PCEs Affected	Appreciable
	only				
Methidathion	Yes	No	No	Yes	No
Methyl parathion	Yes	NA	NA	Yes	No
Many Use Sites					
Dimethoate	Yes	Yes	Not in Idaho	Yes	No
Naled	Yes	Yes	Yes	Yes	No
Phosmet	Yes	Yes	Yes	Yes	Yes
More Limited Use Sites					
Bensulide	Yes	Yes	NA	No	No
Ethoprop	Yes	NA	NA	No	No
Phorate	Yes	NA	NA	Yes	No

1: : Crop-specific analysis was conducted.

Upper Willamette River (UWR) Chinook Salmon

Of 59 assessed watersheds, 22 and 18 are of high and medium conservation value, respectively. The lower Willamette/Columbia River rearing/migration corridor downstream of the spawning range is also of high conservation value. Our GIS analysis indicates 15 and 19 high conservation watersheds are exposed to pesticide applications from agriculture and urban land uses, respectively. Of the medium conservation watersheds, 13 and 12 are also exposed to pesticide applications from the above respective land uses.

Migration and rearing PCEs have been degraded by dams altering migration timing and water management. Migration, rearing, and estuary PCEs are also degraded by the loss of riparian vegetation and instream cover. Water quality is also degraded in floodplain rearing habitat along the lower Willamette River. As elevated water temperature prevents UWR Chinook salmon from inhabiting 2,468 km of waters within its range, PCE conditions in remaining species habitat are important for ensuring long-term conservation for this species.

Co-occurrence of agriculture, urban/residential, and forestry land uses is indicated on Table 171.

The percentage of cultivated and develop lands that overlap with UWR Chinook salmon habitat are 10.5% and 9%, respectively. Spawning, rearing, and migration freshwater (PCEs in these exposed watersheds (including mainstem and floodplain wetlands) likely experience reductions in water quality and prey abundance.

Table 171. Co-occurrence of land use types and likelihood of a reduction in PCEs for Upper Willamette River Chinook Salmon

a.i.	Co-occurrence			Reduction in PCEs	
	Agriculture	Urban/Residential	Forest	PCEs Affected	Appreciable
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Orchard crops only	NA	NA	No	No
Disulfoton	Yes	Yes	Yes	Yes	No
Methamidophos[1]	Potatoes only	NA	NA	No	No
Methidathion	Yes	No	No	Yes	No
Methyl parathion	Yes	NA	NA	Yes	No
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Yes	No
Naled	Yes	Yes	Yes	Yes	Yes
Phosmet	Yes	Yes	Yes	Yes	Yes
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Yes	No
Ethoprop	Yes	NA	NA	Yes	No
Phorate	Yes	NA	NA	Yes	Yes

1: Crop-specific analysis was conducted.

California Coastal (CC) Chinook Salmon

Of 45 occupied watersheds, 27 and 10 are of high and medium conservation value, respectively. Our GIS analysis indicates 8 and 27 high conservation watersheds are exposed to pesticides from agriculture and urban land uses, respectively. Of the medium conservation watersheds, 4 and 10 are exposed to pesticide applications from the above respective land uses.

The spawning PCE in coastal streams have been degraded from timber harvests. Rearing and migration PCEs in the Russian River have also been impacted by agriculture and urban areas. Water management for dams within the Russian and Eel River watersheds maintain high flows and warm water during summer which indirectly benefits the introduced Sacramento pikeminnow, a predatory fish on CC Chinook salmon along migration corridors. The estuary PCE has also been degraded from breaches of the sandbar at the mouth of the Russian River causing periodic mixing of salt water. This condition alters the water quality and salinity conditions for the juvenile physiological transitions between fresh and salt water. Current PCE conditions likely maintain a low population abundance across the ESU.

Co-occurrence of agriculture, urban/residential, and forestry uses is indicated on Table 172. The percentage of cultivated croplands and developed lands that overlap with CC Chinook salmon habitat are 1% and 5.4%, respectively. There is substantial overlap with use sites in the Russian River watershed. Rearing and migration PCEs in freshwater and estuaries in this watershed likely experience reductions in water quality and prey abundance.

Table 172. Co-occurrence of land use types and likelihood of a reduction in PCEs for Central California Chinook Salmon

a.i.	Co-occurrence			Reduction in PCEs	
	Agriculture	Urban/Residential	Forest	PCEs Affected	Appreciable
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Orchard crops only	NA	NA	No	No
Disulfoton	Yes	Yes	Yes	Yes	No
Methamidophos[1]	Potatoes only	NA	NA	No	No
Methidathion	Yes	Yes	No	Yes	No
Methyl parathion	Yes	NA	NA	Yes	No
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Yes	No
Naled	Yes	Yes	Yes	Yes	Yes
Phosmet	Yes	Yes	Yes	Yes	Yes
More Limited Use Sites					

a.i.	Co-occurrence			Reduction in PCEs	
	Agriculture	Urban/Residential	Forest	PCEs Affected	Appreciable
Bensulide	Yes	Yes	NA	Yes	No
Ethoprop	Yes	NA	NA	Yes	No
Phorate	Yes	NA	NA	Yes	Yes

1: Crop specific analysis conducted

Central Valley (CV) Spring-run Chinook Salmon

Of 38 occupied watersheds, 28 and 3 are of high and medium conservation value, respectively. Four of these watersheds comprise portions of the San Francisco-San Pablo-Suisun Bay estuarine complex which provides rearing and migratory habitat for CV Spring-run Chinook salmon. Our GIS analysis indicates 17 and 28 high conservation value watersheds are exposed to pesticides from agriculture and urban land uses, respectively. Of the medium conservation watersheds, two and three watersheds are exposed to pesticide applications from the above land uses as well.

Spawning and rearing PCEs are currently degraded by elevated water temperature and lost access to historic spawning areas in upper watersheds with cool and clean water throughout the summer. The rearing PCE is degraded and is affected by loss of floodplain habitat connectivity from the mainstem of larger rivers through the Sacramento River watershed, thereby reducing effective foraging. The migration PCE is degraded by lack of natural cover along the migration corridors. Juvenile migration is further obstructed by water diversions along the Sacramento River and by two large state and federal water-export facilities in the Sacramento-San Joaquin Delta. Agriculture and urban runoff containing a suite of pollutants further impair water quality of receiving systems used by this species.

Intensive agricultural development occurs in the California Central Valley and may impact waters draining into the Sacramento River. We further expect rearing and migration PCEs in non-natal tributaries, intermittent streams, and floodplain habitats may also experience likely reductions in water quality and prey abundance.

Co-occurrence of agriculture, urban/residential, and forestry uses is indicated on Table 173. The percentage of cultivated croplands and developed lands that overlap with CV Chinook salmon habitat are 21.3% and 10.8%, respectively. Migration PCEs in the San Francisco-San Pablo-Suisan Bay estuaries complex, which are heavily influenced by input from California's Central Valley likely experience reductions in water quality and prey abundance.

Table 173. Co-occurrence of land use types and likelihood of a reduction in PCEs for Central Valley Spring-run Chinook Salmon

a.i.	Co-occurrence			Reduction in PCEs	
	Agriculture	Urban/Residential	Forest	PCEs Affected	Appreciable
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Orchard crops only	NA	NA	No	No
Disulfoton	Yes	Yes	Yes	Yes	No
Methamidophos[1]	Potatoes only	NA	NA	No	No
Methidathion	Yes	No	No	Yes	Yes
Methyl parathion	Yes	NA	NA	Yes	No
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Yes	Yes
Naled	Yes	Yes	Yes	Yes	Yes
Phosmet	Yes	Yes	Yes	Yes	Yes
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Yes	Yes
Ethoprop	Yes	NA	NA	Yes	No
Phorate	Yes	NA	NA	Yes	Yes

1: Crop-specific analysis was conducted.

Sacramento River Winter-run Chinook Salmon

Individual subbasins or river sections were not evaluated for their conservation value. However, the entire Sacramento River and the Delta are considered of high conservation value for spawning, rearing, and migration.

Spawning and rearing PCEs are currently degraded by elevated water temperature and lost access to historic spawning areas in upper watersheds with cool and clean water throughout the summer. The rearing PCE is degraded and is affected by loss of floodplain habitat connection from the mainstem of larger rivers through the Sacramento River watershed, thereby reducing effective foraging. The migration PCE is degraded by lack of natural cover along the migration corridors. Juvenile migration is further obstructed by water diversions along the Sacramento River and by two large state and federal water-export facilities in the Sacramento-San Joaquin Delta.

Co-occurrence of agriculture, urban/residential, and forestry uses is indicated on Table 174. The percentage of cultivated croplands and developed lands that overlap with SR Winter-run Chinook salmon are 25% and 10%, respectively. As agriculture and urban land uses occur in the Sacramento River watershed and in the Sacramento-San Joaquin Delta, we expect rearing and spawning PCEs in floodplain habitat and the Sacramento River may experience reductions in water quality and prey abundance.

Table 174. Co-occurrence of land use types and likelihood of a reduction in PCEs for Sacramento River Winter-run Chinook Salmon

a.i.	Co-occurrence			Reduction in PCEs	
	Agriculture	Urban/Residential	Forest	PCEs Affected	Appreciable
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Orchard crops only	NA	NA	No	No
Disulfoton	Yes	Yes	Yes	Yes	No
Methamidophos[1]	Potatoes only	NA	NA	No	No
Methidathion	Yes	Yes	NA	Yes	Yes
Methyl parathion	Yes	NA	NA	Yes	No
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Yes	Yes
Naled	Yes	Yes	Yes	Yes	Yes
Phosmet	Yes	Yes	Yes	Yes	Yes
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Yes	Yes
Ethoprop	Yes	NA	NA	Yes	Yes
Phorate	Yes	NA	NA	Yes	Yes

1: Crop-specific analysis was conducted.

Hood Canal Summer-run Chum Salmon

Of 12 assessed watersheds, nine and three are of high and medium conservation value, respectively. Five nearshore marine areas were also rated as high conservation value. Many of the watersheds have less than four miles of spawning habitat and none are greater than 8.5 miles in length. Our GIS analysis indicates seven and nine high conservation value watersheds are exposed to pesticides from agriculture and urban land uses, respectively. All three medium conservation watersheds are exposed to both land uses as well.

The spawning PCE is degraded by excessive fine sediment in gravel. The rearing PCE is degraded by loss of access to sloughs in the estuary and nearshore areas and excessive predation. Migration and rearing PCEs in estuaries are impaired by the loss of functional floodplain areas. These degraded conditions likely maintain low population abundance across the ESU.

Co-occurrence with agriculture, urban/residential, and forestry land uses in Table 175. Land use within the range of this ESU is predominantly forested (73%), open water (17%), urban/residential (9%), and agriculture (2%). The percentage of cultivated croplands and developed lands that overall with HC Summer-run chum salmon habitat is about 9.94% and 8.9%, respectively. Most of the agriculture and urban/residential uses occur within rivers and stream valleys in lowland areas. Nearshore marine areas are frequently adjacent to urban/residential areas. Given these uses, spawning and migration PCEs in streams, estuaries, and nearshore marine areas may experience reductions in water quality and prey abundance during allowable pesticide applications adjacent to these systems.

Table 175. Co-occurrence of land use types and likelihood of a reduction in PCEs for Hood Canal Summer-run Chum Salmon

a.i.	Co-occurrence			Reduction in PCEs	
	Agriculture	Urban/Residential	Forest	PCEs Affected	Appreciable
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Orchard crops only	NA	NA	No	No
Disulfoton	Yes	Yes	Yes	Yes	No
Methamidophos[1]	Potatoes	NA	NA	No	No

	only				
Methidathion	Yes	Yes	NA	Yes	No
Methyl parathion	Yes	NA	NA	Yes	No
Many Use Sites					
Dimethoate	Yes	Yes	Yes	No	No
Naled	Yes	Yes	Yes	Yes	Yes
Phosmet	Yes	Yes	Yes	Yes	Yes
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Yes	No
Ethoprop	Yes	NA	NA	No	No
Phorate	Yes	NA	NA	Yes	No

1: Crop-specific analysis was conducted.

Columbia River (CR) Chum Salmon

Of 19 assessed watersheds, 16 and 3 are of high and medium conservation value, respectively. Our GIS analysis indicates all high and medium conservation value watersheds are exposed to pesticide applications from agriculture, developed areas, and forestry adjacent to CR chum salmon habitat.

The migration PCE for this species has been significantly impacted by dams obstructing adult migration and access to historic spawning sites. Water quality and cover for estuary and rearing PCEs have decreased and are not likely to maintain their intended function to conserve the species. Elevated water temperature further prevents CR chum salmon from inhabiting 272.8 km of waters within its range.

Co-occurrence of agriculture, urban/residential, and forestry land uses is indicated on Table 176. The percentage of cultivated croplands, hay/pasture, and developed lands that overlap with CR chum salmon habitat is 2%, 5%, and 15%, respectively. More than 50% of the range of the ESU is covered by deciduous, evergreen, or mixed forests. Within the ESU, agricultural and development are predominantly distributed in the low-lying areas near the Columbia River and its tributaries. Given these uses the rearing and migration PCEs along the edges of the mainstem or in tributaries and side channels of freshwater and estuarine systems may experience reductions in water quality and prey abundance during allowable pesticide applications adjacent to these systems.

Table 176. Co-occurrence of land use types and likelihood of a reduction in PCEs for Columbia River Chum Salmon

a.i.	Co-occurrence			Reduction in PCEs	
	Agriculture	Urban/Residential	Forest	PCEs Affected	Appreciable
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Orchard crops only	NA	NA	Yes	No
Disulfoton	Yes	Yes	Yes	Yes	No
Methamidophos[1]	Potatoes only	NA	NA	Yes	No
Methidathion	Yes	Yes	NA	Yes	No
Methyl parathion	Yes	NA	NA	Yes	No
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Yes	No
Naled	Yes	Yes	Yes	Yes	Yes
Phosmet	Yes	Yes	Yes	Yes	Yes
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Yes	No
Ethoprop	Yes	NA	NA	Yes	No
Phorate	Yes	NA	NA	Yes	No

1:: Crop-specific analysis was conducted.

Oregon Coast (OC) Coho Salmon

Of 80 watersheds, 45 and 27 are of high and medium conservation value, respectively. Our GIS analysis indicates 39 and 44 high conservation watersheds are exposed to pesticides from agriculture and urban areas, respectively. Of the medium conservation watersheds, 18 and 23 are exposed to pesticide applications from the above respective land uses.

The rearing PCE has been degraded by elevated water temperature in 29 of the 80 HUC 5 watersheds. Elevated temperature further prevents OC coho salmon from inhabiting 3,716 km of waters within its range. Twelve watersheds have reduced water quality from contaminants and excessive nutrition.

Co-occurrence of agriculture, urban/residential, and forestry land uses is indicated in Table 177. The percentage of cultivated croplands and developed lands that overlap with OC coho salmon habitat are 0.23% and 6.6%, respectively. Most of the cropland is hay/pasture and is primarily located in the Umpqua watersheds. Given these uses, we expect a low likelihood of freshwater rearing PCE in small streams to experience reductions in water quality and prey abundance.

Table 177. Co-occurrence of land use types and likelihood of a reduction in PCEs for Oregon Coast Coho Salmon

a.i.	Co-occurrence			Reduction in PCEs	
	Agriculture	Urban/Residential	Forest	PCEs Affected	Appreciable
Under Cancellation					
Fenamiphos[1]	NA	NA	NA	NA	NA
Azinphos methyl[2]	Orchard crops only	NA	NA	No	No
Disulfoton	Yes	Yes	Yes	No	No
Methamidophos[2]	Potatoes only	NA	NA	No	No
Methidathion	Yes	Yes	NA	No	No
Methyl parathion	Yes	NA	NA	No	No
Many Use Sites					
Dimethoate	Yes	Yes	Yes	No	No
Naled	Yes	Yes	Yes	Yes	No
Phosmet	Yes	Yes	Yes	Yes	No
More Limited Use Sites					
Bensulide	Yes	Yes	NA	No	No
Ethoprop	Yes	NA	NA	No	No
Phorate	Yes	NA	NA	No	No

1: Crops specific analysis conducted

Southern Oregon/Northern California Coast (SONCC) Coho Salmon

Although watersheds within this ESU were not evaluated for their conservation value, the northern coastal streams that are designated as critical habitat are of good quality. Throughout this ESU's range, the spawning PCE has been degraded by fines in spawning gravel from logging. The rearing PCE has been considerably degraded in many inland watersheds by the loss of riparian vegetation, resulting in unsuitable high temperatures. Rearing and migration PCEs have been reduced by the disconnection of floodplain and off-channel habitats in low gradient reaches of streams. Elevated water temperature further prevents SONCC coho salmon from inhabiting 3,249.2 km of waters within its range.

Co-occurrence of agriculture, urban/residential, and forestry land uses is indicated on Table 178. The percentage of cultivated croplands and developed lands that overlap with SONCC coho salmon habitat are 2.5% and 4.3%, respectively. Areas with more cropland include the Scott and Shasta watersheds in the Klamath basin and the Upper and Middle rough River watersheds. Of the development in this ESU, much is in the rough River basin, with remaining development distributed along the coastline and estuaries. Given these uses, we expect rearing and migration PCEs in backwater, side channels, and shallow channel edge fish habitat may experience reductions in water quality and prey abundance during allowable pesticide applications adjacent to these systems.

Table 178. Co-occurrence of land use types and likelihood of a reduction in PCEs for Southern Oregon Northern California Coast Coho Salmon

a.i.	Co-occurrence			Reduction in PCEs	
	Agriculture	Urban/Residential	Forest	PCEs Affected	Appreciable
Under Cancellation					
Fenamiphos[1]	NA	NA	NA	NA	NA
Azinphos methyl[2]	Orchard crops only	NA	NA	No	No
Disulfoton	Yes	Yes	Yes	Yes	No
Methamidophos[2]	Potatoes only	NA	NA	No	No
Methidathion	Yes	No	No	Yes	No
Methyl parathion	Yes	NA	NA	Yes	No
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Yes	No
Naled	Yes	Yes	Yes	Yes	Yes
Phosmet	Yes	Yes	Yes	Yes	Yes

a.i.	Co-occurrence			Reduction in PCEs	
	Agriculture	Urban/Residential	Forest	PCEs Affected	Appreciable
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Yes	No
Ethoprop	Yes	NA	NA	Yes	No
Phorate	Yes	NA	NA	Yes	No

1: Crop-specific analysis was conducted.

Central California Coast (CCC) Coho Salmon

Individual watersheds have not been evaluated for their conservation value. Nevertheless, there is a distinct trend of increasing degradation in quality and quantity of all PCEs as the habitat progresses south through the species range along the Lost Coast to Navarro Point and the Santa Cruz Mountains. Spawning and incubation substrate and juvenile rearing habitat are generally degraded.

Co-occurrence of agriculture, urban/residential, and forestry land uses is indicated onTable 179.. The percentage of cultivated croplands and developed lands that overlap with CCC coho salmon habitat are 2.3% and 9.4%, respectively. Much of the development is centered around San Francisco Bay, and developed and agricultural areas also occur in the Russian River watershed. The northern, undeveloped watersheds around the Navarro and Big Rivers are used by the majority of this species. Given these land uses, we expect the freshwater rearing PCE may experience reductions in water quality and prey abundance during allowable pesticide applications adjacent to freshwater systems.

Table 179. Co-occurrence of land use types and likelihood of a reduction in PCEs for Central California Coast Coho Salmon

a.i.	Co-occurrence			Reduction in PCEs	
	Agriculture	Urban/Residential	Forest	PCEs Affected	Appreciable
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Orchard crops only	NA	NA	No	No
Disulfoton	Yes	Yes	Yes	Yes	No
Methamidophos[1]	Potatoes only	NA	NA	No	No

a.i.	Co-occurrence			Reduction in PCEs	
	Agriculture	Urban/Residential	Forest	PCEs Affected	Appreciable
Methidathion	Yes	Yes	NA	Yes	Yes
Methyl parathion	Yes	NA	NA	Yes	No
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Yes	No
Naled	Yes	Yes	Yes	Yes	Yes
Phosmet	Yes	Yes	Yes	Yes	Yes
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Yes	No
Ethoprop	Yes	NA	NA	Yes	Yes
Phorate	Yes	NA	NA	Yes	Yes

1: Crop-specific analysis was conducted.

Ozette Lake Sockeye Salmon

The Ozette Lake watershed is of high conservation value. The entire circumference of the lake is within Olympic National Park. Ozette Lake and portions of three tributaries support spawning and rearing PCEs. Ozette River supports rearing and migration PCEs; its river mouth also provides estuarine habitat. Migration habitat is also affected by low water flow in summer and elevated water temperature which pose as a thermal barrier for migration.

Spawning habitat has been affected by the loss of tributary spawning areas, low water levels in summer, and vegetation and sediment that have reduced the quantity and suitability of beaches for spawning. The rearing PCE is degraded by excessive predation, competition with non-native species, and loss of rearing habitat. Migration habitat is affected by high water temperatures and low water flows in summer.

Co-occurrence of urban/residential and forestry uses is indicated on Table 180. Ozette Lake is in a sparsely populated area, with less than 1% of land developed within the range of this ESU. Similarly, there is no cultivated cropland. However, salmonid habitat may be at risk of exposure form forestry-related uses. Land use is primarily forest with private, state, and federal ownership (86% forested, 13% open water, 1% developed land, 0% agriculture). The predominant pesticide use sites (*i.e.*, urban/residential and forestry) overlap with the Lake's freshwater tributaries. Thus, the greatest risk of exposure to freshwater PCEs are in tributary habitats. However, we do not expect a reduction in prey abundance within these tributaries. Although private residences along tributaries may have small, non-commercial crops for pesticide applications, it is unlikely that restricted use pesticides would be applied.

Table 180. Co-occurrence of land use types and likelihood of a reduction in PCEs for Ozette Lake Sockeye Salmon

a.i.	Co-occurrence			Reduction in PCEs	
	Agriculture	Urban/Residential	Forest	PCEs Affected	Appreciable
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	No	NA	NA	No	No
Disulfoton	No	Yes	Yes	Yes	No
Methamidophos[1]	No	NA	NA	No	No
Methidathion	No	Yes	NA	No	No
Methyl parathion	No	NA	NA	No	No
Many Use Sites					
Dimethoate	No	Yes	Yes	Yes	No
Naled	No	Yes	Yes	Yes	No
Phosmet	No	Yes	Yes	Yes	No
More Limited Use Sites					
Bensulide	No	Yes	NA	No	No
Ethoprop	No	NA	NA	No	No
Phorate	No	NA	NA	No	No

1: Crop-specific analysis was conducted.

Snake River Sockeye Salmon

Conservation values of individual watersheds have not been reported. Nevertheless, all areas occupied and used by migrating SR sockeye are considered of high conservation value as this species is limited to a single lake within the SR basin.

The quality and quantity of rearing and migration PCEs have been reduced by land uses that disrupt access to foraging areas, increase the amount of fines in the stream substrate, and reduce instream cover. Water quality is impaired by a suite of anthropogenic pollutants which enter surface waters and riverine sediments from the headwaters of the Salmon River to the Columbia River estuary. The migration PCE is also affected by four dams in the SR basins that obstructs migration and increases mortality of downstream migrating juveniles. Given the migration distance traveled by this species, adequate passage conditions (water quality and quantity available at specific times) is critical.

Co-occurrence of agriculture, urban/residential, and forestry uses is indicated in Table 181. About 1% of the land surrounding Red Fish Lake has been developed, and another 1% is used for agriculture, primarily hay and pasture. More than 50% of range of this ESU is in evergreen forests. Consequently, forestry uses are the major source of exposure in spawning and rearing habitats. Given the limited uses around Red Fish Lake, we expect the migration PCE may experience some reductions in water quality and abundance during allowing pesticide applications adjacent to the Snake and Columbia Rivers leading to the ocean.

Table 181. Co-occurrence of land use types and likelihood of a reduction in PCEs for Snake River Sockeye Salmon

a.i.	Co-occurrence			Reduction in PCEs	
	Agriculture	Urban/Residential	Forest	PCEs Affected	Appreciable
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Orchard crops only	NA	NA	No	No
Disulfoton	Yes	Yes	Yes	No	No
Methamidophos[1]	Potatoes only	NA	NA	No	No
Methidathion	Yes	No	No	No	No
Methyl parathion	Yes	NA	NA	No	No
Many Use Sites					
Dimethoate	Yes	Yes	Yes	No	No
Naled	Yes	Yes	Yes	Yes	Yes
Phosmet	Yes	Yes	Yes	Yes	Yes
More Limited Use Sites					
Bensulide	Yes	Yes	NA	No	No
Ethoprop	Yes	NA	NA	No	No

a.i.	Co-occurrence			Reduction in PCEs	
	Agriculture	Urban/Residential	Forest	PCEs Affected	Appreciable
Phorate	Yes	NA	NA	No	No

1: Crop-specific analysis was conducted.

Lower Columbia River Steelhead

Of 41 watersheds listed as critical habitat for LCR steelhead, 28 and 11 are of high and medium conservation value, respectively. Our GIS analysis indicates 21 and 26 high conservation watersheds are exposed to pesticides from agriculture and urban/residential land uses, respectively. Of the medium conservation watersheds, 11 and 10 are also exposed to pesticide applications from the above respective land uses.

The water quality of the rearing PCE within the lower portion and alluvial valleys of many watersheds has been degraded by agricultural runoff into tributaries reaches and the mainstem Columbia River. Consequently, invertebrate production in these aquatic systems is also affected. Elevated water temperature further prevents LCR steelhead from inhabiting 341.5 km of waters within its range.

Co-occurrence of agriculture, urban/residential, and forestry uses is indicated in Table 182. The percentage of cultivated crop lands that overlap with LCR steelhead is about 7%, with 4.5% as hay/pasture land and 2.5% as cultivated cropland. More than 61% of the range of this DPS is composed of evergreen, deciduous forest, and mixed forests. Urban/residential development lands (12%) were a fairly substantial portion of this DPS. Given these uses, we expect the freshwater rearing PCE in floodplain habitats, and natal rivers and streams, may experience reductions in water quality and prey abundance during allowable pesticide applications to these systems.

Table 182. Co-occurrence of land use types and likelihood of a reduction in PCEs for Lower Columbia River Steelhead

a.i.	Co-occurrence			Reduction in PCEs	
	Agriculture	Urban/Residential	Forest	PCEs Affected	Appreciable
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA

a.i.	Co-occurrence			Reduction in PCEs	
	Agriculture	Urban/Residential	Forest	PCEs Affected	Appreciable
Azinphos methyl[1]	Orchard crops only	NA	NA	No	No
Disulfoton	Yes	Yes	Yes	No	No
Methamidophos[1]	Potatoes	NA	NA	Yes	No
Methidathion	Yes	Yes	NA	Yes	No
Methyl parathion	Yes	NA	NA	Yes	No
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Yes	No
Naled	Yes	Yes	Yes	Yes	Yes
Phosmet	Yes	Yes	Yes	Yes	Yes
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Yes	No
Ethoprop	Yes	NA	NA	Yes	No
Phorate	Yes	NA	NA	Yes	Yes

1: Italics indicate that the a.i. is undergoing cancellation.
2: Crop-specific analysis was conducted.

Upper Willamette River Steelhead

Of the watersheds assessed, 14 and 6 are of high and medium conservation value, respectively. Our GIS analysis indicates all high and medium conservation value watersheds are exposed to pesticide applications from agriculture and urban areas adjacent to UWR steelhead critical habitat.

Existing water quality necessary for juvenile rearing within many watersheds have been impaired by pollutants in agricultural runoff. Consequently, invertebrate production for salmonids in several watersheds and in the mainstem Columbia River is affected. As several dams obstruct migrating fish along the migratory corridor, the migration PCE is also reduced by these features. Elevated water temperature further prevents UWR steelhead from inhabiting 1,668 km of waters within its range.

Co-occurrence of agriculture, urban/residential, and forestry land uses is indicated in Table 183. The percentage of cultivated crop land and developed lands are 14.5% and 10%, respectively.

Given these uses, we expect the freshwater rearing PCE in floodplain habitats, rivers, and streams may experience reductions in water quality and prey abundance during allowable pesticide applications adjacent to these systems.

Table 183. Co-occurrence of land use types and likelihood of a reduction in PCEs for Upper Willamette River Steelhead

a.i.	Co-occurrence			Reduction in PCEs	
	Agriculture	Urban/Residential	Forest	PCEs Affected	Appreciable
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Orchard crops only	NA	NA	No	No
Disulfoton	Yes	Yes	Yes	Yes	No
Methamidophos[1]	Potatoes only	NA	NA	No	No
Methidathion	Yes	Yes	NA	Yes	No
Methyl parathion	Yes	NA	NA	Yes	No
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Yes	No
Naled	Yes	Yes	Yes	Yes	Yes
Phosmet	Yes	Yes	Yes	Yes	Yes
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Yes	No
Ethoprop	Yes	NA	NA	Yes	No
Phorate	Yes	NA	NA	Yes	Yes

1: Crop-specific analysis was conducted.

Middle Columbia River Steelhead

Of the 106 assessed watersheds, 73 and 24 are of high and medium conservation value, respectively. The lower Columbia River rearing/migration corridor downstream of the spawning range is also of high conservation value. Our GIS analysis indicates 67 and 68 high conservation watersheds are exposed to pesticides from agriculture and urban areas, respectively. Of the medium conservation watersheds, 23 and 24 watersheds are also exposed to pesticide applications from the above respective land uses.

The current condition of critical habitat for MCR steelhead is moderately degraded. The water quality attribute for the rearing PCE within many watersheds is reduced. Consequently, invertebrate production in these watersheds and in the mainstem Columbia River is also reduced. Loss of riparian vegetation to grazing has resulted in elevated water temperature in the John Day Basin. Elevated water temperature prevents MCR steelhead from inhabiting 3,727.9 km of waters within its range. In the Yakima River, 72 streams and river segments are also listed as impaired waters and 83% exceed temperature standards. As several dams obstruct fish along their migratory corridor, these features further degrade the migration PCE.

Co-occurrence of agriculture, urban/residential, and forestry land uses is indicated onTable 184. The percentage of cultivated crop lands and developed lands within the range of this DPS are 17% and 3%, respectively. Given the continuous run of steelhead throughout the year, the conditions of the rearing PCE in freshwater habitat is important for adult and rearing juveniles. Given the above land uses, the freshwater rearing PCE in shallow water along the banks of streams or aquatic habitats on stream margins are vulnerable to reductions in water quality and prey abundance during allowable pesticide applications to these systems.

Table 184. Co-occurrence of land use types and likelihood of a reduction in PCEs for Middle Columbia River Steelhead

a.i.	Co-occurrence			Reduction in PCEs	
	Agriculture	Urban	Forest	PCEs Affected	Appreciable
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Orchard crops only	NA	NA	No	No
Disulfoton	Yes	Yes	Yes	Yes	No
Methamidophos[1]	Potatoes only	NA	NA	No	No
Methidathion	Yes	Yes	NA	Yes	Yes
Methyl parathion	Yes	NA	NA	Yes	No
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Yes	No
Naled	Yes	Yes	Yes	Yes	Yes
Phosmet	Yes	Yes	Yes	Yes	Yes

a.i.	Co-occurrence			Reduction in PCEs	
	Agriculture	Urban	Forest	PCEs Affected	Appreciable
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Yes	No
Ethoprop	Yes	NA	NA	Yes	No
Phorate	Yes	NA	NA	Yes	Yes

1: Crop-specific analysis was conducted.

Upper Columbia River Steelhead

Of the 41 watersheds occupied by UCR steelhead, 31 and 7 are of high and medium conservation value, respectively. The lower Columbia River rearing/migration corridor downstream of the species' spawning range is also of high conservation value. Our GIS analysis indicates 28 and 31 high conservation watersheds are exposed to pesticides from agriculture and urban areas, respectively. All seven medium conservation watersheds are exposed to pesticide applications from the above land uses.

The current condition of UCR steelhead critical habitat is moderately degraded. Habitat quality in tributary streams range from excellent to poor. Water quality for the rearing PCEs within many watersheds has been reduced from agriculture runoff. Consequently, invertebrate production in several watersheds and in the mainstem Columbia River is also reduced. Several dams obstruct fish migrating through the migratory corridor and further impact the migration PCEs.

Co-occurrence of agriculture, urban/residential, and forestry uses is indicated in Table 185. The percentage of cultivated crop lands and developed lands within the range of this DPS are 13% and 4%, respectively. There is some agriculture in the spawning and rearing areas in the Wenatchee, Methow, and Okenogan watersheds. Intense agriculture occurs outside the 2.5 km buffer and in the Upper Columbia Irrigation District within the Entiat watershed. The water is heavily used and re-used for irrigation. We expect reductions in water quality and prey abundance for the migration PCE along the Columbia River, where the valley is heavily agricultural. Reductions in water quality and prey abundance are also expected for the freshwater rearing PCE during allowable pesticide applications adjacent to stream margins or cascades.

Table 185. Co-occurrence of land use types and likelihood of a reduction in PCEs for Upper Columbia River Steelhead

a.i.	Co-occurrence			Reduction in PCEs	
	Agriculture	Urban/Residential	Forest	PCEs Affected	Appreciable
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Orchard crops only	NA	NA	Yes	No
Disulfoton	Yes	Yes	Yes	Yes	No
Methamidophos[1]	Potatoes only	NA	NA	Yes	No
Methidathion	Yes	Yes	NA	Yes	Yes
Methyl parathion	Yes	NA	NA	Yes	No
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Yes	No
Naled	Yes	Yes	Yes	Yes	Yes
Phosmet	Yes	Yes	Yes	Yes	Yes
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Yes	No
Ethoprop	Yes	NA	NA	Yes	No
Phorate	Yes	NA	NA	Yes	Yes

1:Crop-specific analysis was conducted.

Snake River Basin Steelhead

Of the watersheds assessed, 229 and 41 are of high and medium conservation value, respectively. The Columbia River migration corridor is also of high conservation value. Our GIS analysis indicates 163 and 99 high conservation watersheds are exposed to pesticides from agriculture and urban areas, respectively. Of the medium conservation watersheds, 34 and 28 are also exposed to pesticide applications from the above land uses.

The current condition of SR basin steelhead critical habitat is moderately degraded. Water quality conditions for rearing PCEs within many watersheds have been degraded from contaminants in agricultural runoff. Consequently, invertebrate communities in several watersheds and in the mainstem Columbia River are negatively impacted. Loss of riparian vegetation to grazing has resulted in elevated water temperature in the John Day basin. These conditions have reduced the rearing PCE. As several dams obstruct adult fish migrating along the migratory corridor, the migration PCE is also negatively impacted. Elevated water temperature further prevents SR basin steelhead from inhabiting 3,282 km of waters within its range.

Co-occurrence of agriculture, urban/residential, and forestry uses is indicated in Table 186. Land uses within the range of this DPS include evergreen forests (52%), agricultural lands including cultivated crops (8%), and hay/pasture (1%), and urban/residential or developed areas (2%). Given these land uses, the rearing PCE in freshwater habitats may be exposed during allowable pesticide applications adjacent to exposed systems. Consequently, we expect reductions in the water quality and prey abundance in these systems.

Table 186. Co-occurrence of land use types and likelihood of a reduction in PCEs for Snake River Basin Steelhead

a.i.	Co-occurrence			Reduction in PCEs	
	Agriculture	Urban/Residential	Forest	PCEs Affected	Appreciable
Under Cancellation					
Fenamiphos[1]	NA	NA	NA	NA	NA
Azinphos methyl[2]	Orchard crops only	NA	NA	No	No
Disulfoton	Yes	Yes	Yes	No	No
Methamidophos[2]	Potatoes only	NA	NA	No	No
Methidathion	Yes	Yes	NA	Yes	No
Methyl parathion	Yes	NA	NA	Yes	No
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Yes	No
Naled	Yes	Yes	Yes	Yes	No
Phosmet	Yes	Yes	Yes	Yes	Yes
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Yes	No
Ethoprop	Yes	NA	NA	Yes	No

a.i.	Co-occurrence			Reduction in PCEs	
	Agriculture	Urban/Residential	Forest	PCEs Affected	Appreciable
Phorate	Yes	NA	NA	Yes	No

1 Crop-specific analysis was conducted.

Northern California Steelhead

Of the 50 assessed watersheds, 27 and 14 are of high and medium conservation value, respectively. Two estuarine habitat areas used for rearing and migration (Humboldt Bay and the Eel River Estuary) are also of high conservation value. Our GIS analysis indicates 10 and 27 high conservation watersheds are exposed to agriculture and urban areas, respectively. Of the medium conservation watersheds, 2 and 14 are also exposed to pesticide applications from the same above land uses, respectively.

The current condition of critical habitat for NC steelhead is moderately degraded. Removal of riparian vegetation within portions of its range promotes elevated water temperature and consequently affects the rearing PCE in freshwater and estuaries. Spawning PCE attributes such as the quality of substrate supporting spawning, incubation, and larval development are degraded by silt and sediment fines in the spawning gravel. Access to tributaries in many watersheds is affected by bridges, culverts, and forest road construction. Consequently, these uses reduce the function of the migration PCE for adults.

Co-occurrence of agriculture, urban/residential, and forestry uses is indicated on Table 187. The percentage of cultivated crop lands and developed lands overlapping with NC steelhead habitat are less than 1% and 19%, respectively. There are few areas of concentrated agriculture and most appear to be hay/pasture and are concentrated in the Lower Eel watershed and some of the other coastal valleys. Development is concentrated primarily near Eureka, on the coast in the Mad River and Redwood Creek watersheds. Much of the land area in this DPS is heavily forested, and there is a number of state and national parks. Given these land uses we expect the rearing PCE in freshwater streams and lagoons may experience some reductions in water quality and prey abundance during allowing pesticide applications adjacent to these systems.

Table 187. Co-occurrence of land use types and likelihood of a reduction in PCEs for Northern California Steelhead

a.i.	Co-occurrence			Reduction in PCEs	
	Agriculture	Urban/Residential	Forest	PCEs Affected	Appreciable
Under Cancellation					
Fenamiphos[1]	NA	NA	NA	NA	NA
Azinphos methyl[2]	Orchard crops only	NA	NA	No	No
Disulfoton	Yes	Yes	Yes	No	No
Methamidophos[2]	Potatoes only	NA	NA	No	No
Methidathion	Yes	Yes	NA	Yes	No
Methyl parathion	Yes	NA	NA	No	No
Many Use Sites					
Dimethoate	Yes	Yes	Yes	No	No
Naled	Yes	Yes	Yes	Yes	No
Phosmet	Yes	Yes	Yes	Yes	No
More Limited Use Sites					
Bensulide	Yes	Yes	NA	No	No
Ethoprop	Yes	NA	NA	No	No
Phorate	Yes	NA	NA	No	No

1: Crop-specific analysis was conducted.

Central California Coast (CCC) Steelhead

Of 47 occupied watersheds, 19 and 15 are of high and medium conservation value, respectively. Our GIS analysis indicates 12 and 15 high conservation watersheds are exposed to pesticide applications from agriculture and urban areas, respectively. Of the medium conservation watersheds, 8 and 13 are also exposed to the above land uses areas, respectively. Throughout the species' range, habitat conditions and quality have been degraded by a lack of channel complexity, eroded banks, turbid and contaminated water, low summer flow and high water temperatures, multiple contaminants found at toxic levels, and restricted access to cooler head waters from migration barriers.

The current condition of designated critical habitat for CCC steelhead is poor. The spawning PCE is impacted by sediment fines in the spawning gravel, which limits the production of aquatic stream insects adapted to running water. Elevated water temperature and impaired water quality have further reduced the quality, quantity, and function of the rearing PCE within most streams.

Co-occurrence of agriculture, urban/residential, and forestry uses is indicated in Table 188. The percentage of cultivated croplands and developed lands that overlap with CCV steelhead habitat are 27% and 10%, respectively. High densities of crop farming occur throughout the San Joaquin Basin, the Delta, and along the lower Sacramento River. Agriculture also occurs in the Russian River valley. Most of the watersheds in this DPS are heavily developed, and/or have intensive agriculture in the river valley. Given these land uses, rearing and migration PCEs in small freshwater tributaries and floodplains and the San Francisco-San Pablo-Suisan Bay estuarine complex may experience reductions in water quality and prey abundance during allowable pesticide applications adjacent to these systems.

Table 188. Co-occurrence of land use types and likelihood of a reduction in PCEs for Central California Coast Steelhead

a.i.	Co-occurrence			Reduction in PCEs	
	Agriculture	Urban/Residential	Forest	PCEs Affected	Appreciable
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Orchard crops only	NA	NA	No	No
Disulfoton	Yes	Yes	Yes	Yes	No
Methamidophos[1]	Potatoes only	NA	NA	No	No
Methidathion	Yes	Yes	NA	Yes	Yes
Methyl parathion	Yes	NA	NA	Yes	No
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Yes	No
Naled	Yes	Yes	Yes	Yes	Yes
Phosmet	Yes	Yes	Yes	Yes	Yes
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Yes	No
Ethoprop	Yes	NA	NA	Yes	No
Phorate	Yes	NA	NA	Yes	Yes

1: Crop-specific analysis was conducted.

California Central Valley (CCV) Steelhead

Of 67 occupied watersheds, 37 and 18 are of high and medium conservation value, respectively. Our GIS analysis indicates 24 and 37 high conservation watersheds are exposed to pesticide applications from agriculture and urban areas, respectively. Of the medium conservation watersheds, 14 and 17 watersheds are exposed to pesticide applications from the above land uses, respectively.

The current condition of CCV steelhead critical habitat is degraded and does not function well for ensuring species recovery. The Sacramento-San Joaquin River Delta serves little function for juvenile CCV steelhead rearing and their physiological transition to salt water. Water flow and temperature, especially during the summer months affect the condition of the spawning PCE in floodplains and flood bypasses. The rearing PCE is degraded by channelized, leveed, and riprapped river reaches and sloughs in the Sacramento-San Joaquin system. Stream channels commonly have elevated water temperature. The current condition of migration corridors is poor. 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 from contaminants in runoff. The RBDD gates obstruct migrating juveniles and adults. State and federal government pumps and associated fish facilities alter flow in the Delta and consequently obstruct migrations along the migratory corridor.

Co-occurrence of agriculture, urban/residential, and forestry uses is indicated in Table 189. The percentage of agriculture, developed, and forested lands that overlap with CCV steelhead habitat are 32%, 10%, and 58%, respectively. Heavy uses of agricultural pesticides and the high probably of mixtures increase the likelihood of negative effects on PCEs and critical habitat. AS there is a continuous run of steelhead throughout the year, the conditions of the rearing PCE in a variety of habitat are important for this DPS. Given these land uses, freshwater rearing and migration PCEs in the Sacramento River, the Delta, tributaries, tidal and non-tidal marshes, and other shallow areas in the Delta may experience reductions in water quality and prey abundance during allowable pesticide applications adjacent to these systems.

Table 189. Co-occurrence of land use types and likelihood of a reduction in PCEs for California Central Valley Steelhead

a.i.	Co-occurrence			Reduction in PCEs	
	Agriculture	Urban/Residential	Forest	PCEs Affected	Appreciable
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Yes	NA	NA	No	No
Disulfoton	Yes	Yes	Yes	Yes	No
Methamidophos[1]	Yes	NA	NA	Yes	No
Methidathion	Yes	Yes	NA	Yes	Yes
Methyl parathion	Yes	NA	NA	Yes	No
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Yes	Yes
Naled	Yes	Yes	Yes	Yes	Yes
Phosmet	Yes	Yes	Yes	Yes	Yes
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Yes	No
Ethoprop	Yes	NA	NA	Yes	No
Phorate	Yes	NA	NA	Yes	Yes

1:
Crop-specific analysis was conducted.

South-Central California Coast (S-CCC) Steelhead

Of 29 occupied watersheds, 12 and 11 are of high and medium conservation value, respectively. Our GIS analysis indicates all high conservation watersheds are exposed to pesticide applications from agriculture and urban areas. Of the medium conservation watersheds, 9 and 11 watersheds are exposed to pesticide applications from agriculture and urban areas, respectively.

Migration and rearing PCEs are degraded throughout critical habitat by elevated water temperature and contaminants from urban and agricultural runoff. The estuarine PCE is further affected when estuaries are breached and receive contaminant inputs from runoff.

Co-occurrence of agriculture, urban/residential, and forestry uses is indicated inTable 190. The percentage of cultivated crop land and developed lands that overlap with this DPS' range are 7%

and 10%, respectively. Agriculture is the dominant land use in the Salinas River valley, and there are areas of intense agriculture in the Pajaro watershed as well. Areas higher in the Salinas and Pajaro watersheds and along some of the coastal areas are less affected. Crops for which phorate may be used appear unlikely to be planted in this area. Given these uses, we expect the rearing PCE in smaller freshwater tributaries and floodplain habitats may be exposed during pesticide application adjacent to these systems. Consequently, reductions in water quality and prey abundance are expected.

Table 190. Co-occurrence of land use types and likelihood of a reduction in PCEs for South-Central California Coast Steelhead

a.i.	Co-occurrence			Reduction in PCEs	
	Agriculture	Urban/Residential	Forest	PCEs Affected	Appreciable
Under Cancellation					
Fenamiphos	NA	NA	NA	NA	NA
Azinphos methyl[1]	Orchard crops only	NA	NA	No	No
Disulfoton	Yes	Yes	Yes	Yes	No
Methamidophos[1]	Potatoes only	NA	NA	No	No
Methidathion	Yes	Yes	NA	Yes	Yes
Methyl parathion	Yes	NA	NA	Yes	No
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Yes	No
Naled	Yes	Yes	Yes	Yes	Yes
Phosmet	Yes	NA	NA	Yes	Yes
More Limited Use Sites					
Bensulide	Yes	Yes	NA	Yes	No
Ethoprop	Yes	NA	NA	Yes	No
Phorate	Yes	NA	NA	Yes	No

1:Crop-specific analysis was conducted.

Southern California (SC) Steelhead

Of 29 freshwater and estuarine watersheds, 21 and 5 are of high and medium conservation value, respectively. Our GIS analysis indicates 15 and 21 high conservation watersheds are exposed to pesticide applications from agriculture and urban areas, respectively. Of the medium conservation watersheds, all five watersheds are exposed to pesticide applications from the same above land uses.

All PCEs are affected by degraded water quality from pollutants in urban and agricultural runoff. Elevated water temperature and low water flow impact rearing and migration PCEs. The spawning PCE is affected by erosive geology and land use activities that result in an excessive amount of fines in the spawning gravel of most rivers.

Co-occurrence of agriculture, urban/residential, and forestry uses is indicated in Table 191. The percentage of cultivated crop land and developed lands within SC steelhead habitat are about 5% and 34%, respectively. Three counties within this DPS are included within the fire ant use area for phosmet. Although presumably treated areas are small, the application rate for this use is high (listed in sq. ft., but equivalent to 379 lbs a.i./A when scaled up). All of the rivers are affected by anthropogenic inputs. Given these land uses we expect some level of exposure for rearing and migration PCEs in fresh water, coastal lagoons, and downstream migration habitats.

Table 191. Co-occurrence of land use types and likelihood of adverse effects for Southern California Steelhead

a.i.	Co-occurrence			Likelihood of Adverse Effects	
	Agriculture	Urban/Residential	Forest	PCEs	Critical Habitat
Under Cancellation					
Fenamiphos[1]	NA	NA	NA	NA	NA
Azinphos methyl[2]	Orchard crops only	NA	NA	No	No
Disulfoton	Yes	Yes	Yes	Yes	No
Methamidophos[2]	Potatoes only	NA	NA	Yes	No
Methidathion	Yes	Yes	NA	Yes	Yes
Methyl parathion	Yes	NA	NA	Yes	No
Many Use Sites					
Dimethoate	Yes	Yes	Yes	Yes	Yes
Naled	Yes	Yes	Yes	Yes	Yes
Phosmet	Yes	Yes	Yes	Yes	Yes
More Limited Use Sites					

a.i.	Co-occurrence			Likelihood of Adverse Effects	
	Agriculture	Urban/Residential	Forest	PCEs	Critical Habitat
Bensulide	Yes	Yes	NA	Yes	Yes
Ethoprop	Yes	NA	NA	Yes	Yes
Phorate	Yes	NA	NA	Yes	Yes

1:Crop-specific analysis was conducted.

Table 192: Critical habitat calls for a.i.s undergoing cancellation

Species	ESU	Undergoing Cancellation					
		Fenamiphos	Azinphos-methyl	Disulfoton	Methamidophos	Methidathion	Methyl parathion
Chinook	Puget Sound	No	No	No	No	Yes	No
	Lower Columbia River	No	No	No	No	No	No
	Upper Columbia River Spring - Run	No	No	No	No	Yes	Yes
	Snake River Fall - Run	No	No	No	No	No	No
	Snake River Spring/Summer - Run	No	No	No	No	No	No
	Upper Willamette River	No	No	No	No	No	No
	California Coastal	No	No	No	No	No	No
	Central Valley Spring - Run	No	No	No	No	Yes	No
	Sacramento River Winter - Run	No	No	Yes	No	Yes	No
Chum	Hood Canal Summer - Run	No	No	No	No	No	No
	Columbia River	No	No	No	No	No	No
Coho	Lower Columbia River	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*
	Oregon Coast	No	No	No	No	No	No
	Southern Oregon and Northern California Coast	No	No	No	No	No	No
	Central California Coast	No	No	No	No	Yes	No
Sockeye	Ozette Lake	No	No	No	No	No	No
	Snake River	No	No	No	No	No	No
Steelhead	Puget Sound	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*
	Lower Columbia River	No	No	No	No	No	No
	Upper Willamette River	No	No	No	No	No	No
	Middle Columbia River	No	No	No	No	Yes	No
	Upper Columbia River	No	No	No	No	Yes	No
	Snake River	No	No	No	No	No	No
	Northern California	No	No	No	No	No	No
	Central California Coast	No	No	No	No	Yes	No
	California Central Valley	No	No	No	No	Yes	No
	South-Central California Coast	No	No	No	No	Yes	No
	Southern California	No	No	No	No	Yes	No

Table 193: Critical Habitat calls for static a.i.s

Species	ESU	Many Uses			Limited Uses		
		Dimethoate	Naled	Phosmet	Bensulide	Ethoprop	Phorate
Chinook	Puget Sound	No	Yes	Yes	No	No	No
	Lower Columbia River	No	Yes	Yes	No	No	Yes
	Upper Columbia River Spring - Run	Yes	Yes	Yes	No	No	Yes
	Snake River Fall - Run	No	No	Yes	No	No	No
	Snake River Spring/Summer - Run	No	No	Yes	No	No	No
	Upper Willamette River	No	Yes	Yes	No	No	Yes
	California Coastal	No	Yes	Yes	No	No	Yes
	Central Valley Spring - Run	Yes	Yes	Yes	Yes	No	Yes
	Sacramento River Winter - Run	Yes	Yes	Yes	Yes	Yes	Yes
Chum	Hood Canal Summer - Run	No	Yes	Yes	No	No	No
	Columbia River	No	Yes	Yes	No	No	No
Coho	Lower Columbia River	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*
	Oregon Coast	No	No	No	No	No	No
	Southern Oregon and Northern California Coast	No	Yes	Yes	No	No	No
	Central California Coast	No	Yes	Yes	No	Yes	Yes
Sockeye	Ozette Lake	No	No	No	No	No	No
	Snake River	No	Yes	Yes	No	No	No
Steelhead	Puget Sound	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*
	Lower Columbia River	No	Yes	Yes	No	No	Yes
	Upper Willamette River	No	Yes	Yes	No	No	Yes
	Middle Columbia River	No	Yes	Yes	No	No	Yes
	Upper Columbia River	No	Yes	Yes	No	No	Yes
	Snake River	No	No	Yes	No	No	No
	Northern California	No	No	No	No	No	No
	Central California Coast	No	Yes	Yes	No	No	Yes
	California Central Valley	Yes	Yes	Yes	No	No	Yes
	South-Central California Coast	No	Yes	Yes	No	No	No
	Southern California	Yes	Yes	Yes	Yes	Yes	Yes

Conclusion

In the *Integration and Synthesis of Effects to Threatened and Endangered Pacific Salmon* section, we described NMFS' assessment of the likelihood of negative effects posed to the survival and recovery of listed Pacific salmonids as a result of EPA's registration of azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, methamidophos, methidathion, methyl parathion, naled, phorate, and phosmet. The likelihood of negative effects assigned to each ESU/DPS for each a.i. reflects NMFS' evaluation of the likelihood that a compound will cause reductions in species viability. This likelihood informs NMFS' determination of whether the action is likely to jeopardize the continued survival and recovery of listed salmonids.

Separate determinations were made for each species / a.i. pair. NMFS considered the extent and location of use sites, relevant cancellation information, and all stressors of the action. The high, medium, and low likelihood of effects translated directly into NMFS's final determination on the effects of the action. The status of the species (threatened or endangered) was also taken into consideration. Species that had a high likelihood of negative effects for a given a.i. translated to a jeopardy determination for that pairing. As species listed as 'endangered' are more vulnerable, for these species a medium likelihood of negative effects also translated to a jeopardy determination. 'Threatened' species/a.i. combinations that resulted in a medium likelihood of effects were considered not jeopardized by the stressors of the action. All species/a.i. pairings with a low likelihood of negative effects were considered to not jeopardize the survival and recovery of that ESU/DPS. Jeopardy determinations for each species/a.i. pair are given below in Table 194 and Table 195.

In the *Integration and Synthesis of Effects to Critical Habitat* section, we described NMFS' assessment of the likelihood of an appreciable reduction in the conservation value of designated critical habitat and PCEs as a result of EPA's registration of azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, methamidophos, methidathion, methyl parathion, naled, phorate, and phosmet. This likelihood allows NMFS to determine whether critical habitat is likely to be destroyed or adversely modified by the proposed action. For most ESUs/DPSs, the PCEs identified during the critical habitat designation process are types of habitat that support various life stages and functions of salmonids (e.g., spawning, rearing, migratory habitat). These PCEs have attributes that allow them to support the listed salmonids. NMFS considered reductions in the attributes of water quality and prey availability in evaluating the effects of the proposed action to critical habitat.

Separate determinations were made for all critical habitat/a.i. pairs. We expect that water quality and prey availability would be negatively affected if use sites are located within critical habitat. To determine whether the conservation value of the PCES would be reduced appreciably, NMFS considered the extent and location of use sites, relevant cancellation information, and all stressors of the action. ESU/DPSs whose designated critical habitat was likely to experience a reduction in the conservation value due to exposure to an a.i. translates to an adverse modification determination for that pairing. Final determinations for the adverse modification of critical habitat are given below in Table 196 and Table 197.

Table 194: Jeopardy determinations for a.i.s undergoing cancellation

Species	ESU	Undergoing Cancellation					
		Fenamiphos	Azinphos-methyl	Disulfoton	Methamidophos	Methidathion	Methyl parathion
Chinook	Puget Sound	No	No	No	No	Jeopardy	No
	Lower Columbia River	No	No	No	No	No	No
	Upper Columbia River Spring - Run	No	No	No	No	Jeopardy	No
	Snake River Fall - Run	No	No	No	No	No	No
	Snake River Spring/Summer - Run	No	No	No	No	No	No
	Upper Willamette River	No	No	No	No	No	No
	California Coastal	No	No	No	No	No	No
	Central Valley Spring - Run	No	No	No	No	Jeopardy	No
	Sacramento River Winter - Run	No	No	No	No	Jeopardy	No
Chum	Hood Canal Summer - Run	No	No	No	No	No	No
	Columbia River	No	No	No	No	No	No
Coho	Lower Columbia River	No	No	No	No	No	No
	Oregon Coast	No	No	No	No	No	No
	Southern Oregon and Northern California Coast	No	No	No	No	No	No
	Central California Coast	No	No	No	No	Jeopardy	No
Sockeye	Ozette Lake	No	No	No	No	No	No
	Snake River	No	No	No	No	No	No
Steelhead	Puget Sound	No	No	No	No	Jeopardy	No
	Lower Columbia River	No	No	No	No	No	No
	Upper Willamette River	No	No	No	No	No	No
	Middle Columbia River	No	No	No	No	Jeopardy	No
	Upper Columbia River	No	No	No	No	Jeopardy	No
	Snake River	No	No	No	No	No	No
	Northern California	No	No	No	No	No	No
	Central California Coast	No	No	No	No	Jeopardy	No
	California Central Valley	No	No	No	No	Jeopardy	No
	South-Central California Coast	No	No	No	No	Jeopardy	No
	Southern California	No	No	No	No	Jeopardy	No

Table 195: Jeopardy determinations for static a.i.s

Species	ESU	Many Uses			Limited Uses		
		Dimethoate	Naled	Phosmet	Bensulide	Ethoprop	Phorate
Chinook	Puget Sound	No	Jeopardy	Jeopardy	No	No	No
	Lower Columbia River	No	Jeopardy	Jeopardy	No	No	Jeopardy
	Upper Columbia River Spring - Run	Jeopardy	Jeopardy	Jeopardy	No	No	Jeopardy
	Snake River Fall - Run	No	No	No	No	No	No
	Snake River Spring/Summer - Run	No	No	No	No	No	No
	Upper Willamette River	No	Jeopardy	Jeopardy	No	No	Jeopardy
	California Coastal	No	Jeopardy	Jeopardy	No	No	Jeopardy
	Central Valley Spring - Run	Jeopardy	Jeopardy	Jeopardy	Jeopardy	No	Jeopardy
	Sacramento River Winter - Run	Jeopardy	Jeopardy	Jeopardy	Jeopardy	Jeopardy	Jeopardy
Chum	Hood Canal Summer - Run	No	Jeopardy	No	No	No	No
	Columbia River	No	Jeopardy	No	No	No	No
Coho	Lower Columbia River	No	Jeopardy	Jeopardy	No	No	Jeopardy
	Oregon Coast	No	No	No	No	No	No
	Southern Oregon and Northern California Coast	No	Jeopardy	Jeopardy	No	No	No
	Central California Coast	No	Jeopardy	Jeopardy	No	Jeopardy	Jeopardy
Sockeye	Ozette Lake	No	No	No	No	No	No
	Snake River	No	Jeopardy	No	No	No	No
Steelhead	Puget Sound	No	Jeopardy	Jeopardy	No	No	No
	Lower Columbia River	No	Jeopardy	Jeopardy	No	No	Jeopardy
	Upper Willamette River	No	Jeopardy	Jeopardy	No	No	Jeopardy
	Middle Columbia River	No	Jeopardy	Jeopardy	No	No	Jeopardy
	Upper Columbia River	No	Jeopardy	Jeopardy	No	No	Jeopardy
	Snake River	No	No	Jeopardy	No	No	No
	Northern California	No	No	No	No	No	No
	Central California Coast	No	Jeopardy	Jeopardy	No	No	Jeopardy
	California Central Valley	Jeopardy	Jeopardy	Jeopardy	No	No	Jeopardy
	South-Central California Coast	No	Jeopardy	Jeopardy	No	No	No
	Southern California	Jeopardy	Jeopardy	Jeopardy	Jeopardy	Jeopardy	Jeopardy

Table 196: Adverse modification determinations for a.i.s undergoing cancellation

Species	ESU	Undergoing Cancellation					
		Fenamiphos	Azinphos-methyl	Disulfoton	Methamidophos	Methidathion	Methyl parathion
Chinook	Puget Sound	No	No	No	No	Ad Mod	No
	Lower Columbia River	No	No	No	No	No	No
	Upper Columbia River Spring - Run	No	No	No	No	Ad Mod	No
	Snake River Fall - Run	No	No	No	No	No	No
	Snake River Spring/Summer - Run	No	No	No	No	No	No
	Upper Willamette River	No	No	No	No	No	No
	California Coastal	No	No	No	No	No	No
	Central Valley Spring - Run	No	No	No	No	Ad Mod	No
	Sacramento River Winter - Run	No	No	No	No	Ad Mod	No
Chum	Hood Canal Summer - Run	No	No	No	No	No	No
	Columbia River	No	No	No	No	No	No
Coho	Lower Columbia River	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*
	Oregon Coast	No	No	No	No	No	No
	Southern Oregon and Northern California Coast	No	No	No	No	No	No
	Central California Coast	No	No	No	No	Ad Mod	No
Sockeye	Ozette Lake	No	No	No	No	No	No
	Snake River	No	No	No	No	No	No
Steelhead	Puget Sound	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*
	Lower Columbia River	No	No	No	No	No	No
	Upper Willamette River	No	No	No	No	No	No
	Middle Columbia River	No	No	No	No	Ad Mod	No
	Upper Columbia River	No	No	No	No	Ad Mod	No
	Snake River	No	No	No	No	No	No
	Northern California	No	No	No	No	No	No
	Central California Coast	No	No	No	No	Ad Mod	No
	California Central Valley	No	No	No	No	Ad Mod	No
	South-Central California Coast	No	No	No	No	Ad Mod	No
	Southern California	No	No	No	No	Ad Mod	No

Table 197: Adverse modification determinations for static a.i.s

Species	ESU	Many Uses			Limited Uses		
		Dimethoate	Naled	Phosmet	Bensulide	Ethoprop	Phorate
Chinook	Puget Sound	No	Ad Mod	Ad Mod	No	No	No
	Lower Columbia River	No	Ad Mod	Ad Mod	No	No	Ad Mod
	Upper Columbia River Spring - Run	Ad Mod	Ad Mod	Ad Mod	No	No	Ad Mod
	Snake River Fall - Run	No	No	Ad Mod	No	No	No
	Snake River Spring/Summer - Run	No	No	Ad Mod	No	No	No
	Upper Willamette River	No	Ad Mod	Ad Mod	No	No	Ad Mod
	California Coastal	No	Ad Mod	Ad Mod	No	No	Ad Mod
	Central Valley Spring - Run	Ad Mod	Ad Mod	Ad Mod	Ad Mod	No	Ad Mod
	Sacramento River Winter - Run	Ad Mod	Ad Mod	Ad Mod	Ad Mod	Ad Mod	Ad Mod
Chum	Hood Canal Summer - Run	No	Ad Mod	Ad Mod	No	No	No
	Columbia River	No	Ad Mod	Ad Mod	No	No	No
Coho	Lower Columbia River	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*
	Oregon Coast	No	No	No	No	No	No
	Southern Oregon and Northern California Coast	No	Ad Mod	Ad Mod	No	No	No
	Central California Coast	No	Ad Mod	Ad Mod	No	Ad Mod	Ad Mod
Sockeye	Ozette Lake	No	No	No	No	No	No
	Snake River	No	Ad Mod	Ad Mod	No	No	No
Steelhead	Puget Sound	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*
	Lower Columbia River	No	Ad Mod	Ad Mod	No	No	Ad Mod
	Upper Willamette River	No	Ad Mod	Ad Mod	No	No	Ad Mod
	Middle Columbia River	No	Ad Mod	Ad Mod	No	No	Ad Mod
	Upper Columbia River	No	Ad Mod	Ad Mod	No	No	Ad Mod
	Snake River	No	No	Ad Mod	No	No	No
	Northern California	No	No	No	No	No	No
	Central California Coast	No	Ad Mod	Ad Mod	No	No	Ad Mod
	California Central Valley	Ad Mod	Ad Mod	Ad Mod	No	No	Ad Mod
	South-Central California Coast	No	Ad Mod	Ad Mod	No	No	No
	Southern California	Ad Mod	Ad Mod	Ad Mod	Ad Mod	Ad Mod	Ad Mod

Reasonable and Prudent Alternatives

Regulations (50 CFR §402.02) implementing section 7 of the ESA define reasonable and prudent alternatives as alternative actions, identified during formal consultation, that: (1) can be implemented in a manner consistent with the intended purpose of the action; (2) can be implemented consistent with the scope of the action agency's legal authority and jurisdiction; (3) are economically and technologically feasible; and (4) NMFS believes would avoid the likelihood of jeopardizing the continued existence of listed species or resulting in the destruction or adverse modification of critical habitat.

This Opinion has concluded that EPA's proposed registration of pesticides containing bensulide, dimethoate, ethoprop, methidathion, naled, phorate, and phosmet are each likely to jeopardize the continued existence of one or more of the 28 endangered and threatened Pacific salmonids and are each likely to destroy or adversely modify designated critical habitat for one or more of the 28 threatened and endangered salmonids. "Jeopardize the continued existence of" means "to engage in an action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species" (50 CFR §402.02).

NMFS reached this conclusion because predicted concentrations of these seven a.i.s in salmonid habitats, particularly in floodplain habitats[17], are likely to cause adverse effects to at least one ESU or DPS of listed Pacific salmonids including significant reductions in growth or survival.

As a result, twenty-three ESUs/DPSs of listed Pacific salmonids are likely to suffer reductions in viability from at least one of the a.i.s given the severity of expected changes

[17] Floodplain 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 tributaries.

Main channel –the stream channel that includes the thalweg (longitudinal continuous deepest portion of the channel.

in abundance and productivity associated with the proposed action. These adverse effects are expected to appreciably reduce the likelihood of both the survival and recovery of these listed Pacific salmonids. EPA's proposed registration of bensulide, dimethoate, ethoprop, methidathion, naled, phorate, and phosmet likely to jeopardize 23 ESUs and not likely to jeopardize 5 ESUs. EPA's proposed registration of bensulide, dimethoate, ethoprop, methidathion, naled, phorate, and phosmet is also likely to result in the destruction or adverse modification of critical habitat for 25 affected ESUs/DPSs because of adverse effects from at least one active ingredient on salmonid prey and water quality in freshwater rearing, spawning, and foraging areas.

The Reasonable and Prudent Alternative (RPA) accounts for the following issues: (1) the action will result in exposure to other chemical stressors in addition to the a.i. that may increase the risk of the action to listed species, including unspecified inert ingredients, adjuvants, and tank mixes; (2) exposure to chemical mixtures containing the a.i.s and other cholinesterase-inhibiting compounds result in additive and synergistic responses; and (3) exposure to other chemicals and physical stressors (*e.g.*, temperature) in the baseline habitat will likely intensify response to the a.i.s.

The action as implemented under the RPA will remove the likelihood of jeopardy and of destruction or adverse modification of critical habitat by reducing the concentrations of each of these a.i.s to below concentrations predicted cause significant declines in model population lambdas, (a measure of abundance and productivity). In the proposed RPA, NMFS does not attempt to ensure there is no take of listed species. NMFS believes take will occur, and has provided an incidental take statement exempting that take from the take prohibitions, so long as the action is conducted according to the RPA and reasonable and prudent measures (RPM). Avoiding take altogether would most likely entail canceling registration, or prohibiting use in watersheds inhabited by salmonids. NMFS recognizes the registration of methidathion is canceled and exposure to this a.i.s will decline accordingly. However, the terms of the cancellation for this a.i.s have provisions allowing for pesticide product sales and use to continue for several years, with no specific end date. The RPA therefore applies to methidathion in the geographic area of those ESUs or DPSs for which NMFS determined that there was likely jeopardy or likely adverse modification or both. The goal of the RPA is to reduce exposure to ensure that the action is not likely to jeopardize listed species or destroy or adversely modify critical habitat.

The RPA is comprised of five required elements that must be implemented in its entirety within one year of the EPA's receipt of this Opinion to ensure the registration of these pesticides is not likely to jeopardize endangered or threatened Pacific salmonids under the jurisdiction of NMFS or destroy or adversely modify critical habitat designated for these species. For each active ingredient, the elements of the RPA apply only to those ESUs/DPSs where NMFS has determined that registration of that a.i. causes likely jeopardy or the destruction or adverse modification of critical habitat (Table 194 through Table 197). These elements rely upon recognized practices for reducing drift and runoff of pesticide products into aquatic habitats.

Specific Elements of the Reasonable and Prudent Alternative

Elements 1-4 shall be specified on FIFRA labels of all pesticide products containing bensulide, dimethoate, ethoprop, methidathion, naled, phorate, and phosmet. Alternatively, the label could direct pesticide users to the EPA's Endangered Species Protection Program (ESPP) bulletins that specify elements 1-4. For purposes of this RPA salmonid habitats are defined as freshwaters, estuarine habitats, and nearshore marine habitats including bays within the ESU/DPS ranges including migratory corridors. The freshwater habitats include intermittent streams and other habitats temporally connected to salmonid-bearing waters when those habitats contain water. Freshwater habitats also include all known types of floodplain habitats as well as drainages, ditches, and other man-made conveyances to salmonid habitats that lack salmonid exclusion devices (*e.g.*, screens).

*Element 1***.** Do not apply when wind speeds are greater than or equal to 10 mph.

Element 2. For all uses do not apply pesticide products when soil moisture is at field capacity, or when a storm event likely to produce runoff from the treated area is forecasted by to occur within 48 h following application by NOAA/NWS (National Weather Service) or other similar forecasting service.

Element 3. EPA will implement NMFS approved risk reduction measures to ensure maximum concentrations of the a.i.s predicted in salmonid habitats will not exceed the values specified in Table 198 for any allowed use. These values represent the highest concentrations that may be achieved in salmonid habitats, rather than time-weighted average concentrations, considering the range in potential droplet size spectrum, release heights, wind speeds, and wind directions that may be associated with all labeled application methods (*e.g.*, agricultural applications, vector control in public health programs, *etc*.). The maximum predicted concentrations shall account for potential contributions from both runoff and drift to salmonid habitats, as appropriate. Risk reduction measures shall account for the predicted maximum concentrations in all salmonid habitats, including a modeled floodplain habitat of 1-2 m wide and 0.1 m deep. They shall also account for potential increases in aquatic concentrations associated with the maximum application rate and the maximum number of times an a.i. may be applied per season according to label restrictions. Risk reductions measures may include, but are not limited to:

a) Buffers – Example: Do not apply pesticide products containing the a.i. within specified distances of salmonid habitats. Buffers only apply when water exists in the stream or habitat and shall be measured from the water's edge of salmonid habitat, including floodplain, to the point of deposition (below spray nozzle).
b) Vegetated filter strips- Example: Provide a 20 ft (6.1 m) minimum strip of non-crop vegetation (on which no pesticides shall be applied) on the downhill side of the application site immediately adjacent to any surface waters that have a connection to salmonid-bearing waters. This includes drainage systems that have salmonid exclusion devices, but drain to salmonid-bearing waters.
c) Reduction in the maximum single application rate, or maximum seasonal application rate - Example: Do not apply more than 1.5 lbs a.i./A/application or more than 4.5 lbs a.i./A/season.
d) Reduction in the number of applications allowed, or increase in the minimum application interval. Example: Do not apply more this a.i. more than 10 times per season. Allow a minimum of 7 days between applications.
e) Restrictions on application methods- Example: Apply by ground application methods only.
f) Restrictions on use sites- Example: prohibit applications of a.i. on high risk use sites such as "swamps" and "tidal marshes."

Table 198. Maximum concentration limits for active ingredients in salmonid habitat

Active Ingredient	Maximum Concentration Limit for salmonid habitat µg/L
Bensulide	200
Dimethoate	60
Ethoprop	20
Methidathion	0.3
Naled	0.2
Phorate	0.1
Phosmet	0.5

The maximum concentration limits in Table 198 are approximately two-fold lower than concentrations associated with significant decreases in population growth rates (lambda). These values were selected by considering the likelihood that model estimates accurately predict reductions in population growth rate by weighing the model assumptions, model limitations, and other pesticide-specific considerations (Table 199). For example, some of the model assumptions increase the likelihood that risk is overestimated (*e.g.*, all individuals of the population are exposed) while others increase the likelihood that risk is underestimated (*e.g.*, reproductive impacts will not contribute to declines in population growth rate. The maximum concentrations limits were established by weighing model assumptions (as shown in Table 199) and other considerations regarding the risk associated with the use of pesticide product containing the a.i.s.

Table 199. Considerations for developing maximum concentration limits for salmonid habitats

<table>
<tr><th>Model assumptions and other assumptions and considerations</th><th>Increase likelihood that risk of significant reduction in population growth rate is overestimated</th><th>Increase likelihood that risk of significant reduction in population growth rate is underestimated</th></tr>
<tr><td>4-day exposure assumed versus maximum concentration limit</td><td>X</td><td></td></tr>
<tr><td>Assumption that all individuals of the population exposed</td><td>X</td><td></td></tr>
<tr><td>Assumption that toxicity inputs accurately reflect sensitivity of listed salmonids and their prey</td><td colspan="2">May either overestimate or underestimate risk</td></tr>
<tr><td>Control population assumptions (survival rate, reproductive contributions, etc.)</td><td colspan="2">May either overestimate or underestimate risk</td></tr>
<tr><td>Uncertainty associated with effectiveness of risk reduction method employed (e.g. buffers)</td><td colspan="2">May either overestimate or underestimate risk</td></tr>
<tr><td>Assumption that population will experience a single exposure event</td><td></td><td>X</td></tr>
</table>

Model assumptions and other assumptions and considerations	Increase likelihood that risk of significant reduction in population growth rate is overestimated	Increase likelihood that risk of significant reduction in population growth rate is underestimated
Assumption that lethality or somatic growth may impact population growth rate, but not both concurrently		X
Assumption that no baseline stressors (*e.g.* temp) will increase response		X
Assumption that exposure to other AChE inhibitors will not occur or increase response		X
Assumption that other a.i.s in pesticide formulations will not increase response		X
Assumption that inerts ingredients in the pesticide formulation will not increase response		X
Assumption that tank mixture ingredients will not increase response		X
Assumption that other known, unknown, or uncertain effects will not contribute to declines in population growth rates (*e.g.* impacts to reproductive endpoints)		X

Element 4. Report all incidents of fish mortality that occur within the vicinity of the treatment area, including areas downstream and downwind, in the four days following application of and of these a.i.s to EPA OPP (703-305-7695). Alternatively, these incidents may be reported to the pesticide manufacturer through the phone number on the product label once EPA modifies FIFRA 6(a)2 to require registrants to report all fish kills immediately, regardless of incident classification (*i.e.* both minor and major incidents). EPA shall submit an annual report to NMFS OPR that identifies the total number of fish affected and incident locations.

*Element 5***.** In addition to the labeling requirements above, EPA shall develop and implement a NMFS-approved effectiveness monitoring plan for floodplain habitats, and produce annual reports of the results. The plan shall identify representative floodplain habitats prone to drift and runoff of pesticides within agricultural areas. The representative floodplain habitat sampling sites shall include floodplain habitats currently used by threatened and endangered Pacific salmonids, as identified by NMFS biologists. Sampling sites include at least two sites for each general species (*i.e.*, coho salmon, chum salmon, steelhead, sockeye salmon, and ocean-type Chinook and stream-type Chinook salmon). Sampling shall consist of daily collection of surface water samples for seven consecutive days during three periods of high application for these a.i.s. Collected water samples will be analyzed for current-use OPs and carbamates following USGS schedule for analytical chemistry. The report shall be submitted to NMFS OPR and will summarize annual monitoring data and provide all raw data.

Because this Opinion has found jeopardy and destruction or adverse modification to designated critical habitat, the EPA is required to notify NMFS of its final decision on the implementation of the reasonable and prudent alternatives (50 CFR §402.15(b)).

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.

Amount or Extent of Take Anticipated

As described earlier in this Opinion, this is a consultation on the EPA's registration of pesticide products containing azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, methamidophos, methyl parathion, naled, phorate, or phosmet, and their formulations as they are used in the Pacific Northwest and California and the impacts of these applications on listed ESUs/DPSs of Pacific salmonids. The EPA authorizes use of these pesticide products for pest control purposes across multiple landscapes. 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. However, even though the court's order did not address the two more recently listed ESUs and DPSs, NMFS analyzed the impacts of EPA's actions to them because they belong to the same taxon and the analysis requires consideration of the same information. 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 across the states of California, Idaho, Oregon, and Washington to 28 listed Pacific 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 RPAs are designed to reduce this exposure but not eliminate it. Pesticide runoff and drift of azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, methamidophos, methidathion, methyl parathion, naled, phorate, and phosmet are most likely to reach streams and other aquatic sites when they are applied to crops and other land use settings located adjacent to wetlands, riparian areas, ditches, off-channel habitats, and intermittent streams. These inputs into aquatic habitats are especially high when rainfall immediately follows applications. The effects of pesticides and other contaminants found in urban runoff, especially from areas with a high degree of impervious surfaces, may also exacerbate degraded water quality conditions of receiving waters used by salmon. Urban runoff is also generally warmer in temperature, and elevated water temperature poses negative effects on certain life history phases for salmon. The range of effects of the 12 a.i.s on salmonids includes reductions in growth, prey capture, and swimming ability, impaired olfaction affecting homing and reproductive behaviors, and increased susceptibility to predation and disease. Thus, we expect some exposed fish will respond to these effects by changing normal behaviors. In some cases, fish may die, be injured, or suffer sublethal effects. 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. Incomplete information on the proposed action (*i.e.,* no master label summarizing all authorized uses of pesticide products azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, methamidophos, methyl parathion, naled, phorate, and phosmet);
2. Limited use and exposure data on stressors of the action for non-agricultural uses of these pesticides;
3. Minimal information on exposure and toxicity for pesticide formulations, adjuvants, and other/inert ingredients within registered formulations;
4. No information on permitted tank mixtures and associated exposure estimates;
5. Limited data on toxicity of environmental mixtures;
6. No known method to predict synergistic responses from exposure to combinations of the 12 a.i.s;
7. Annual variable conditions regarding land use, crop cover, and pest pressure;
8. Variable temporal and spatial conditions within each ESU, especially at the population-level; and
9. 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 azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, methamidophos, methyl parathion, naled, phorate, or phosmet, 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 these a.i.s than many other species of fish, so that if there are kills of other freshwater fishes attributed to use of these pesticides, 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 one of these 12 ingredients, its metabolites, or degradates, if the a.i is known to have been applied in the vicinity and may reasonably be supposed to have run off or drifted into the affected area, and if surface water samples, AChE measurement, or pathology indicate lethal levels of the a.i.(s).

NMFS notes that with increased monitoring and study of the impact of these pesticides on water quality, particularly water quality in off-channel 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.

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 permit or grant document, the protective coverage of section 7(o)(2) may lapse. In order to monitor the impact of incidental take, the EPA must report the progress of the action and its impact on the species to 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 cumulative impacts of its long-term registration of pesticide products containing azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, methamidophos, methyl parathion, naled, phorate, or phosmet; (b) evaluate the direct, indirect, or cumulative impacts of pesticide misapplications in the aquatic habitats in which they occur; and (c) the consequences of those effects on listed Pacific salmonids under NMFS' jurisdiction. 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 from use of pesticide products containing azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, methamidophos, methidathion, methyl parathion, naled, phorate, or phosmet by reducing the potential of chemicals to reach salmon-bering waters;
2. Monitor any incidental take or surrogate measure of take that occurs from the action; and
3. Report annually to NMFS OPR on the monitoring results from the previous year.

Terms and Conditions

To be exempt from the prohibitions of section 9 of the ESA, within one year 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. a. Do not apply pesticide products when wind speeds are greater than or equal to 10 mph.
   b. Do not apply pesticide products when soil moisture is at field capacity, or when a storm event likely to produce runoff from the treated area is forecasted by to occur within 48 h following application by NOAA/NWS (National Weather Service) or other similar forecasting service.

2. a. EPA shall include the following instructions requiring reporting of fish kills either on the labels for all products containing azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, methamidophos, methyl parathion, naled, phorate, and phosmet or in ESPP Bulletins:

   NOTICE: Incidents where salmon appear injured or killed as a result of pesticide applications shall be reported to NMFS OPR at 301-713-1401 and EPA at 703-305-7695. 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.

b. EPA shall report to NMFS OPR any incidences regarding azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, methamidophos, methyl parathion, naled, phorate, or phosmet effects on aquatic ecosystems added to its incident database that it has classified as probable or highly probable.

3. EPA shall provide OPR a commencement date for annual reporting of monitoring results.

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 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 off-channel habitats; and
3. Develop models to estimate pesticide concentrations in aquatic habitats associated with non-agricultural applications, particularly in residential and industrial environments.

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.

Reinitiation Notice

This concludes formal consultation on the EPA's proposed registration of pesticide products containing azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, methamidophos, methyl parathion, naled, phorate, and phosmet 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. If reinitiation of consultation appears warranted due to one or more of the above circumstances, EPA must contact NMFS OPR. In the event reinitiation conditions (1), (2), or (3) is met, reinitiation will be only for the a.i.(s) which meet that condition, not for all 12 a.i.s considered in the Opinon. 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.

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Appendix 1: Population Modeling

Introduction

To assess the potential for adverse impacts of the anticholinesterase insecticides on Pacific salmon populations, a model was developed that explicitly links impairments in the biochemistry, behavior, prey availability and somatic growth of individual salmon to the productivity of salmon populations. More specifically, the model connects known effects of the pesticides on salmon physiology and behavior with community-level effects on salmon prey to estimate population-level effects on salmon. The model used here is an extension of one developed for investigating the direct effects of pesticides on the biochemistry, behavior and growth of ocean-type Chinook salmon (Baldwin et al., 2009).

In the freshwater portion of their life, Pacific salmon may be exposed to insecticides that act by inhibiting acetylcholinesterase (AChE). Acetylcholinesterase is a crucial enzyme in the proper functioning of cholinergic synapses in the central and peripheral nervous systems of vertebrates and invertebrates. Of consequence to salmon, anticholinesterase insecticides have been shown to interfere with salmon swimming behavior (Beauvais et al. 2000, Brewer et al. 2001, Sandahl et al. 2005), feeding behavior (Sandahl et al. 2005), foraging behavior (Morgan and Kiceniuk 1990), homing behavior (Scholz et al. 2000), antipredator behaviors (Scholz et al. 2000) and reproductive physiology (Moore and Waring 1996, Waring and Moore 1997, Scholz et al. 2000).

Anticholinesterase insecticides have also been found to reduce benthic densities of aquatic invertebrates and alter the composition of aquatic communities (Liess and Schulz 1999, Schulz and Liess 1999, Schulz et al. 2002, Fleeger et al. 2003, Schulz 2004, Chang et al. 2005, Relyea 2005). Spray drift and runoff from agricultural and urban areas can expose aquatic invertebrates to relatively low concentrations of insecticides for as little as minutes or hours, but populations of many taxa can take months or even years to recover to pre-exposure or reference densities (Wallace et al. 1991, Liess and Schulz 1999, Anderson et al. 2003, Stark et al. 2004). For example, when an aquatic macroinvertebrate community in a German stream was exposed to runoff containing parathion (an acetylcholinesterase inhibitor) and fenvalerate (another commonly used insecticide), eight of eleven abundant species disappeared and the remaining three were reduced in abundance (Liess and Schulz 1999). Long-term changes in invertebrate densities and community composition likely result in reductions in salmon prey availability. Therefore, in addition to the direct impacts that acetylcholinesterase inhibitors have on salmon, there may also be, independently, significant indirect effects to salmon via their prey (Peterson et al. 2001a). Wild juvenile salmon feed primarily on invertebrates in the water column and those trapped on the water's surface, actively selecting the largest items available (Healey 1991, Quinn 2005). Salmon are often found to be food limited (Quinn 2005), suggesting that a reduction in prey number or size due to insecticide exposure may further stress salmon. For example, Davies and Cook (1993) found that several months following a spray drift event, benthic and drift densities were still reduced in exposed stream reaches. Consequently, brown trout in the exposed reaches fed less and grew at a slower rate compared to those in unexposed stream reaches (Davies and Cook 1993). Although the insecticide in their study was cypermethrin (a pyrethroid), similar reductions in macroinvertebrate density and recovery times have been found in studies with acetylcholinesterase inhibitors (Liess and Schulz 1999, Schulz et al. 2002), suggesting indirect effects to salmon via prey availability may be similar.

One likely biological consequence of reduced swimming, feeding, foraging, and prey availability is a reduction in food uptake and, subsequently, a reduction in somatic growth of exposed fish. Juvenile growth is a critical determinant of freshwater and marine survival for Chinook salmon (Higgs et al. 1995). Reductions in the somatic growth rate of salmon fry and smolts are believed to result in increased size-dependent mortality (Healey 1982, West and Larkin 1987, Zabel and Achord 2004). Zabel and Achord (2004) observed size-dependent survival for juvenile salmon during the freshwater phase of their outmigration. Mortality is also higher among smaller and slower growing salmon because they are more susceptible to predation during their first winter (Healey 1982, Holtby et al. 1990, Beamish and Mahnken 2001). These studies suggest that factors affecting the organism and reducing somatic growth, such as anticholinesterase insecticide exposure, could result in decreased first-year survival and, thus, reduce population productivity.

Changes to the size of juvenile salmon from exposure to anticholinesterase pesticides were linked to salmon population demographics. We used size-dependent survival of juveniles during a period of their first year of life. We did this by constructing and analyzing general life-history matrix models for coho salmon (*Oncorhynchus kisutch*), sockeye salmon (*O. nerka*) and ocean-type and stream-type Chinook salmon (*O. tshawytscha*). A steelhead (*O. mykiss*) life-history model was not constructed due to the lack of demographic information relating to the proportions of resident and anadromous individuals, the freshwater residence time of steelhead, and rates of repeated spawning. The basic salmonid life history modeled consisted of hatching and rearing in freshwater, smoltification in estuaries, migration to the ocean, maturation at sea, and returning to the natal freshwater stream for spawning followed shortly by death. Differences between the modeled strategies are lifespan of the female, time to reproductive maturity, and the number and relative contribution of the reproductive age classes (Figure 1). The coho females we modeled reach reproductive maturity at age 3 and provide all of the reproductive contribution. Sockeye females reach maturity at age 4 or 5, but the majority of reproductive contributions are provided by age 4 females. Chinook females can mature at age 3, 4 or 5, with the majority of the reproductive contribution from ages 4 and 5. The primary difference between the ocean-type and stream-type Chinook is the juvenile freshwater residence with ocean-type juveniles migrating to the ocean as subyearlings and stream-type overwintering in freshwater and migrating to the ocean as yearlings. The models depicted general populations representing each life-history strategy and were constructed based upon literature data described below. Specific populations were not modeled due to the difficulty in finding sufficient demographic and reproductive data for a single population. Our modeled populations were not designed to represent particular salmon population segments, and they did not incorporate potentially influential life history information that may vary among populations. This includes, for example, density-dependent effects on juvenile growth and survival as well as the effects of adult migration (*i.e.,* straying) on adult spawner abundance. Our results using a more simplified and generic model for show how improving water quality conditions by reducing the pesticide load could potentially impact population viability and rate of recovery. This should allow resource managers to consider pesticides at the same biological scale as physical and biological stressors when prioritizing habitat restoration activities.

A separate acute toxicity model was constructed that estimated the population-level impacts of juvenile mortality resulting from exposure to lethal concentrations of the 12 pesticides. These models excluded sublethal and indirect effects of the pesticide exposures and focused on the population-level outcomes resulting from an annual exposure of juveniles to a pesticide. The lethal impact was implemented as a change in first year survival for each of the salmon life-history strategies.

The overall model endpoint used to assess population-level impacts for both the growth and acute lethality models was the percent change in the intrinsic population growth rate (lambda, λ) resulting from the pesticide exposure. Change in λ is an accepted population parameter often used in evaluating population productivity, status, and viability. The National Marine Fisheries Service uses changes in λ when estimating the status of species, conducting risk and viability assessments, developing Endangered Species Recovery Plans, composing Biological Opinions, and communicating with other federal, state and local agencies (McClure et al. 2003). While values of $\lambda<1.0$ indicate a declining population, negative changes in lambda greater than the natural variability for the population indicate a loss of productivity. This can be a cause for concern since the decline could make a population more susceptible to dropping below 1.0 due to impacts from multiple stressors.

The following models were developed to serve as a means to assess the potential effects on ESA-listed salmon populations from exposure to AChE inhibiting pesticides, including n-methyl carbamates and organophosphorus insecticides. The growth model focuses on the impacts to prey abundance and a salmon's ability to feed which are integrated into reductions in juvenile growth. Assessing the results from different pesticide exposure scenarios relative to a control (i.e. unexposed) scenario can indicate the potential for sublethal pesticide exposures to lead to changes in the somatic growth and survival of individual subyearling salmon. Consequently, subsequent changes in salmon population dynamics as indicated by percent change in a population's intrinsic rate of increase assists us in forecasting the potential population-level impacts to listed populations. Also, the model helps us understand the potential influence of life-history strategies that might explain differential results within the species modeled.

Methods

The model consists of two parts, an organismal portion and a population portion. The organismal portion of the model links AChE inhibition and reduced prey abundance due to insecticide exposure to potential reductions in the growth of individual fish. The population portion of the model links the sizes of individual subyearling salmon to their survival and the subsequent growth of the population. Models were constructed using MATLAB 7.7.0 (R2008b) (The MathWorks, Inc. Natick, MA).

Organismal Model

For the organismal model a relationship between AChE activity and somatic growth of salmonid fingerlings was developed using a series of relationships between pesticide exposure, AChE activity, feeding behavior, food uptake, and somatic growth rate (Figures 2-4). The model incorporates empirical data when available. Since growth and toxicity data are limited, extrapolation from one salmon species to the others was done with the assumption that the salmon stocks would exhibit similar physiological and toxicological responses. Sigmoidal dose-response relationships based upon the AChE inhibition EC50 values and their slopes are used to determine the level of AChE activity (Figure 2A, 2B, 2C) from the exposure concentration of each pesticide exposure or pulse.

A linear relationship based on empirical data related AChE activity to feeding behavior (Sandahl et al. 2005, Figure 2D). Feeding behavior was then assumed to be directly proportional to food uptake, defined as potential ration (Figure 2E). The potential ration expresses the amount of food the organism can consume when prey abundance is not limiting. Potential ration over time (Figure 2F) depicts how the food intake of individual fish changes in response to the behavioral effects of the pesticide exposure over the modeled growth period. Potential ration is equal to final ration if no effects on prey abundance are incorporated (Figure 4). If effects of pesticide exposure on prey abundance are incorporated, final ration is the product of potential ration (relating to the fish’s ability to capture prey, Figure 2) and the relative abundance of prey available following exposure (Figure 3). Next, additional empirical data (e.g. Weatherley and Gill 1995) defined the relationship between final ration and somatic growth rate (Figure 4C).

While the empirical relationship is more complex (e.g. somatic growth rate plateaus at rations above maximum feeding), a linear model was considered sufficient for the overall purpose of this model. Finally, the model combines these linear models relating AChE activity to feeding behavior, feeding behavior to potential ration, and final ration to somatic growth rate to produce a linear relationship between AChE activity and somatic growth rate (Figure 4D). One important assumption of the model is that the relationships are stable, i.e. do not change with time. The relationships would need to be modified to incorporate time as a variable if, for example, fish are shown to compensate over time for reduced AChE activity to improve their feeding behavior and increase food uptake.

Selection of aquatic invertebrate toxicity values to represent salmonid prey items:
The model requires an EC50 for each pesticide (defined as a 50% reduction in the biomass of salmonid prey items) and a corresponding slope. The term "EC50" will be used in this section to describe short-term survival data for aquatic invertebrates (death and immobility). To determine what levels of the OPs reduce aquatic invertebrate numbers, we reviewed the available field and laboratory studies. We found a wide spectrum of available data for the 10 a.i.s (2 a.i.s were not modeled: fenamiphos and bensulide). We did not locate a field study that measured aquatic community response to a range of concentrations of these pesticides. Therefore, we did not select concentration data from field experiments as we did in NMFS' 2008 Opinion on the registration of chlorpyrifos, diazinon, and malathion {Kuivila, 1995 #2083}. Due to the scarcity of data for many of the a.i.s, we did not develop probability plots. Instead, we selected the lowest available survival EC50 for D. magna for each a.i. to represent the salmonid prey community EC50 because D. magna data were available for all a.i.s (Table 5).

The models allow exposures that can include multiple AChE-inhibiting pesticides over various time pulses. Sigmoidal dose-response relationships, at steady-state, between each single pesticide exposure and 1) AChE activity and 2) relative prey abundance are modeled using specific EC50s and EC50s and slopes (Figure 2B and 3B). The timecourse for each exposure was built into the model as a pulse with a defined start and end during which the exposure remained constant (Figure 2A and 3A). The timecourse for AChE activity, on the other hand, was modeled using two single-order exponential functions, one for the time required for the exposure to reach full effect and the other for time required for complete recovery following the end of the exposure (time-to-effect$_{\text{AChE activity}}$ and time-to-recovery$_{\text{AChE activity}}$, respectively; Figure 2C). The apparent activity level was back-calculated to result in a relative concentration (concentration/ AChE inhibition EC50) for each day of the growth period for each pulse. The relative concentration for each day was summed across all the pulses to result in a total apparent concentration for each day. The sigmoid slope used in the calculation of AChE activity using the apparent concentration was the arithmetic mean of the sigmoid slopes for each pesticide present on each day. The timecourse for relative prey abundance was modeled incorporating a one day spike in prey drift relative to the toxicity and available prey base followed by a drop in abundance due to the toxic impacts (Figure 3C). Recovery is assumed to be due to a constant influx of invertebrates from connected habitats (aquatic and terrestrial) that are not exposed to the pesticide. Incoming organisms are subject to toxicity if pesticides are still present and this alters the rate of recovery during exposures. Incorporating dynamic effects and recovery variables allows the model to simulate differences in the pharmacokinetics (e.g. the rates of uptake from the environment and of detoxification) of various pesticides and simulate differences in invertebrate community response and recovery rates (see below).

The relationship between final ration and somatic growth rate (Figure 4C) produces a relationship representing somatic growth rate over time (Figure 4D), which is then used to model individual growth rate and size over time. The growth models were run for 1000 individual fish, with initial weight selected from a normal distribution with a mean of 1.0 g and standard deviation of 0.1 g. The size of 1.0 g was chosen to represent subyearling size in the spring prior to the onset of pesticide application. For each iteration of the model (one day for the organismal model), the somatic growth rate is calculated for each fish by selecting the parameter values from normal distributions with specified means and standard deviations (Table 1). The weight for each fish is then adjusted based on the calculated growth rate to generate a new weight for the next iteration. The length (days) to run the growth portion of the model was selected to represent the time from when the fish enter the linear portion of their growth trajectory in the mid to late spring until they change their growth pattern in the fall due to reductions in temperature and resources or until they migrate out of the system. The outputs of the organismal model that are handed to the population models consist of mean weights (with standard deviations) after the species-appropriate growth period (Table 2). A sensitivity analysis was run to determine the influence of the parameter values on the output of the growth model.

The option of exposing only a specified percent of the population to the pesticide(s) during the somatic growth period is provided. The exposed percent of the population is applied to the number of individuals run in the individual growth model. After running all 1000 individual growth trajectories (with X% exposed and 100-X% control) the mean weight and standard deviation of the whole is determined and handed to the population model to run as the size distribution of the impacted population.

The parameter values defining control conditions that are constant for all the modeled species are listed in Table 1. Model parameters such as the length of the growth period and control daily growth rate that are species specific are listed in Table 2. Each exposure scenario was defined by a concentration and exposure time for each pesticide. The duration of time until full effect for the pesticides was assumed to be within a few days (Ferrari et al. 2004), with a half-life of 0.5 days.

For prey, it is assumed there is a constant, independent influx of prey from upstream habitats that will eventually (depending on the rate selected) return prey abundance to 1. As mentioned above, however, these invertebrates are subject to exposure once added to the system, and therefore prey recovery rate is a product of the influx rate as well as the exposure scenario. While recovery rates reported in the literature vary, it is assumed a 1% recovery rate is ecologically realistic (Ward et al. 1995, Van den Brink et al. 1996, Colville et al. 2008). It was also assumed that regardless of the exposure scenario, relative prey abundance would not drop below a specific floor (Figure 3B). This assumption depends on a minimal yet constant terrestrial subsidy of prey and/or an aquatic community with tolerant individuals that would be available as prey, regardless of pesticide exposure and in addition to the constant recovery rate. No studies specify floors per se, but studies quantifying invertebrate densities following highly toxic exposures indicate a floor of 0.2 is ecologically realistic (i.e. regardless of the exposure, 20% of a fish's ration will be available daily; e.g., Cuffney et al. 1984). Finally, because prey availability has been found to increase dramatically albeit briefly following pesticide exposures (due to immediate mortality and/or emigration of benthic prey into the water column; Davies and Cook 1993, Schulz 2004), a one-day prey spike is included for the day following an exposure. The relative magnitude of the spike is calculated as the product of the standing prey availability the day prior to exposure (minus the floor), the toxicity of the exposure, and a constant of 20. This calculation therefore accounts for the potential prey that are available and the severity of the exposure. The spike will be greater when more prey are available and/or the toxicity of the exposure is greater; alternatively, the spike will be small when few prey are available and/or the exposure toxicity is low.

Below are the mathematical equations used to derive Figures 2, 3, and 4.

Figures 2A and 3A use a step function:

time < start; exposure = 0

start ≤ time ≤ end; exposure = exposure concentration(s)

time > end; exposure = 0.

Figures 2B and 3B use a sigmoid function:

y = bottom + (top – bottom)/(1 + (exposure concentration/EC50)^slope).

For 2B, y = AChE activity, top = Ac, bottom = 0.

For Figure 3B, y = prey abundance, top = Pc (in this case 1), bottom = Pf.

Figures 2D, 2E, and 4C use a linear function (the point-slope form of a line):

y = m*(x – x1) + y1.

For 2D, m = Mfa, x1 = Ac, and y1 = Fc.

For 2E, m = Mrf (computed as Rc/Fc), x1 = Fc, and y1 = Rc.

For 4C, m = Mgr, x1 = Rc, and y1 = Gc.

Figure 2C uses a series of exponential functions:

time < start; y = c

start ≤ time ≤ end; y = c – (c – i)*(1 – exp(-ke*(time – start)))

time > end; ye = c – (c – i)*(1 – exp(-ke*(end – start)))

y = ye + (c – ye)*(1 – exp(-kr*(time – end))).

For Figure 2C, c = Ac, i = Ai, ke = ln(2)/AChE effect half-life, kr = ln(2)/AChE recovery half-life. For Figure 2C the value of ye is calculated to determine the amount of inhibition that is reached during the exposure time, which may not be long enough to reach the maximum level of inhibition.

For Figure 3C, an exposure pulse would result in a 1-day spike followed by a decline to the impacted level based upon the prey toxicity. During exposures resulting in low prey toxicity, toxicity-limited recovery can occur. After exposure ends a constant rate of recovery proceeds until control drift is reached or another exposure occurs

```
    preyavail=preydrift(day-1)-floor;
    preytox=1/(1+(concentration)^preyslope);
    preyrecrate=0.01;
    preydriftrec = preyrecrate*preytox.
    time=start; spike=(-1+10^(1.654*preyavail))*(1-preytox)
        preydrift =preydrift+spike
    start ≤ time ≤ end;  preydrift=(preyavail*preytox)+preyrdriftrec+floor;
    time>end; preydrift = preydrift(day-1)+preydriftrec
```

Figure 2F is generated by using the output of Figure 2C for a given time as the input for 2D and using the resulting output of 2D as the input for 2E. The resulting output of 2E produces a single time point in the relationship in 2F. Performing this series of computations across multiple days produces the entire relationship in 2F. 4D is generated by taking the outputs of 4A and 4B for the same day. Note the relationship of 4A is equivalent to 2F. The resulting outputs of 4A and 4B are multiplied to produce a final ration for a given day. The prey abundance (4B) available for consumption during a prey spike is capped at a maximum of 1.5*control drift to provide a limited benefit to the individual fish. The final ration is used as input for 4C to generate 4D.

Population Model

The weight distributions from the organismal growth portion of the model are used to calculate size-dependent first-year survival for a life-history matrix population model for each species and life-history type. This incorporates the impact that reductions in size could have on population growth rate and abundance. The first-year survival element of the transition matrix incorporates a size-dependent survival rate for a three- or four-month interval (depending upon the species) which takes the juveniles up to 12 months of age. This time represents the 4-month early winter survival in freshwater for stream-type Chinook, coho, and sockeye models. For ocean-type Chinook, it is the 3-month period the subyearling smolt spend in the estuary and nearshore habitats (i.e. estuary survival). The weight distributions from the organismal model are converted to length distributions by applying condition factors from data for each modeled species (cf; 0.0095 for sockeye and 0.0115 for all others) as shown in Equation L.

Equation L: length(mm) = ((fish weight(g)/cf)^(1/3))*10

The relationship between length and early winter or estuary survival rate was adapted from Zabel and Achord (2004) to match the survival rate for each control model population (Howell et al. 1985, Kostow 1995, Myers et al. 2006). The relationship is based on the length of a subyearling salmon relative to the mean length of other competing subyearling salmon of the same species in the system, Equation D, and relates that relative difference to size-dependent survival based upon Equation S. The values for α and resulting size-dependent survival (survival ϕ) for control runs for each species are listed in Table 2. The constant α is a species-specific parameter defined such that it produces the correct control survival ϕ value when Δlength equals zero.

Equation D: Δlength = fish length(mm) – mean length(mm)

Equation S: Survival $\phi = (e^{(\alpha+(0.0329*\Delta length))}) / (1 + e^{(\alpha+(0.0329*\Delta length))})$

Randomly selecting length values from the normal distribution calculated from the organismal model output size and applying equations 1 and 2 generates a size-dependent survival probability for each fish. This process was replicated 1000 times for each exposure scenario and simultaneously 1000 times for the paired control scenario and results in a mean size-dependent survival rate for each population. The resulting size-dependent survival rates are inserted in the calculation of first-year survival in the respective control and pesticide-exposed transition matrices.

The investigation of population-level responses to pesticide exposures uses life-history projection matrix models. Individuals within a population exhibit various growth, reproduction, and survivorship rates depending on their developmental or life-history stage or age. These age specific characteristics are depicted in the life-history graph (Figure 1A-D) in which transitions are depicted as arrows. The nonzero matrix elements represent transitions corresponding to reproductive contribution or survival, located in the top row and the subdiagonal of the matrix, respectively (Figure 1E). The survival transitions in the life-history graph are incorporated into the n x n square matrix (A) by assigning each age a number (1 through n) and each transition from age i to age j becomes the element a_{ij} of matrix A (i = row, j = column) and represent the proportion of the individuals in each age passing to the next age as a result of survival. The reproductive element (a_{1j}) gives the number of offspring that hatch per individual in the contributing age, j. The reproductive element value incorporates the proportion of females in each age, the proportion of females in the age that are sexually mature, fecundity, fertilization success, and hatch success.

In order to understand the relative impacts of a short-term pesticide exposure on exposed vs. unexposed fish, we used parameters for an idealized control population that exhibits an increasing population growth rate. All characteristics exhibit density independent dynamics. The models assume closed systems, allowing no migration impact on population size. No stochastic impacts are included beyond natural variability as represented by selecting parameter values from a normal distribution about a mean each model iteration (year). Ocean conditions, freshwater habitat, fishing pressure, and marine resource availability were assumed constant and density independent.

In the model an individual fish experiences an exposure scenario once as a subyearling (during its first spring) and never again. The pesticide exposure is assumed to occur annually. All subyearlings within a given population are assumed to be exposed to the pesticide. No other age classes experience the exposure. The model integrates this as every brood class being exposed as subyearlings and thus the vital demographic rates of the transition matrix are continually impacted in the same manner. Regardless of the level of AChE inhibition due to the direct exposure, only the sublethal effects are incorporated in the models.

The model recalculates first-year survival for each run using a size-dependent survival value selected from a normal distribution with the mean and standard deviation produced by Equation S. Population model output consists of the percent change in lambda from the unexposed control populations derived from the mean of two thousand calculations of both the unexposed control population and the pesticide exposed population. Change in lambda, representing alterations to the population productivity, was selected as the primary model output for reasons outlined previously.

A prospective analysis of the transition matrix, A, (Caswell 2001) explored the intrinsic population growth rate as a function of the vital rates. The intrinsic population growth rate, λ, equals the dominant eigenvalue of A and was calculated using matrix analysis software (MATLAB version 7.7.0 by The Math Works Inc., Natick, MA). Therefore λ is calculated directly from the matrix and running projections of abundances over time is redundant and unnecessary. The stable age distribution, the proportional distribution of individuals among the ages when the population is at equilibrium, is calculated as the right normalized eigenvector corresponding to the dominant eigenvalue λ. Variability was integrated by repeating the calculation of λ 2000 times selecting the values in the transition matrix from their normal distribution defined by the mean standard deviation. The influence of each matrix element, a_{ij}, on λ was assessed by calculating the sensitivity values for A. The sensitivity of matrix element a_{ij} equals the rate of change in λ with respect to a_{ij}, defined by $\delta\lambda/ \delta a_{ij}$. Higher sensitivity values indicate greater influence on λ. The elasticity of matrix element a_{ij} is defined as the proportional change in λ relative to the proportional change in a_{ij}, and equals (a_{ij}/λ) times the sensitivity of a_{ij}. One characteristic of elasticity analysis is that the elasticity values for a transition matrix sum to unity (one). The unity characteristic also allows comparison of the influence of transition elements and comparison across matrices.

Due to differences in the life-history strategies, specifically lifespan, age at reproduction and first year residence and migration habits, four life-history models were constructed. This was done to encompass the different responses to freshwater pesticide exposures and assess potentially different population-level responses. Separate models were constructed for coho, sockeye, ocean-type and stream-type Chinook. In all cases transition values were determined from literature data on survival and reproductive characteristics of each species.

A life-history model was constructed for coho salmon (*O. kisutch*) with a maximum age of 3. Spawning occurs in late fall and early winter with emergence from March to May. Fry spend 14-18 months in freshwater, smolt and spend 16-20 months in the saltwater before returning to spawn (Pess et al. 2002). Survival numbers were summarized in Knudsen et al. (2002) as follows. The average fecundity of each female is 4500 with a standard deviation of 500. The observed number of males:females was 1:1. Survival from spawning to emergence is 0.3 (0.07). Survival from emergence to smolt is 0.0296 (0.00029) and marine survival is 0.05 (0.01). All parameters followed a normal distribution (Knudson et al. 2002). The calculated values used in the matrix are listed in Table 3. The growth period for first year coho was set at 180 days to represent the time from mid-spring to mid-fall when the temperatures and resources drop and somatic growth slows (Knudson et al. 2002).

Life-history models for sockeye salmon (*O. nerka*) were based upon the lake wintering populations of Lake Washington, Washington, USA. These female sockeye salmon spend one winter in freshwater, then migrate to the ocean to spend three to four winters before returning to spawn at ages 4 or 5. Jacks return at age 2 after only one winter in the ocean. The age proportion of returning adults is 0.03, 0.82, and 0.15 for ages 3, 4 and 5, respectively (Gustafson et al.1997). All age 3 returning adults are males. Hatch rate and first year survival were calculated from brood year data on escapement, resulting presmolts and returning adults (Pauley et al. 1989) and fecundity (McGurk 2000). Fecundity values for age 4 females were 3374 (473) and for age 5 females were 4058 (557) (McGurk 2000). First year survival rates were 0.737/month (Gustafson et al. 1997). Ocean survival rates were calculated based upon brood data and the findings that 90% of ocean mortality occurs during the first 4 months of ocean residence (Pauley et al. 1989). Matrix values used in the sockeye baseline model are listed in Table 3. The 168 day growth period represents the time from lake entry to early fall when the temperature drops and somatic growth slows (Gustafson et al. 1997).

A life-history model was constructed for ocean-type Chinook salmon (*O. tshawytscha*) with a maximum female age of 5 and reproductive maturity at ages 3, 4 or 5. Ocean-type Chinook migrate from their natal stream within a couple months of hatching and spend several months rearing in estuary and nearshore habitats before continuing on to the open ocean. Transition values were determined from literature data on survival and reproductive characteristics from several ocean-type Chinook populations in the Columbia River system (Healey and Heard 1984, Howell et al. 1985, Roni and Quinn 1995, Ratner et al. 1997, PSCCTC 2002, Green and Beechie 2004). The sex ratio of spawners was approximately 1:1. Estimated size-based fecundity of 4511(65), 5184(89), and 5812(102) was calculated based on data from Howell et al., 1985, using length-fecundity relationships from Healy and Heard (1984). Control matrix values for the Chinook model are listed in Table 3. The growth period of 140 days encompasses the time the fish rear in freshwater prior to entering the estuary and open ocean. The first three months of estuary/ocean survival are the size-dependent stage. Size data for determining subyearling Chinook condition indices came from data collected in the lower Columbia River and estuary (Johnson et al. 2007).

An age-structured life-history matrix model for stream-type Chinook salmon with a maximum age of 5 was defined based upon literature data on Yakima River spring Chinook from Knudsen et al. (2006) and Fast et al. (1988), with sex ratios of 0.035, 0.62 and 0.62 for females spawning at ages 3, 4, and 5, respectively. Length data from Fast et al. (1988) was used to calculate fecundity from the length-fecundity relationships in Healy and Heard (1984). The 184-day growth period produces control fish with a mean size of 96mm, within the observed range documented in the fall prior to the first winter (Beckman et al. 2000). The size-dependent survival encompasses the 4 early winter months, up until the fish are 12 months old.

Acute Toxicity Model

In order to estimate the population-level responses of exposure to lethal pesticide concentrations, acute mortality models were constructed based upon the control life-history matrices described above. The acute responses are modeled as direct reduction in the first year survival rate (S1). Two options are available to run, direct mortality estimates and exposure scenarios. Direct mortality can be input as percent mortality and is multiplied by the first-year survival rate in the transition matrix. For the exposure scenarios all subyearling salmon are assumed to be exposed in each scenario. Exposures are assumed to result in a cumulative reduction in survival as defined by the concentration and the dose-response curve as defined by the LC50 and slope for each pesticide. A sigmoid dose-response relationship is used to accurately handle responses well away from LC50 and to be consistent with other does-response relationships. The model inputs for each scenario are the exposure concentration and acute fish LC50, as well as the sigmoid slope for the LC50. For a given concentration a pesticide survival rate (1-mortality) is calculated and is multiplied by the control first-year survival rate, producing an exposed scenario first-year survival for the life-history matrix. Variability is incorporated as described above using mean and standard deviation of normally distributed survival and reproductive rates and model output consists of the percent change in lambda from unexposed control populations derived from the mean of 10000 calculations of both the unexposed control population and the pesticide exposed population. The percent change in lambda is considered different from control when the difference is greater than the percent of one standard deviation from the control lambda.

Results

Sensitivity Analysis

A sensitivity analysis conducted on the organismal model revealed that changes in the control somatic growth rate had the greatest influence on the final weights (Table 1). While this parameter value was experimentally derived for another species (sockeye salmon; Brett et al. 1969), this value was adapted for each model species and is within the variability reported in the literature for other salmonids (reviewed in Weatherley and Gill 1995). Other parameters related to the daily growth rate calculation, including the growth to ration slope (Mgr) and the control ration produced strong sensitivity values. Initial weight, the prey recovery rate and the prey floor also strongly influenced the final weight values (Table 1). Large changes (0.5 to 2X) in the other key parameters produced proportionate changes in final weight.

The sensitivity analysis of all four of the control population matrices predicted the greatest changes in population growth rate (λ) result from changes in first-year survival. Parameter values and their corresponding sensitivity values are listed in Table 4. The elasticity values for the transition matrices also corresponded to the driving influence of first-year survival, with contributions to lambda of 0.33 for coho, 0.29 for ocean-type Chinook, 0.25 for stream-type Chinook, and 0.24 for sockeye.

Model Output

Organismal and population model outputs for all scenarios are summarized as graphs in the main text of the Opinion. As expected, greater changes in population growth resulted from longer exposures to the pesticides. The factors driving the level of change in lambda were the Prey Drift and relative AChE Activity parameters determined by the toxicity values for each pesticide (Table 3).

Output from the acute toxicity models was presented in the Risk Characterization section of the main text. Increases in direct mortality during the first year of life produced large impacts on the population growth rates for all the life-history strategies.

While strong trends in effects were seen for each pesticide across all four life-history strategies modeled, some slight differences were apparent. The similarity in patterns likely stems from using the same toxicity values for all four models, while the differences are consequences of distinctions between the life-history matrices. The stream-type Chinook and sockeye models produced very similar results as measured as the percent change in population growth rate. The ocean-type Chinook and coho models output produced the greatest changes in lambda resulting from the pesticide exposures. When looking for similarities in parameters to explain the ranking, no single life history parameter or characteristic, such as lifespan, reproductive ages, age distribution, lambda and standard deviation, or first-year survival show a pattern that matches this consistent output. Combining these factors into the transition matrix for each life-history and conducting the sensitivity and elasticity analyses revealed that changes in first-year survival produced the greatest changes in lambda. In addition, the elasticity analysis can be used to predict relative contribution to lambda from changes in first-year survival on a per unit basis. As detailed by the elasticity values reported above, the same change in first-year survival will produce a slightly greater change in the population growth rate for coho and ocean-type Chinook than for stream-type Chinook and sockeye. While some life-history characteristics may lead a population to be more vulnerable to an impact, the culmination of age structure, survival and reproductive rates as a whole strongly influences the population-level response.

Figure 1: Life-History Graphs and Transition Matrix for coho (A), sockeye (B) and Chinook (C) salmon. The life-history graph for a population labeled by age, with each transition element labeled according to the matrix position, a_{ij}, i row and j column. Dashed lines represent reproductive contribution and solid lines represent survival transitions. D) The transition matrix for the life-history graph depicted in C.

Figure 2: Relationships used to link anticholinesterase exposure to the organism's ability to acquire food (potential ration). See text for details. Relationships in B, C, and D utilize empirical data. Closed circles represent control conditions. Open circles represent the exposed (inhibited) condition. A) Representation of a constant level of anticholinesterase pesticide exposure (either a single compound or mixtures). B) Sigmoidal relationship between exposure concentration and steady-state acetylcholinesterase (AChE) activity showing a dose-dependent reduction defined by control activity (horizontal line, Ac), sigmoidal (i.e. hille) slope (AChE slope), and the concentration producing 50% inhibition (vertical line, EC50). C) Timecourse of acetylcholinesterase inhibition based on modeling the time-to-effect and time-to-recovery as single exponential curves with different time-constants. At the start of the exposure AChE activity will be at control and then decline toward the inhibited activity (Ai) based on Panel B. D) Linear model relating acetylcholinesterase activity to feeding behavior using a line that passes through the feeding (Fc) and activity (Ac) control conditions with a slope of Mfa. E) The relationship between feeding behavior and the potential ratio an organism could acquire (if not food limited) used a line passing through the control conditions (Fc as in Panel D and the control ration, Rc) and through the origin producing a slope (Mrf) equal to Rc/Fc. F) Timecourse for effect of exposure to anticholinesterase on potential ration produced by combining C & E.

Figure 3: Relationships used to link anticholinesterase exposure to the availability of prey. See text for details. Relationships in B and C utilize empirical data. Closed circles represent control conditions. Open circles represent the exposed (inhibited) condition. A) Representation of a constant level of anticholinesterase pesticide exposure (either single compound or mixtures). B) Sigmoidal relationship between exposure concentration and relative prey abundance showing a dose-dependent reduction defined by control abundance (horizontal line at 1, Pc), sigmoid (i.e. hille) slope (prey slope), the concentration producing a 50% reduction in prey (vertical line,

EC_{50}), and a minimum abundance always present (horizontal line denoted as floor, Pf). C) Timecourse of prey abundance including a 1-day spike in prey drift relative to the available prey and the level of toxicity followed by a drop to the level of impact or the floor whichever is greater. During extended exposures at low toxicity recovery can begin at the constant prey influx rate multiplied by the current level of toxicity. After exposure recovery to control prey drift is at the constant rate of influx from upstream habitats.

Figure 4: Relationships used to link anticholinesterase exposure to growth rate relating to long-term weight gain of each fish. See text for details. Relationships in A, B, and C utilize empirical data. Closed circles represent control conditions. Open circles (e.g. Ai) represent the exposed (inhibited) condition. A&B) Relationships describing the Timecourse of the effects of anticholinesterase exposure on the organisms ability to capture food (Panel A, potential ration) and the availability of food to capture (Panel B, relative prey abundance). The figures are the same as those in Figures 2F and 3C, respectively. For a given exposure concentration and time, multiplying potential ration by relative prey abundance yields the final ration acquired by the organism. C) A linear model was used to relate final ration to growth rate using a line passing through the control conditions and through the maintenance condition with a slope denoted by Mgr. D) Timecourse for effect of exposure to anticholinesterase on growth rate produced by combining A, B, & C.

Table 1. List of values used for control parameters to model organismal growth and the model sensitivity to changes in the parameter.

Parameter	Value[1]	Error[2]	Sensitivity[3]
acetylcholinesterase activity (Ac)	1.0[4,5]	0.06[5]	-0.167
feeding (Fc)	1.0[4,5]	0.05[5]	0.088
ration (Rc)	5% weight/day[6]	0.05[7]	-0.547
feeding vs. activity slope (Mfa)	1.0[5]	0.1[5]	-0.047
ration vs. feeding slope (Mrf)	5 (Rc/Fc)	-	-
growth vs. ration slope (Mgr)	0.35[6]	0.02[6]	-0.547
growth vs. activity slope (Mga)	1.75 (Mfa*Mrf*Mgr)	-	-
initial weight	1 gram[8]	0.1[8]	1.00
control prey drift	1.0[4]	0.05[11]	0.116
AChE impact time-to-effect ($t_{1/2}$)	0.5 day[9]	n/a	0.005
AChE time-to-recovery ($t_{1/2}$)	30 days[10]	n/a	-0.0001
prey floor	0.20[11]	n/a	0.178
prey recovery rate	0.01[12]	n/a	0.323
somatic growth rate (Gc)	1.3[13]	0.06[6]	2.531

[1] mean value of a normal distribution used in the model or constant value when no corresponding error is listed

[2] standard deviation of the normal distribution used in the model

[3] mean sensitivity when baseline parameter is changed over range of 0.5 to 2-fold

[4] other values relative to control

[5] derived from Sandahl et al. 2005

[6] derived from Brett et al. 1969

[7] data from Brett et al. 1969 has no variability (ration was the independent variable) so a variability of 1% was selected to introduce some variability

[8] consistent with field-collected data for juvenile Chinook (Nelson et al. 2004)

[9] estimated from Ferrari et al. 2004

[10] consistent with Eder et al., 2007; Ferrari et al., 2004; Chambers et al., 2002

[11] estimated from Van den Brink et al. 1996

[12] derived from Ward et al. 1995, Van den Brink et al. 1996, Colville et al. 2008

[13] derived from Brett et al. 1969 and adapted for ocean-type Chinook, used for sensitivity analysis

Table 2. Species specific control parameters to model organismal growth and survival rates. Growth period and survival rate are determined from the literature data listed for each species. Gc and α were calculated to make the basic model produce the appropriate size and survival values from the literature.

	Chinook Stream-type[1]	Chinook Ocean-type[2]	Coho[3]	Sockeye[4]
days to run organismal growth model	184	140	184	168
growth rate % body wt/day (Gc)	1.28	1.30	0.90	1.183
α from equation S	-0.33	-1.99	-0.802	-0.871
Control Survival ϕ	0.418	0.169	0.310	0.295

[1] Values from data in Healy and Heard 1984, Fast et al. 1988, Beckman et al. 2000, Knudsen et al. 2006

[2] Values from data in Healey and Heard 1984, Howell et al. 1985, Roni and Quinn 1995, Ratner et al. 1997, PSCCTC 2002, Green and Beechie 2004, Johnson et al. 2007

[3] Values from data in Pess et al. 2002, Knudsen et al. 2002

[4] Values from data in Pauley et al. 1989, Gustafson et al. 1997, McGurk 2000

Table 3. Effects values (ug/L) and slopes for AChE activity, acute fish lethality, and prey abundance dose-response curves.

compound	AChE Activity EC_{50}[1] ug/L	AChE Activity slope	Fish lethality LC_{50}[2] ug/L	Fish lethality slope[3]	Prey Abundance EC_{50}[4] ug/L	Prey Abundance Slope[5]
Azinphos-methyl	0.1639	0.99	1.2	3.63	1.13	3.63
Bensulide	180	0.99	720	3.63	580	3.63

Dimethoate	273.4	0.99	6200	3.63	3320	3.63
Disulfoton	485.5	0.99	1850	3.63	13	3.63
Ethoprop	90.62	0.99	1020	3.63	44	3.63
Fenamiphos			68	3.63	1.3	3.63
Methamidophos	10000	0.99	25000	3.63	26	3.63
Methidathion	1.123	0.99	6.6	3.63	3	3.63
Methyl parathion	28.75	0.99	1850	3.63	0.14	3.63
Naled	7.848	0.99	87	3.63	0.3	3.63
Phorate	0.5697	0.99	13	3.63	37	3.63
Phosmet	3.25	0.99	150	3.63	5.6	3.63

[1] Values from Laetz et al.

[2] Values from EPA BEs

[3] sigmoidal slope that produces responses with a probit slope of Peterson et al. 2001a, see text.

[4] Values from analysis of global search of reported LC50 and EC50s reported in EPA's Ecotox database. See text.

[5] Values from Peterson et al. 2001a

Table 4. Matrix transition element and sensitivity (S) and elasticity (E) values for each model species. These control values are listed by the transition element taken from the life-history graphs as depicted in Figure 1 and the literature data described in the method text. Blank cells indicate elements that are not in the transition matrix for a particular species. The influence of each matrix element on λ was assessed by calculating the sensitivity (S) and elasticity (E) values for A. The sensitivity of matrix element a_{ij} equals the rate of change in λ with respect to the transition element, defined by $\delta\lambda / \delta a$. The elasticity of transition element a_{ij} is defined as the proportional change in λ relative to the proportional change in a_{ij}, and equals (a_{ij}/λ) times the sensitivity of a_{ij}. Elasticity values allow comparison of the influence of individual transition elements and comparison across matrices.

Transition Element	Chinook Stream-type			Chinook Ocean-type			Coho			Sockeye		
	Value[1]	S	E	Value[2]	S	E	Value[3]	S	E	Value[4]	S	E
S1	0.0643	3.844	0.247	0.0056	57.13	0.292	0.0296	11.59	0.333	0.0257	9.441	0.239
S2	0.1160	2.132	0.247	0.48	0.670	0.292	0.0505	6.809	0.333	0.183	1.326	0.239
S3	0.17005	1.448	0.246	0.246	0.476	0.106				0.499	0.486	0.239
S4	0.04	0.319	0.0127	0.136	0.136	0.0168				0.1377	0.322	0.0437
R3	0.5807	0.00184	0.0011	313.8	0.0006	0.186	732.8	0.000469	0.333			
R4	746.73	0.000313	0.233	677.1	0.000146	0.0896				379.57	0.000537	0.195
R5	1020.36	1.25E-05	0.0127	1028	1.80E-05	0.0168				608.7	7.28E-05	0.0437

[1] Value calculated from data in Healy and Heard 1984, Fast et al. 1988, Beckman et al. 2000, Knudsen et al. 2006

[2] Value calculated from data in Healey and Heard 1984, Howell et al. 1985, Roni and Quinn 1995, Ratner et al. 1997, PSCCTC 2002, Green and Beechie 2004, Johnson et al. 2007

[3] Value calculated from data in Pess et al. 2002, Knudsen et al. 2002

[4] Value calculated from data in Pauley et al. 1989, Gustafson et al. 1997, McGurk 2000

Table 5 48 h survival EC50s of *Daphnia magna*

Organophosphate	*Daphnia magna* 48 hr EC50 (µg/L) (95% CI)	Data Source
Azinphos methyl	1.13	MRID 00068678
Dimethoate	3320 (1730-4120)	Song, M.Y., J.D. Stark, J.J. Brown. 1997. Comparative Toxicity of Four Insecticides, Including Imidacloprid and Tebufenozide, to Four Aquatic. Environ.Toxicol.Chem. 16(12):2494-2500
Disulfoton	13	MRID 00143401
Ethoprop	44	MRID 00068325
Methamidophos	26 (20-34)	MRID 00041311)
Methidathion	3	MRID 42081704 (1991)
Methyl parathion	0.14 (0.09-0.2)	MRID 40094602
Naled	0.3	MRID BA0NAL02
Phorate	37 (30-44)	MRID 0161825
Phosmet	5.6	MRID 00063194,

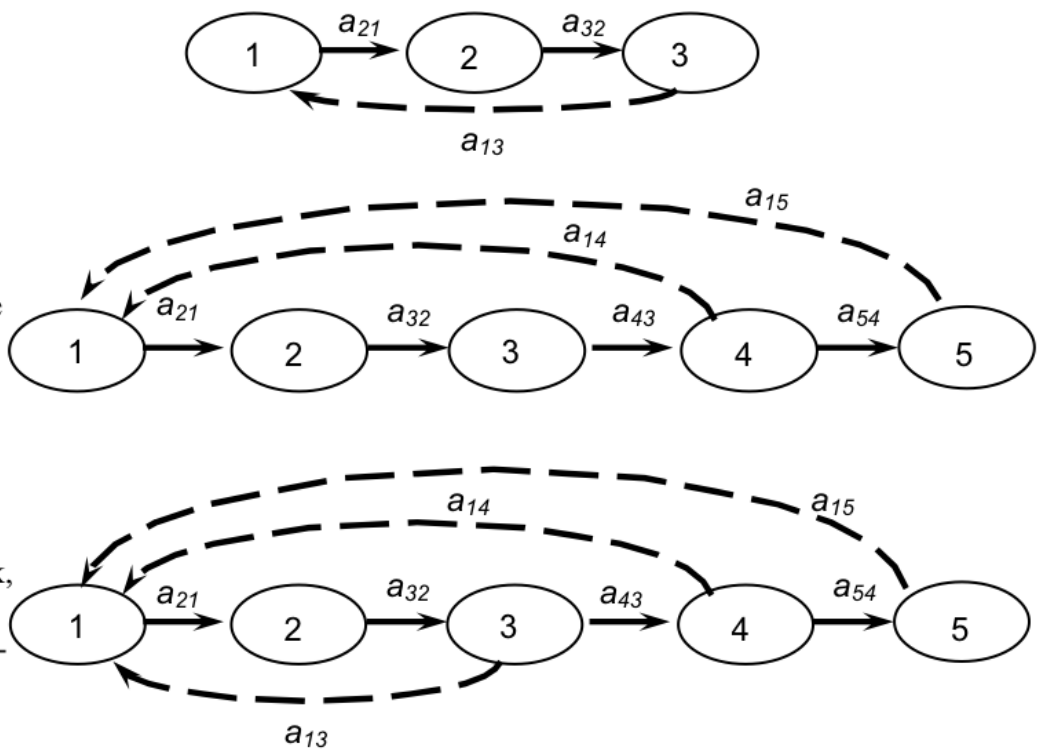
For this table, unless noted otherwise EC50 from EPA documents, and is referred to by MRID number.

Figure 1.

A. Coho

B. Sockeye

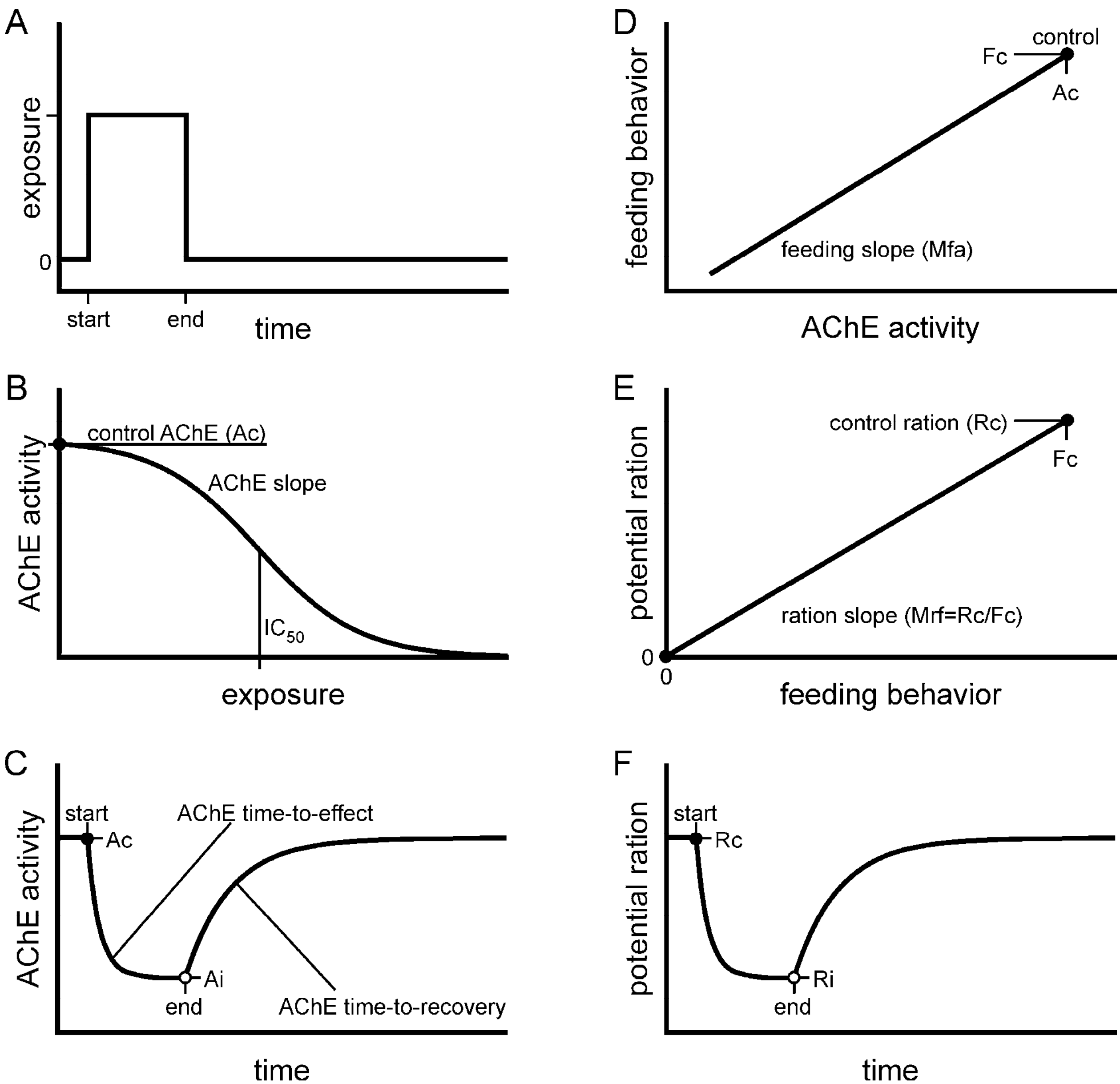
C. Chinook, ocean-type and stream-type



D. Transition Matrix

$$A = \begin{bmatrix} 0 & 0 & R3=a_{13} & R4=a_{14} & R5=a_{15} \\ S1=a_{21} & 0 & 0 & 0 & 0 \\ 0 & S2=a_{32} & 0 & 0 & 0 \\ 0 & 0 & S3=a_{43} & 0 & 0 \\ 0 & 0 & 0 & S4=a_{54} & 0 \end{bmatrix}$$

Figure 2.


A
exposure
0
start
end
time
B
AChE activity
control AChE (Ac)
AChE slope
IC50
exposure
C
AChE activity
start
Ac
AChE time-to-effect
Ai
end
AChE time-to-recovery
time
D
feeding behavior
control
Fc
Ac
feeding slope (Mfa)
AChE activity
E
potential ration
control ration (Rc)
Fc
ration slope (Mrf=Rc/Fc)
0
0
feeding behavior
F
potential ration
start
Rc
Ri
end
time

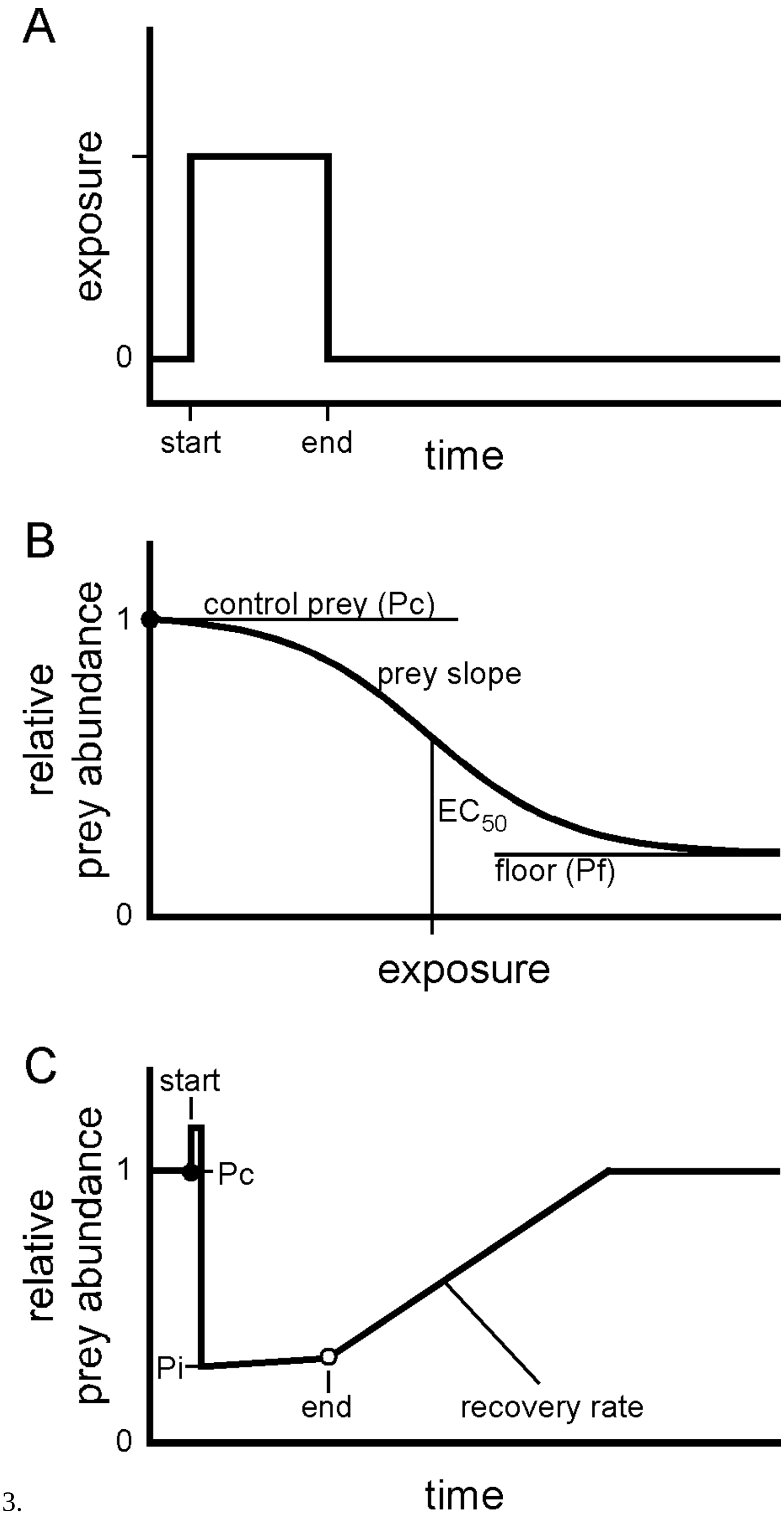
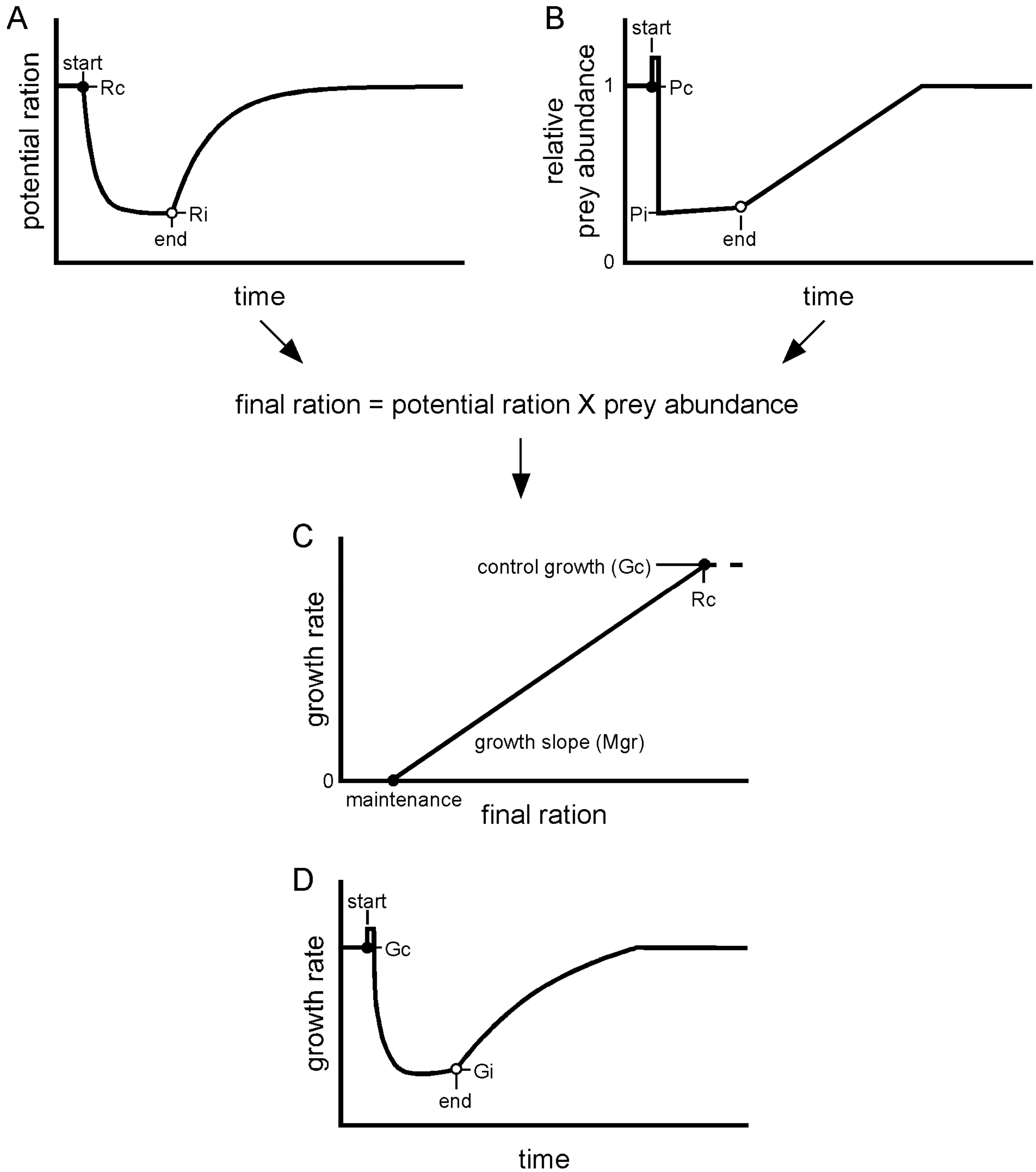

A
exposure
0
start
end
time
B
relative
prey abundance
1
0
control prey (Pc)
prey slope
EC50
floor (Pf)
exposure
C
relative
prey abundance
start
1
Pc
Pi
0
end
recovery rate
time

Figure 3.



A
potential ration
start
Rc
Ri
end
time
B
relative
prey abundance
start
1
Pc
Pi
end
0
time
final ration = potential ration X prey abundance
C
growth rate
control growth (Gc)
Rc
growth slope (Mgr)
0
maintenance
final ration
D
growth rate
start
Gc
Gi
end
time

Figure 4.

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Appendix 2. Species and Population Annual Rates of Growth

Chinook Salmon

ESU	Population	λ - H=0	95% CI -lower	95% CI - upper
California Coastal	Eel River	N/A	N/A	N/A
	Redwood Creek	N/A	N/A	N/A
	Mad River	N/A	N/A	N/A
	Humboldt Bay tributaries	N/A	N/A	N/A
	Bear River	N/A	N/A	N/A
	Mattole River	N/A	N/A	N/A
	Tenmile to Gualala	N/A	N/A	N/A
	Russain River	N/A	N/A	N/A
Central Valley Spring - Run (Good et al., 2005 - 90% CI)	Butte Creek - spring run	1.300	1.060	1.600
	Deer Creek - spring run	1.170	1.040	1.350
	Mill Creek - spring run	1.190	1.000	1.470
Lower Columbia River (Good et al., 2005) (# = McElhany et al., 2007)	Youngs Bay	N/A	N/A	N/A
	Grays River - fall run	0.944	0.739	1.204
	Big Creek	N/A	N/A	N/A
	Elochoman River - fall run	1.037	0.813	1.323
	Clatskanie River #	0.990	0.824	1.189
	Mill, Abernathy, Germany Creeks - fall run	0.981	0.769	1.252
	Scappose Creek	N/A	N/A	N/A
	Coweeman River - fall run	1.092	0.855	1.393
	Lower Cowlitz River - fall run	0.998	0.776	1.282
	Upper Cowlitz River - fall run	N/A	N/A	N/A
	Toutle River - fall run	N/A	N/A	N/A
	Kalamaha River - fall run	0.937	0.763	1.242
	Salmon Creek / Lewis River - fall run	0.984	0.771	1.256
	Clackamas River - fall run	N/A	N/A	N/A
	Washougal River - fall run	1.025	0.803	1.308
	Sandy River - fall run	N/A	N/A	N/A
	Lower Gorge tributaries	N/A	N/A	N/A
	Upper Gorge tributaries - fall run	0.959	0.751	1.224
	Hood River - fall run	N/A	N/A	N/A
	Big White Salmon River - fall run	0.963	0.755	1.229
	Sandy River - late fall run	0.943	0.715	1.243
	North Fork Lewis River - late fall run	0.968	0.756	1.204
	Upper Cowlitz River - spring run	N/A	N/A	N/A
	Cispus River	N/A	N/A	N/A
	Tilton River	N/A	N/A	N/A
	Toutle River - spring run	N/A	N/A	N/A
	Kalamaha River - spring run	N/A	N/A	N/A
	Lewis River - spring run	N/A	N/A	N/A
	Sandy River - spring run #	0.961	0.853	1.083
	Big White Salmon River - spring run	N/A	N/A	N/A
	Hood River - spring run	N/A	N/A	N/A

Chinook Salmon (continued)

ESU	Population	λ - H=0	95% CI -lower	95% CI - upper
Upper Columbia River Spring - Run (FCRPS)	Methow River	1.100	N/A	N/A
	Twisp River	N/A	N/A	N/A
	Chewuch River	N/A	N/A	N/A
	Lost / Early River	N/A	N/A	N/A
	Entiat River	0.990	N/A	N/A
	Wenatchee River	1.010	N/A	N/A
	Chiawawa River	N/A	N/A	N/A
	Nason River	N/A	N/A	N/A
	Upper Wenatchee River	N/A	N/A	N/A
	White River	N/A	N/A	N/A
	Little Wenatchee River	N/A	N/A	N/A
Puget Sound (only have λ where hatchery fish = native fish), (Good et al., 2005)	Nooksack - North Fork	0.750	0.680	0.820
	Nooksack - South Fork	0.940	0.880	0.990
	Lower Skagit	1.050	0.960	1.140
	Upper Skagit	1.050	0.990	1.110
	Upper Cascade	1.060	1.010	1.110
	Lower Sauk	1.010	0.890	1.130
	Upper Sauk	0.960	0.900	1.020
	Suiattle	0.990	0.930	1.050
	Stillaguamish - North Fork	0.920	0.880	0.960
	Stillaguamish - South Fork	0.990	0.970	1.010
	Skykomish	0.870	0.840	0.900
	Snoqualmie	1.000	0.960	1.040
	North Lake Washington	1.070	1.000	1.140
	Cedar	0.990	0.920	1.060
	Green	0.670	0.610	0.730
	White	1.160	1.100	1.220
	Puyallup	0.950	0.890	1.010
	Nisqually	1.040	0.970	1.110
	Skokomish	1.040	1.000	1.080
	Dosewallips	1.170	1.070	1.270
	Duckabush	N/A	N/A	N/A
	Hamma Hamma	N/A	N/A	N/A
	Mid Hood Canal	N/A	N/A	N/A
	Dungeness	1.090	0.980	1.200
	Elwha	0.950	0.840	1.060
Sacramento River Winter - Run (Good, 2005 - 90% CI))	Sacramento River - winter run	0.970	0.870	1.090

Chinook Salmon (continued)

ESU	Population	λ - H=0	95% CI -lower	95% CI - upper
Snake River Fall - Run (Good, 2005)	Lower Snake River	1.024	N/A	N/A
Snake River Spring/Summer - Run (FCRPS)	Tucannon River	1.000	N/A	N/A
	Wenaha River	1.100	N/A	N/A
	Wallowa River	N/A	N/A	N/A
	Lostine River	1.050	N/A	N/A
	Minam River	1.050	N/A	N/A
	Catherine Creek	0.970	N/A	N/A
	Upper Grande Ronde River	N/A	N/A	N/A
	South Fork Salmon River	1.110	N/A	N/A
	Secesh River	1.070	N/A	N/A
	Johnson Creek	N/A	N/A	N/A
	Big Creek Spring Run	1.090	N/A	N/A
	Big Creek Summer Run	1.090	N/A	N/A
	Loon Creek	N/A	N/A	N/A
	Marsh Creek	1.080	N/A	N/A
	Bear Valley / Elk Creek	1.100	N/A	N/A
	North Fork Salmon River	N/A	N/A	N/A
	Lemhi River	1.020	N/A	N/A
	Pahsimeroi River	1.080	N/A	N/A
	East Fork Salmon Spring Run	1.040	N/A	N/A
	East Fork Salmon Summer Run	1.040	N/A	N/A
	Yankee Fork Spring Run	N/A	N/A	N/A
	Yankee Fork Summer Run	N/A	N/A	N/A
	Valley Creek Spring Run	N/A	N/A	N/A
	Valley Creek Summer Run	N/A	N/A	N/A
	Upper Salmon Spring Run	1.060	N/A	N/A
	Upper Salmon Summer Run	1.060	N/A	N/A
	Alturas Lake Creek	N/A	N/A	N/A
	Imnaha River	1.050	N/A	N/A
	Big Sheep Creek	N/A	N/A	N/A
	Lick Creek	N/A	N/A	N/A
Upper Williamette River (McElhany et al., 2007)	Clackamas River	0.967	0.849	1.102
	Molalla River	N/A	N/A	N/A
	North Santiam River	N/A	N/A	N/A
	South Santiam River	N/A	N/A	N/A
	Calapooia River	N/A	N/A	N/A
	McKenzie River	0.927	0.761	1.129
	Middle Fork Williamette River	N/A	N/A	N/A
	Upper Fork Williamette River	N/A	N/A	N/A

Chum Salmon

ESU	Population	λ - H=0	95% CI -lower	95% CI - upper
Columbia River	Youngs Bay	N/A	N/A	N/A
	Grays River	0.954	0.855	1.064
	Big Creek	N/A	N/A	N/A
	Elochoman River	N/A	N/A	N/A
	Clatskanie River	N/A	N/A	N/A
	Mill, Abernathy and German Creeks	N/A	N/A	N/A
	Scappose Creek	N/A	N/A	N/A
	Cowlitz River	N/A	N/A	N/A
	Kalama River	N/A	N/A	N/A
	Lewis River	N/A	N/A	N/A
	Salmon Creek	N/A	N/A	N/A
	Clackamus River	N/A	N/A	N/A
	Sandy River	N/A	N/A	N/A
	Washougal River	N/A	N/A	N/A
	Lower Gorge tributaries	0.984	0.883	1.096
	Upper Gorge tributaries	N/A	N/A	N/A
Hood Canal Summer - Run (only have λ where hatchery fish reproductive potential = native fish; Good et. al., 2005)	Jimmycomelately Creek	0.850	0.690	1.010
	Salmon / Snow Creeks	1.230	1.130	1.330
	Big / Little Quilcene rivers	1.390	1.170	1.610
	Lilliwaup Creek	1.190	0.750	1.630
	Hamma Hamma River	1.300	1.110	1.490
	Duckabush River	1.100	0.930	1.270
	Dosewallips River	1.170	0.930	1.410
	Union River	1.150	1.050	1.250
	Chimacum Creek	N/A	N/A	N/A
	Big Beef Creek	N/A	N/A	N/A
	Dewetto Creek	N/A	N/A	N/A

Coho Salmon

ESU	Population	λ - H=0	95% CI -lower	95% CI - upper
Central California Coast	Ten Mile River	N/A	N/A	N/A
	Noyo River	N/A	N/A	N/A
	Big River	N/A	N/A	N/A
	Navarro River	N/A	N/A	N/A
	Garcia River	N/A	N/A	N/A
	Other Mendacino County Rivers	N/A	N/A	N/A
	Gualala River	N/A	N/A	N/A
	Russain River	N/A	N/A	N/A
	Other Sonoma County Rivers	N/A	N/A	N/A
	Martin County	N/A	N/A	N/A
	San Mateo County	N/A	N/A	N/A
	Santa Cruz County	N/A	N/A	N/A
	San Lorenzo River	N/A	N/A	N/A
Lower Columbia River (Good et al., 2005)	Youngs Bay	N/A	N/A	N/A
	Grays River	N/A	N/A	N/A
	Elochoman River	N/A	N/A	N/A
	Clatskanie River	N/A	N/A	N/A
	Mill, Abernathy, Germany Creeks	N/A	N/A	N/A
	Scappose Creek	N/A	N/A	N/A
	Cispus River	N/A	N/A	N/A
	Tilton River	N/A	N/A	N/A
	Upper Cowlitz River	N/A	N/A	N/A
	Lower Cowlitz River	N/A	N/A	N/A
	North Fork Toutle River	N/A	N/A	N/A
	South Fork Toutle River	N/A	N/A	N/A
	Coweeman River	N/A	N/A	N/A
	Kalama River	N/A	N/A	N/A
	North Fork Lewis River	N/A	N/A	N/A
	East Fork Lewis River	N/A	N/A	N/A
	Upper Clackamas River	1.028	0.898	1.177
	Lower Clackamas River	N/A	N/A	N/A
	Salmon Creek	N/A	N/A	N/A
	Upper Sandy River	1.102	0.874	1.172
	Lower Sandy River	N/A	N/A	N/A
	Washougal River	N/A	N/A	N/A
	Lower Columbia River gorge tributaries	N/A	N/A	N/A
	White Salmon	N/A	N/A	N/A
	Upper Columbia River gorge tributaries	N/A	N/A	N/A
	Hood River	N/A	N/A	N/A

Coho Salmon (continued)

ESU	Population	λ - H=0	95% CI -lower	95% CI - upper
Southern Oregon and Northern California Coast	Southern Oregon and Northern California Coast	N/A	N/A	N/A
Oregon Coast	Necanicum	N/A	N/A	N/A
	Nehalem	N/A	N/A	N/A
	Tillamook	N/A	N/A	N/A
	Nestucca	N/A	N/A	N/A
	Siletz	N/A	N/A	N/A
	Yaquima	N/A	N/A	N/A
	Alsea	N/A	N/A	N/A
	Siuslaw	N/A	N/A	N/A
	Umpqua	N/A	N/A	N/A
	Coos	N/A	N/A	N/A
	Coquille	N/A	N/A	N/A

Sockeye Salmon

ESU	Population	λ - H=0	95% CI -lower	95% CI - upper
Ozette Lake	Ozette Lake	N/A	N/A	N/A
Snake River	Snake River	N/A	N/A	N/A

Steelhead

DPS	Population	λ - H=0	95% CI -lower	95% CI - upper
Central California Coast (Good et al., 2005)	Russain River	N/A	N/A	N/A
	Lagunitas	N/A	N/A	N/A
	San Gregorio	N/A	N/A	N/A
	Waddell Creek	N/A	N/A	N/A
	Scott Creek	N/A	N/A	N/A
	San Vincente Creek	N/A	N/A	N/A
	San Lorenzo River	N/A	N/A	N/A
	Soquel Creek	N/A	N/A	N/A
	Aptos Creek	N/A	N/A	N/A
California Central Valley (Good et al., 2005)	Sacramento River	0.950	0.900	1.020
Lower Columbia River (Good et al., 2005)	Cispus River	N/A	N/A	N/A
	Tilton River	N/A	N/A	N/A
	Upper Cowlitz River	N/A	N/A	N/A
	Lower Cowlitz River	N/A	N/A	N/A
	Coweeman River	0.908	0.792	1.041
	South Fork Toutle River	0.938	0.830	1.059
	North Fork Toutle River	1.062	0.915	1.233
	Kalama River - winter run	1.010	9.130	1.117
	Kalama River - summer run	0.981	0.889	1.083
	North Fork Lewis River - winter run	N/A	N/A	N/A
	North Fork Lewis River - summer run	N/A	N/A	N/A
	East Fork Lewis River - winter run	N/A	N/A	N/A
	East Fork Lewis River - summer run	N/A	N/A	N/A
	Salmon Creek	N/A	N/A	N/A
	Washougal River - winter run	N/A	N/A	N/A
	Washougal River - summer run	1.003	0.884	1.138
	Clackamas River	0.971	0.901	1.047
	Sandy River	0.945	0.850	1.051
	Lower Columbia gorge tributaries	N/A	N/A	N/A
	Upper Columbia gorge tributaries	N/A	N/A	N/A

Steelhead (continued)

DPS	Population	λ - H=0	95% CI -lower	95% CI - upper
Middle Columbia River (Good et al., 2005)	Klickitat River	N/A	N/A	N/A
	Yakima River	1.009	N/A	N/A
	Fifteenmile Creek	0.981	N/A	N/A
	Deschutes River	1.022	N/A	N/A
	John Day - upper main stream	0.975	N/A	N/A
	John Day - lower main stream	0.981	N/A	N/A
	John Day - upper north fork	1.011	N/A	N/A
	John Day - lower north fork	1.013	N/A	N/A
	John Day - middle fork	0.966	N/A	N/A
	John Day - south fork	0.967	N/A	N/A
	Umatilla River	1.007	N/A	N/A
	Touchet River	0.961	N/A	N/A
Northern California (Good et al., 2005)	Redwood Creek	N/A	N/A	N/A
	Mad River - winter run	1.000	0.930	1.050
	Eel River - summer run	0.980	0.930	1.040
	Mattole River	N/A	N/A	N/A
	Ten Mile river	N/A	N/A	N/A
	Noyo River	N/A	N/A	N/A
	Big River	N/A	N/A	N/A
	Navarro River	N/A	N/A	N/A
	Garcia River	N/A	N/A	N/A
	Gualala River	N/A	N/A	N/A
	Other Humboldt County streams	N/A	N/A	N/A
	Other Mendocino County streams	N/A	N/A	N/A
Puget Sound*	Puget Sound	N/A	N/A	N/A
Snake River (Good et al., 2005)	Tucannon River	0.886	N/A	N/A
	Lower Granite run	0.994	N/A	N/A
	Snake A run	0.998	N/A	N/A
	Snake B run	0.927	N/A	N/A
	Asotin Creek	N/A	N/A	N/A
	Upper Grande Ronde River	0.967	N/A	N/A
	Joseph Creek	1.069	N/A	N/A
	Imnaha River	1.045	N/A	N/A
	Camp Creek	1.077	N/A	N/A
South-Central California Coast	South-Central California Coast	N/A	N/A	N/A
Southern California	Santa Ynez River	N/A	N/A	N/A
	Ventura River	N/A	N/A	N/A
	Matilija River	N/A	N/A	N/A
	Creek River	N/A	N/A	N/A
	Santa Clara River	N/A	N/A	N/A

Steelhead (continued)

DPS	Population	λ - H=0	95% CI -lower	95% CI - upper
Upper Columbia River (Good et al., 2005)	Wenatchee / Entiat Rivers	1.067	N/A	N/A
	Methow / Okanogan Rivers	1.086	N/A	N/A
Upper Williamette River (McElhany et al., 2007)	Molalla River	0.988	0.790	1.235
	North Santiam River	0.983	0.789	1.231
	South Santiam River	0.976	0.855	1.114
	Calapooia River	1.023	0.743	1.409

Appendix 3: Abbreviations

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 Divsion
BLM	Bureau of Land Management
BMP	Best Management Practices
BOR	Bureau of Reclaimation
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

Appendix 4: 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 fro an significant to the remainder of its species based n 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 Salmon	This salmon stock returns from the ocean in late summer and early 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 an technologically feasible, an 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 an 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 judgement.
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 Daily Load (TMDL)	defines how much of a pollutant a water body can tolerate (absorb) 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.
Unique Mixture	A specific combination of 2 or more compounds, regardless of the presence of other compounds.
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

Appendix 5: Exposure Modeling

GENEEC RUNS - September 28, 2009

```
RUN No.   1 FOR ethoprop         ON   potatoes      * INPUT VALUES *
--------------------------------------------------------------------
RATE (#/AC)   No.APPS &   SOIL  SOLUBIL   APPL TYPE  NO-SPRAY INCORP
 ONE(MULT)    INTERVAL     Kd   (PPM )    (%DRIFT)   ZONE(FT)  (IN)
--------------------------------------------------------------------
12.000( 12.000)   1   1       2.1  843.0   GRANUL(   .0)    .0   4.0


FIELD AND STANDARD POND HALFLIFE VALUES (DAYS)
--------------------------------------------------------------------
METABOLIC  DAYS UNTIL  HYDROLYSIS   PHOTOLYSIS   METABOLIC  COMBINED
 (FIELD)   RAIN/RUNOFF   (POND)     (POND-EFF)    (POND)     (POND)
--------------------------------------------------------------------
 300.00        2          N/A       .00-     .00   600.00    600.00


GENERIC EECs (IN MICROGRAMS/LITER (PPB))     Version 2.0 Aug 1, 2001
--------------------------------------------------------------------
    PEAK      MAX 4 DAY     MAX 21 DAY    MAX 60 DAY    MAX 90 DAY
    GEEC      AVG GEEC       AVG GEEC      AVG GEEC      AVG GEEC
--------------------------------------------------------------------
   127.29     127.02        125.50        122.15        119.70
```

RUN No. 2 FOR methyl parathion ON potatoes * INPUT VALUES *

RATE (#/AC) ONE(MULT)	No.APPS & INTERVAL		SOIL Koc	SOLUBIL (PPM)	APPL TYPE (%DRIFT)	NO-SPRAY (FT)	INCORP (IN)
1.500(3.519)	4	7	486.0	60.0	AERL_B(13.0)	.0	.0

FIELD AND STANDARD POND HALFLIFE VALUES (DAYS)

METABOLIC (FIELD)	DAYS UNTIL RAIN/RUNOFF	HYDROLYSIS (POND)	PHOTOLYSIS (POND-EFF)	METABOLIC (POND)	COMBINED (POND)
11.25	2	N/A	2.00- 248.00	12.30	11.72

GENERIC EECs (IN MICROGRAMS/LITER (PPB)) Version 2.0 Aug 1, 2001

PEAK GEEC	MAX 4 DAY AVG GEEC	MAX 21 DAY AVG GEEC	MAX 60 DAY AVG GEEC	MAX 90 DAY AVG GEEC
120.15	114.94	87.96	52.47	38.32

RUN No. 3 FOR phorate ON potatoes * INPUT VALUES *

RATE (#/AC) ONE(MULT)	No.APPS & INTERVAL	SOIL Koc	SOLUBIL (PPM)	APPL TYPE (%DRIFT)	NO-SPRAY (FT)	INCORP (IN)
2.310(2.310)	1 1	50.0	500.0	GRANUL(.0)	.0	1.2

FIELD AND STANDARD POND HALFLIFE VALUES (DAYS)

METABOLIC (FIELD)	DAYS UNTIL RAIN/RUNOFF	HYDROLYSIS (POND)	PHOTOLYSIS (POND-EFF)	METABOLIC (POND)	COMBINED (POND)
137.00	2	N/A	1.10- 136.40	21.00	18.20

GENERIC EECs (IN MICROGRAMS/LITER (PPB)) Version 2.0 Aug 1, 2001

PEAK GEEC	MAX 4 DAY AVG GEEC	MAX 21 DAY AVG GEEC	MAX 60 DAY AVG GEEC	MAX 90 DAY AVG GEEC
98.41	95.36	80.29	56.31	44.44

RUN No.   4 FOR methidathion   ON   oranges   * INPUT VALUES *

RATE (#/AC) ONE(MULT)	No.APPS & INTERVAL	SOIL Koc	SOLUBIL (PPM)	APPL TYPE (%DRIFT)	NO-SPRAY (FT)	INCORP (IN)
5.000(5.841)	2 45	364.0	2500.0	ORCHAR(3.8)	50.0	.0

FIELD AND STANDARD POND HALFLIFE VALUES (DAYS)

METABOLIC (FIELD)	DAYS UNTIL RAIN/RUNOFF	HYDROLYSIS (POND)	PHOTOLYSIS (POND-EFF)	METABOLIC (POND)	COMBINED (POND)
17.50	2	N/A	10.00- 1240.00	35.00	34.04

GENERIC EECs (IN MICROGRAMS/LITER (PPB))   Version 2.0 Aug 1, 2001

PEAK GEEC	MAX 4 DAY AVG GEEC	MAX 21 DAY AVG GEEC	MAX 60 DAY AVG GEEC	MAX 90 DAY AVG GEEC
201.05	197.47	178.06	142.46	121.62

RUN No. 5 FOR naled ON oranges * INPUT VALUES *

```
--------------------------------------------------------------------
 RATE (#/AC)   No.APPS &   SOIL  SOLUBIL   APPL TYPE NO-SPRAY INCORP
  ONE(MULT)    INTERVAL    Koc   (PPM )    (%DRIFT)   (FT)     (IN)
--------------------------------------------------------------------
1.875(  2.321)   3   7     37.015600.0   ORCHAR(  3.8)   50.0    .0
```

FIELD AND STANDARD POND HALFLIFE VALUES (DAYS)

```
--------------------------------------------------------------------
METABOLIC  DAYS UNTIL  HYDROLYSIS   PHOTOLYSIS   METABOLIC  COMBINED
 (FIELD)   RAIN/RUNOFF   (POND)     (POND-EFF)    (POND)     (POND)
--------------------------------------------------------------------
   3.00        2          N/A     10.00- 1240.00    6.00      5.97
```

GENERIC EECs (IN MICROGRAMS/LITER (PPB)) Version 2.0 Aug 1, 2001

```
--------------------------------------------------------------------
    PEAK      MAX 4 DAY     MAX 21 DAY    MAX 60 DAY    MAX 90 DAY
    GEEC      AVG GEEC      AVG GEEC      AVG GEEC      AVG GEEC
--------------------------------------------------------------------
    83.11      75.99         47.42         21.97         14.92
```

```
RUN No.   6 FOR Phosmet         ON   Oranges       * INPUT VALUES *
--------------------------------------------------------------------
RATE (#/AC)   No.APPS &   SOIL  SOLUBIL   APPL TYPE  NO-SPRAY INCORP
 ONE(MULT)    INTERVAL     Kd   (PPM )    (%DRIFT)   ZONE(FT)  (IN)
--------------------------------------------------------------------
2.100(  4.078)   2   7      10.7  250.0   ORCHAR(  9.7)    .0    .0


FIELD AND STANDARD POND HALFLIFE VALUES (DAYS)
--------------------------------------------------------------------
METABOLIC  DAYS UNTIL  HYDROLYSIS   PHOTOLYSIS   METABOLIC  COMBINED
 (FIELD)   RAIN/RUNOFF   (POND)     (POND-EFF)    (POND)     (POND)
--------------------------------------------------------------------
  81.00        2           .39     .00-     .00     .00        .39


GENERIC EECs (IN MICROGRAMS/LITER (PPB))     Version 2.0 Aug 1, 2001
--------------------------------------------------------------------
    PEAK      MAX 4 DAY     MAX 21 DAY    MAX 60 DAY    MAX 90 DAY
    GEEC      AVG GEEC       AVG GEEC      AVG GEEC      AVG GEEC
--------------------------------------------------------------------
    90.22      35.52          6.90          2.42          1.61
```

RUN No.   7 FOR AZM   ON   Cherrries   * INPUT VALUES *

RATE (#/AC) ONE(MULT)	No.APPS & INTERVAL		SOIL Kd	SOLUBIL (PPM)	APPL TYPE (%DRIFT)	NO-SPRAY ZONE(FT)	INCORP (IN)
.750(.750)	1	1	7.6	25.1	ORCHAR(3.4)	60.0	.0

FIELD AND STANDARD POND HALFLIFE VALUES (DAYS)

METABOLIC (FIELD)	DAYS UNTIL RAIN/RUNOFF	HYDROLYSIS (POND)	PHOTOLYSIS (POND-EFF)		METABOLIC (POND)	COMBINED (POND)
95.40	2	N/A	3.19-	395.56	190.80	128.71

GENERIC EECs (IN MICROGRAMS/LITER (PPB))   Version 2.0 Aug 1, 2001

PEAK GEEC	MAX 4 DAY AVG GEEC	MAX 21 DAY AVG GEEC	MAX 60 DAY AVG GEEC	MAX 90 DAY AVG GEEC
21.26	21.11	20.24	18.42	17.19

```
RUN No.   8 FOR methidathion     ON   cherries      * INPUT VALUES *
--------------------------------------------------------------------
 RATE (#/AC)   No.APPS &   SOIL  SOLUBIL   APPL TYPE NO-SPRAY INCORP
  ONE(MULT)    INTERVAL    Koc   (PPM )    (%DRIFT)   (FT)     (IN)
--------------------------------------------------------------------
3.000(  3.000)   1   1     364.0 2500.0   ORCHAR(  5.5)   25.0    .0


FIELD AND STANDARD POND HALFLIFE VALUES (DAYS)
--------------------------------------------------------------------
METABOLIC  DAYS UNTIL  HYDROLYSIS   PHOTOLYSIS   METABOLIC  COMBINED
 (FIELD)   RAIN/RUNOFF   (POND)     (POND-EFF)    (POND)     (POND)
--------------------------------------------------------------------
  17.50        2          N/A     10.00- 1240.00    35.00     34.04


GENERIC EECs (IN MICROGRAMS/LITER (PPB))     Version 2.0 Aug 1, 2001
--------------------------------------------------------------------
    PEAK      MAX 4 DAY     MAX 21 DAY    MAX 60 DAY    MAX 90 DAY
    GEEC      AVG GEEC       AVG GEEC      AVG GEEC      AVG GEEC
--------------------------------------------------------------------
   103.68     101.84         91.83         73.47         62.72
```

RUN No.   9 FOR phosmet   ON   cherries   * INPUT VALUES *

```
--------------------------------------------------------------------
RATE (#/AC)   No.APPS &   SOIL  SOLUBIL   APPL TYPE  NO-SPRAY INCORP
 ONE(MULT)    INTERVAL     Kd   (PPM )    (%DRIFT)   ZONE(FT)  (IN)
--------------------------------------------------------------------
.930(   .930)   1   1      10.7  250.0   ORCHAR(  9.7)    .0    .0
```

FIELD AND STANDARD POND HALFLIFE VALUES (DAYS)

```
--------------------------------------------------------------------
METABOLIC  DAYS UNTIL  HYDROLYSIS   PHOTOLYSIS   METABOLIC  COMBINED
 (FIELD)   RAIN/RUNOFF   (POND)     (POND-EFF)    (POND)     (POND)
--------------------------------------------------------------------
  81.00        2           .39     .00-     .00     .00        .39
```

GENERIC EECs (IN MICROGRAMS/LITER (PPB))   Version 2.0 Aug 1, 2001

```
--------------------------------------------------------------------
    PEAK      MAX 4 DAY     MAX 21 DAY    MAX 60 DAY    MAX 90 DAY
    GEEC      AVG GEEC      AVG GEEC      AVG GEEC      AVG GEEC
--------------------------------------------------------------------
    20.70       8.18          1.59          .56           .37
```

AGDISP Input Data Summary 08-19-2010

--General--

Title: Dibrom Applications (Mosquito ULV)

Notes:

Calculations Done: Yes

Run ID: AGDISP Dibrom Mosquito ULV.ag 8.17 08-19-2010 17:28:05

--Aircraft--	----------------------------	
Name	Air Tractor AT-401	
Type	Library	
Boom Height (ft)	200	
Spray Lines	20	
Optimize Spray Lines	No	
Spray Line Reps	#	Reps
	1	1
	2	1
	3	1
	4	1
	5	1
	6	1
	7	1
	8	1
	9	1
	10	1
	11	1
	12	1
	13	1
	14	1
	15	1

16	1
17	1
18	1
19	1
20	1

Wing Type	Fixed-Wing
Semispan (ft)	24.5
Typical Speed (mph)	119.99
Biplane Separation (ft)	0
Weight (lbs)	6000
Planform Area (ft²)	294
Propeller RPM	2000
Propeller Radius (ft)	4.5
Engine Vert Distance (ft)	-1.2
Engine Fwd Distance (ft)	11.9

--Aerial Application Type--	----------------------------
Aerial Application Type	Liquid

--Drop Size Distribution--	----------------------------
Name	Aerosol to Very Fine
Type	Reference

Drop Categories

#	Diam (um)	Frac
1	10.77	0.0280
2	16.73	0.0350
3	19.39	0.0300
4	22.49	0.0200
5	26.05	0.0210
6	30.21	0.0460
7	35.01	0.0780
8	40.57	0.1060
9	47.03	0.1250

10	54.50	0.1240
11	63.16	0.0860
12	73.23	0.0610
13	84.85	0.0700
14	98.12	0.0710
15	113.71	0.0540
16	131.73	0.0270
17	152.79	0.0060
18	177.84	0.0020
19	205.84	0.0100

--Nozzle Distribution-- ----------------------------

Boom Length (%) 65.06

Nozzle Locations

#	Hor(ft)	Ver(ft)	Fwd(ft)
1	-15.94	0	0
2	-15.16	0	0
3	-14.38	0	0
4	-13.61	0	0
5	-12.83	0	0
6	-12.05	0	0
7	-11.27	0	0
8	-10.5	0	0
9	-9.72	0	0
10	-8.94	0	0
11	-8.16	0	0
12	-7.39	0	0
13	-6.61	0	0
14	-5.83	0	0
15	-5.05	0	0
16	-4.28	0	0
17	-3.5	0	0
18	-2.72	0	0

19	-1.94	0	0
20	-1.17	0	0
21	-0.3888	0	0
22	0.3888	0	0
23	1.17	0	0
24	1.94	0	0
25	2.72	0	0
26	3.5	0	0
27	4.28	0	0
28	5.05	0	0
29	5.83	0	0
30	6.61	0	0
31	7.39	0	0
32	8.16	0	0
33	8.94	0	0
34	9.72	0	0
35	10.5	0	0
36	11.27	0	0
37	12.05	0	0
38	12.83	0	0
39	13.61	0	0
40	14.38	0	0
41	15.16	0	0
42	15.94	0	0

--Swath--	----------------------------
Swath Width	400 ft
Swath Displacement	0 ft

--Spray Material--	----------------------------
Name	Oil
Type	Reference

Nonvolatile Fraction	0.103
Active Fraction	0.103
Spray Volume Rate (gal/ac)	0.0078
--Meteorology--	----------------------------
Wind Speed (mph)	2
Wind Direction (deg)	-90
Temperature (deg F)	65
Relative Humidity (%)	50
--Atmospheric Stability--	----------------------------
Atmospheric Stability	Overcast
--Transport--	----------------------------
Flux Plane Distance (ft)	0
--Canopy--	----------------------------
Type	None
--Terrain--	----------------------------
Surface Roughness (ft)	0.0246
Upslope Angle (deg)	0
Sideslope Angle (deg)	0
--Advanced--	----------------------------
Wind Speed Height (ft)	6.56
Max Compute Time (sec)	600
Max Downwind Dist (ft)	2608.24
Vortex Decay Rate (IGE) (mph)	1.25
Vortex Decay Rate (OGE) (mph)	0.3355
Aircraft Drag Coeff	0.1
Propeller Efficiency	0.8

Ambient Pressure (in hg)	29.91
Save Trajectory Files	No
Half Boom	No
Default Swath Offset	1/2 Swath
Specific Gravity (Carrier)	1
Specific Gravity (Nonvolatile)	1
Evaporation Rate (µm²/deg C/sec)	84.76

Appendix 6: Toxicity of 12 OPs to Juvenile Coho Salmon (NOAA 2009)

UNITED STATES DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
NATIONAL MARINE FISHERIES SERVICE
Northwest Fisheries Science Center
2725 Montlake Boulevard East
Seattle, WA 98112-2097

March 11, 2010

To: F/PR3 - Angela Somma, Division Chief, Office of Protected Resources

From: F/NWC - Tracy K. Collier, Director, Environmental Conservation Division

Re: Pesticide Reports in Support of Biological Opinion

Enclosed are two reports titled "The Combined Influence of Temperature and Pesticides on the Brain AChE Activity of Juvenile Coho Salmon" and "Toxicity of Twelve Organophosphate Pesticides to Juvenile Coho Salmon". This work was requested by NOAA's Office of Protected Resources in support of a biological opinion. These reports detail work that was conducted on the effects of pesticides on salmon, and both reports have been extensively reviewed by staff at the Northwest Fisheries Science Center.

Toxicity of Twelve Organophosphate Pesticides to Juvenile Coho Salmon

Introduction

This paper reports the results of an experiment that was conducted for NOAA Fisheries, Office of Protected Resources, in support of a biological opinion regarding organophosphate pesticides and species of threatened and endangered salmon. Juvenile coho salmon were exposed to twelve different organophosphate pesticides, and the subsequent inhibition of the enzyme actylcholinesterase (AChE) was used as a measure of toxicity. AChE is critical for nerve transmission in animals including mammals and fish. Symptoms of cholinergic poisoning include excitability, lethargy, loss of orientation, and increased mucus production. Inhibition of this enzyme can impair swimming and feeding behaviors in juvenile salmon (Sandahl et al., 2005), and can ultimately cause death (Fulton and Key, 2001). AChE data were used to calculate an EC_{50}, or the concentration causing a 50% decrease in AChE activity, for each pesticide. Two of the pesticides, bensulide and fenamiphos, showed no AChE inhibition, but did produce qualitative symptoms of cholinergic poisoning at the highest concentrations tested. One pesticide, methamidophos, showed neither AChE inhibition nor symptoms of poisoning over the range of concentrations tested. The remaining nine pesticides produced concentration-dependent AChE inhibition. All pesticide exposures were intended to be at sublethal concentrations. Incidences of mortality are clearly shown on the figures, but those data were not used in any calculation of EC_{50}.

Methods

Fish

Fertilized coho eggs were obtained from the University of Washington Hatchery in the fall of 2008 and raised at the NWFSC Hatchery. In March 2009, fish were transported to Washington State University's Puyallup Campus and housed there until experiments started in April 2009. Fish were held in recirculating tanks filled with chilled, dechlorinated city water (temperature 12 °C, pH 7.0-8.0, hardness as $CaCO_3$ 120 ppm, alkalinity 40-80 mg/l, and dissolved oxygen 75-90%). Fish were fed commercial fish pellets daily, and exposed to a 12 hour light-dark cycle. Fish used in experiments (n = 584) averaged (mean ± standard deviation) 6.0 ± 0.8 cm in length and 2.2 ± 0.85 g in weight.

Pesticide exposures

Fish (in groups of n = 8) were exposed to a range of concentrations of 12 different organophosphate pesticides (Table 1). The highest concentration of each pesticide was approximately an order of magnitude lower than the reported LC_{50} for salmonids (Table 1). Water-only and methanol control fish (n = 8) were exposed along with each pesticide. Pesticide stock solutions were prepared in methanol and added in 1 ml aliquots to 25 l of hatchery water in 30-l glass aquaria, producing a final carrier concentration of 0.004%. Two stock solutions were used in the bensulide, disulfoton and naled exposures to cover the range of concentrations used. Fish were exposed for 96 hrs with static water renewals conducted every 24 hours. Obvious changes in fish behavior (including excitability, lethargy, and loss of orientation) were noted during daily water changes of the exposure tanks. Following exposures, fish were terminally anaesthetized in MS-222 (tricaine methanesulfonate) until gill activity ceased. Brain tissue (and in some cases muscle tissue) was removed and stored at -80 °C until AChE enzyme analysis.

Analytical chemistry

Three replicate 500 ml water samples were collected from each exposure concentration (one tank of 8 fish at each concentration) to confirm accurate dosing. Prior to sample collection, amber glass bottles were washed in distilled water and rinsed in acetone. Samples were collected at either 48 or 72 hours immediately following that day's water change. Bottles were submersed in the exposure tanks and filled leaving no air space. Five drops of acetic acid solution (1:1 glacial acetic acid to distilled water) were added to all water samples to adjust the pH to between 6.0 and 7.0, thereby improving the chemical stability of the pesticides. Samples were stored under refrigeration until delivery to Washington State University's Food and Environmental Quality Laboratory in Richland, WA, under the direction of Dr. Vincent Hebert, for analysis. Pesticide recoveries from the stock solutions used to dose the exposure tanks averaged 108 ± 19% (Table 2). Pesticide recoveries from exposure tanks averaged 117 ± 31% for all chemicals combined (Table 3). Naled was excluded fro the mean calculation because it was found to not be stable in aqueous solution over the duration of sample storage (data in analytical summary report). Throughout this report, pesticide concentrations are presented as nominal values.

AChE enzyme assay

Determination of AChE enzyme activity followed previously published methods (Laetz et al., 2009). Briefly, whole brains were homogenized at 50 mg/ml in 0.1 M sodium phosphate buffer with 0.1% Triton X-100. Homogenates were centrifuged, and 15 μl of the supernatant were combined with 685 μl of 10 mM phosphate buffered saline, 50 μl of 6 mM DTNB (5,5'-dithio-bis(2-nitrobenzoic acid)) and 30 μl of 100 mM acetylthiocholine iodide. Triplicate 200 μl samples were transferred to a 96-well plate, and the change in absorbance at 412 nm was measured on a plate reader at 12 s intervals for 5 min at 25 °C. For each exposure, AChE activity was quantified as mOD/min/g tissue and reported as a percentage of the enzyme activity measured from the corresponding methanol control fish. For all exposures, AChE activities from water controls did not differ from methanol controls (ANOVA, $p = 0.68$).

Table 1. Nominal exposure concentrations and reported LC_{50} values of all twelve pesticides.

Pesticide	**Nominal Exposure Concentrations (μg/l)**	**LC_{50} (μg/l)**
azinphos-methyl	0.05, 0.1, 0.2, 0.3, 0.45, 0.6, 1	4-8
bensulide	5, 10, 50, 100, 200, 300, 400, 500	720-3200
dimethoate	30, 60, 150, 300, 450, 600	6200-8600
disulfoton	10, 30, 100, 300, 1000	3000
ethoprop	10, 25, 50, 100, 250, 500	700-13,800
fenamiphos	0.3, 1, 3, 10, 30, 100	68-560
methamidophos	15, 30, 60, 120, 300, 600, 1000	25,000
methidathion	0.2, 0.6, 1.2, 2, 4, 6	10-26
methyl parathion	3, 10, 20, 30, 100, 300	3000-6000
naled	0.6, 2, 6, 20, 60	200
phorate	0.06, 0.2, 0.6, 1.2, 2, 6	20
phosmet	0.6, 2, 6, 20, 60	200-1500

Table 2. Nominal and actual concentrations of pesticide stock solutions used to dose exposure tanks.

Pesticide	Nominal (mg/l)	Actual (mg/l)	Recovery (%)
azinphos-methyl	0.025	0.0333	133
bensulide-stock 1	2.5	2.7	107
bensulide-stock 2	12.5	12.4	99
dimethoate	15.2	21.9	145
disulfoton-stock 1	7.6	10.3	136
disulfoton-stock 2	25.4	29.2	115
ethoprop	12.7	15.2	120
fenamiphos	2.5	2.6	103
methamidophos	25	24	96
methidathion	0.5	0.5	96
methyl parathion	7.5	8.02	107
naled-stock 1	0.5	0.4	79
naled-stock 2	1.5	1.2	83
phorate	0.15	0.152	101
phosmet	1.5	1.5	102
mean			**108**
sd (n = 15)			**19**

Table 3. Nominal and actual pesticide recoveries from exposure tanks. For each chemical, one tank was sampled at each exposure concentration. Naled was excluded from the mean calculation because it was not stable over the duration of sample storage.

Pesticide	Mean Recovery (%)	Tanks (#)	sd
azinphos-methyl	165	6	13.3
bensulide	164	4	4
dimethoate	119	6	7.7
disulfoton	127	7	21
ethoprop	142	6	12.8
fenamiphos	100	5	18.1
methamidophos	79	3	4.1
methidathion	104	5	5.8
methyl parathion	119	6	28.9
naled	38	5	10
phorate	86	5	26.4
phosmet	79	5	26.4
mean	**117**		
sd (n = 11)	**31**		

Results

Nine of the pesticides (azinphos-methyl, dimethoate, disulfoton, ethoprop, methidathion, methyl parathion, naled, phorate, and phosmet), showed a clear trend of concentration-dependent AChE inhibition. For these chemicals, EC_{50} values were calculated using non-linear regression of the form $y = 100/1+(x/EC_{50})^{slope}$ (sigmoidal dose-response with variable slope; parameters listed in Table 4). Three of the pesticides (bensulide, fenamiphos and methamidophos), showed no inhibition of brain or muscle AChE over the range of exposure concentrations. For these chemicals, EC_{50} values were not calculated. Concentration-response curves for individual pesticides are presented on the following pages (Figures 1 – 12). For all figures, AChE data is presented as the average percent activity relative to methanol control fish.

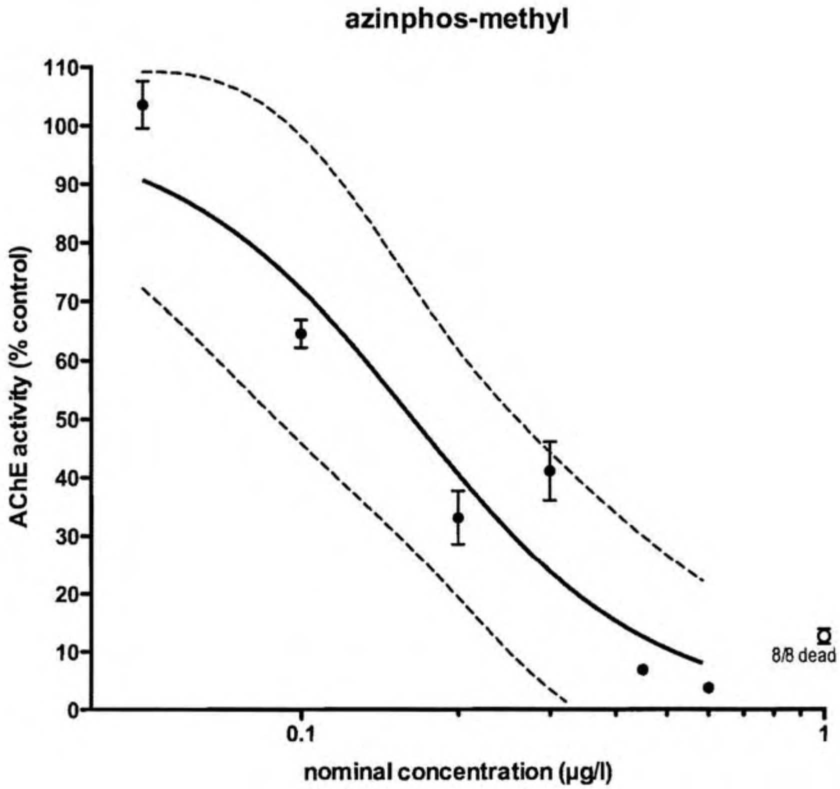
Table 4. Calculated parameters of the regression analysis of AChE activity data from nine of the pesticide exposures.

Pesticide	**EC_{50} (µg/l)**	**95% confidence interval (µg/l)**	**Slope**	**95% confidence interval**	**R^2**
azinphos-methyl	0.16	0.10 to 0.26	-1.9	-3.4 to -0.4	0.91
dimethoate	273	196 to 382	-0.86	-1.2 to -0.52	0.95
disulfoton	488	112 to 2118	-0.32	-0.57 to -0.07	0.65
ethoprop	90.6	69.5 to 118.2	-1.3	-1.8 to -0.89	0.98
methidathion	1.1	0.47 to 2.7	-0.92	-1.9 to 0.1	0.46
methyl-parathion	28.8	21.2 to 39.0	-0.7	-0.88 to -0.51	0.98
naled	7.8	6.5 to 9.5	-1.3	-1.6 to -1.0	0.99
phorate	0.57	0.43 to 0.76	-1.6	-2.3 to -0.91	0.99
phosmet	3.3	2.5 to 4.2	-1.0	-1.3 to -0.78	0.99

Azinphos-methyl

Azinphos-methyl showed a clear trend of decreasing AChE activity with increasing concentration. A total of 7 concentrations (n = 8 fish per concentration) were measured. The calculated EC_{50} is 0.16 µg/l, making it the most toxic of the pesticides studied. The exposure at 1.0 µg/l was excluded from the calculation of EC_{50} because all of the fish died before the end of the 96-hour exposure period (Fig. 1). Closed circles are means ± standard error. The solid line is the regression of the data, dashed lines show the 95% confidence interval of the regression.

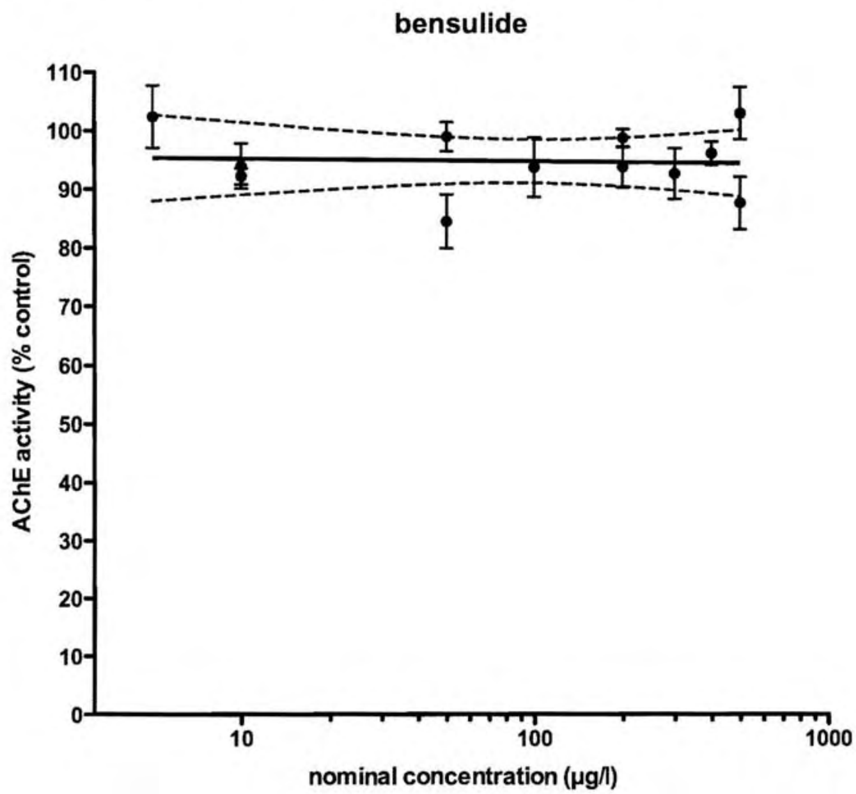
Figure 1. Brain AChE activity of juvenile coho exposed to azinphos-methyl.



Bensulide

Bensulide did not show any AChE inhibition in either brain or muscle tissue over the concentration range tested (Fig. 2; muscle data not shown). A total of 8 concentrations (n = 8 fish per concentration) were tested. Four concentrations were replicated. Although no AChE inhibition was measured, fish at the highest exposure concentrations (300, 400 and 500 µg/l) showed symptoms of cholinergic poisoning including lethargy, excitability, and loss of orientation. Closed circles are means ± standard error. The solid line is the regression of the data, dashed lines show the 95% confidence interval of the regression.

Figure 2. Brain AChE activity in juvenile coho following bensulide exposure.

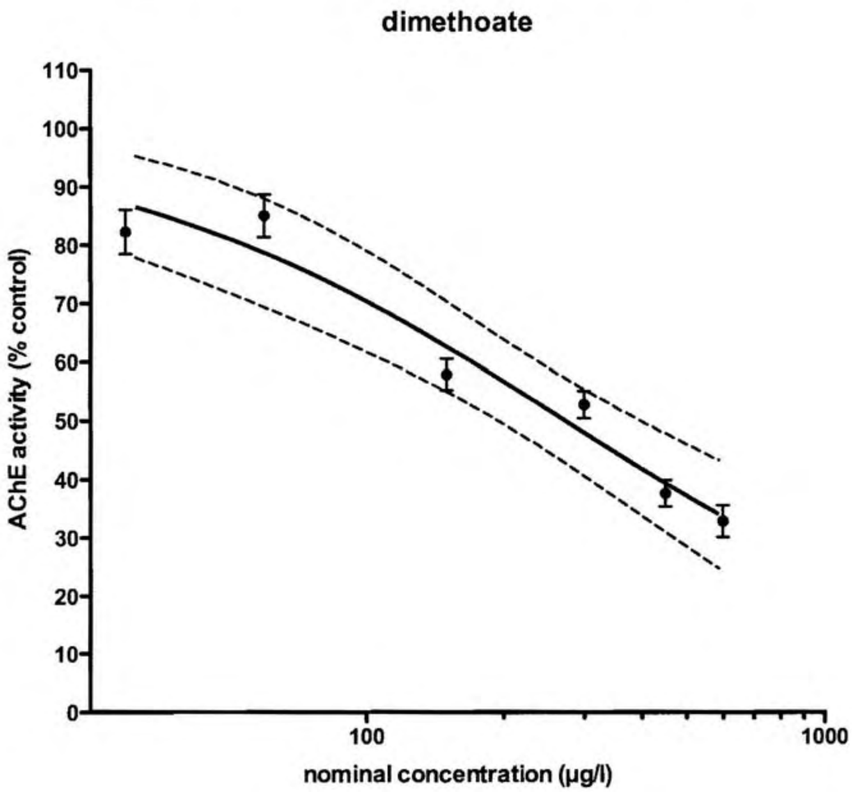


6

Dimethoate

Dimethoate showed a trend of decreasing AChE activity with increasing concentration (Fig. 3). A total of 6 concentrations (n = 8 fish per concentration) were tested. The calculated EC_{50} is 273.4 µg/l. Closed circles are means ± standard error. The solid line is the regression of the data, dashed lines show the 95% confidence interval of the regression.

Figure 3. Brain AChE activity in juvenile coho following dimethoate exposure.

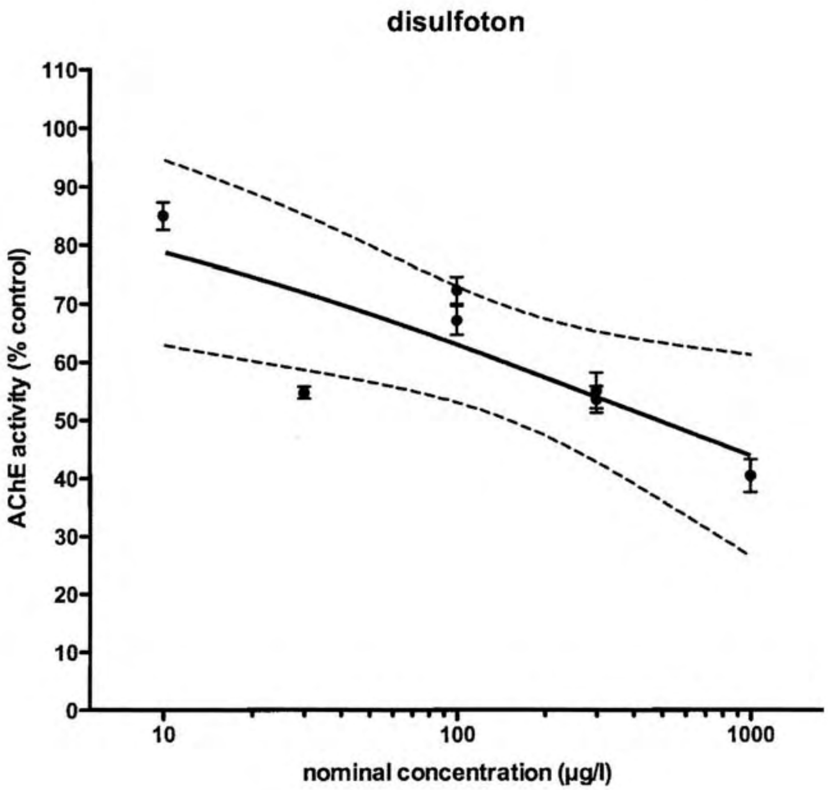


7

Disulfoton

Disulfoton showed a trend of decreasing AChE activity with increasing concentration (Fig. 4). A total of 5 concentrations (n = 8 fish per concentration) were tested, with replicate exposures at 30, 100 and 300 µg/l. The calculated EC_{50} is 487.7 µg/l. AChE activities were lowered to only about 40% of controls. Higher concentrations were not tested because those expected concentrations were approaching the published LC_{50} value of 3000 µg/l. Fish at the highest exposure level (1000 µg/l) were noticeably lethargic, a symptom of cholinergic poisoning. Closed circles are means ± standard error. The solid line is the regression of the data, dashed lines show the 95% confidence interval of the regression.

Figure 4. Brain AChE activity in juvenile coho following disulfoton exposure.

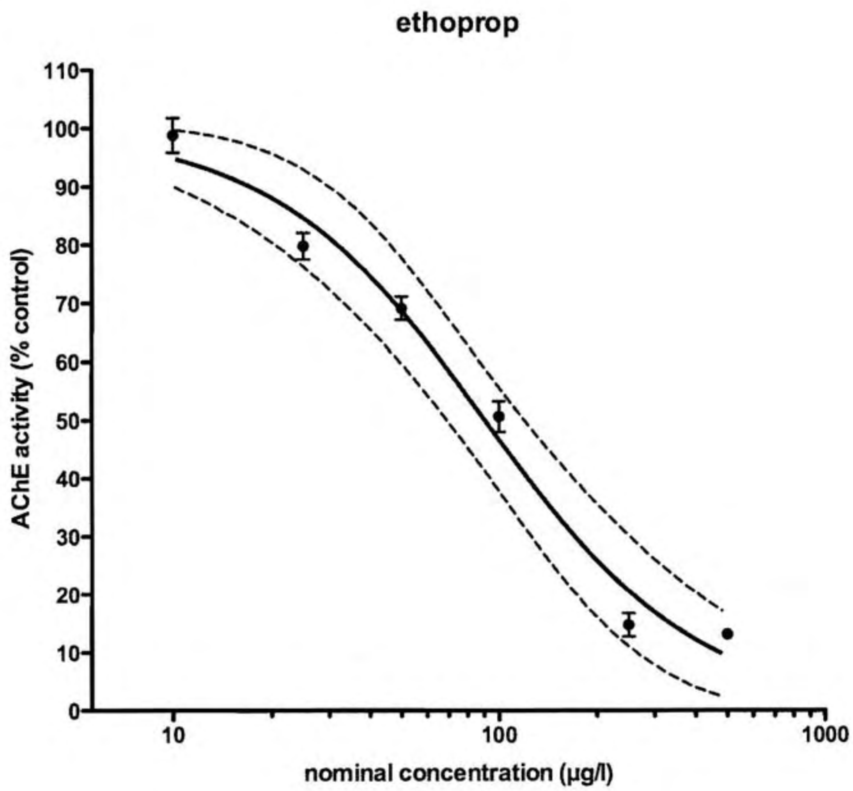


8

Ethoprop

Ethoprop showed a clear trend of decreasing AChE activity with increasing pesticide concentration (Fig. 5). A total of 6 concentrations (n = 8 fish per concentration) were tested. Fish at 250 and 500 µg/l showed excitability and lethargy, symptoms of cholinergic poisoning. The calculated EC_{50} is 90.62 µg/l. Closed circles are means ± standard error. The solid line is the regression of the data, dashed lines show the 95% confidence interval of the regression.

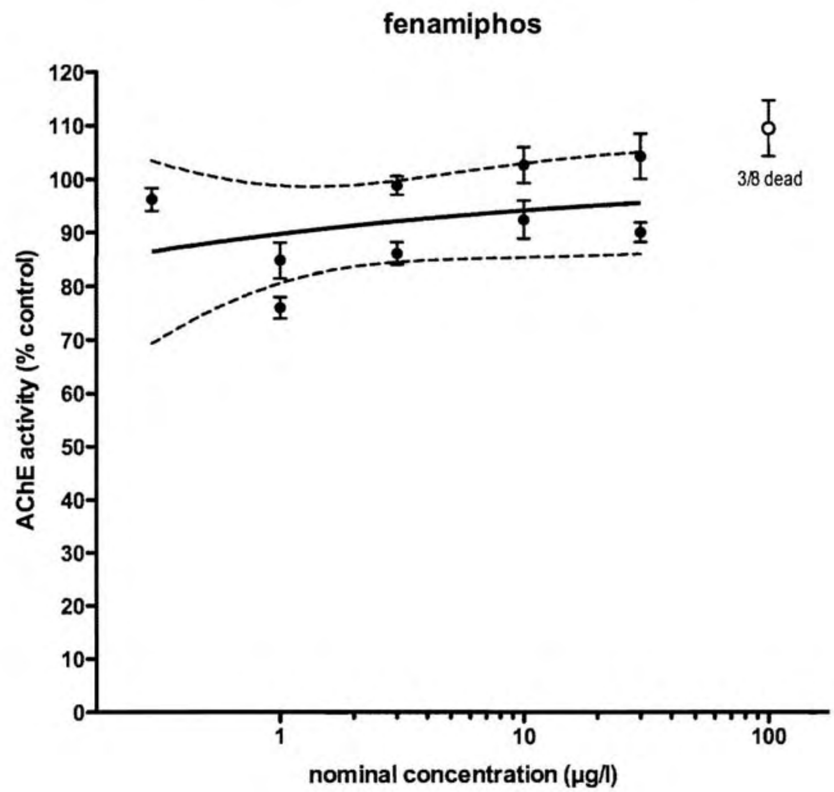
Figure 5. Brain AChE activity in juvenile coho following ethoprop exposure.



Fenamiphos

Fenamiphos did not show any AChE inhibition in either brain or muscle tissue over the range of concentrations tested, despite clear symptoms of cholinergic poisoning and mortality (Fig. 6; muscle data not shown). A total of 6 concentrations (n = 8 fish per concentration) were tested, with replicate exposures at 1, 3, 10, and 30 µg/l. At 100 µg/l three of the eight exposed fish died before the end of the 96-hour exposure period. Fish in the 10, 30 and 100 µg/l exposures showed clear signs of cholinergic poisoning including excitability, lethargy and loss of orientation. Closed circles are means ± standard error. The solid line is the regression of the data, dashed lines show the 95% confidence interval of the regression.

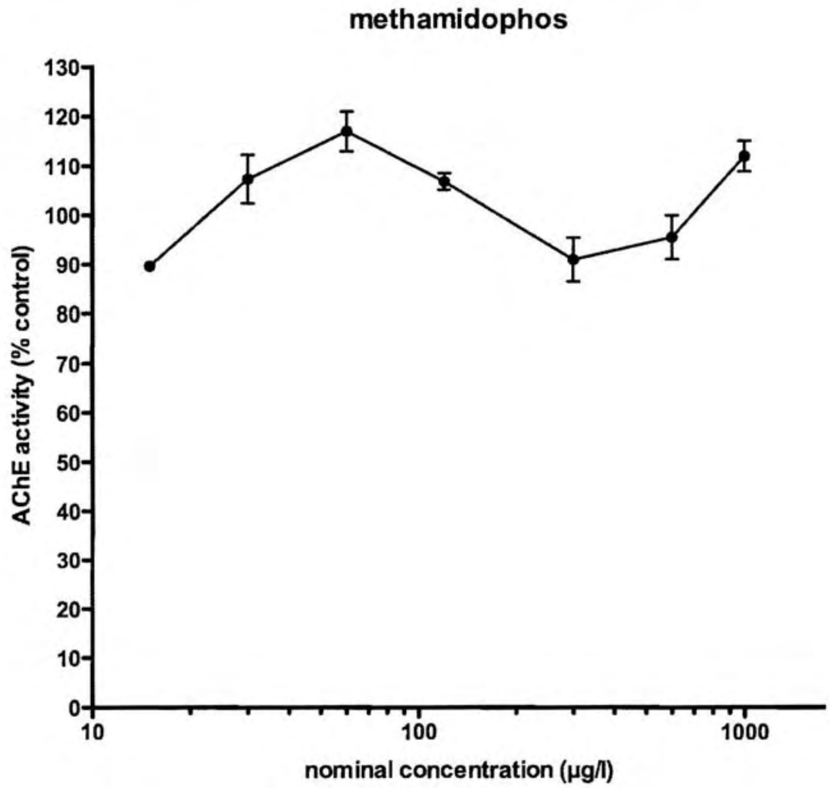
Figure 6. Brain AChE activity in juvenile coho following fenamiphos exposure.



Methamidophos

Methamidophos did not show a trend of AChE inhibition over the range of concentrations tested (Fig. 7). No behavioral indications of toxicity were observed. A total of 7 concentrations (n = 8 fish per concentration) were tested. The reported LC_{50} is about 25 mg/l, indicating that this chemical is not very toxic to fish at expected environmental concentrations. An EC_{50} was not calculated, and a regression analysis was not performed on the data. Closed circles are means ± standard error.

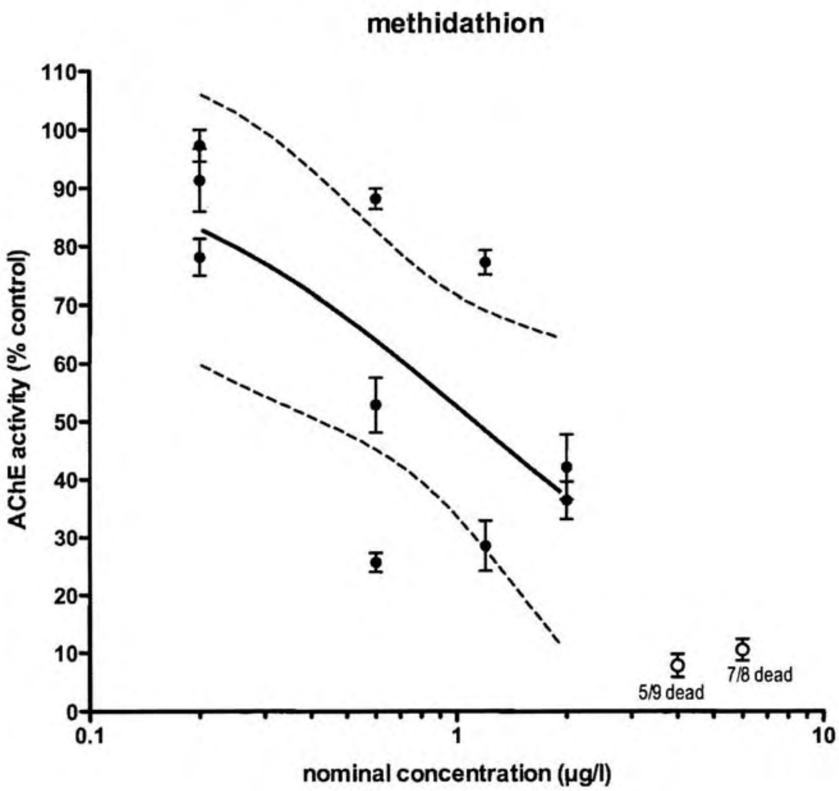
Figure 7. Brain AChE activity in juvenile coho following methamidophos exposure.



Methidathion

Methidathion did show a trend of decreasing AChE activity with increasing concentration (Fig. 8). A total of 6 concentrations (n = 8 fish per concentration) were tested, with replicate exposures at all concentrations except the two highest (4 and 6 µg/l) where mortality occurred. AChE activities from replicate exposures were quite variable, possibly due to the small differences between absolute exposure concentrations. The calculated EC_{50} value is 1.123 µg/l. Closed circles are means $\pm$ standard error. The solid line is the regression of the data, dashed lines show the 95% confidence interval of the regression.

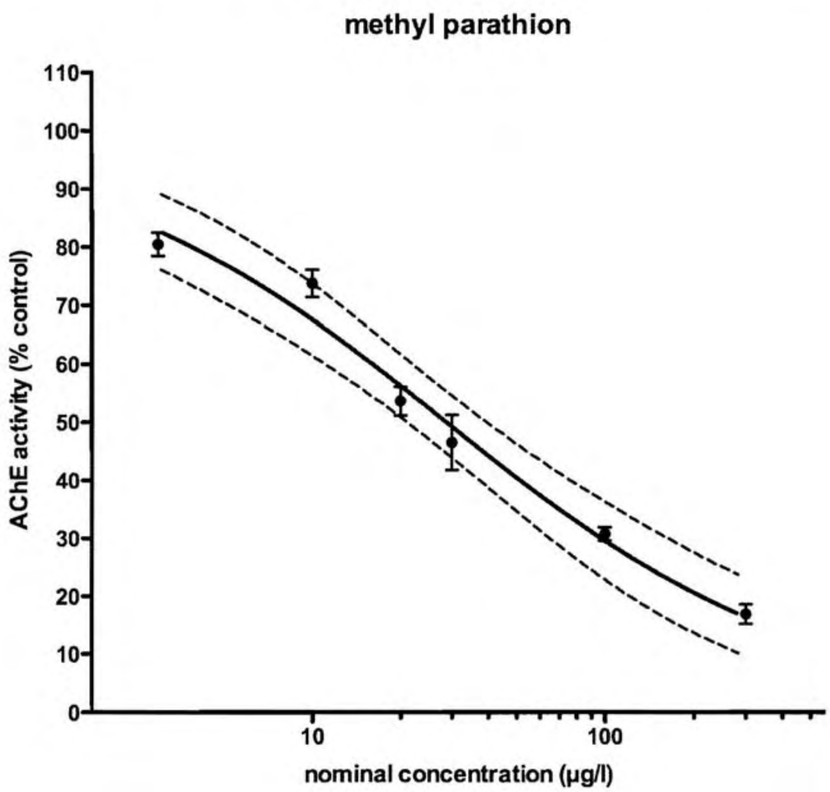
Figure 8. Brain AChE activity in juvenile coho following methidathion exposure.



Methyl-parathion

Methyl parathion showed a clear trend of decreasing AChE activity with increasing concentration (Fig. 9). A total of 6 concentrations (n = 8 fish per concentration) were tested. Fish at the highest exposure concentration (300 µg/l) were lethargic, a symptom of cholinergic poisoning. The calculated EC_{50} value is 28.75 µg/l. Closed circles are means ± standard error. The solid line is the regression of the data, dashed lines show the 95% confidence interval of the regression.

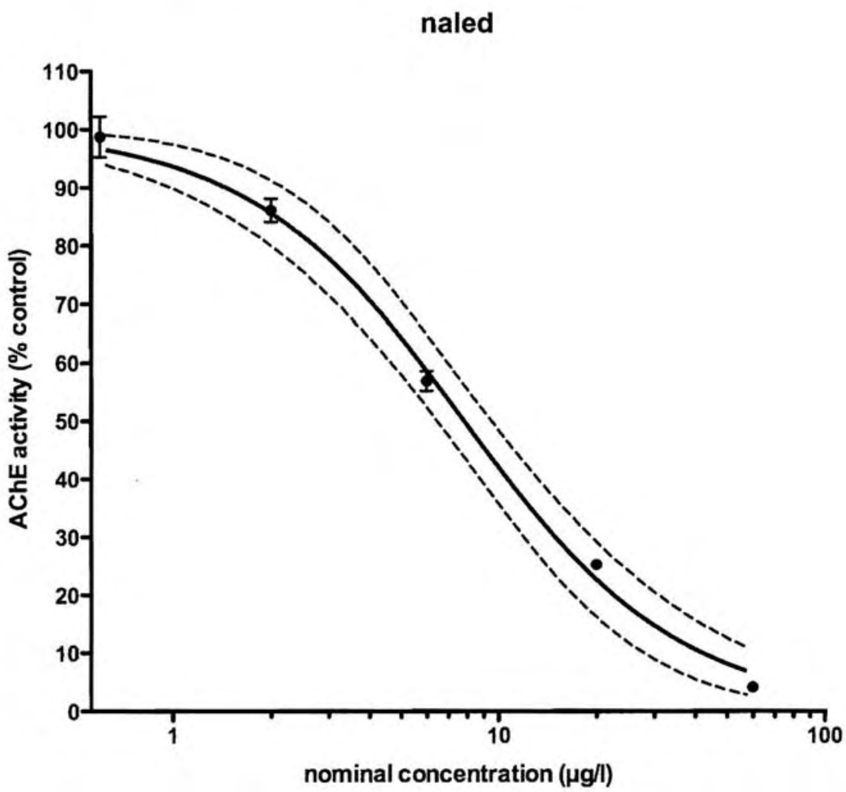
Figure 9. Brain AChE activity in juvenile coho following methyl-parathion exposure.



Naled

Naled showed a clear trend of decreasing AChE activity with increasing concentration (Fig. 10). A total of 5 concentrations (n = 8 fish per concentration) were tested. Fish at the highest exposure concentration (60 µg/l) were clearly lethargic, a symptom of cholinergic poisoning. The calculated EC_{50} value is 7.848 µg/l. Closed circles are means ± standard error. The solid line is the regression of the data, dashed lines show the 95% confidence interval of the regression.

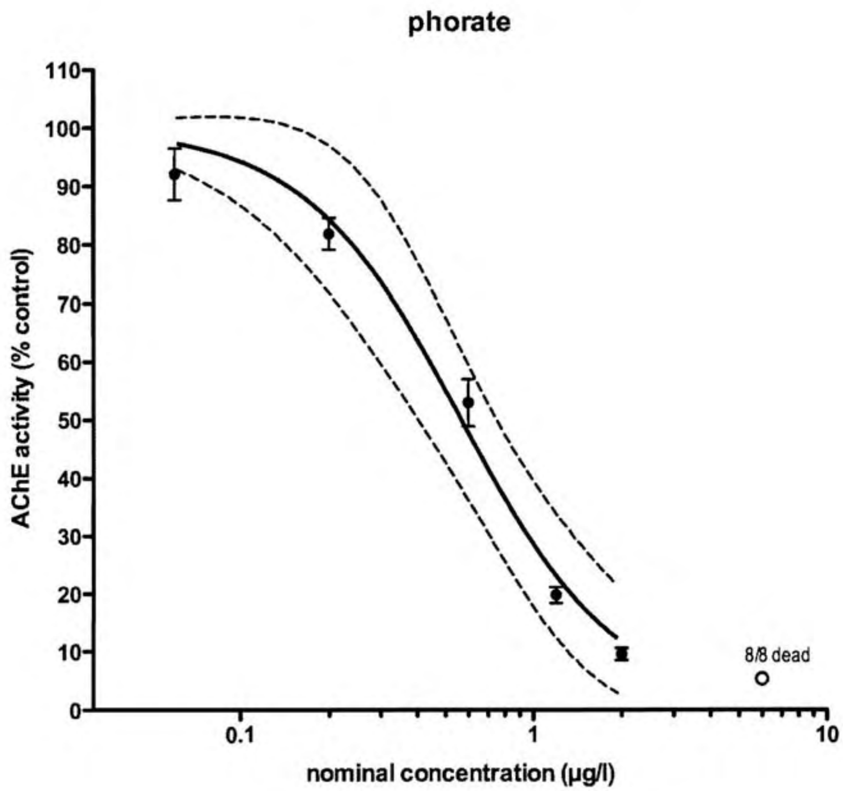
Figure 10. Brain AChE activity in juvenile coho following naled exposure.



Phorate

Phorate showed a clear trend of decreasing AChE activity with increasing concentration (Fig. 11). A total of 6 concentrations (n = 8 fish per concentration) were tested. Fish at the two highest exposures (2 and 6 µg/l) showed clear signs of cholinergic poisoning including lethargy and loss of orientation. Fish at 6 µg/l all died before the end of the 96-hour exposure period. The calculated EC_{50} value is 0.57 µg/l, making it the second most toxic pesticide tested. Closed circles are means ± standard error. The solid line is the regression of the data, dashed lines show the 95% confidence interval of the regression.

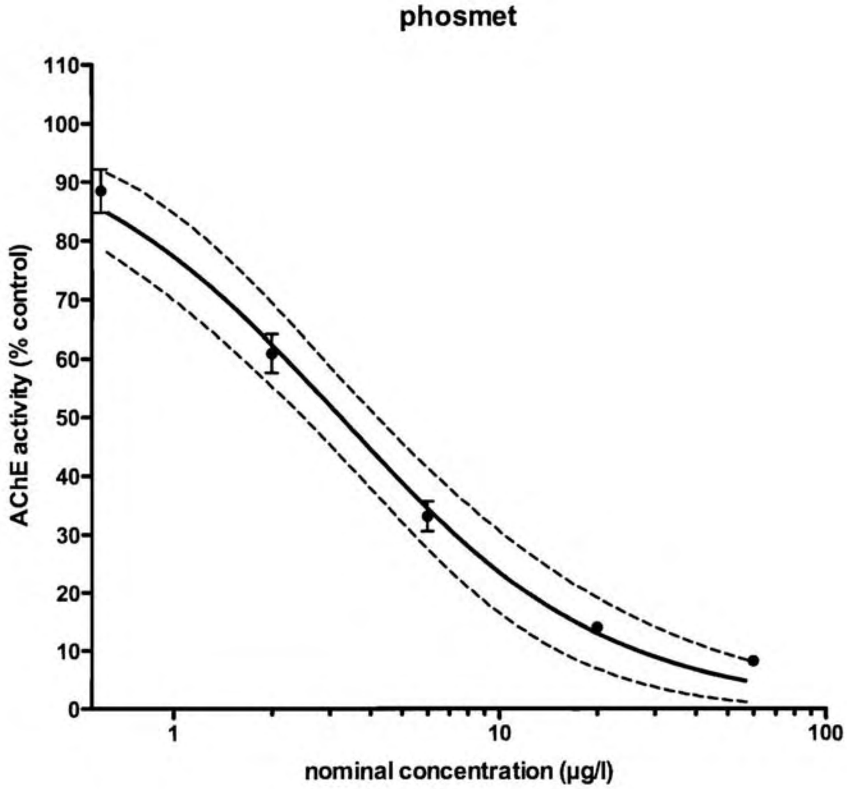
Figure 11. Brain AChE activity in juvenile coho following phorate exposure.



Phosmet

Phosmet showed a clear trend of decreasing AChE activity with increasing concentration (Fig. 12). A total of 5 concentrations (n = 8 fish per concentration) were tested. Fish in the highest exposure (60 µg/l) were lethargic, a symptom of cholinergic poisoning. The calculated EC_{50} value is 3.25 µg/l. Closed circles are means ± standard error. The solid line is the regression of the data, dashed lines show the 95% confidence interval of the regression.

Figure 12. Brain AChE activity in juvenile coho following phosmet exposure.



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12-30-09

The Combined Influence of Temperature and Pesticides on the Brain AChE Activity of Juvenile Coho Salmon

Introduction

Salmon populations in the Pacific Northwest are faced with many environmental stressors that may impact their health, behavior, and ultimately their survival. One such stressor is exposure to organophosphate (OP) and carbamate (CB) pesticides. These two groups of pesticides are widely used on agricultural, residential, urban, and public lands throughout the Northwest, and as a result are commonly found in salmon habitats. These chemicals are rarely detected in isolation, and are more commonly found in freshwater habitats as complex mixtures of multiple chemicals (Gilliom, 2007). OP and CB pesticides inhibit acetylcholinesterase (AChE), an enzyme critical for nerve transmission in animals including mammals, insects, and fish. Inhibition of AChE has been shown to impair swimming and feeding behavior in juvenile coho salmon (Sandahl et al., 2005), and can ultimately lead to death (Fulton and Key, 2001).

Another common stressor in salmonid habitats is elevated water temperature during summer months. Juvenile salmon require cool water and are susceptible to stress from elevated water temperatures, especially species such as coho that spend months to years in streams and rivers before migrating to the ocean. Juvenile salmon have a preferred temperature range of about 12 – 14 °C, and may avoid water above 15 – 18 °C (Madej et al., 2006). Elevated temperatures can adversely affect the growth, distribution, abundance, and survival of coho. Furthermore, coho could be more sensitive to pesticides at elevated temperatures due to elevated metabolic processes such as biotransformation (Heugens et al., 2001).

This paper reports the results of a pilot experiment that was conducted for the Protected Resources Division of NOAA Fisheries in support of a biological opinion. We exposed juvenile coho to 3 different pesticide treatments each at 4 temperatures, measured the resulting AChE activity in brain tissue, and looked for changes in toxicity due to temperature.

Methods

Fish

Juvenile coho were obtained from the University of Washington Hatchery in March 2008. Fish were transported to Washington State University's Puyallup Campus and housed there until experiments were conducted in November 2008. Fish were held in recirculating tanks filled with chilled, dechlorinated city water (temperature 12 °C, pH 6.5-7.0, hardness as $CaCO_3$ 120 ppm, alkalinity 40-80 mg/l, and dissolved oxygen 75-90 %). Water quality was monitored daily. Fish were fed commercial fish pellets daily, and exposed to a 12 hour light-dark cycle. Fish used in experiments (n = 160) averaged ($\pm$ sd) 7.7 $\pm$ 0.7 cm in length and 5.5 $\pm$ 4.6 g in weight.

Pesticide exposures

Fish (n = 8) were exposed to pesticides at their respective EC_{50} concentrations, or the concentration estimated from previous experiments to produce a 50% decrease in AChE activity. Fish were exposed to the single EC_{50} concentrations of chlorpyrifos and carbaryl, as well as to a binary mixture of diazinon and malathion at the predicted cumulative EC_{50} concentration (i.e., each pesticide at one-half its EC_{50}), at 4 temperatures (8, 12, 14 and 15.5

°C). Fish were not acclimated to the different temperature regimes before exposure. Nominal exposure concentrations are shown in Table 1. Water-only and methanol control fish (n = 8) were also exposed at each temperature. Pesticide-containing stock solutions were prepared in methanol. Chlorpyrifos and carbaryl stocks were added in 1 ml aliquots, while diazinon and malathion stocks were added in 500 μl aliquots, to 25 l of hatchery water in 30-l glass aquaria. Final carrier concentrations were 0.004%. For each pesticide treatment at each temperature regime, 8 fish were exposed for 96 hrs with static water renewals conducted every 24 hours. Following exposures, fish were terminally anaesthetized in MS-222 (tricaine methanesulfonate) until gill activity ceased. Brain tissue was removed, put into a plastic microcentrifuge tube, and placed on ice until storage at -80 °C for subsequent AChE analysis.

Table 1. Nominal EC_{50} pesticide concentrations used in exposures.

Pesticide	Concentration (μg/l)
Diazinon + Malathion	1.27 + 0.65
Chlorpyrifos	2.02
Carbofuran	58.42

AChE enzyme assay

Determination of AChE enzyme activity followed previously published methods (Laetz et al., 2009). Briefly, whole brains were homogenized at 50 mg/ml in 0.1 M sodium phosphate buffer with 0.1% Triton X-100. Homogenates were centrifuged, and 15 μl of the supernatant were combined with 685 μl of 10 mM phosphate buffered saline, 50 μl of 6 mM DTNB (5,5'-dithio-bis(2-nitrobenzoic acid)) and 30 μl of 100 mM acetylthiocholine iodide. Triplicate 200 μl samples were transferred to a 96-well plate, and the change in absorbance at 412 nm was measured at 12 s intervals for 5 min at 25 °C. AChE activity was quantified as mOD/min/g tissue and reported as a percentage of the enzyme activity for carrier (methanol) controls.

Results

The AChE activities from all exposures and treatments are shown in Table 2. The AChE activities of coho brains in both the water-only and methanol controls at each of the four treatment temperatures were not significantly different (ANOVA, $p > 0.005$). Therefore, AChE activities from all pesticide treatments at any given temperature were normalized to percent of methanol control at that temperature (Figure 1). Exposure to carbofuran significantly decreased AChE activities from control levels (n = 8, ANOVA, Tukey-Kramer post-hoc, $p < 0.0001$), but showed no trend with temperature (linear regression, $p = 0.41$). Fish exposed to chlorpyrifos showed about 75 - 90 % of methanol control AChE activity. However, there was no apparent trend with temperature (linear regression, $p = 0.67$). Fish exposed to the binary mixture of diazinon and malathion had AChE activities ranging from 98 % to 41 %, and showed a strong trend of declining AChE activity with increasing temperature (linear regression, $p < 0.0001$) (Figure 2).

Table 2. AChE activity (mOD/min/g tissue) in coho brains (n = 8). Values are reported as mean ± standard error.

	8 °C	12 °C	14 °C	15.5 °C
Water control	98 ± 3.1	104 ± 2.5	100 ± 3.2	103 ± 2.3
Methanol control	96 ± 2.3	109 ± 2.8	104 ± 2.6	106 ± 2.3
Carbofuran	34 ± 0.7	36 ± 0.8	38 ± 1.4	38 ± 1.0
Chlorpyrifos	88 ± 2.2	81 ± 2.4	90 ± 3.5	101 ± 5.2
Diazinon+Malathion	98 ± 0.8	78 ± 7.9	66 ± 6.6	41 ± 12.4

Figure 1. AChE activities, normalized to % of methanol control for the respective temperature, in all exposure groups and temperature treatments. Values are means (± standard error) of 8 fish.

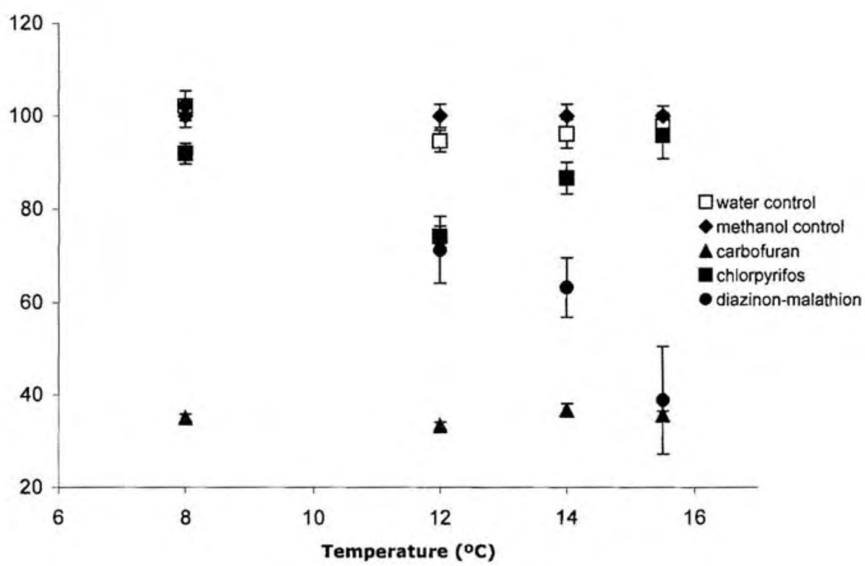

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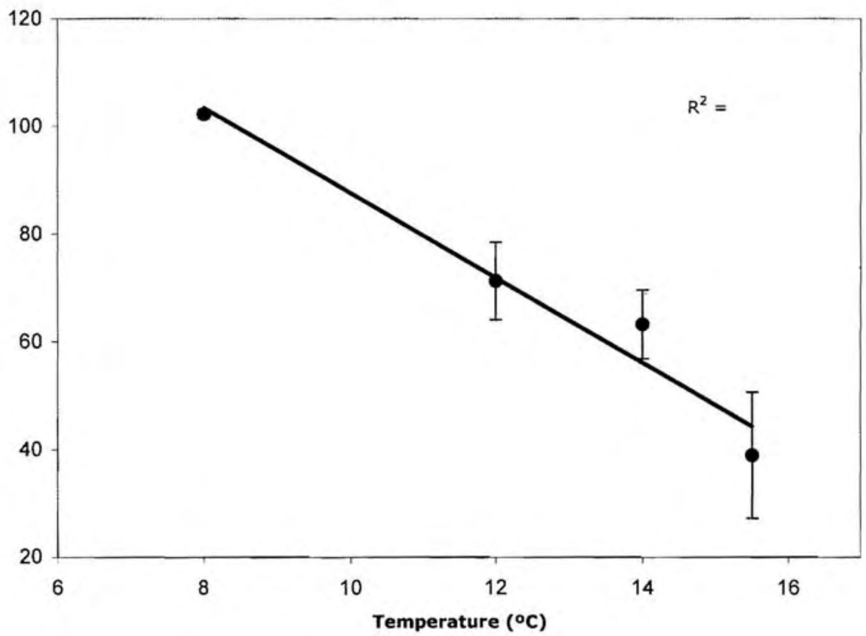
Figure 2. Brain AChE activity (normalized to % of methanol control) decreases with increasing water temperature in juvenile coho exposed to the diazinon-malathion mixture. Values are means (± standard error) of 8 fish.



Conclusions

In this experiment, differences were observed between the carbamate and organophosphate groups of pesticides. In particular, carbofuran (the carbamate) toxicity did not show a trend with increasing temperature. This was expected based on the matabolism – independent activity of carbamates. As a fish's metabolism increases with increasing ambient temperature, rates of biotransformation are also expected to increase. However, because biotransformation is not necessary for carbamates, the total amount of pesticide present to inhibit AChE would not be expected to increase with increasing temperature.

Chlorpyrifos toxicity also did not increase with rising temperature. This was unexpected, since temperature-dependent toxicity of other organophosphates has been observed in fish (Osterauer and Kohler, 2008). However, the concentration of chlorpyrifos used in these exposures was based on dose-response studies conducted previously at 12 °C with considerably smaller fish (mean ± sd of 4.9 ± 1.0 cm and 1.3 ± 0.9 g; Laetz et al., 2009). This large size difference could have shifted the dose-response curve, thereby changing the EC_{50} (the concentration that produces a 50 % decrease in AChE activity). Therefore, there may not have been enough of a toxic effect from the concentration of chlorpyrifos used in these exposures to observe any trend with temperature.

While the results of this pilot experiment didn't demonstrate an effect of temperature on the toxicity of single pesticides, it did indicate that increased temperature enhances the synergistic toxicity of a mixture of two current-use organophosphate pesticides (diazinon and malathion). Previous studies have shown that multiple stressors can lead to synergistic, or greater than expected, pesticide toxicity (Relyea and Mills, 2001; Osterauer and Kohler, 2008). While the exact mechanisms of synergy are unknown, the observation of enhanced toxicity at environmentally relevant concentrations and temperatures is a novel finding worthy of future study. Based on this preliminary study, cumulative stressors like temperature and pesticides may affect the health, fitness, and viability of salmon populations.

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Appendix 7: Co-occurrence Analysis for Integration and Synthesis

Our species viability assessment considers the spatial, temporal, and biological overlap of ESA-listed species with the stressors of the action. Where there is co-occurrence, salmonids may be exposed to and affected by the a.i. and its associated stressors.

Because pesticides are registered for specific uses, we determine where specific portions of the proposed action may be carried out based on the type of use. National Land Cover Database (NLCD) land use categories were used as a surrogate for use sites: cultivated crops or hay/pasture for a specific crop or crops; developed areas for residential and urban uses, pest control, and disease vector control; and managed forests for forestry applications. While cropping patterns may shift or lands may become fallow over a longer period of time, the NLCD dataset is the most relevant method of estimating exposure. As we cannot determine where a certain crop will be cultivated, we assume that any pesticide registered for use on an agricultural crop could be applied in an area defined as agricultural land use.

We used the GIS program ArcView to overlay the NLCD data on ESUs/DPSs range and distribution shapefiles to determine areas of potential co-occurrence of pesticide use and ESA-listed salmon. Species range shapefiles were developed by NMFS Northwest Regional Office. These files exist for every ESU and consist of polygons encompassing the hydrologic units where that species can be found. In some cases, these polygons include areas that are not currently occupied, but are accessible and are part of the historic range of the species. We also assessed distribution data for each ESU/DPS. Distribution files were developed by the Northwest and Southwest regional offices in the process of identifying and designating critical habitat for 19 species in 2005.

The remaining ESUs/DPSs did not have existing distribution layers. They were created for this consultation by overlaying datasets from other sources with the NMFS range polygons. The data is largely presence/absence data collected by governmental agencies and university researchers. Information on Idaho, Oregon, and Washington species was compiled and presented by Streamnet (www.streamnet.org) while California data came from CalFish (www.calfish.org). Streams where fish were present within the range polygon were exported to a new distribution file. This method was used to create files for Snake River Fall-run Chinook salmon, Snake River Spring-run Chinook salmon, Sacramento River Winter-run Chinook salmon, Snake River sockeye salmon, Ozette Lake Sockeye salmon, Lower Columbia River Coho salmon, Southern Oregon Northern California Coho salmon, Central California Coast Coho salmon, and Puget Sound Steelhead salmon.

For all ESUs/DPSs, a 2.5 km “buffer” was created on each side of salmonid aquatic habitat. This distance was selected by the team as it is large enough to account for discrepancies between GIS layers due to channel alteration / migration, but not so large that it would encompass the entire range of an ESU. We expect pesticide applications in these areas are most relevant to concentrations experienced by salmonids via pesticide runoff and drift. If land in any of the relevant NLCD categories was within the buffer we determined that salmon and the a.i. could co-occur. Over the 15-year duration of the proposed action, we expect some individuals within each of the listed ESUs/DPSs in the action area will be exposed to these a.i.s during their life cycle. Given that these pesticides can be used across the landscape, and that temporal and spatial distribution of listed salmonids are both highly variable, we expect exposure is also highly variable among both individuals and populations of listed salmon.

Once co-occurrence is determined via GIS for each a.i., we evaluated the spatial and temporal extent of potential exposure for the ESU/DPS, given the life history of the species. In many cases, fish may be in the system for prolonged periods of time, and there is generally no specific seasonal restriction on application of pesticides. Additionally, species are made up of “runs” which spawn at different times of the year. Thus, the spatial and biological overlap is of greater importance in analyzing this action than the temporal component.

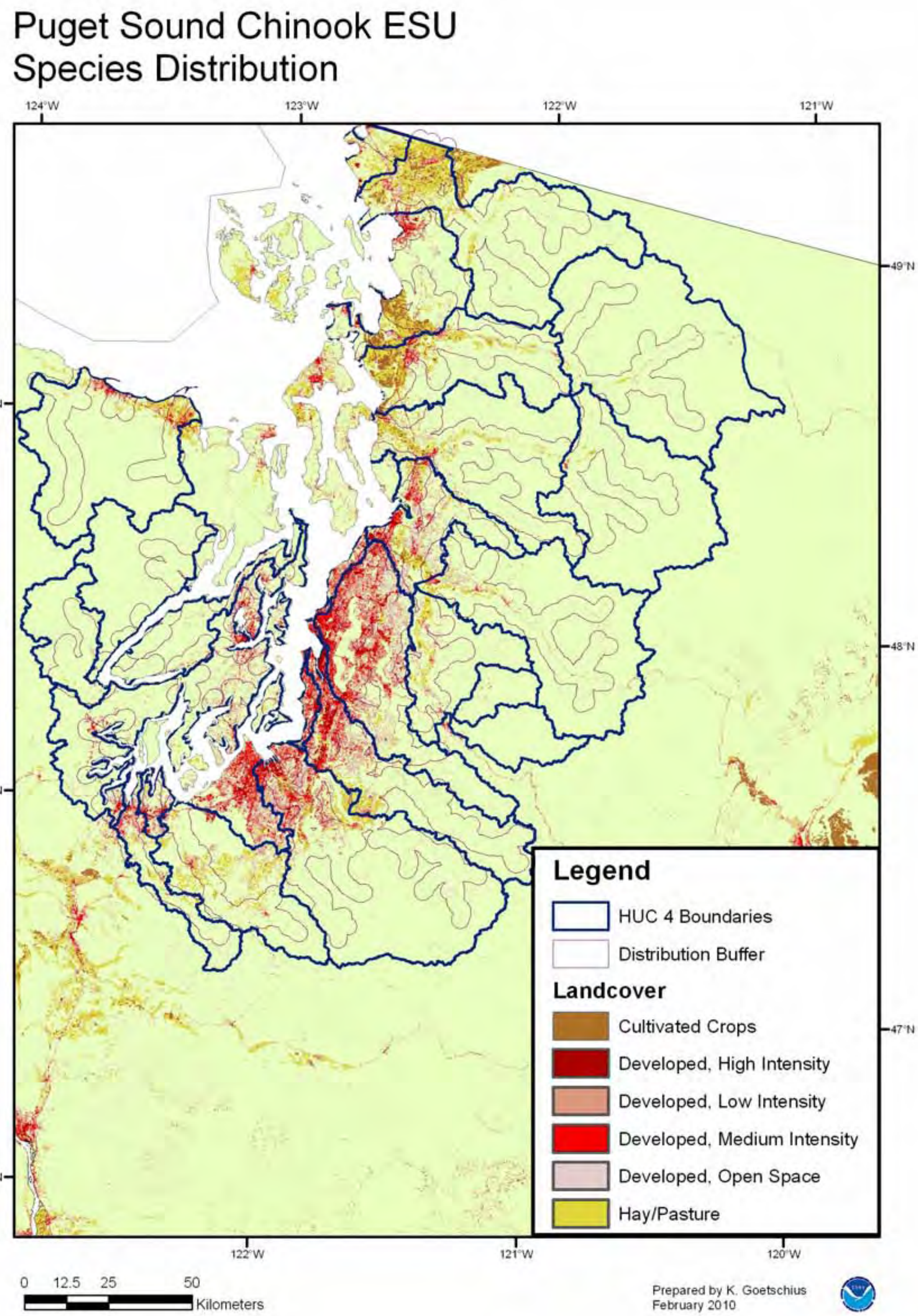
We further considered the existing environmental mixtures, seasonally elevated water temperatures, and other factors which influence the survival of the species, such as loss of habitat features, hydropower and water management conditions, and invasive species or predators.

To illustrate the co-occurrence analysis process, this appendix includes two maps for each ESU/DPS. The first map shows the range of the ESU with each HUC 4 outlined in blue, the 2.5 km buffer in burgundy, and relevant categories from the NLCD land use layer. This map aided in the

Species analyses. The second map was used in the critical habitat analysis. For 19 of the species, conservation values have been assigned to the HUC 5 level units. In Idaho, Oregon, and Washington, these units are referred to as watersheds, while California uses the term "hydrological sub-area" or HSA. The Critical Habitat maps show either, (a) all designated HUC5s and their conservation values, or (b) the species map with the buffer removed. The exceptions to this are Snake River Fall-Run Chinook and Ozette Lake Sockeye, as they cover such small areas, and the two species for which critical habitat has not been designated (Columbia River Coho and Puget Sound Steelhead). These four species each only have one map.

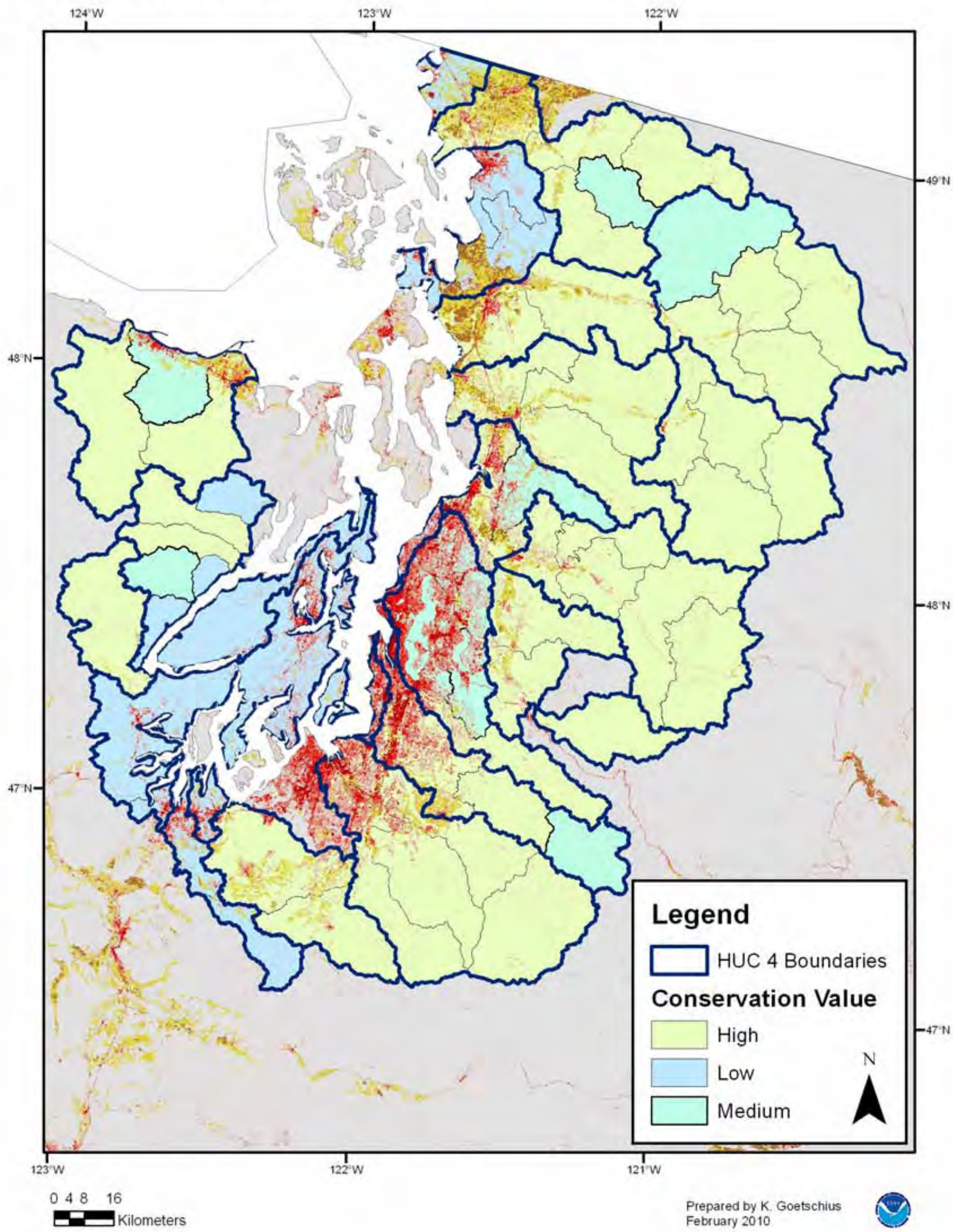
The following species have conservation values assigned by HUC5:

1. Puget Sound Chinook
2. Lower Columbia River Chinook
3. Upper Columbia River Spring Run Chinook
4. Upper Willamette River Chinook
5. California Coastal Chinook
6. Central Valley Spring Run Chinook
7. Columbia River Chum
8. Hood Canal Chum
9. Oregon Coast Coho
10. Lower Columbia River Steelhead
11. Middle Columbia River Steelhead
12. Upper Columbia River Steelhead
13. Upper Willamette River Steelhead
14. Snake River Steelhead
15. Northern California Steelhead
16. Central California Coast Steelhead
17. California Central Valley Steelhead
18. South-Central California Coast Steelhead
19. Southern California Steelhead

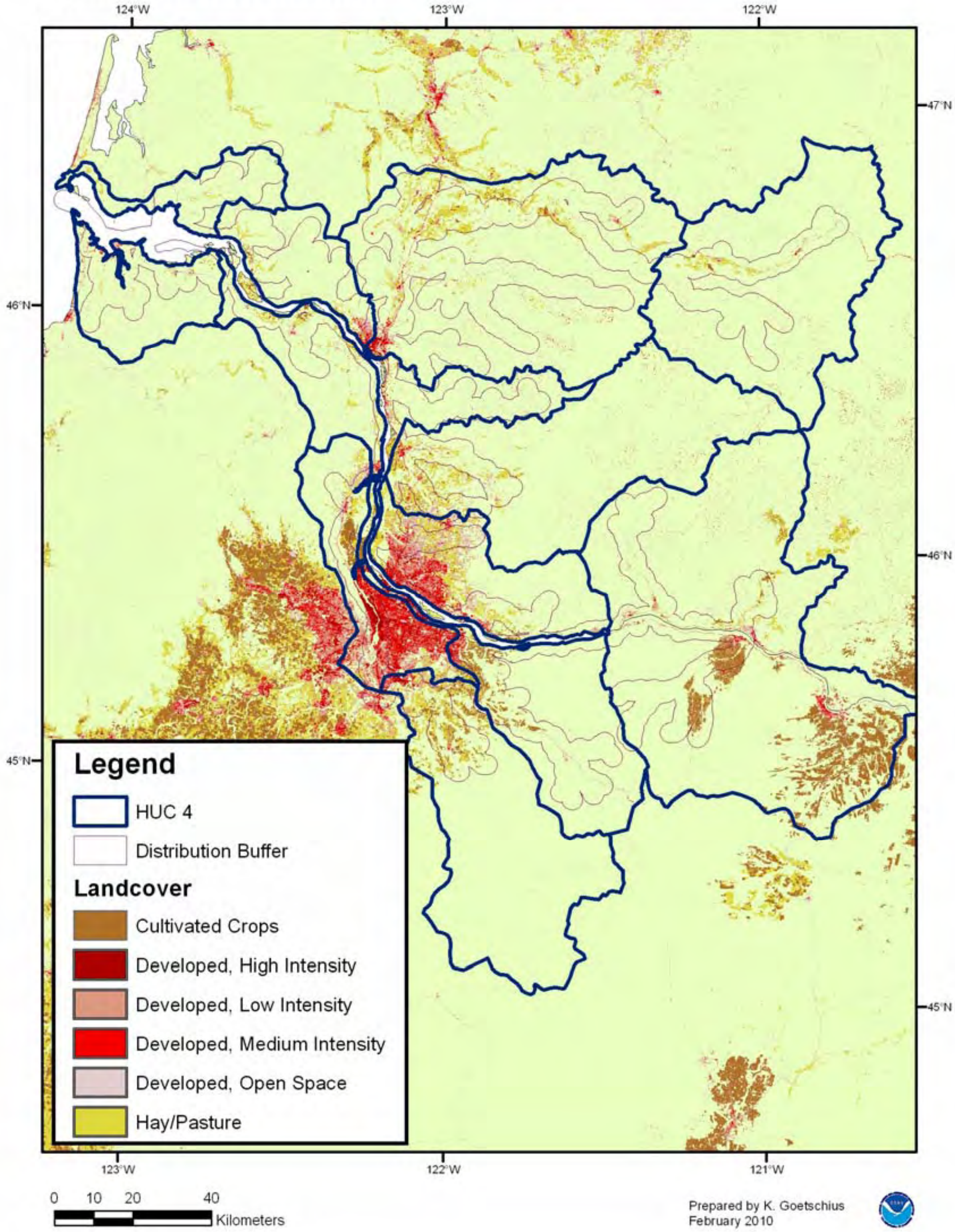


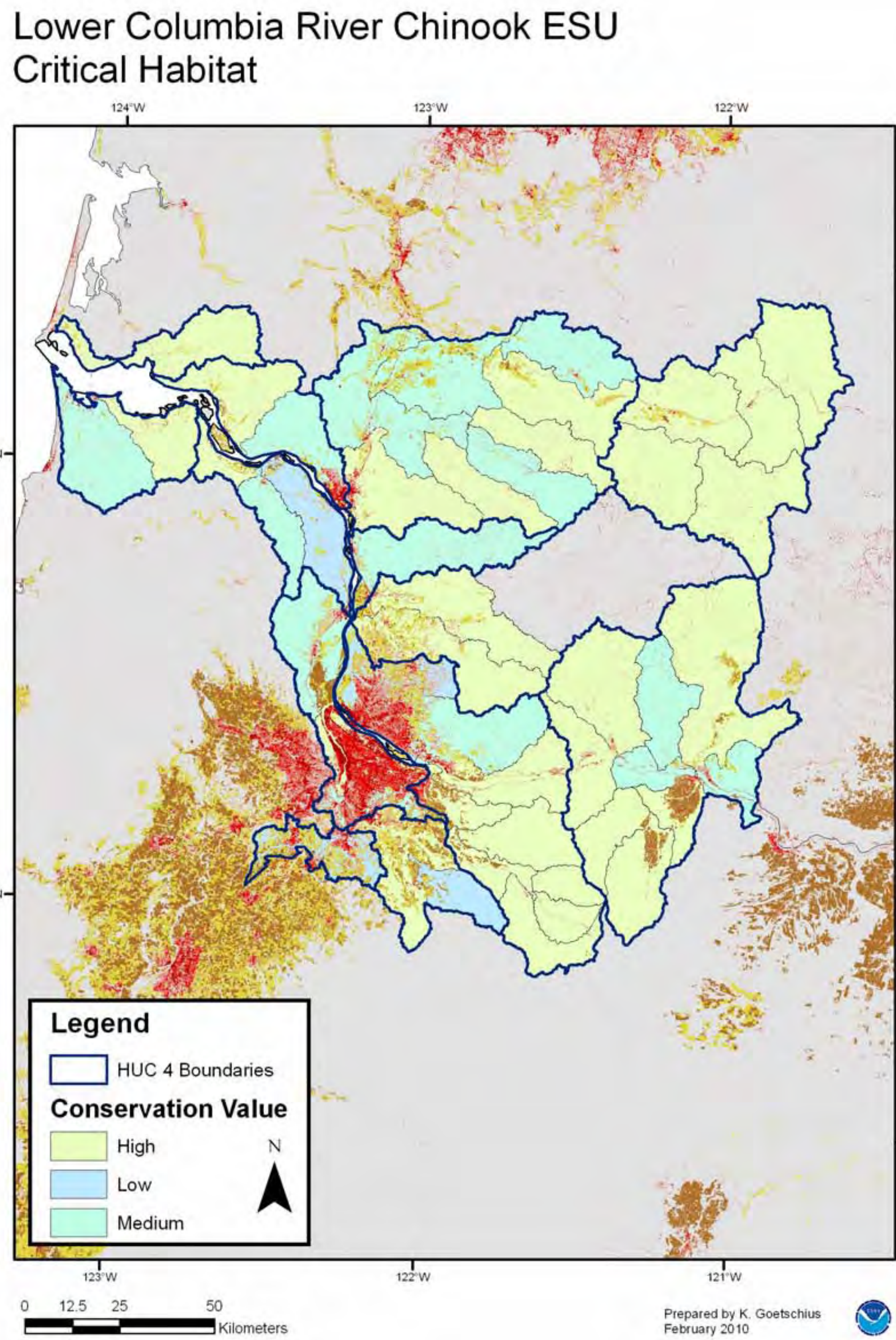
Puget Sound Chinook ESU
Species Distribution
124°W
123°W
122°W
121°W
49°N
48°N
47°N
46°N
122°W
121°W
120°W
48°N
47°N
Legend
HUC 4 Boundaries
Distribution Buffer
Landcover
Cultivated Crops
Developed, High Intensity
Developed, Low Intensity
Developed, Medium Intensity
Developed, Open Space
Hay/Pasture
0 12.5 25 50
Kilometers
Prepared by K. Goetschius
February 2010

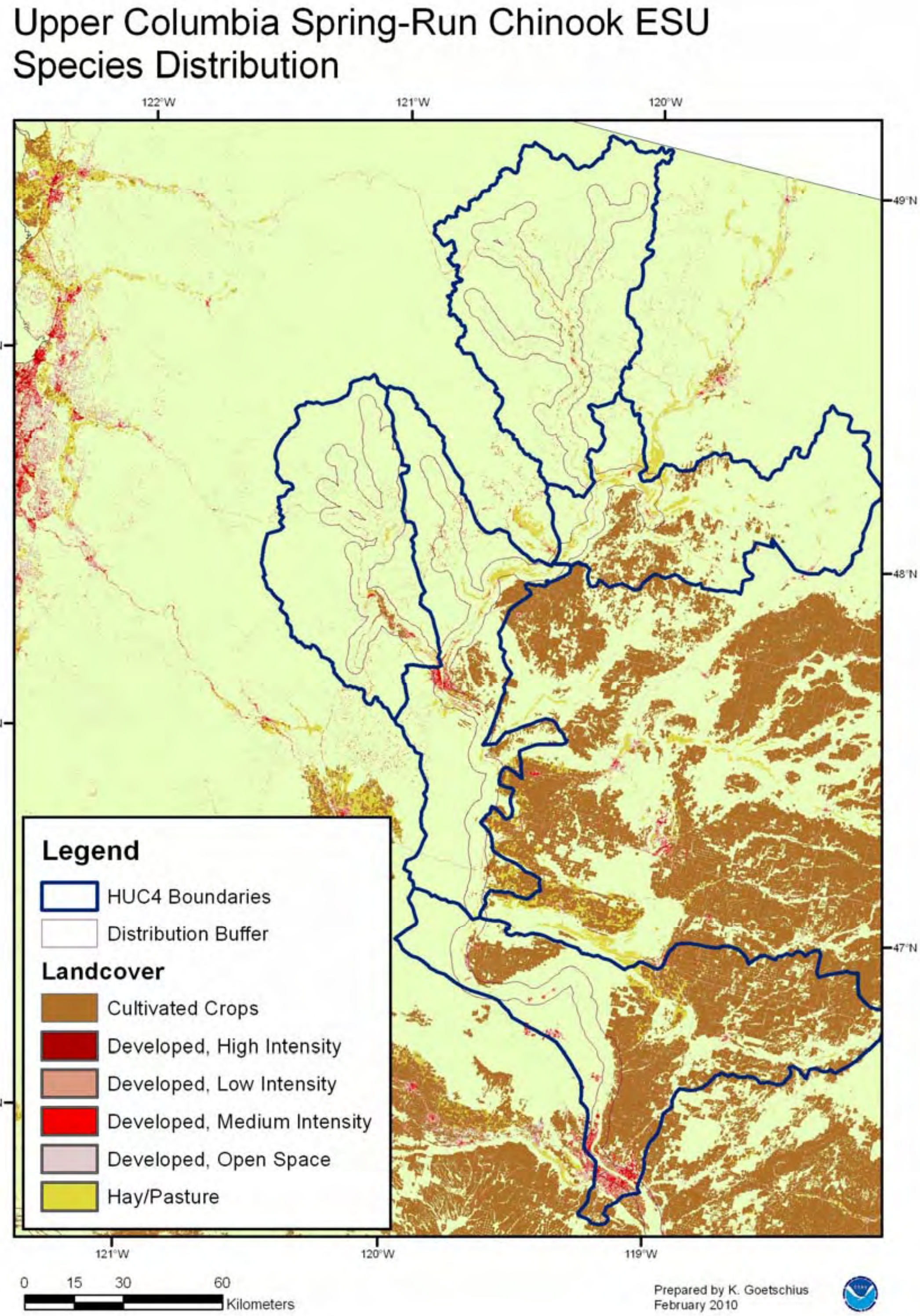
Puget Sound Chinook ESU
Critical Habitat



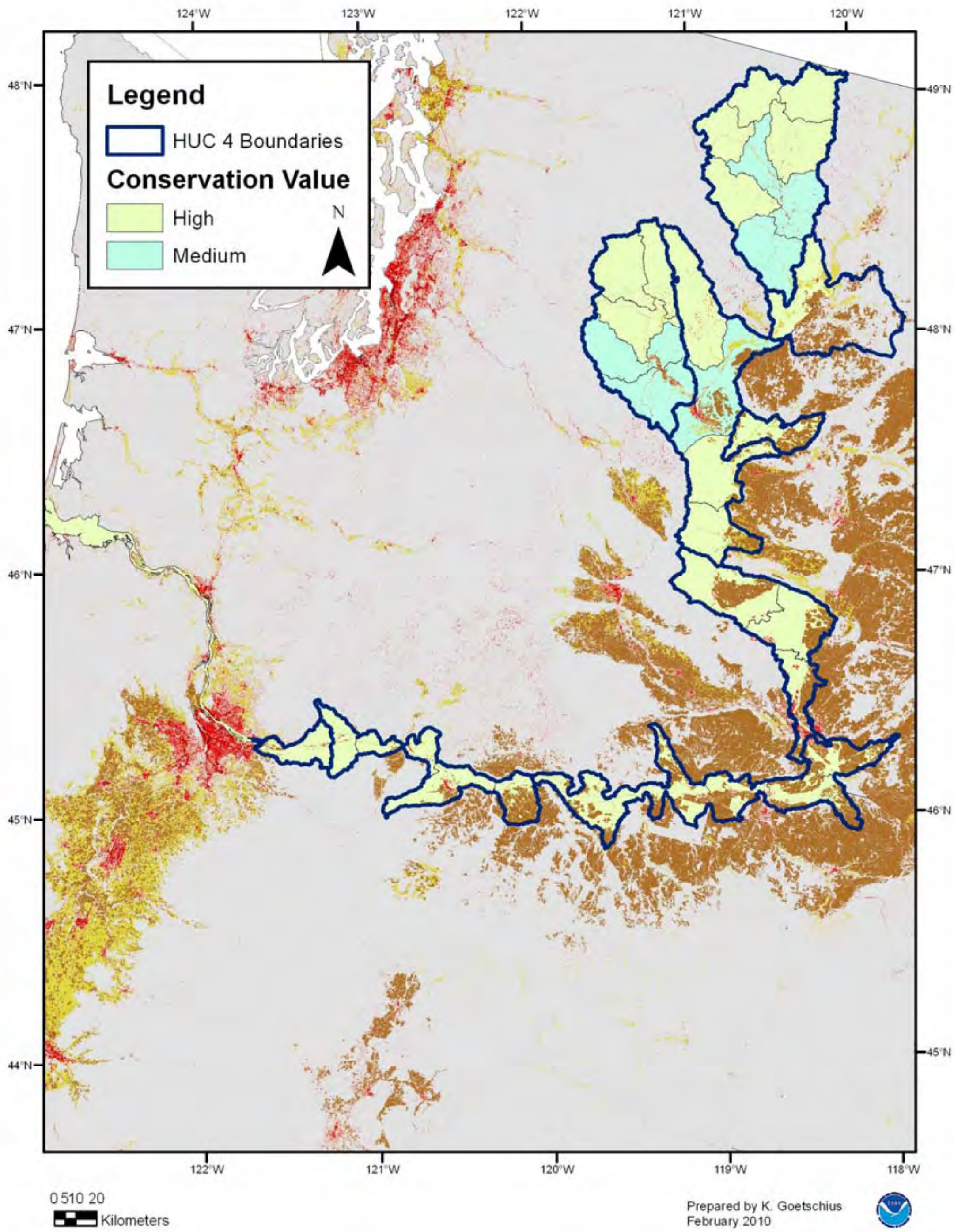
Lower Columbia River Chinook ESU
Species Distribution

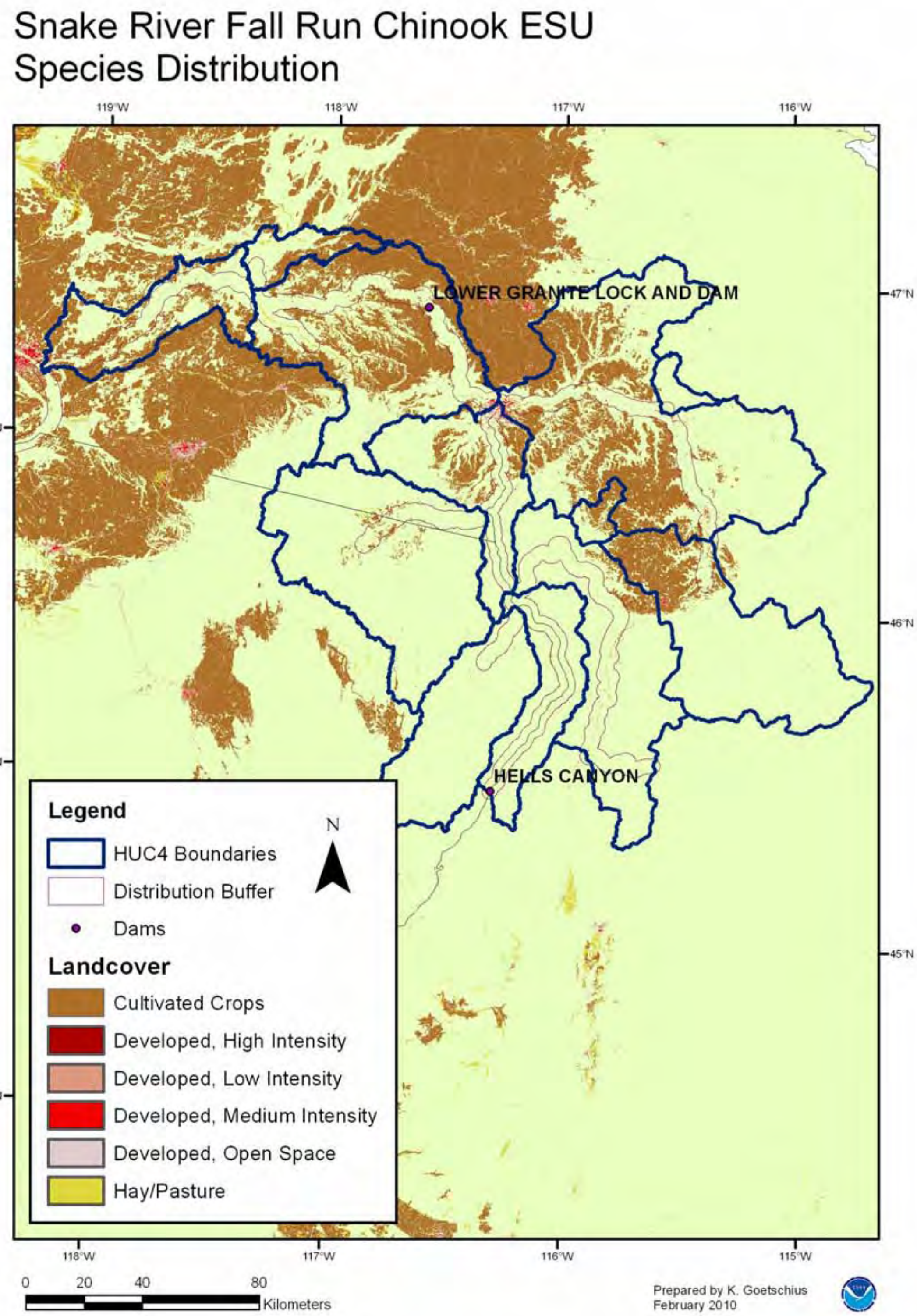



Lower Columbia River Chinook ESU
Critical Habitat
124°W
123°W
122°W
47°N
46°N
45°N
Legend
HUC 4 Boundaries
Conservation Value
High
Low
Medium
N
121°W
0 12.5 25 50
Kilometers
Prepared by K. Goetschius
February 2010

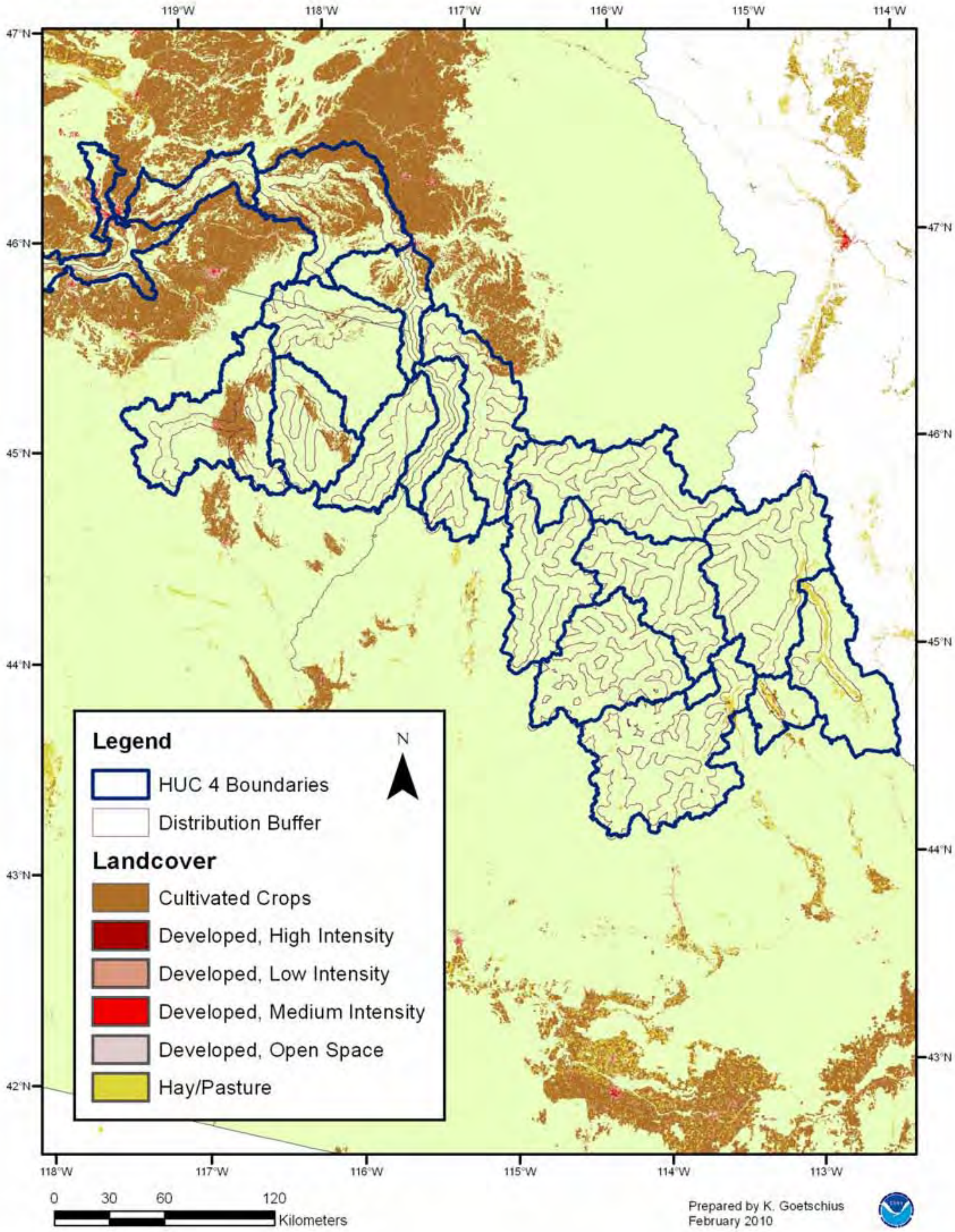

Upper Columbia Spring-Run Chinook ESU
Species Distribution
122°W
121°W
120°W
49°N
48°N
47°N
46°N
119°W
Legend
HUC4 Boundaries
Distribution Buffer
Landcover
Cultivated Crops
Developed, High Intensity
Developed, Low Intensity
Developed, Medium Intensity
Developed, Open Space
Hay/Pasture
0 15 30 60
Kilometers
Prepared by K. Goetschius
February 2010

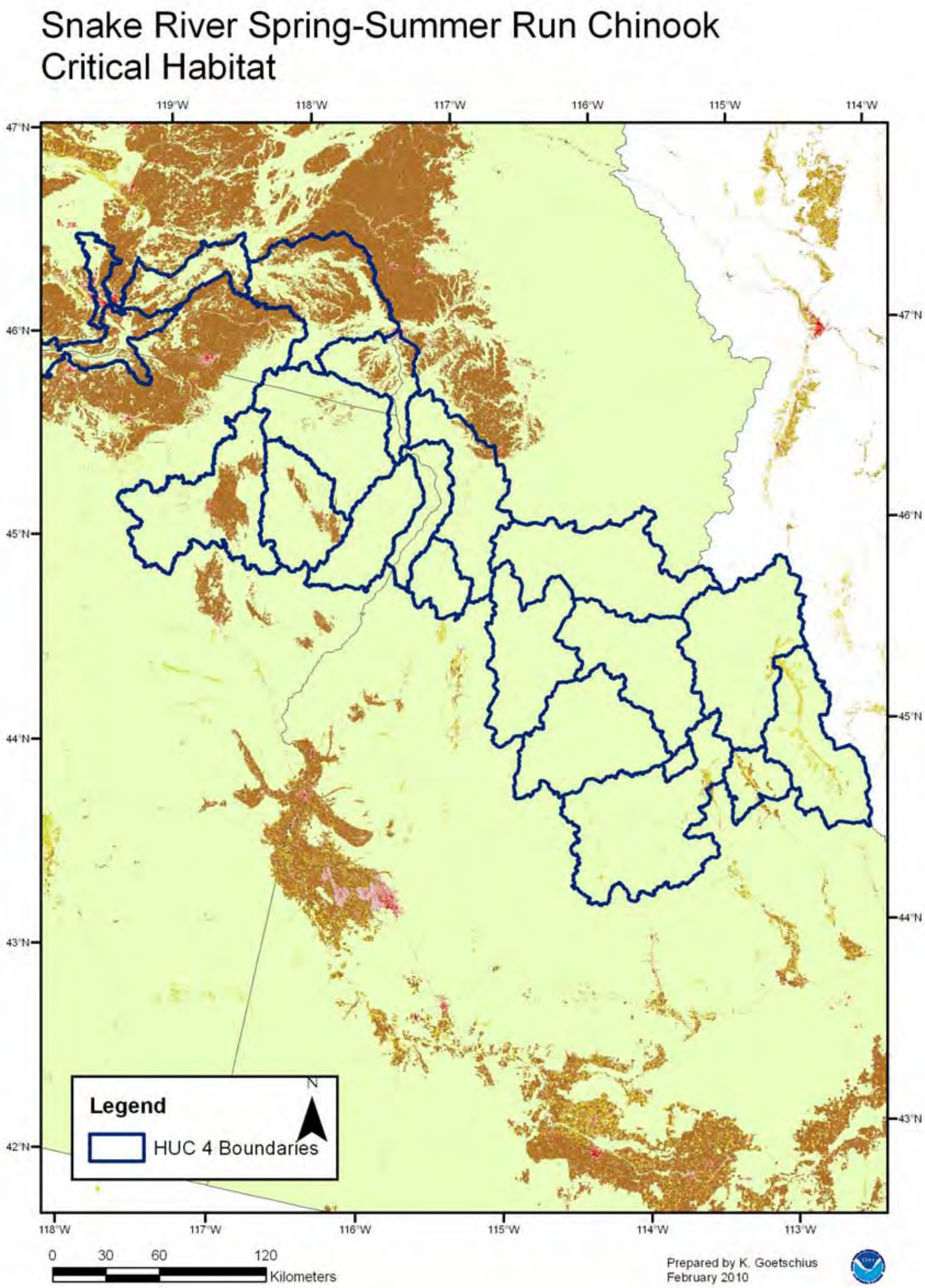
Upper Coulmbia River Chinook ESU
Critical Habitat



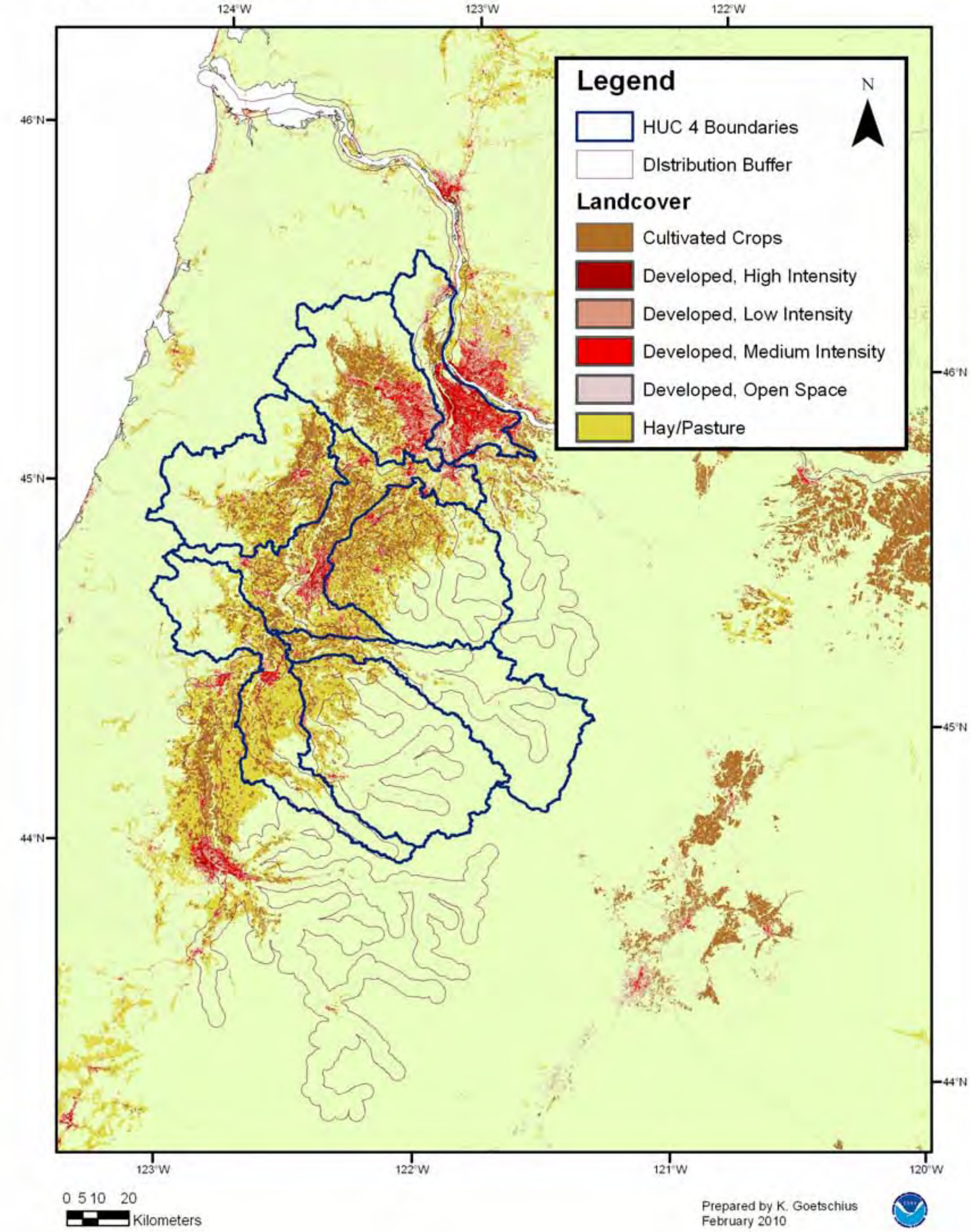

Snake River Fall Run Chinook ESU
Species Distribution
119°W
118°W
117°W
116°W
47°N
46°N
45°N
44°N
118°W
117°W
116°W
115°W
LOWER GRANITE LOCK AND DAM
HELLS CANYON
Legend
HUC4 Boundaries
Distribution Buffer
Dams
Landcover
Cultivated Crops
Developed, High Intensity
Developed, Low Intensity
Developed, Medium Intensity
Developed, Open Space
Hay/Pasture
N
0
20
40
80
Kilometers
Prepared by K. Goetschius
February 2010

Snake River Spring-Summer Run Chinook Species Distribution

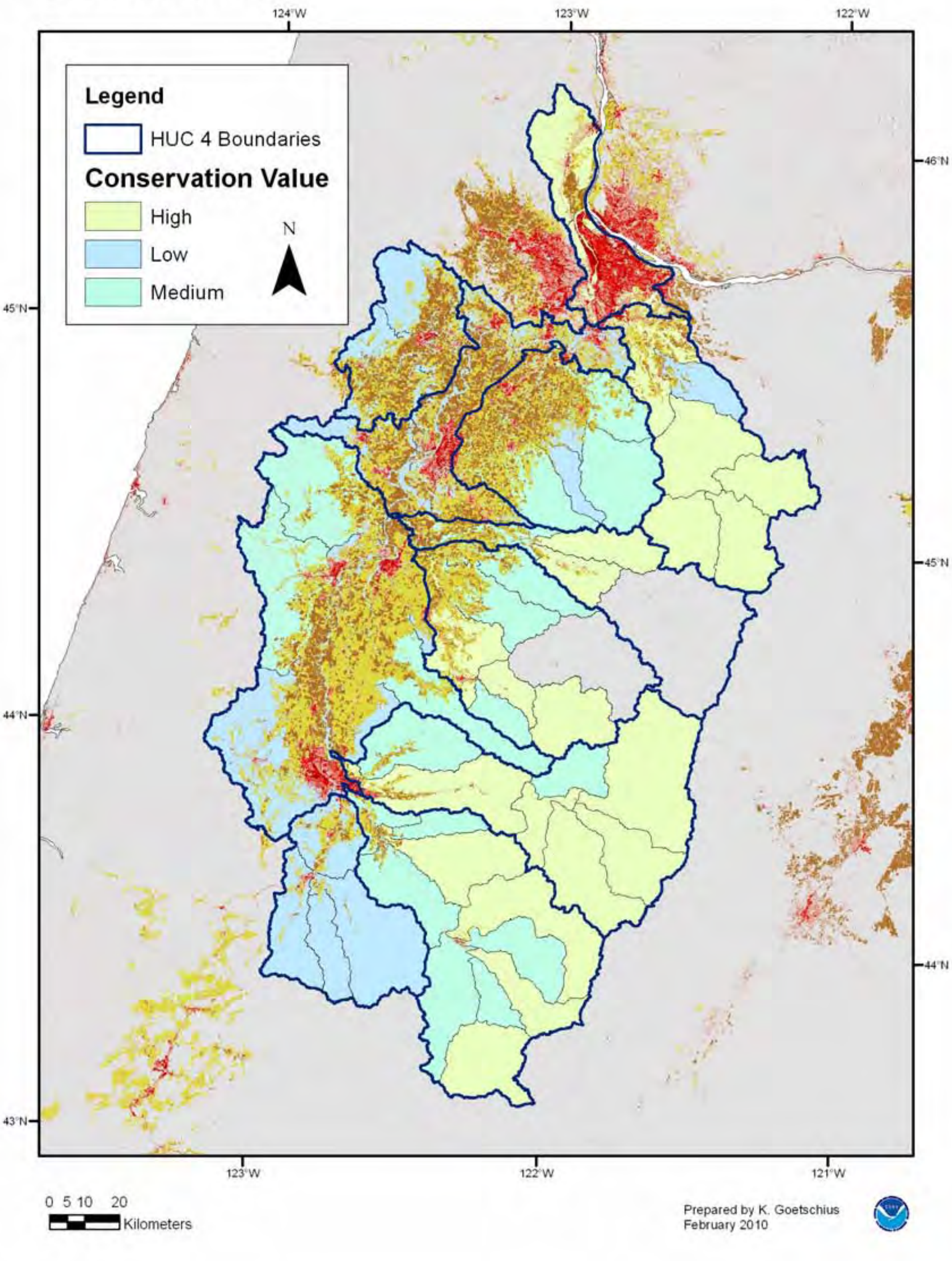


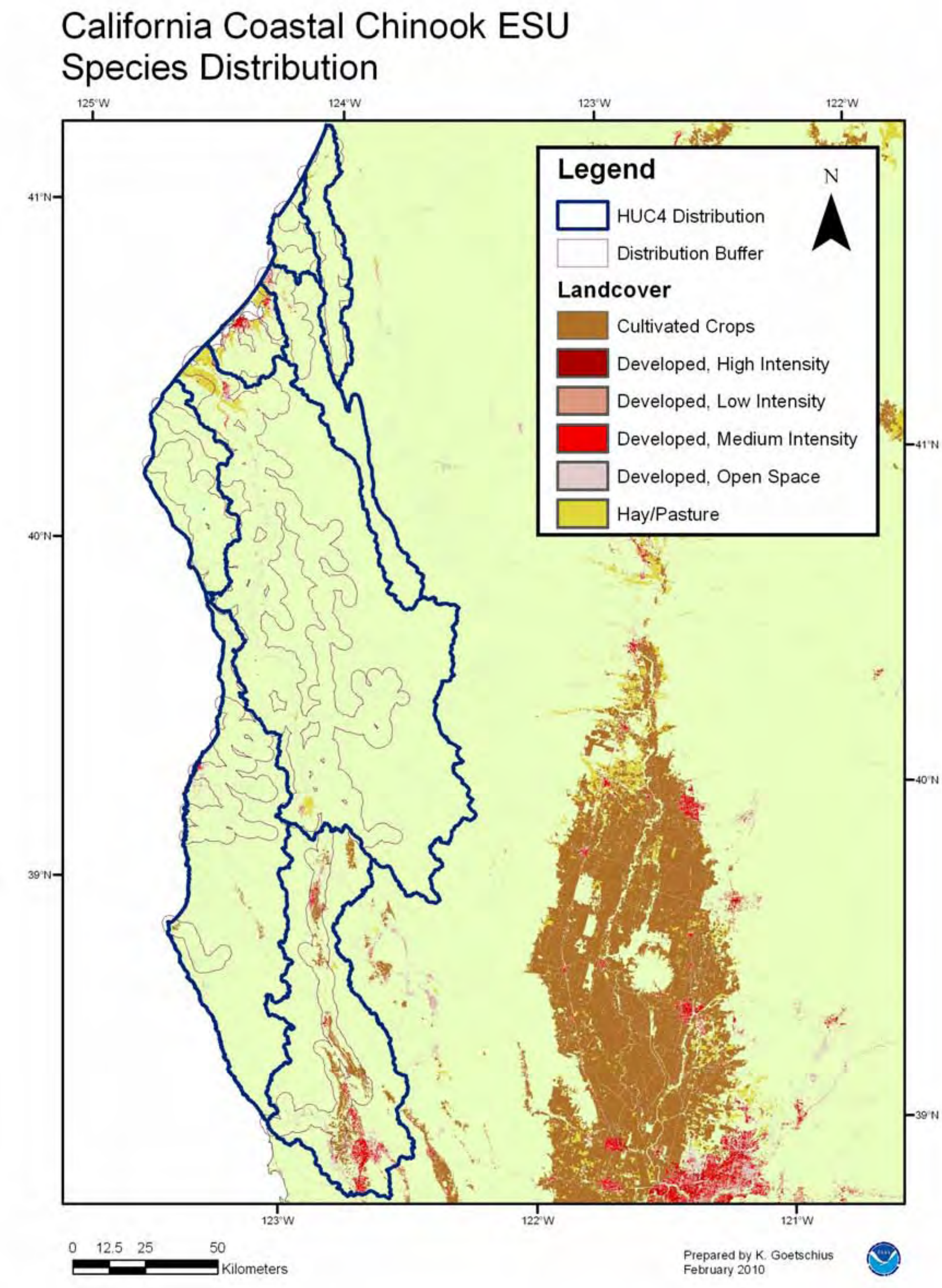

Snake River Spring-Summer Run Chinook
Critical Habitat
119°W
118°W
117°W
116°W
115°W
114°W
47°N
46°N
45°N
44°N
43°N
42°N
118°W
117°W
116°W
115°W
114°W
113°W
Legend
HUC 4 Boundaries
N
0 30 60 120
Kilometers
Prepared by K. Goetschius
February 2010

Upper Willamette River Chinook ESU
Species Distribution

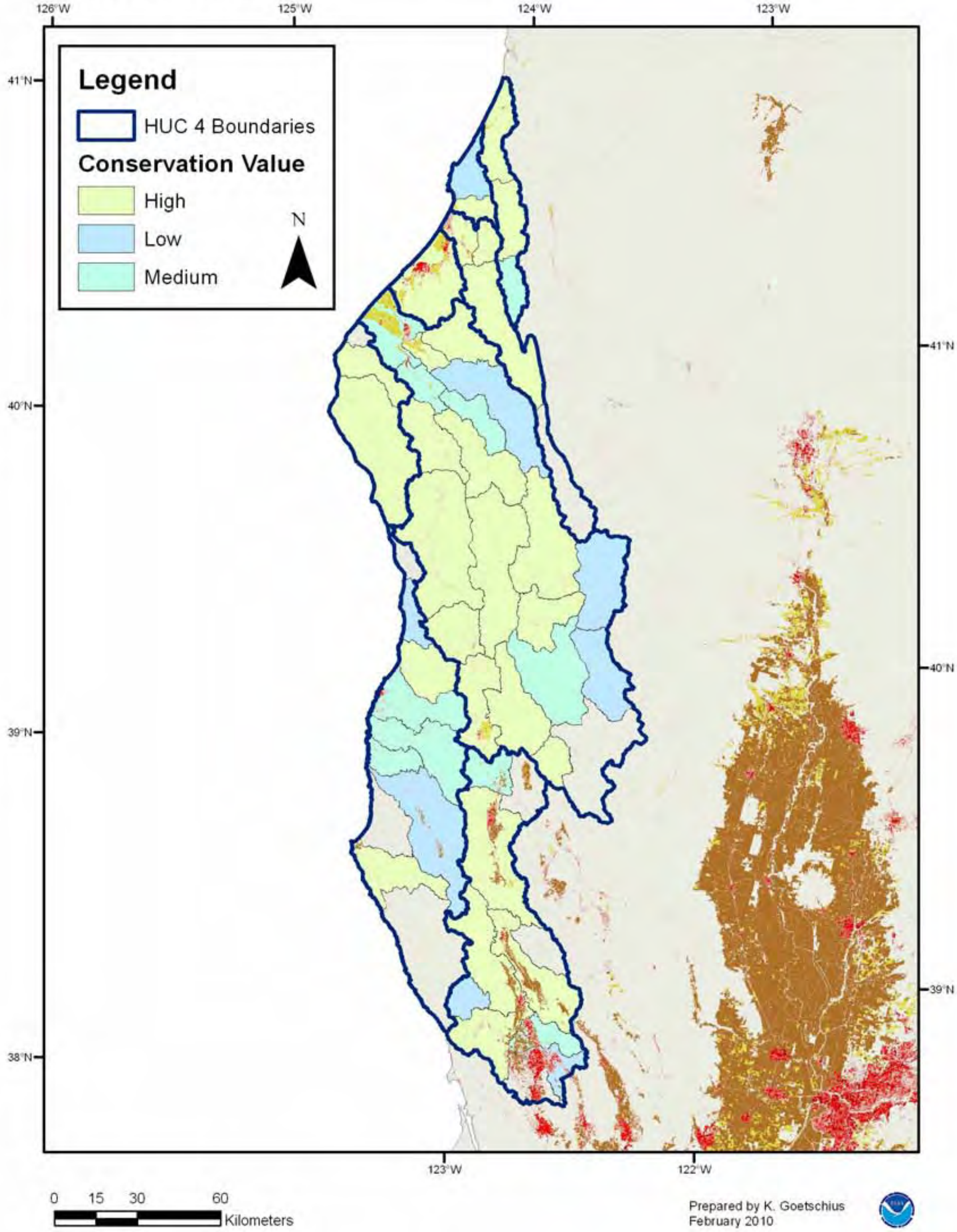


Upper Willamette River Chinook ESU
Critical Habitat

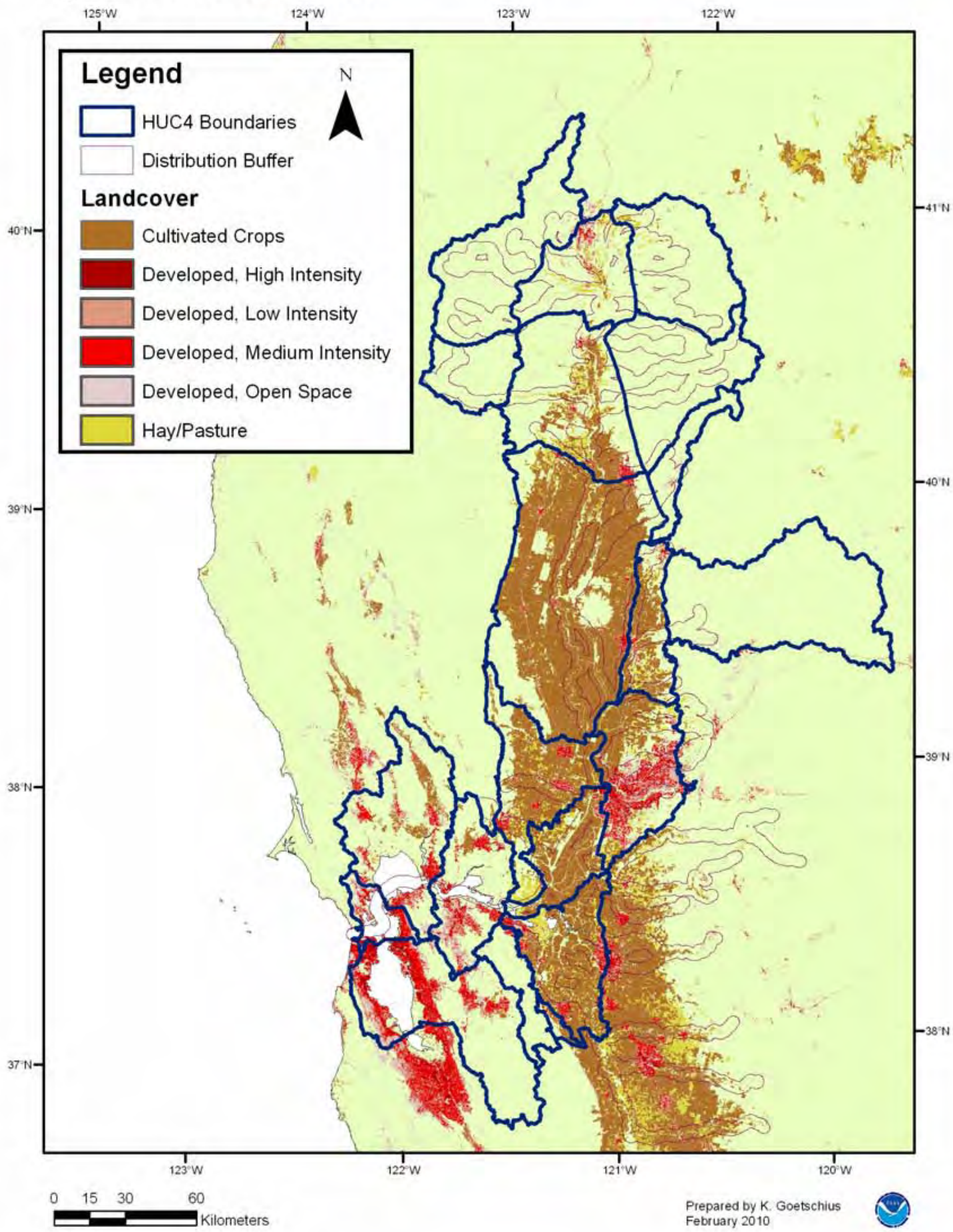



California Coastal Chinook ESU
Species Distribution
Legend
N
HUC4 Distribution
Distribution Buffer
Landcover
Cultivated Crops
Developed, High Intensity
Developed, Low Intensity
Developed, Medium Intensity
Developed, Open Space
Hay/Pasture
125°W
124°W
123°W
122°W
121°W
41°N
40°N
39°N
0 12.5 25 50 Kilometers
Prepared by K. Goetschius
February 2010

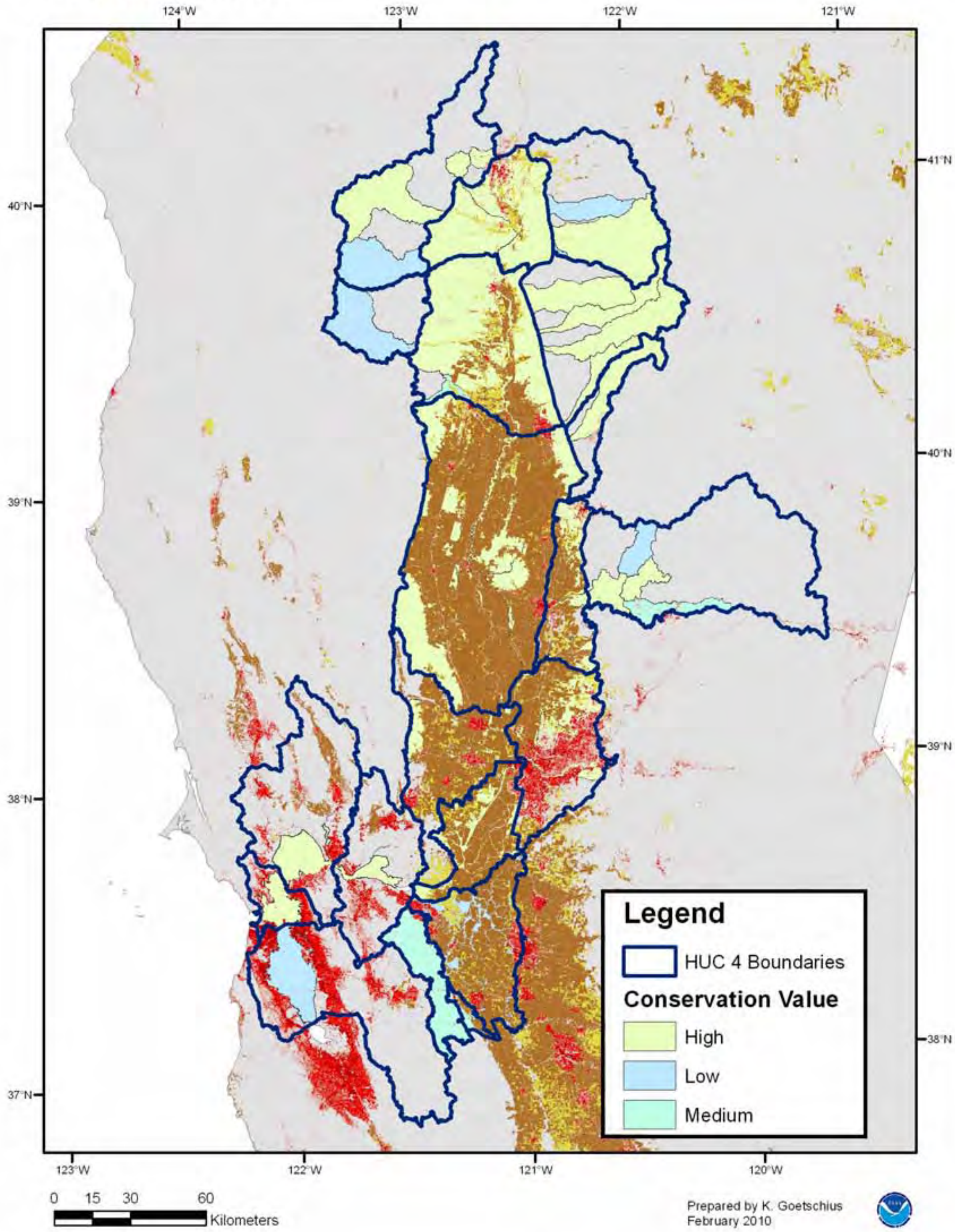
California Coastal Chinook ESU Critical Habitat



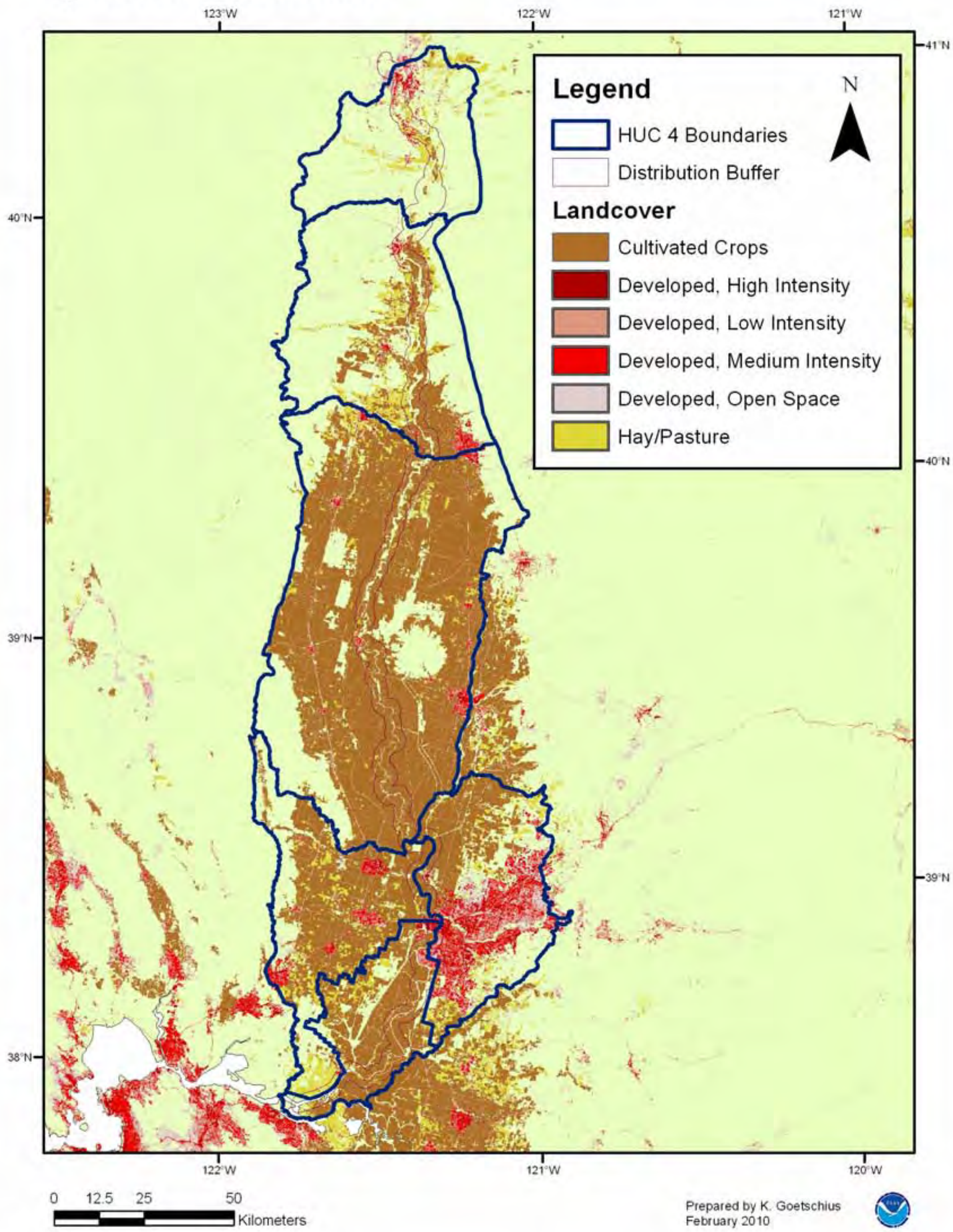
Central Valley Spring-Run Chinook ESU
Species Distribution



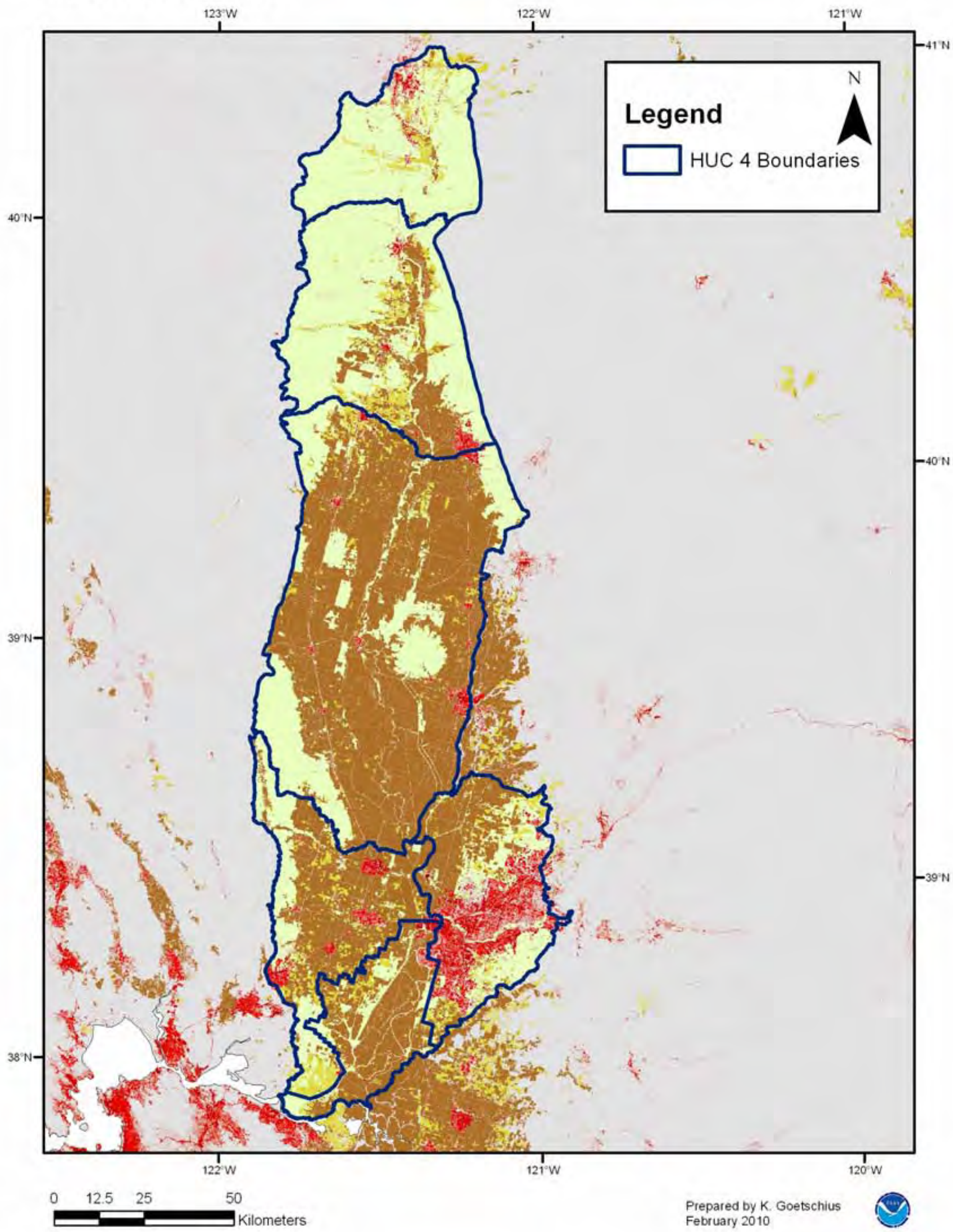
Central Valley Spring-Run Chinook ESU
Critical Habitat

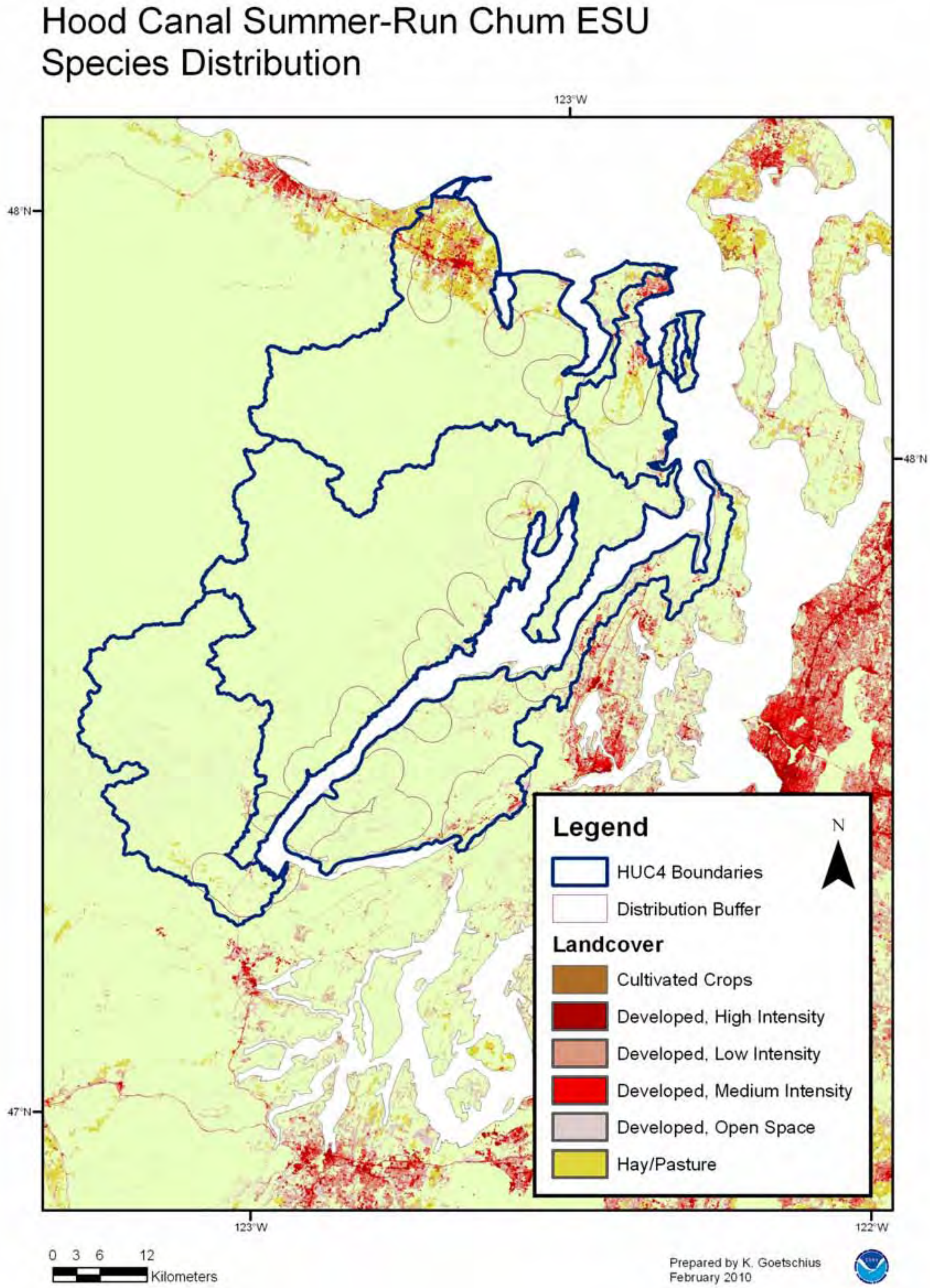


Sacramento River Winter Run Chinook ESU Species Distribution

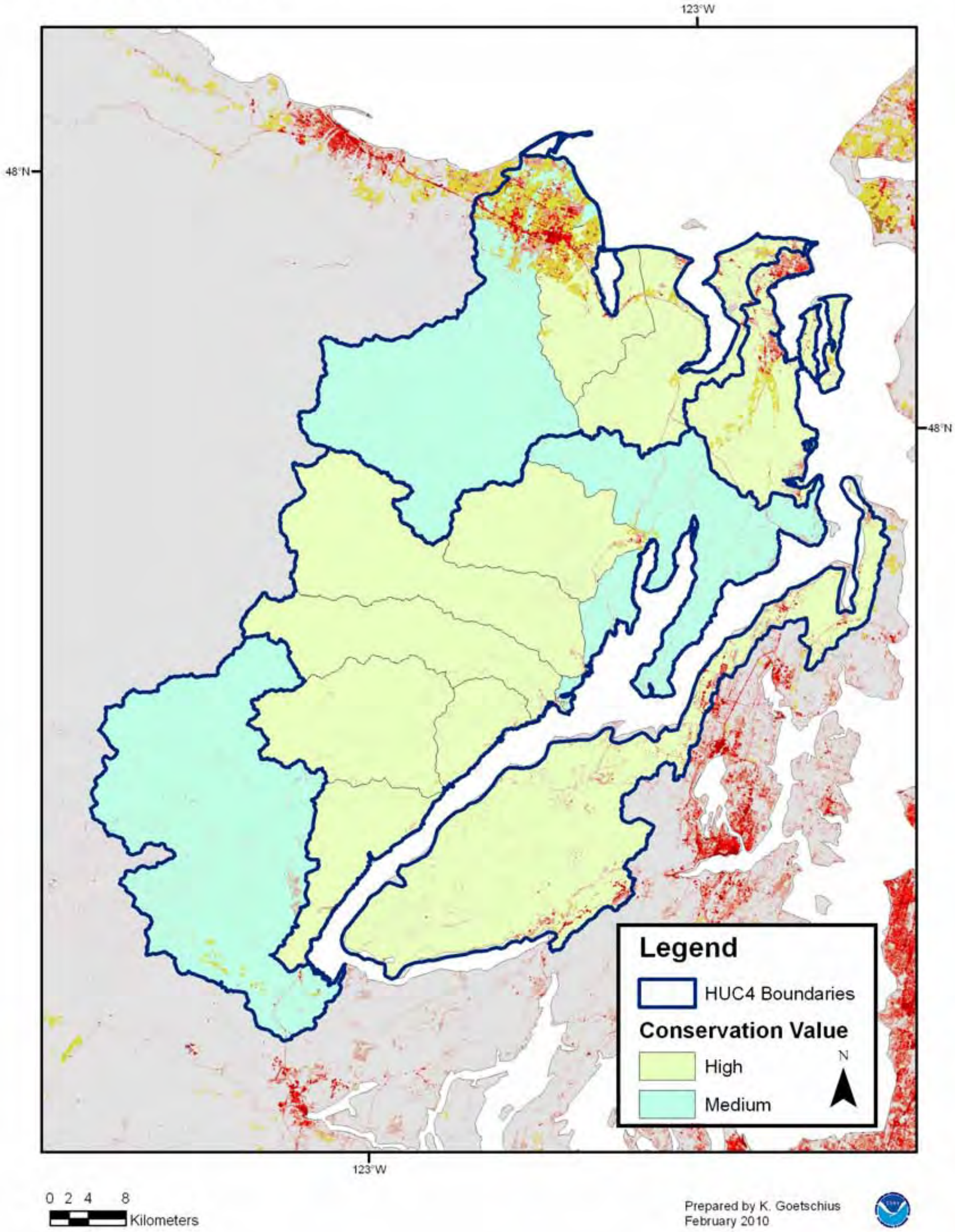


Sacramento River Winter Run Chinook ESU Critical Habitat

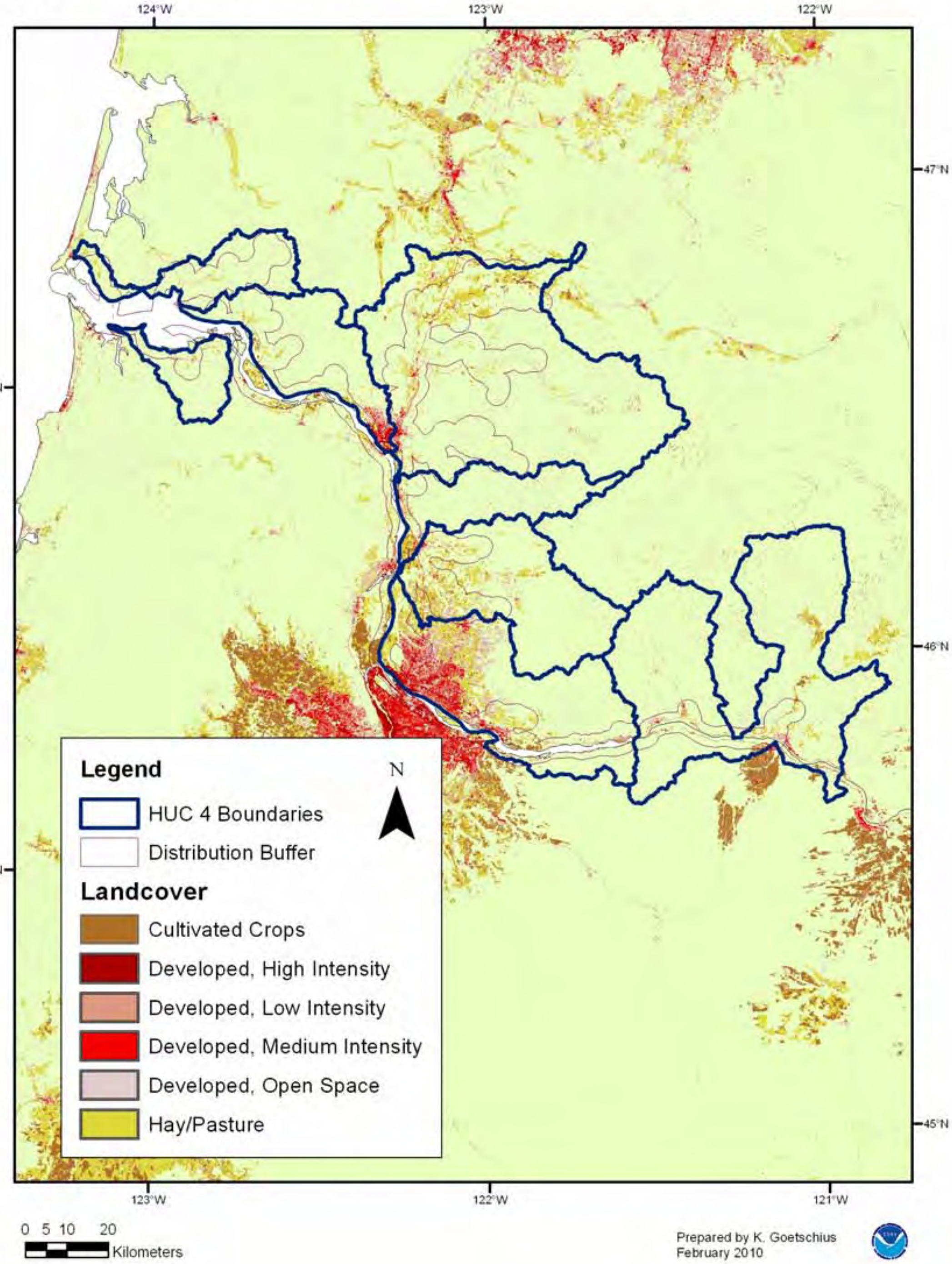



Hood Canal Summer-Run Chum ESU
Species Distribution
123°W
48°N
47°N
122°W
Legend
N
HUC4 Boundaries
Distribution Buffer
Landcover
Cultivated Crops
Developed, High Intensity
Developed, Low Intensity
Developed, Medium Intensity
Developed, Open Space
Hay/Pasture
0 3 6 12
Kilometers
Prepared by K. Goetschius
February 2010

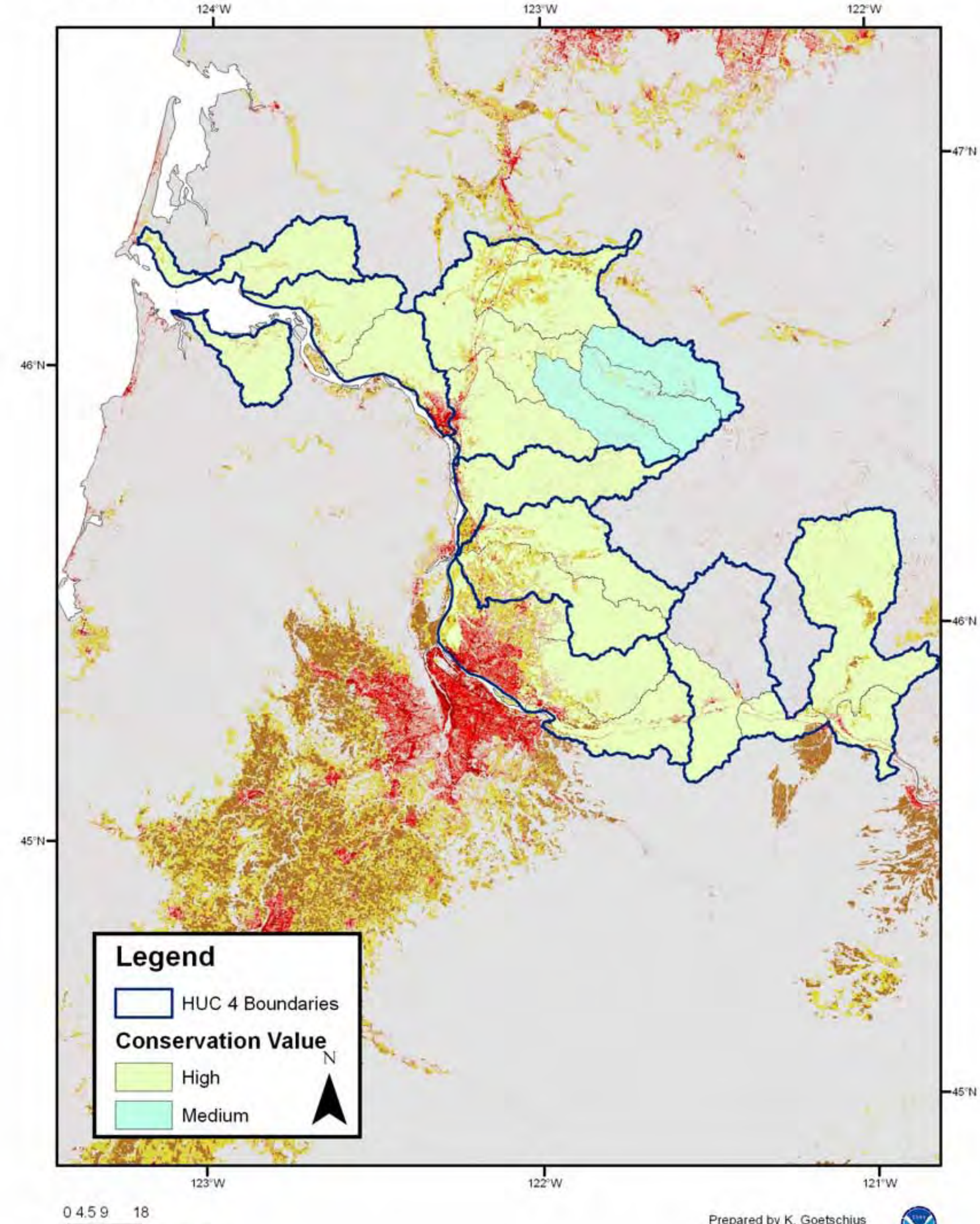
Hood Canal Summer-Run Chum ESU
Critical Habitat



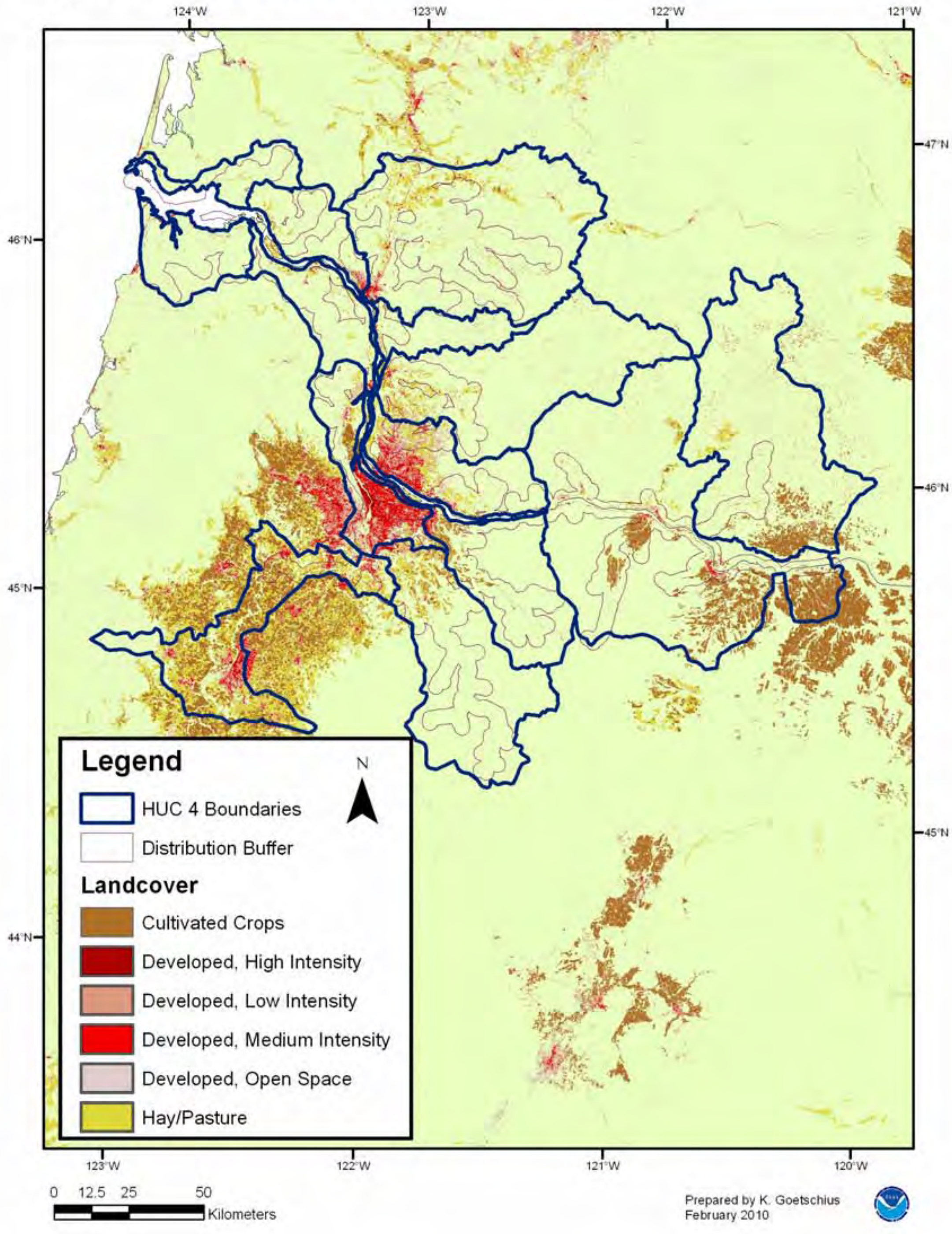
Columbia River Chum ESU
Species Distribution

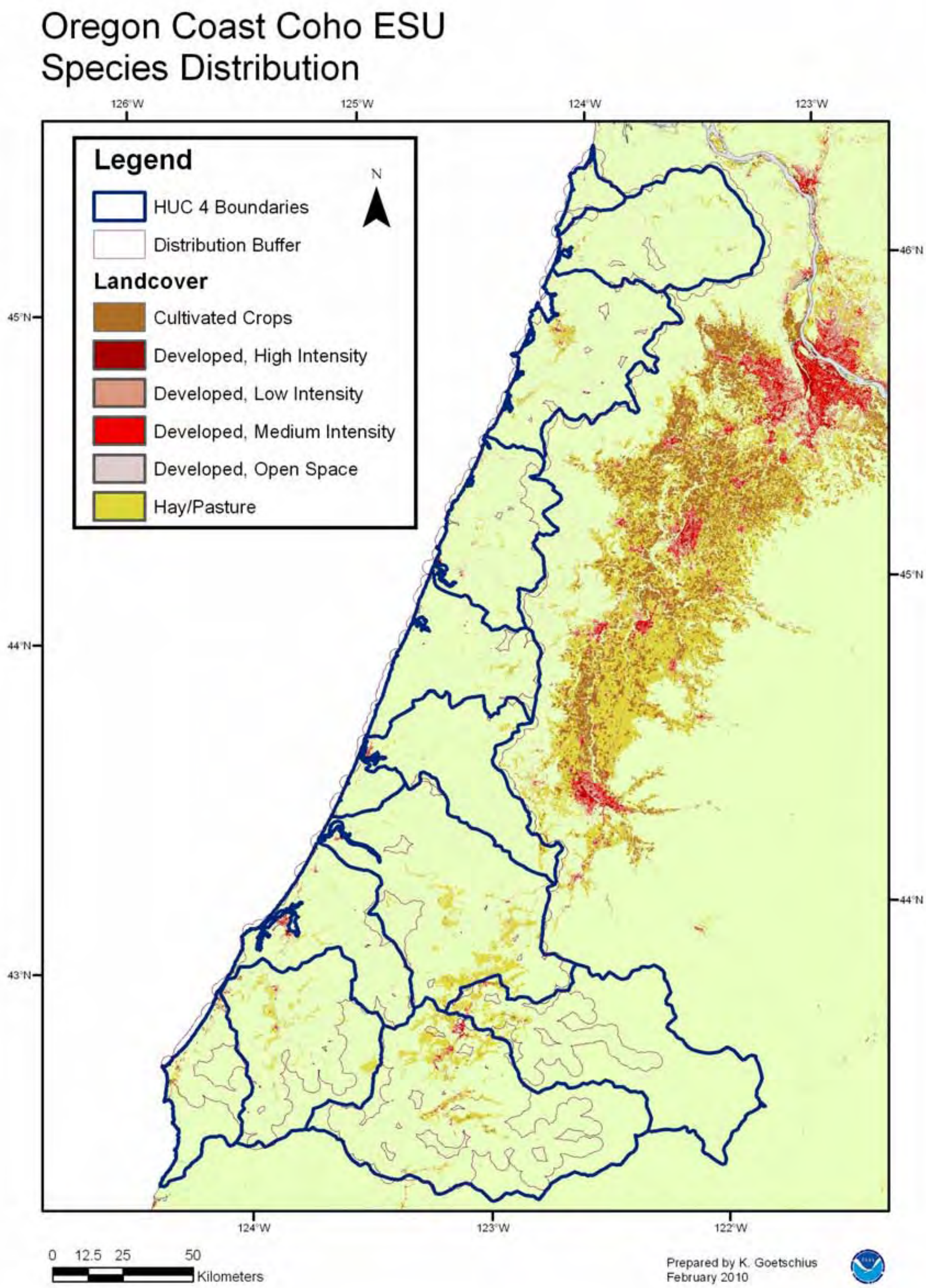


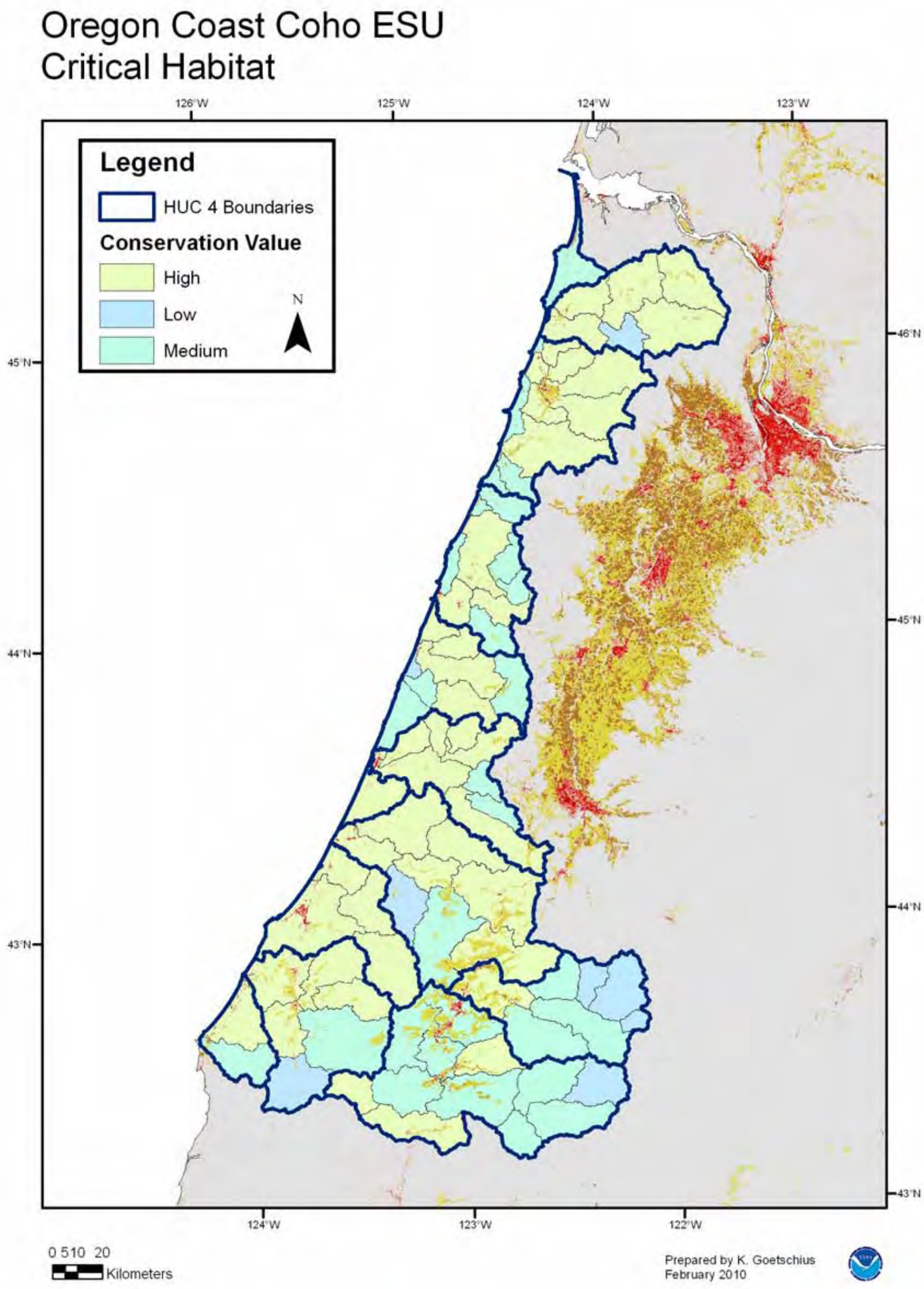
Columbia River Chum ESU Critical Habitat



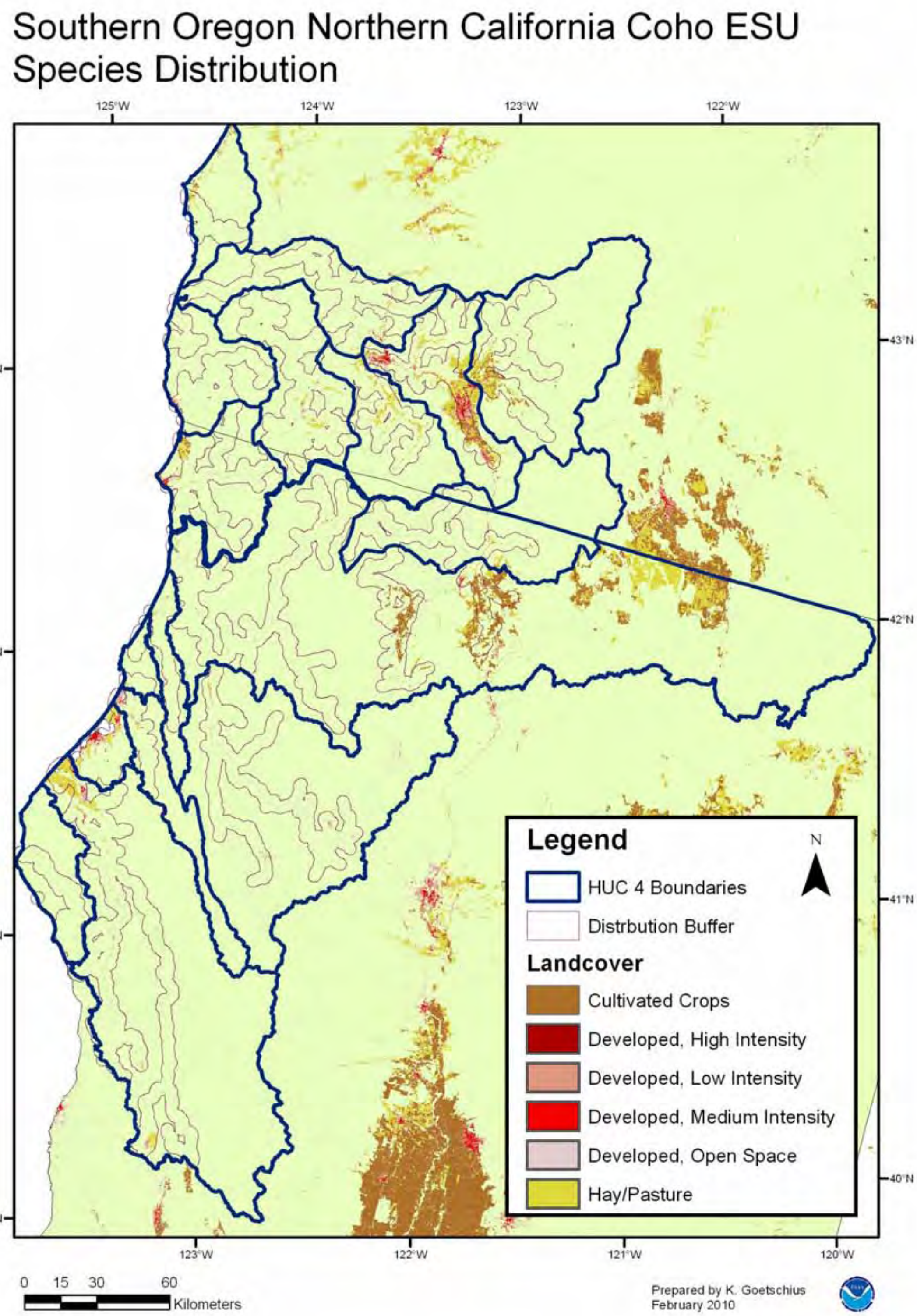
Lower Columbia River Coho ESU
Species Distribution



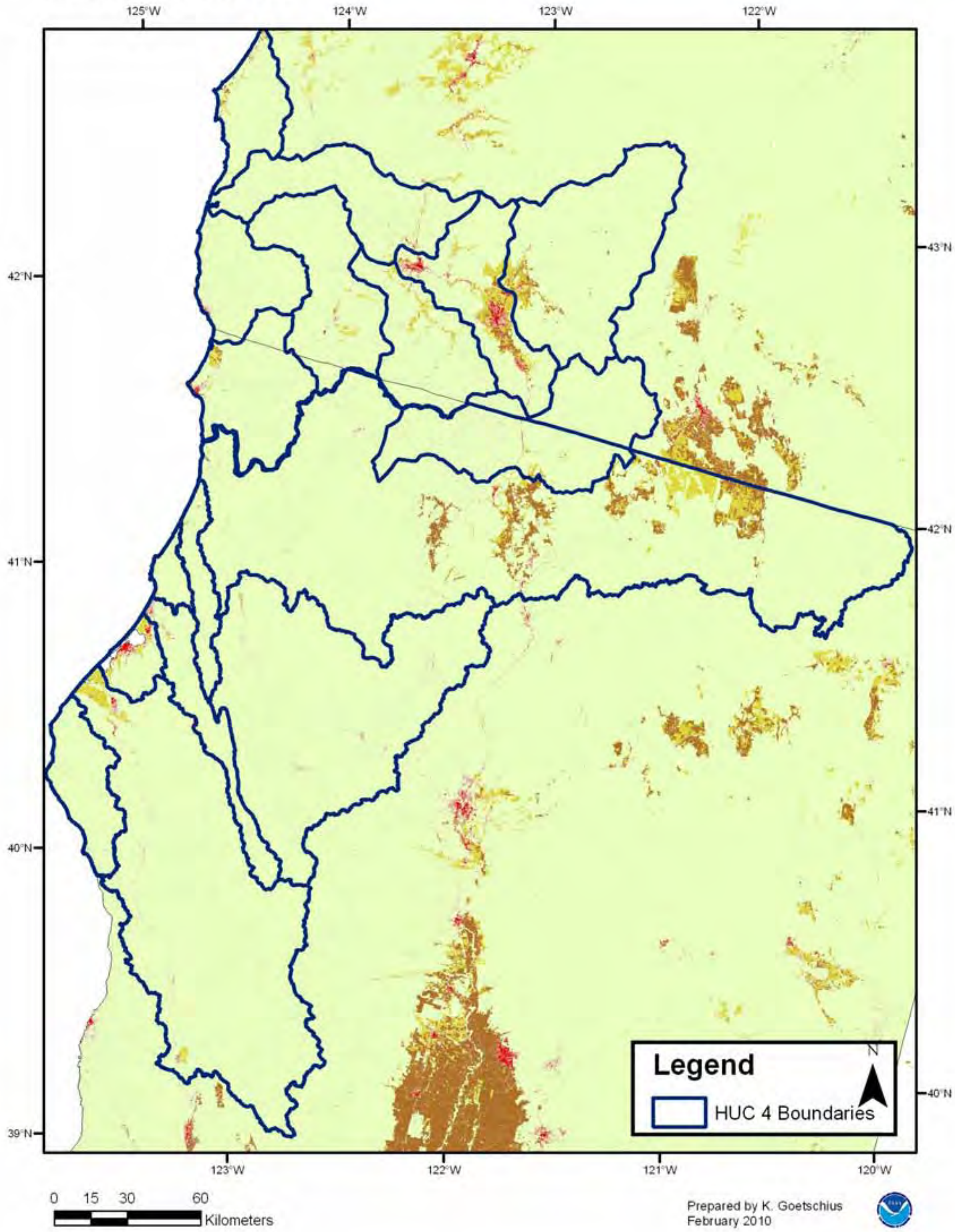

Oregon Coast Coho ESU
Species Distribution
Legend
HUC 4 Boundaries
Distribution Buffer
Landcover
Cultivated Crops
Developed, High Intensity
Developed, Low Intensity
Developed, Medium Intensity
Developed, Open Space
Hay/Pasture
N
126°W
125°W
124°W
123°W
122°W
46°N
45°N
44°N
43°N
0 12.5 25 50
Kilometers
Prepared by K. Goetschius
February 2010

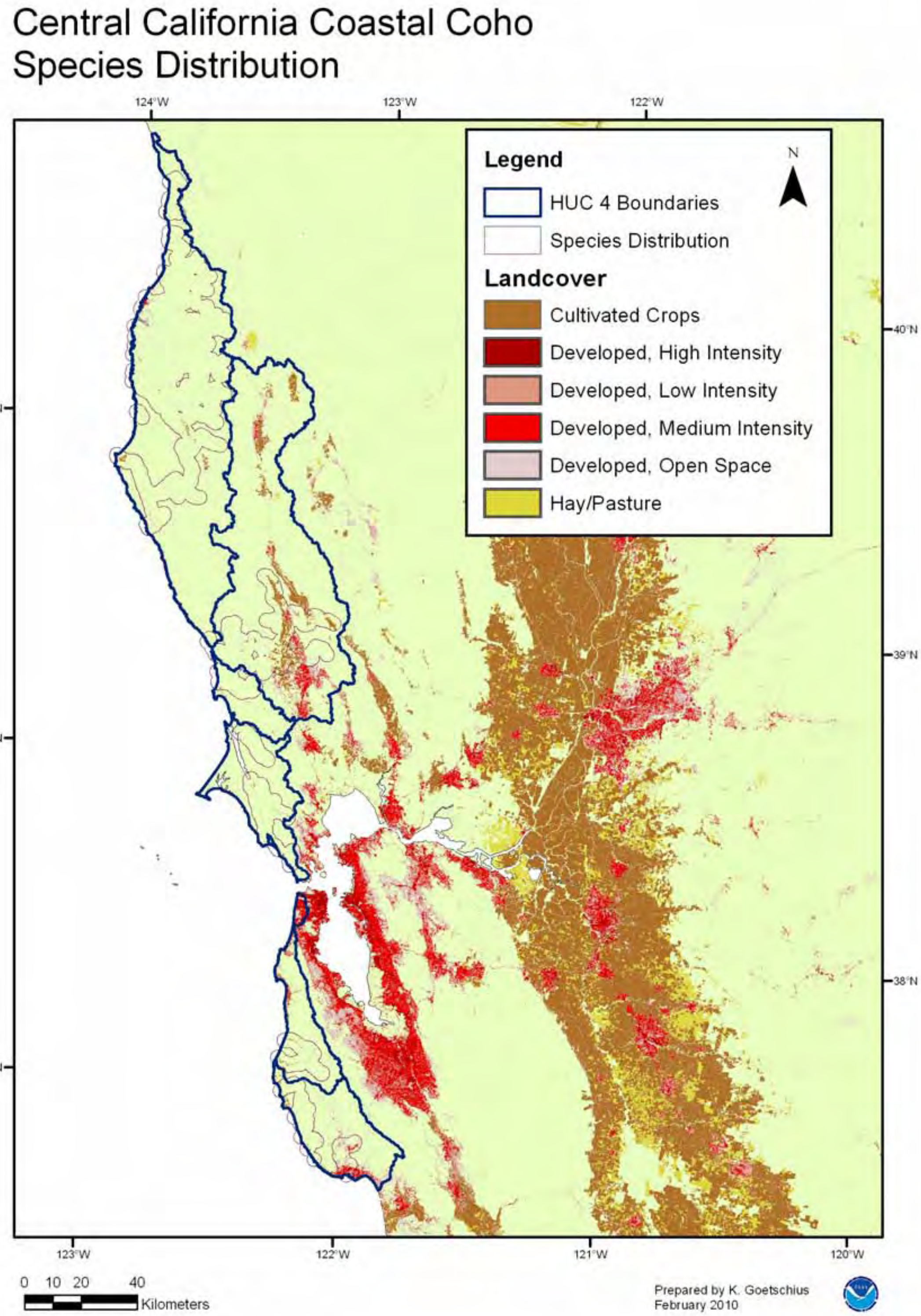


Oregon Coast Coho ESU
Critical Habitat
Legend
HUC 4 Boundaries
Conservation Value
High
Low
Medium
N
126°W
125°W
124°W
123°W
122°W
46°N
45°N
44°N
43°N
0 5 10 20
Kilometers
Prepared by K. Goetschius
February 2010

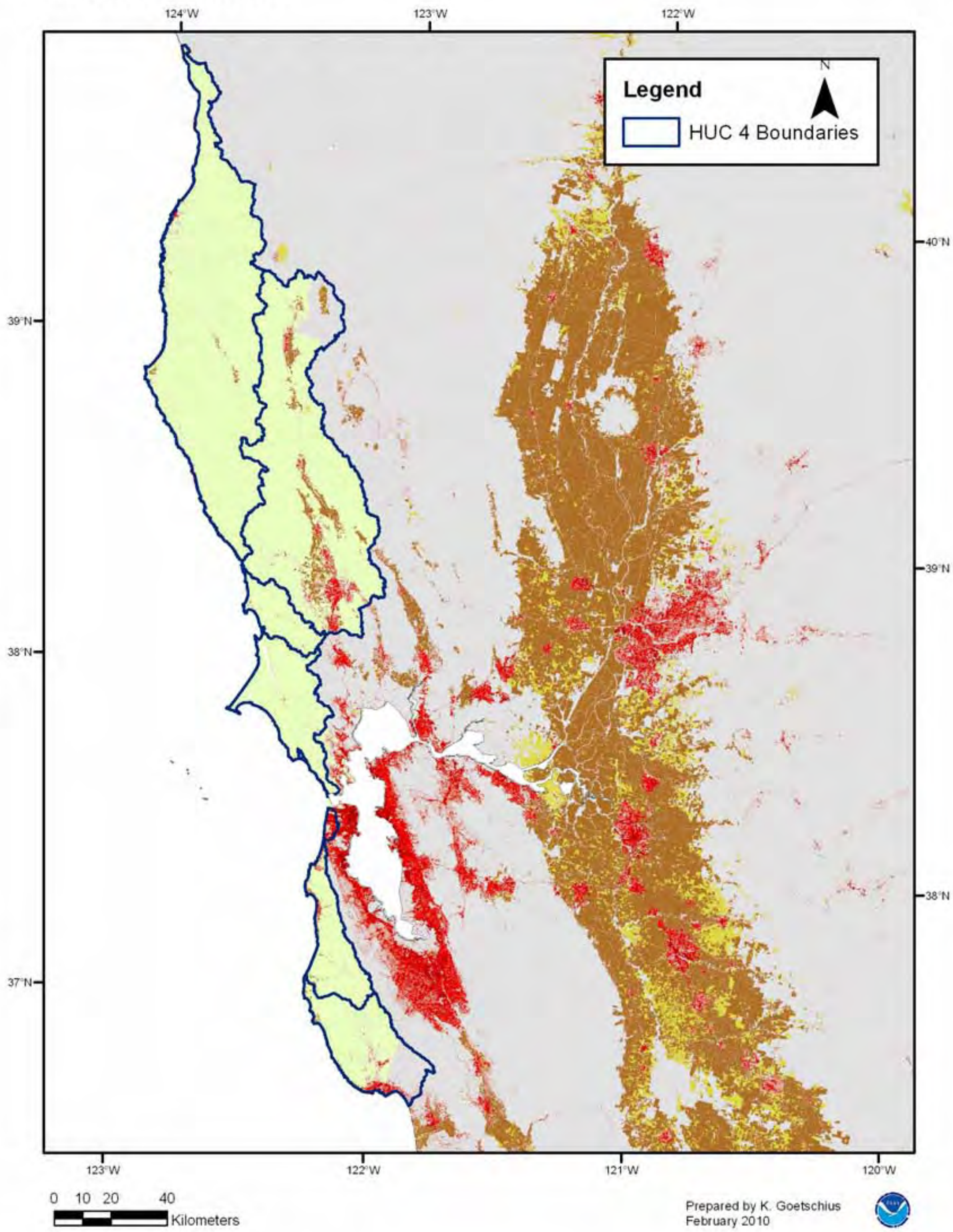

Southern Oregon Northern California Coho ESU
Species Distribution
125°W
124°W
123°W
122°W
43°N
42°N
41°N
40°N
39°N
121°W
120°W
Legend
N
HUC 4 Boundaries
Distrbution Buffer
Landcover
Cultivated Crops
Developed, High Intensity
Developed, Low Intensity
Developed, Medium Intensity
Developed, Open Space
Hay/Pasture
0 15 30 60
Kilometers
Prepared by K. Goetschius
February 2010

Southern Oregon Northern California Coho ESU Critical Habitat

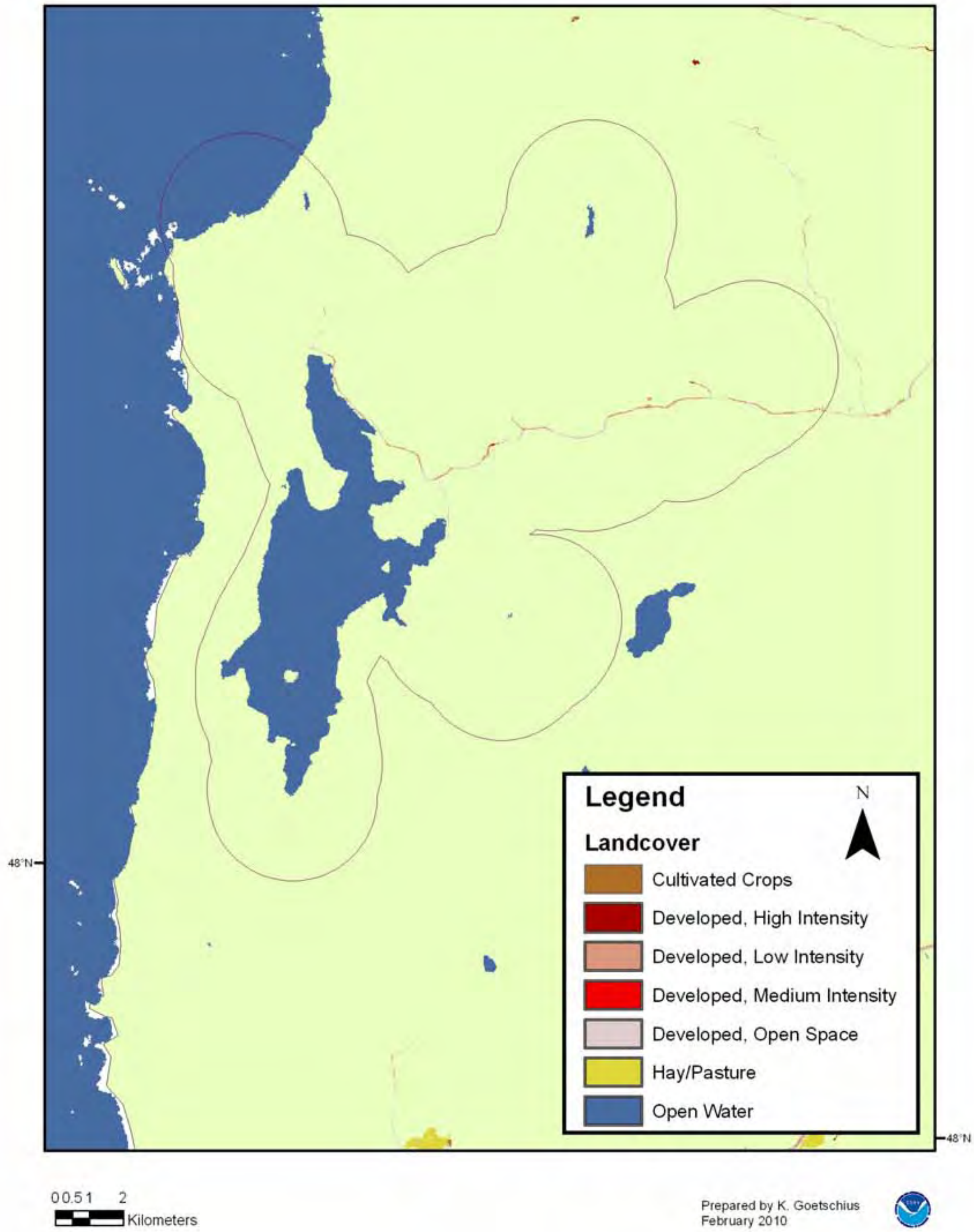



Central California Coastal Coho
Species Distribution
Legend
HUC 4 Boundaries
Species Distribution
Landcover
Cultivated Crops
Developed, High Intensity
Developed, Low Intensity
Developed, Medium Intensity
Developed, Open Space
Hay/Pasture
N
124°W
123°W
122°W
121°W
120°W
40°N
39°N
38°N
37°N
0 10 20 40
Kilometers
Prepared by K. Goetschius
February 2010

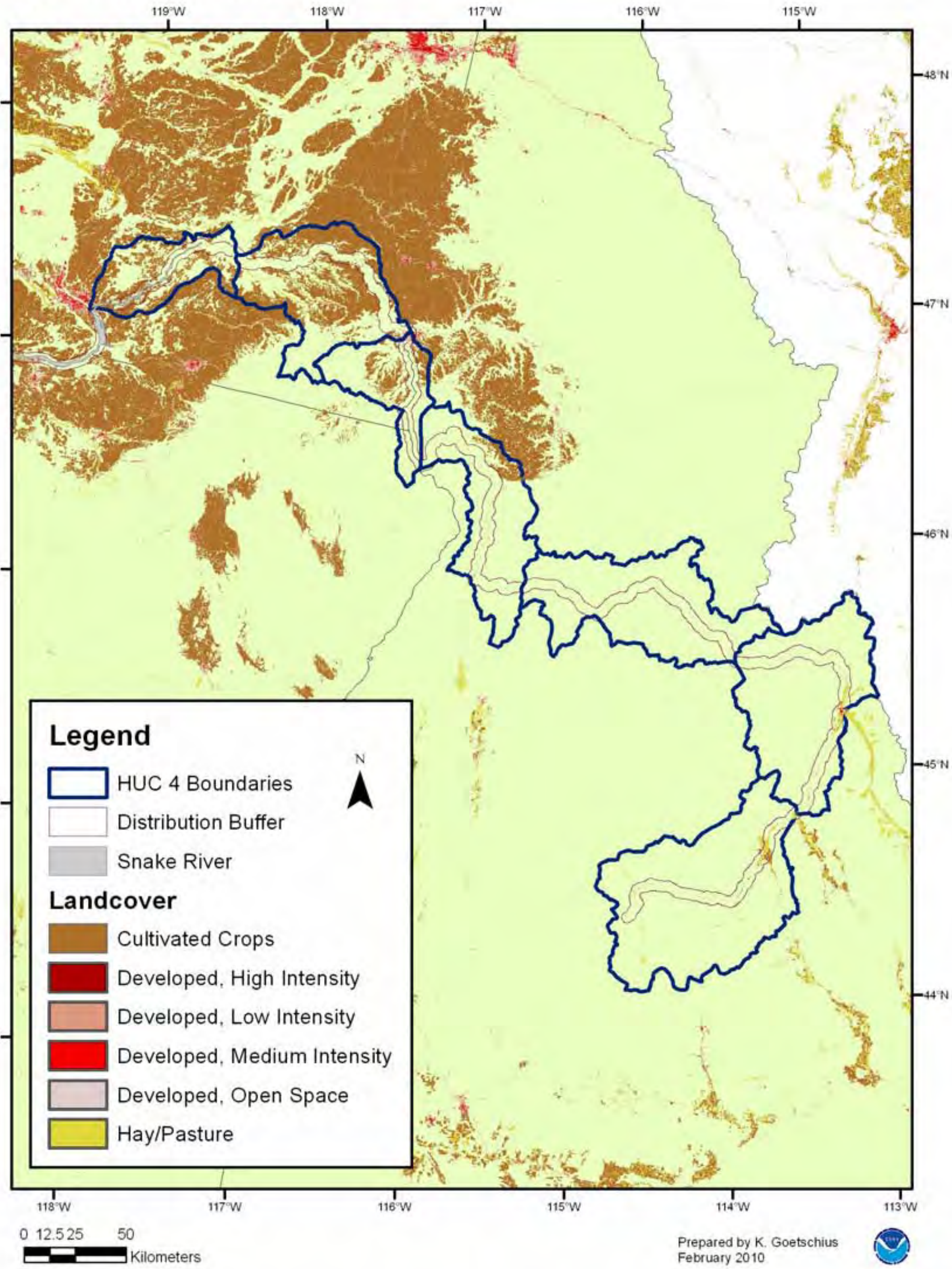
Central California Coastal Coho Critical Habitat



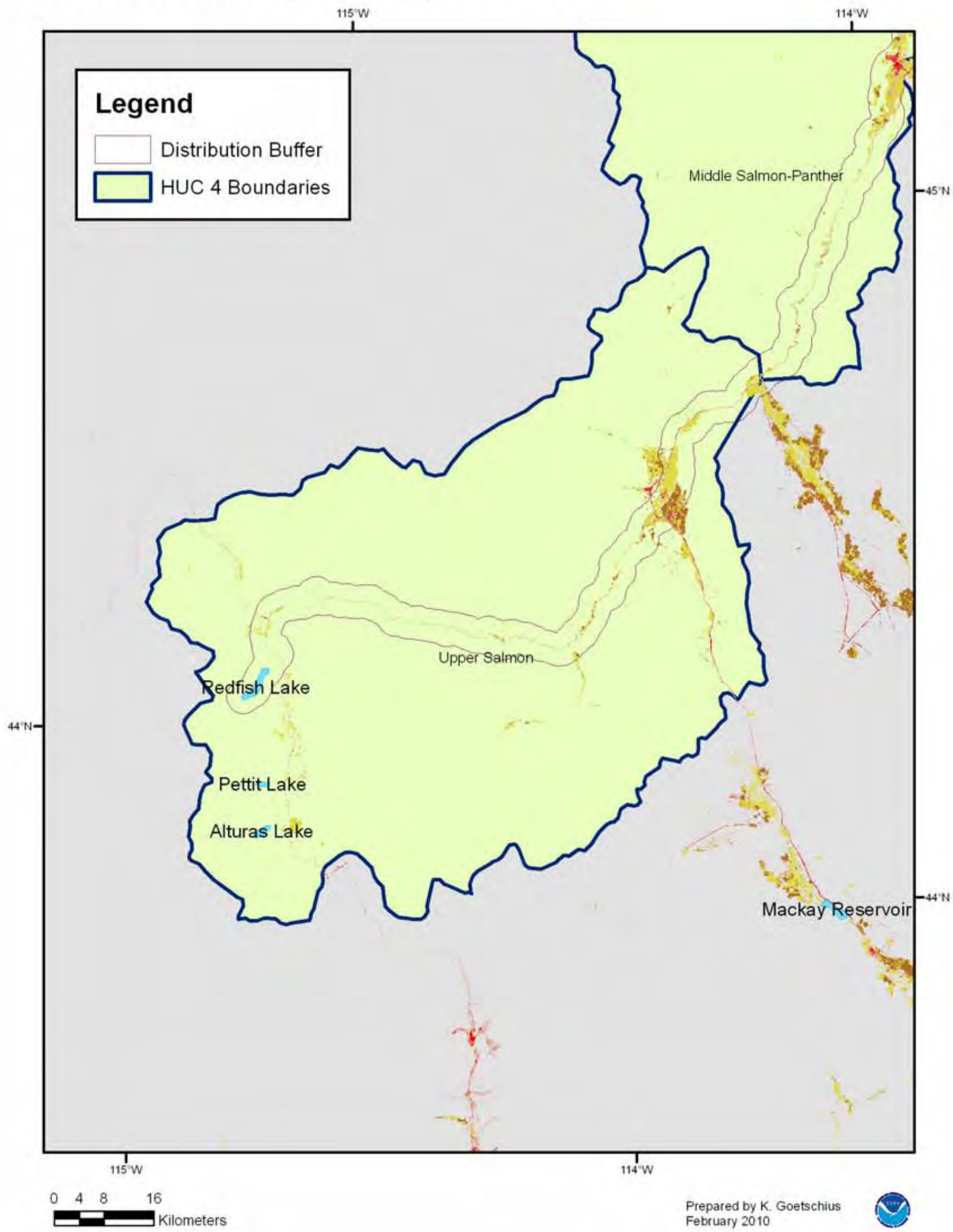
Ozette Lake Sockeye
Species Distribution and Critical Habitat

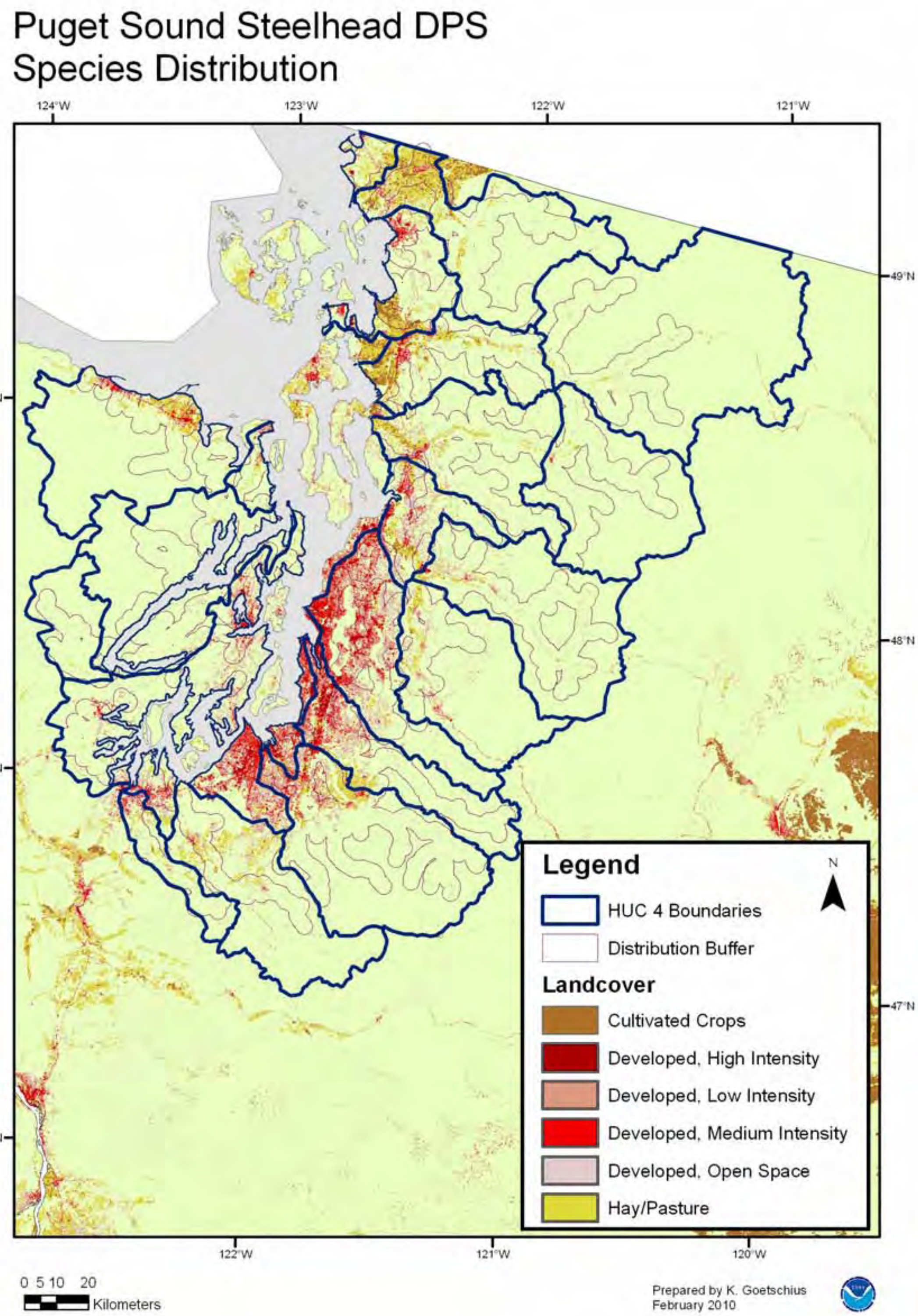


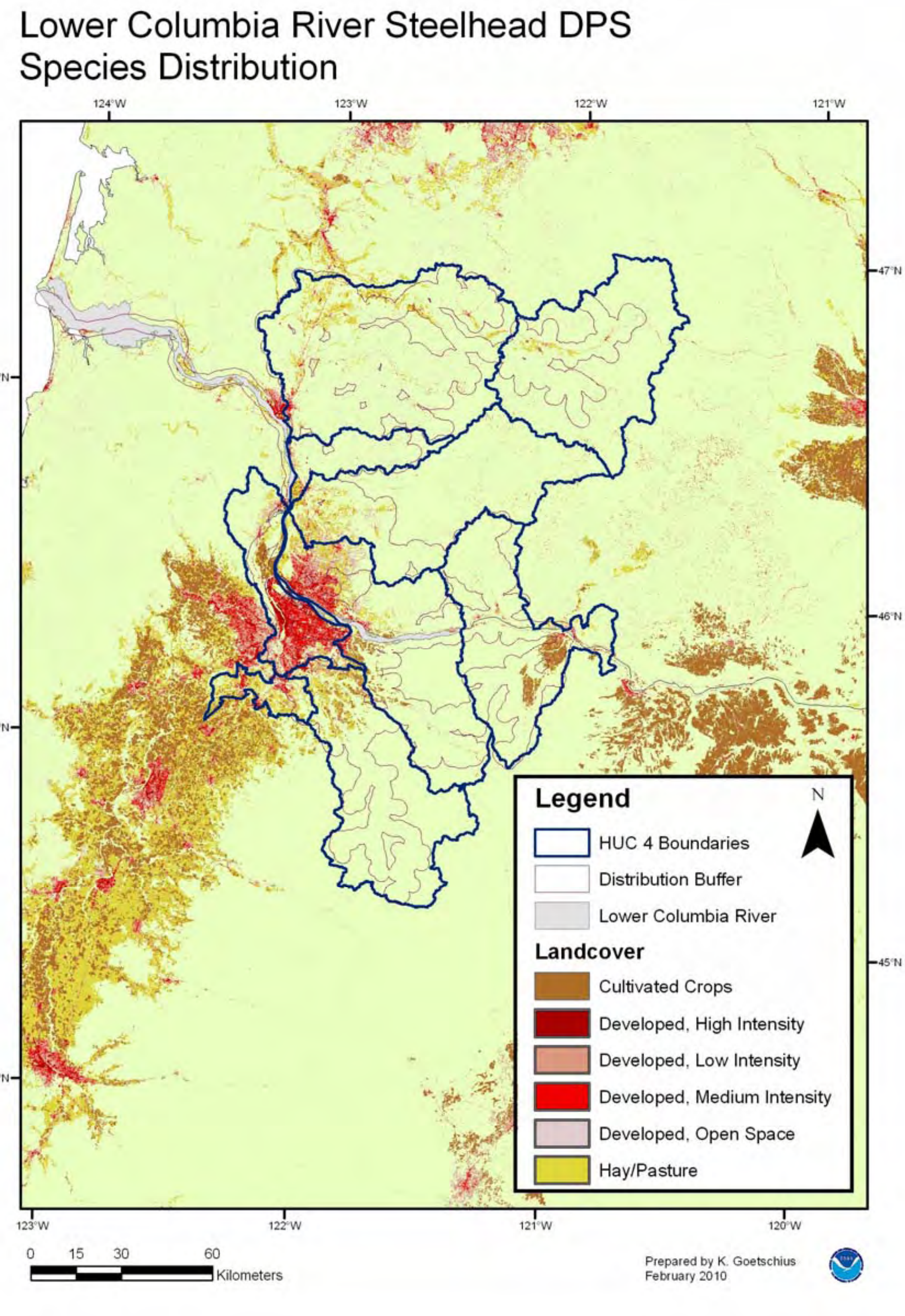
Snake River Sockeye ESU
Species Distribution



Snake River Sockeye Critical Habitat, close up of lakes

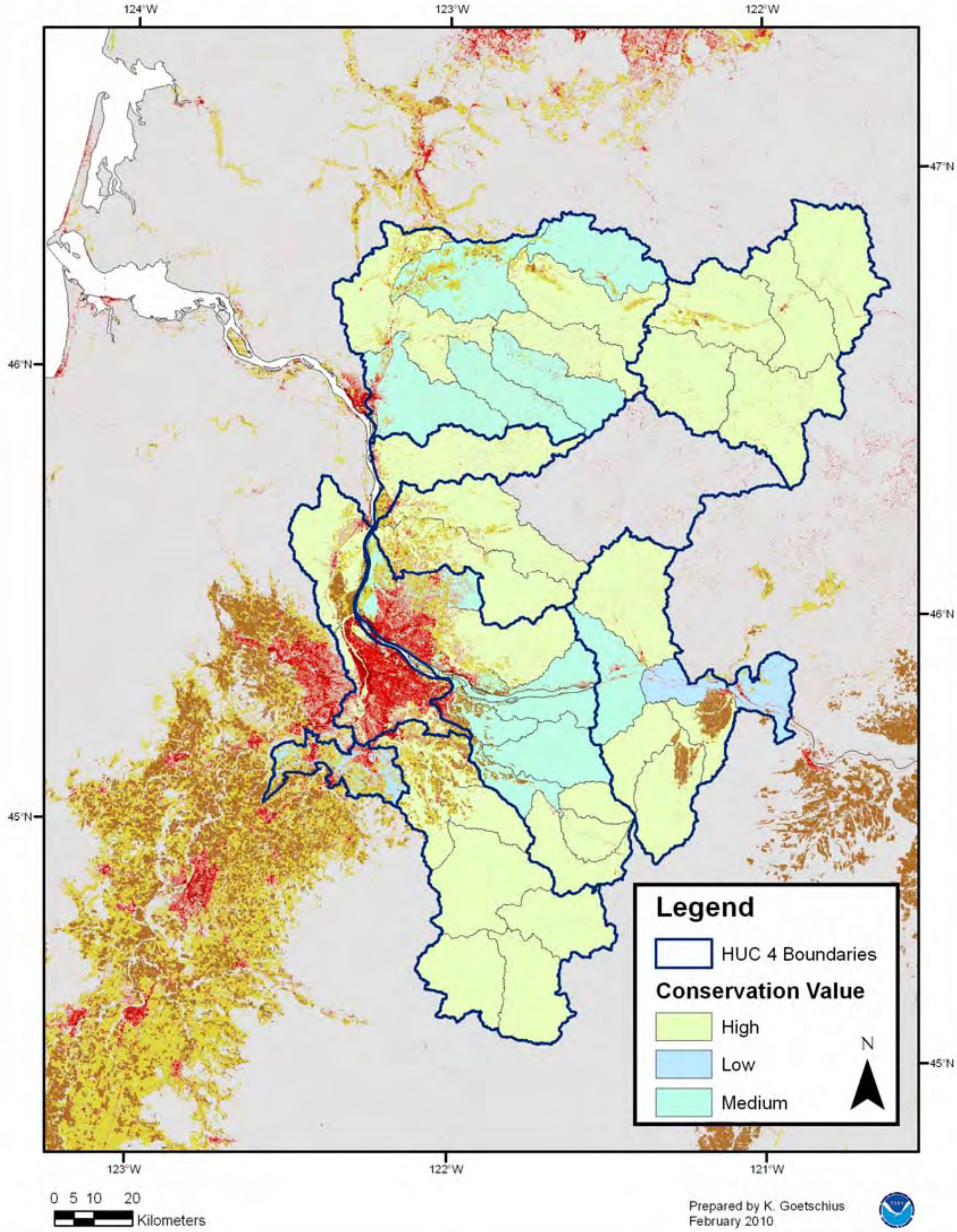



Puget Sound Steelhead DPS
Species Distribution
124°W
123°W
122°W
121°W
49°N
48°N
47°N
46°N
122°W
121°W
120°W
Legend
N
HUC 4 Boundaries
Distribution Buffer
Landcover
Cultivated Crops
Developed, High Intensity
Developed, Low Intensity
Developed, Medium Intensity
Developed, Open Space
Hay/Pasture
0 5 10 20
Kilometers
Prepared by K. Goetschius
February 2010

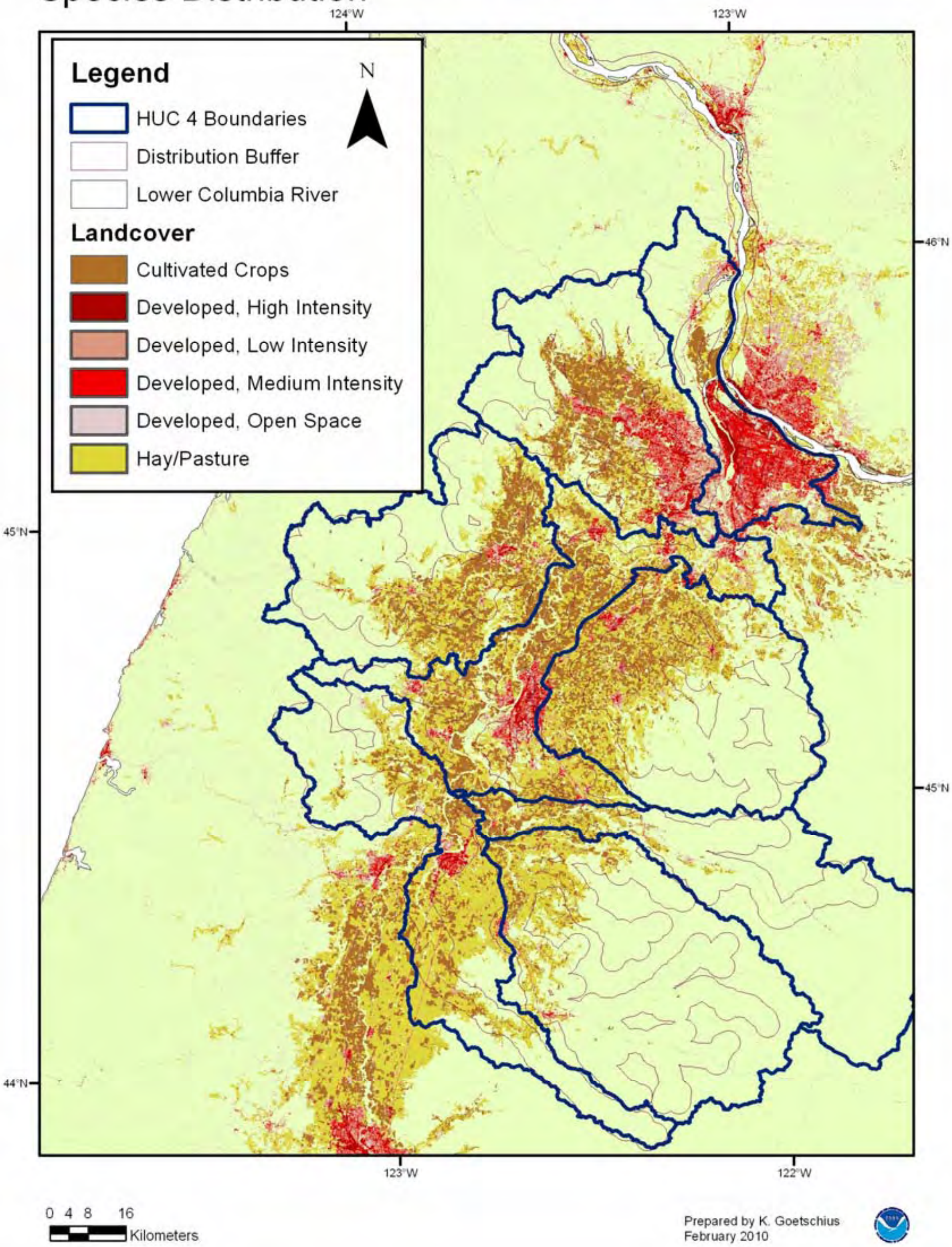


Lower Columbia River Steelhead DPS
Species Distribution
124°W
123°W
122°W
121°W
47°N
46°N
45°N
44°N
123°W
122°W
121°W
120°W
Legend
N
HUC 4 Boundaries
Distribution Buffer
Lower Columbia River
Landcover
Cultivated Crops
Developed, High Intensity
Developed, Low Intensity
Developed, Medium Intensity
Developed, Open Space
Hay/Pasture
0 15 30 60
Kilometers
Prepared by K. Goetschius
February 2010

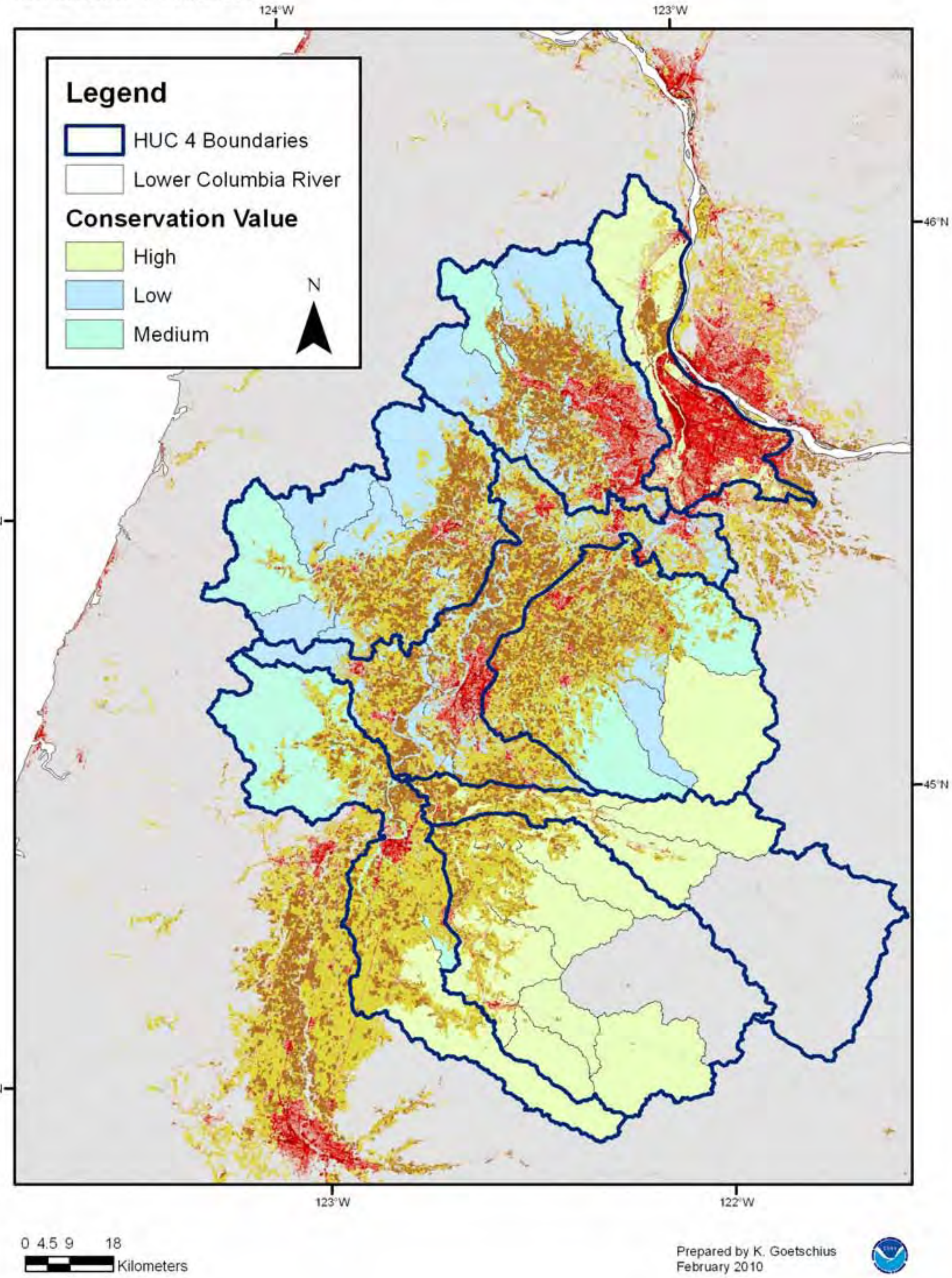
Lower Columbia River Steelhead DPS
Critical Habitat



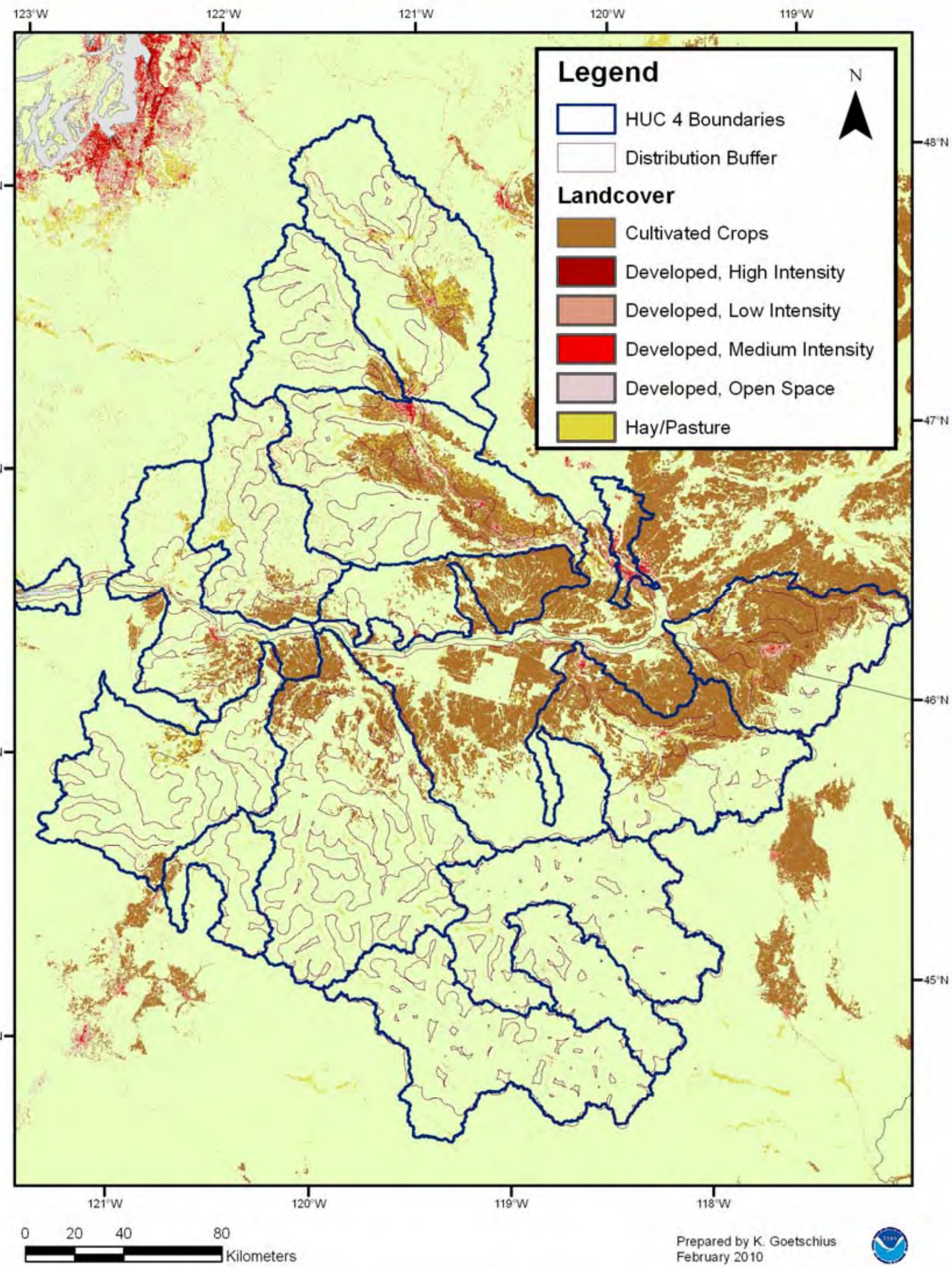
Upper Willamette River Steelhead DPS
Species Distribution



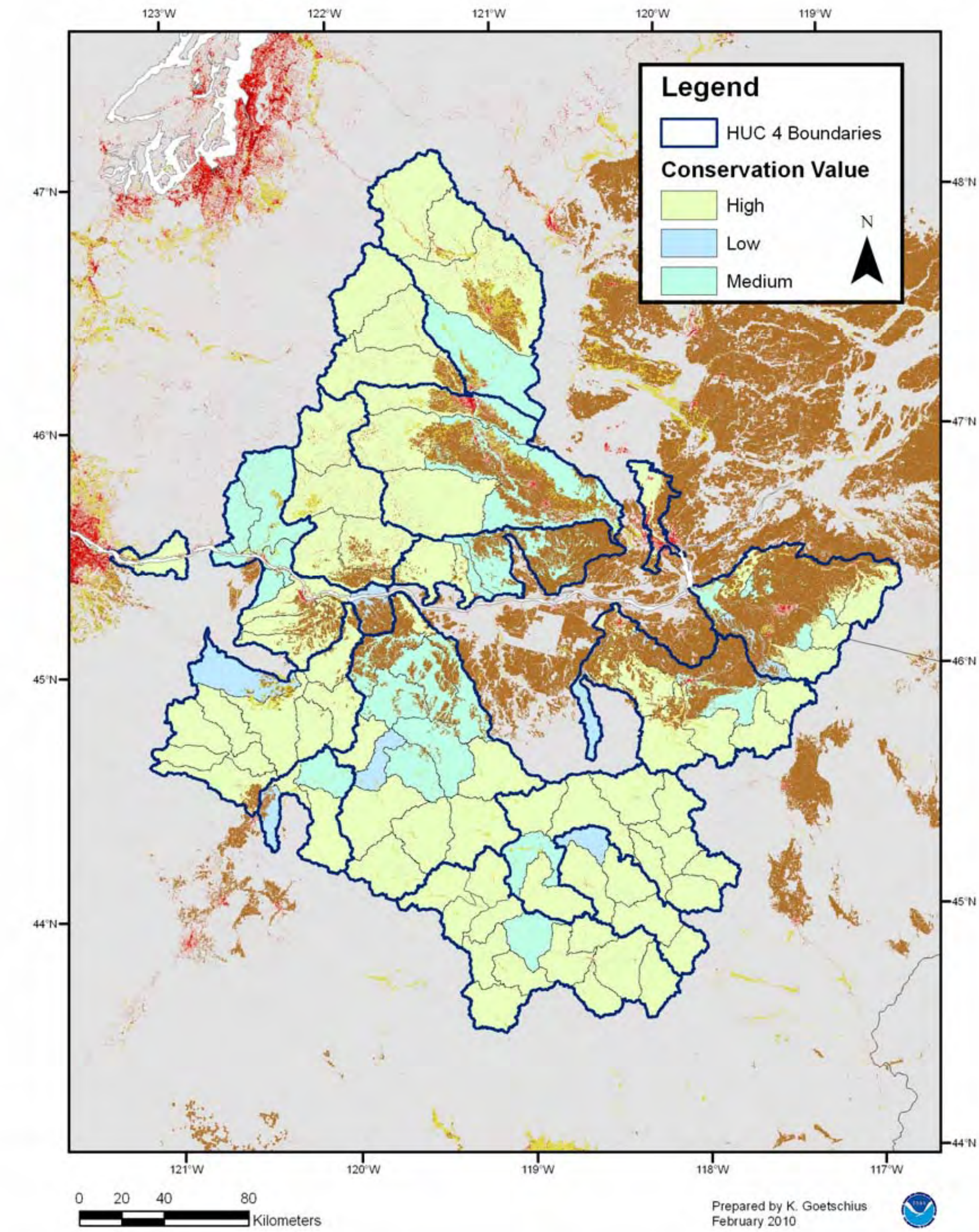
Upper Willamette River Steelhead DPS Critical Habitat



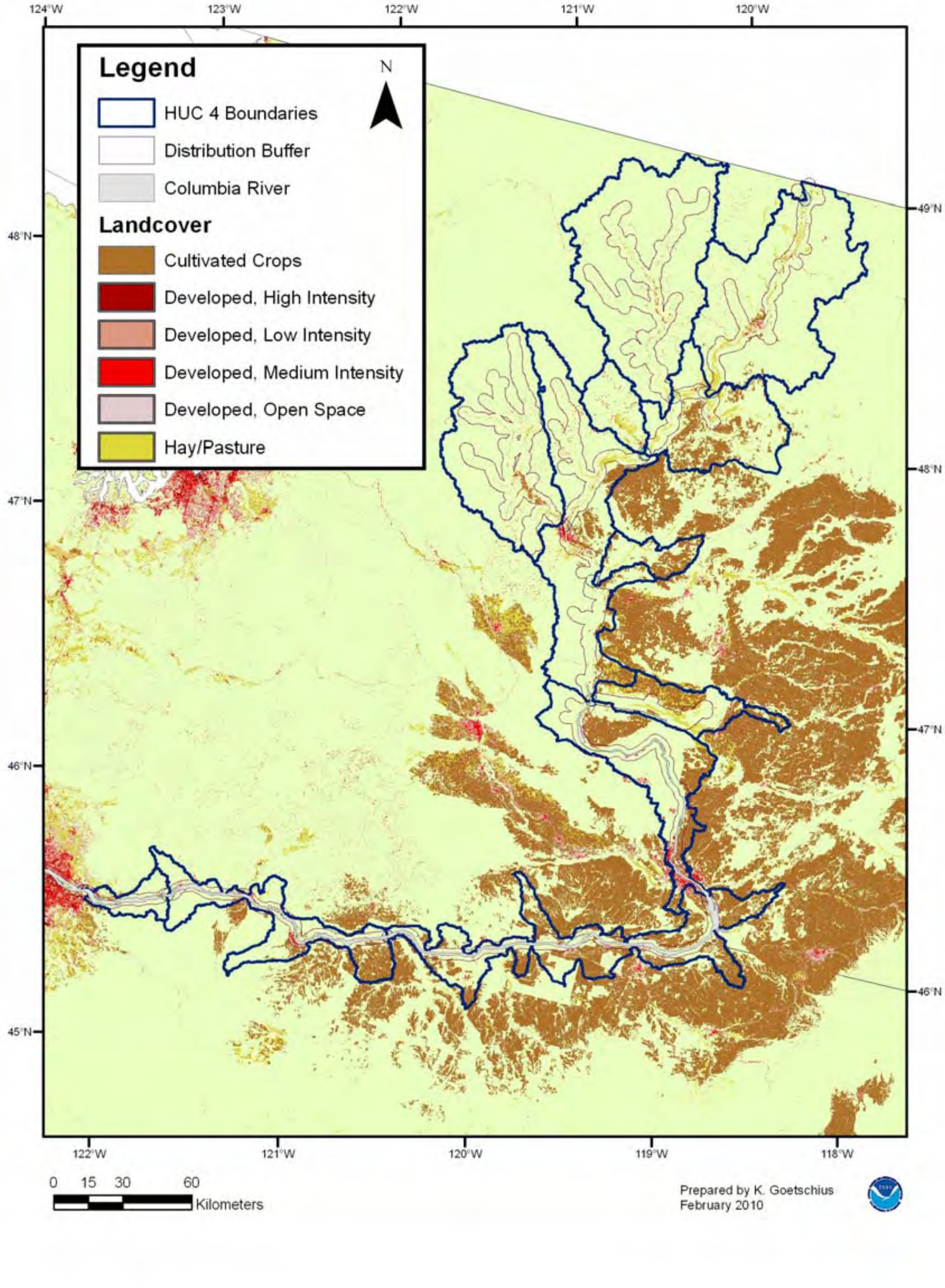
Middle Columbia River Steelhead DPS Species Distribution

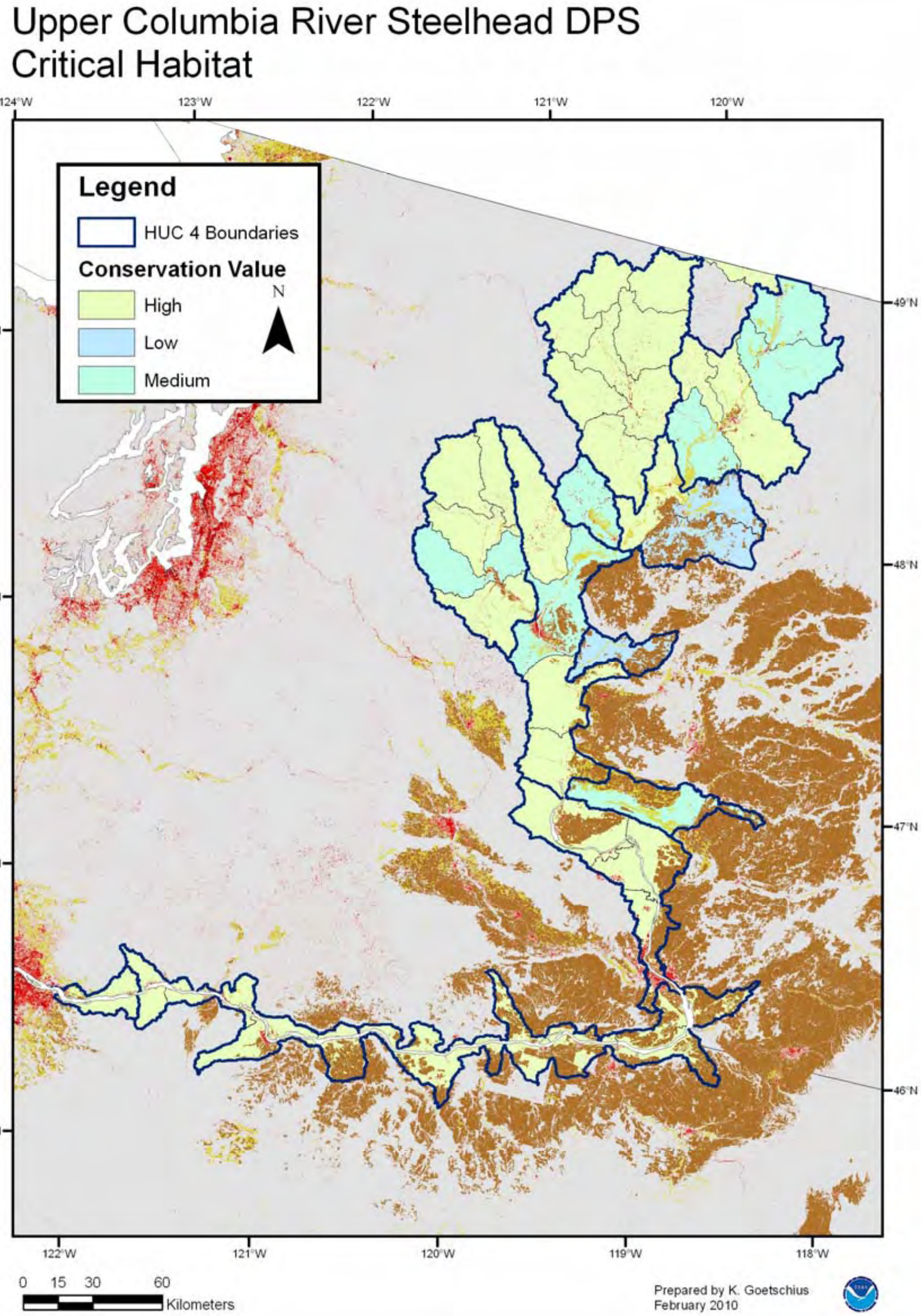


Middle Columbia River Steelhead DPS Critical Habitat

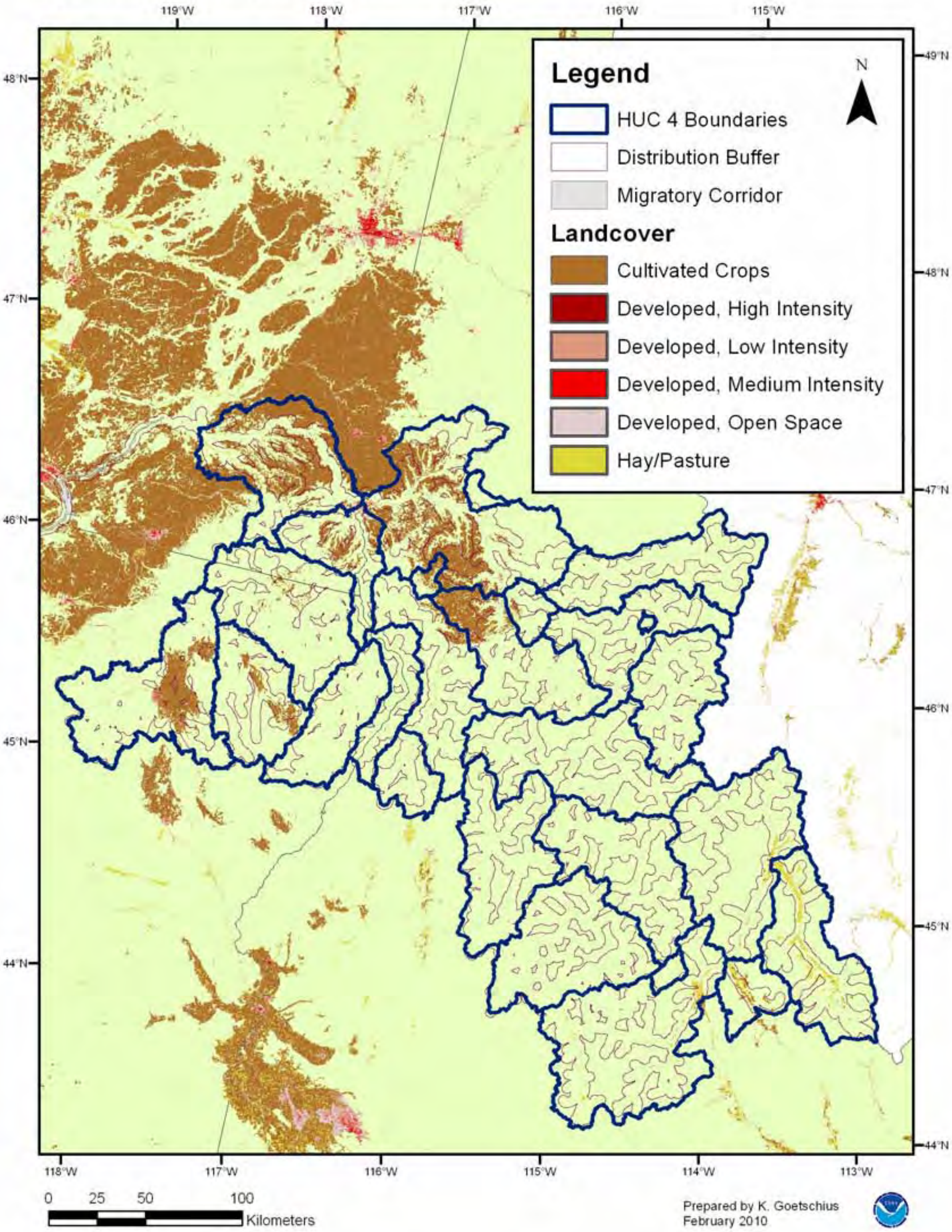


Upper Columbia River Steelhead DPS
Species Distribution

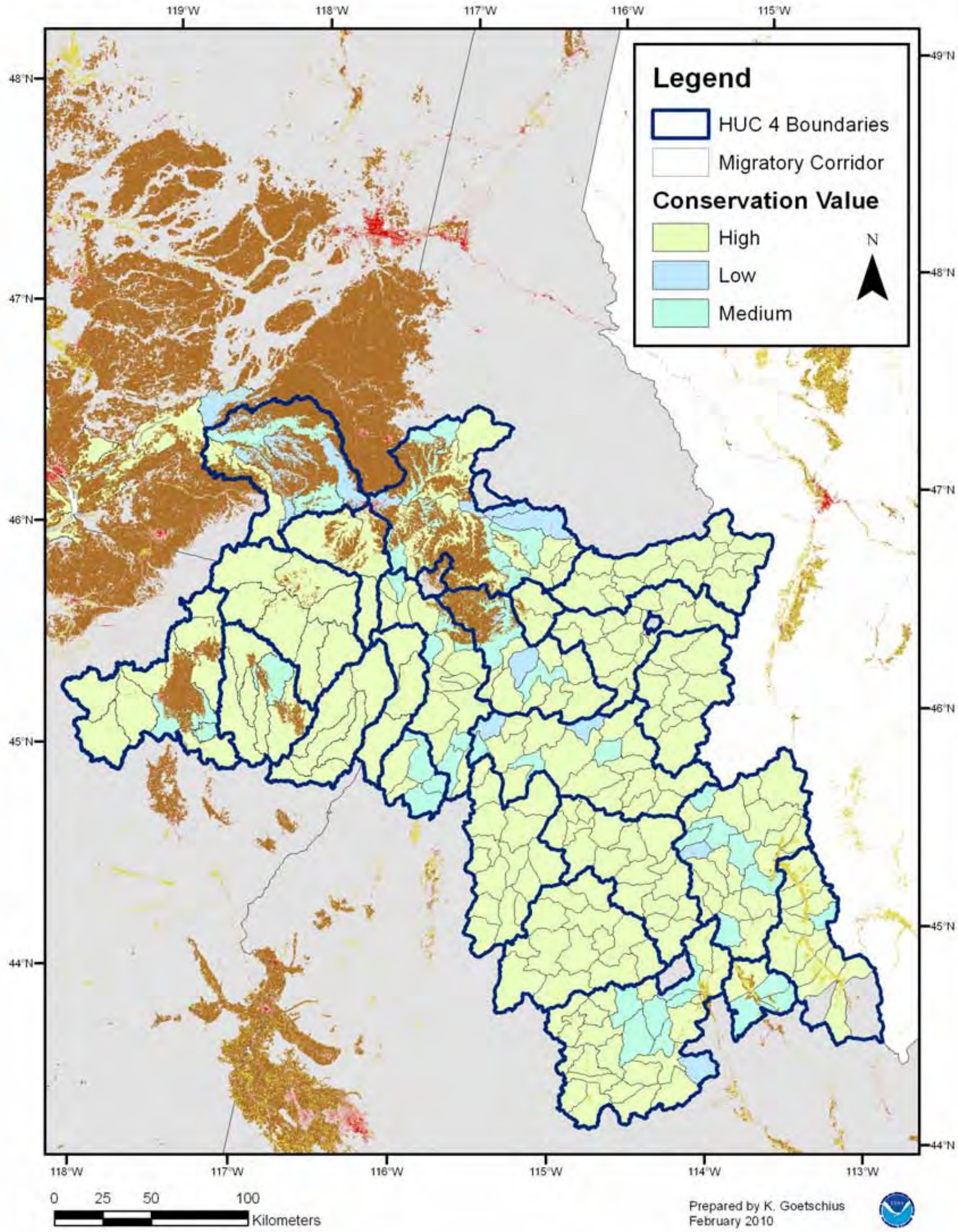


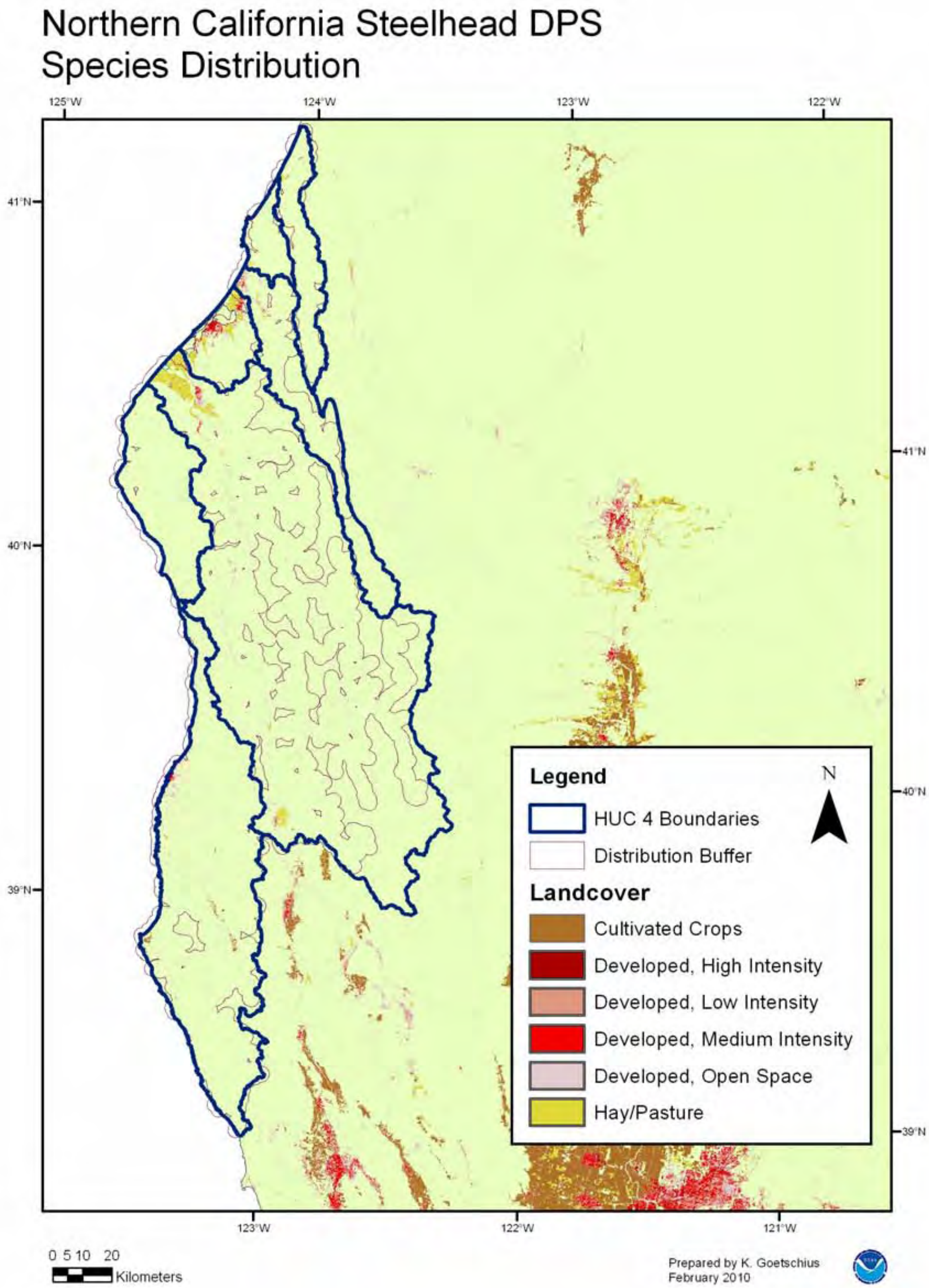

Upper Columbia River Steelhead DPS
Critical Habitat
Legend
HUC 4 Boundaries
Conservation Value
High
Low
Medium
N
124°W
123°W
122°W
121°W
120°W
49°N
48°N
47°N
46°N
45°N
119°W
118°W
0 15 30 60
Kilometers
Prepared by K. Goetschius
February 2010

Snake River Steelhead DPS
Species Distribution



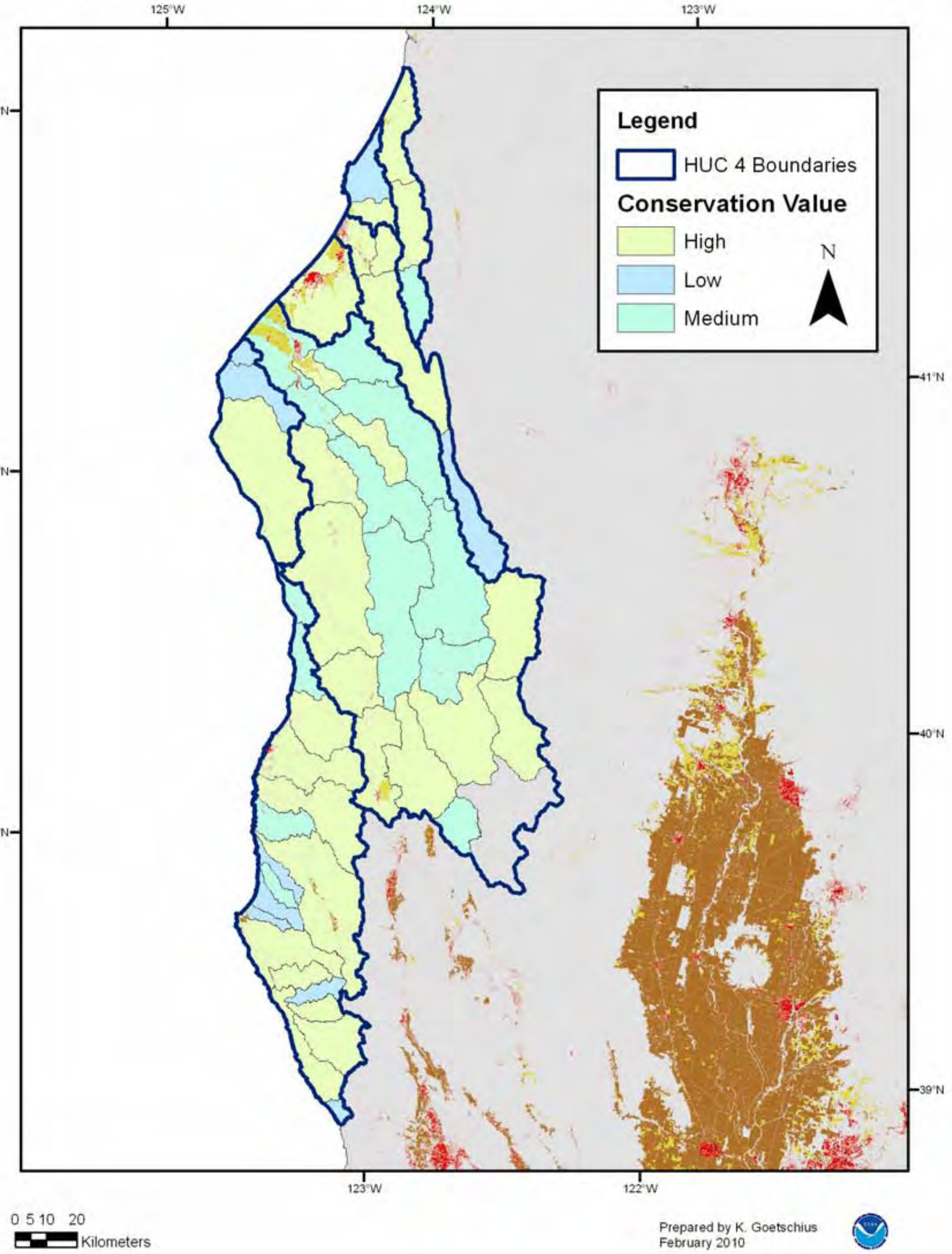
Snake River Steelhead DPS Critical Habitat



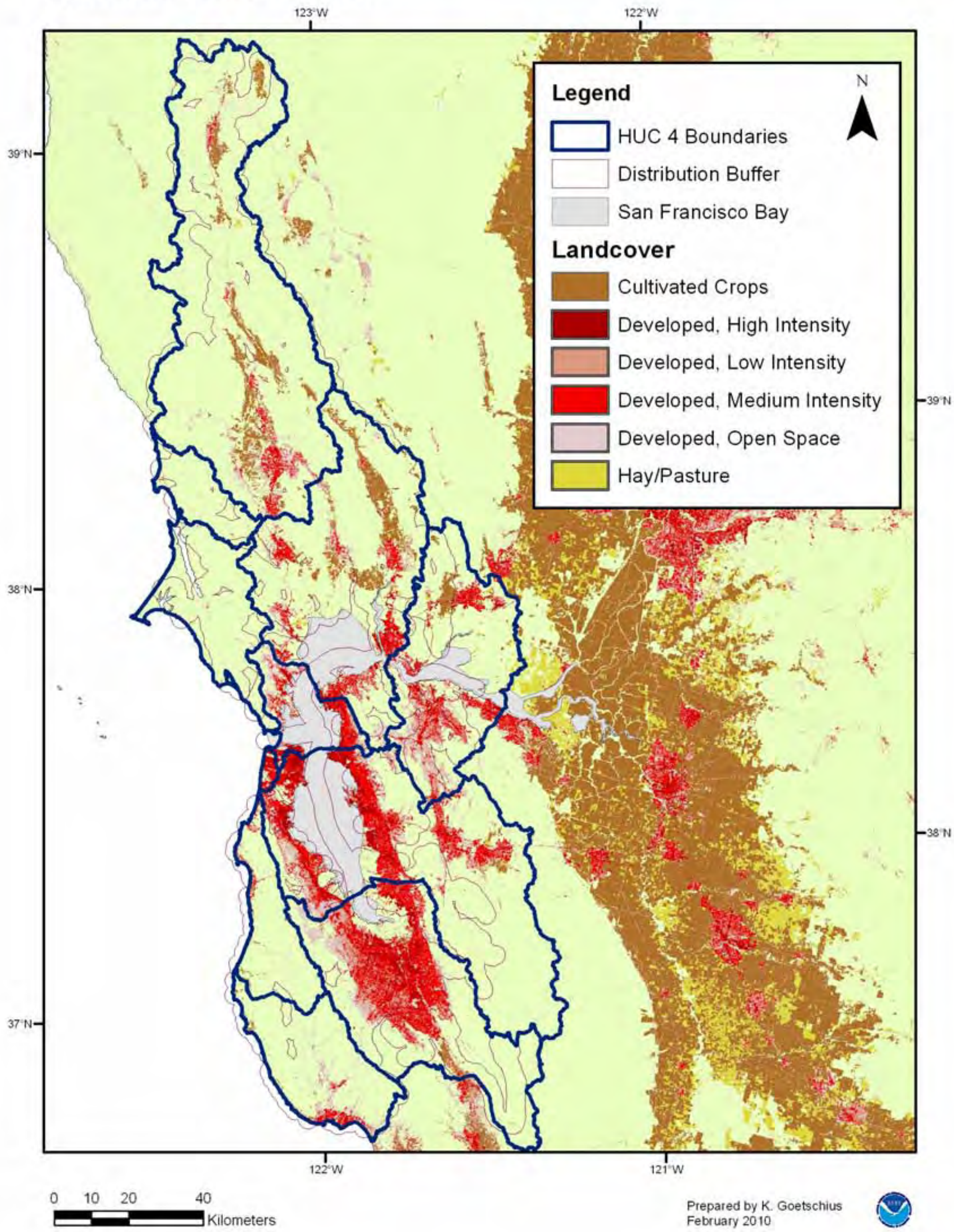


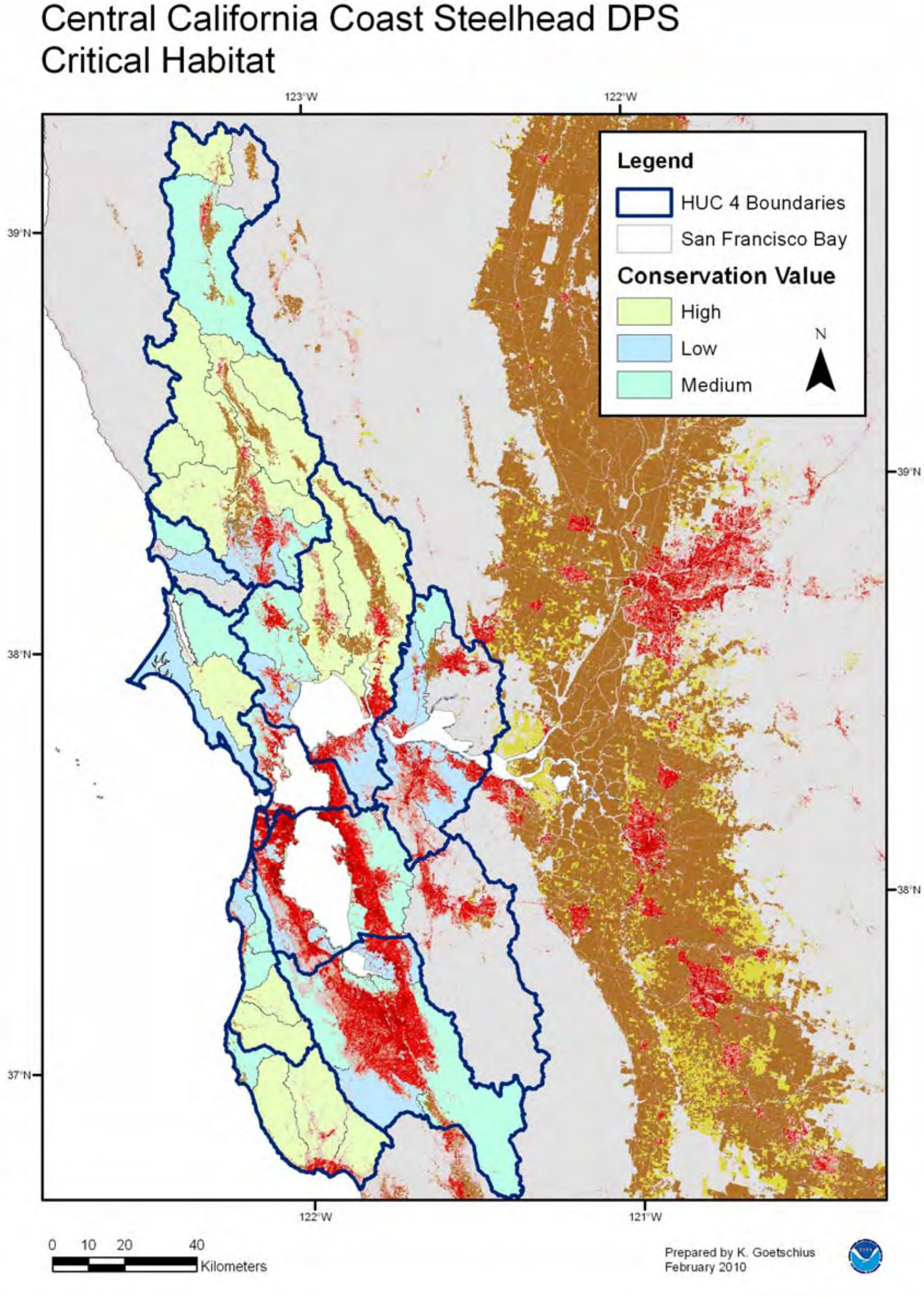
Northern California Steelhead DPS
Species Distribution
125°W
124°W
123°W
122°W
121°W
41°N
40°N
39°N
Legend
N
HUC 4 Boundaries
Distribution Buffer
Landcover
Cultivated Crops
Developed, High Intensity
Developed, Low Intensity
Developed, Medium Intensity
Developed, Open Space
Hay/Pasture
0 5 10 20
Kilometers
Prepared by K. Goetschius
February 2010

Northern California Steelhead DPS Critical Habitat



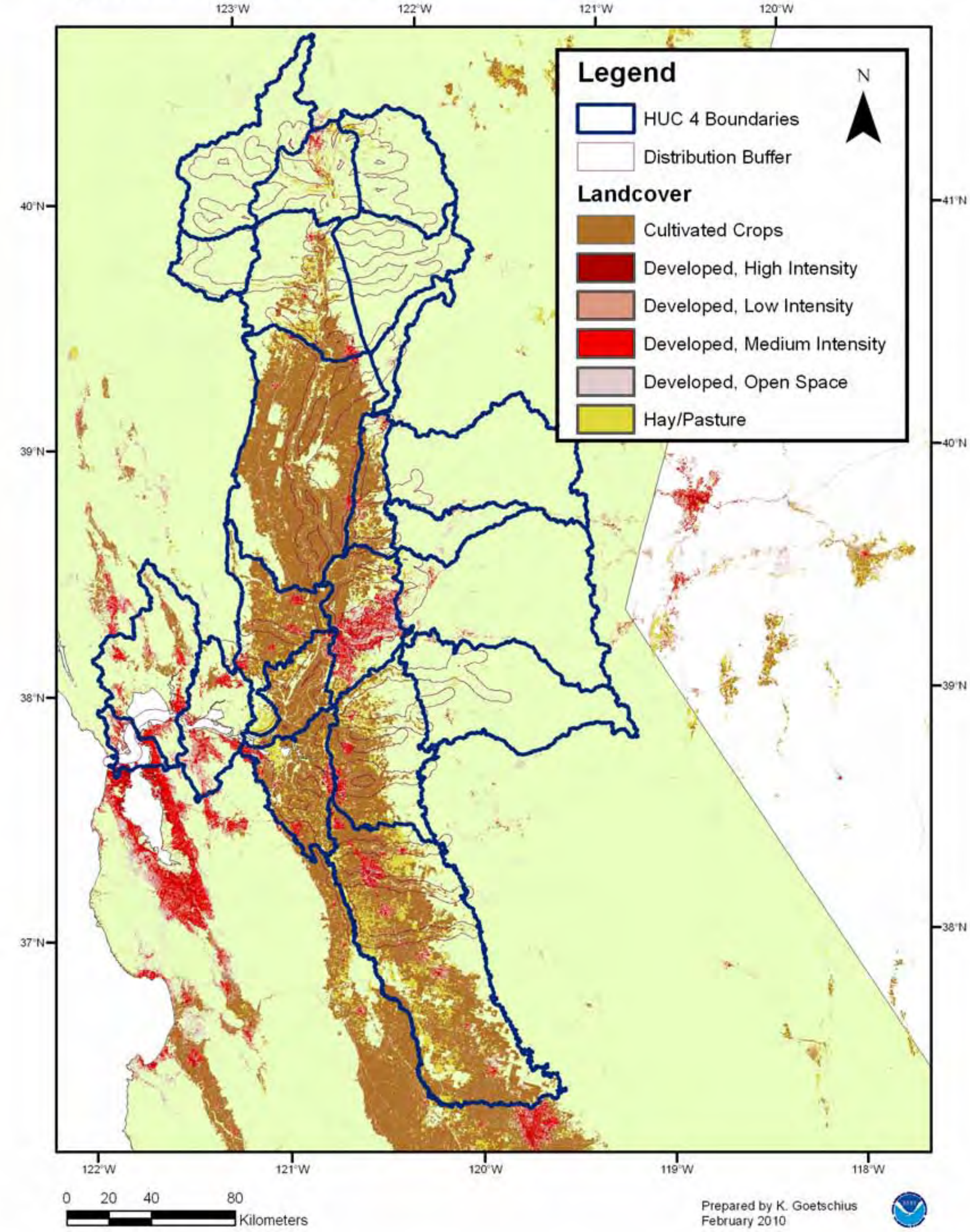
Central California Coast Steelhead DPS
Species Distribution



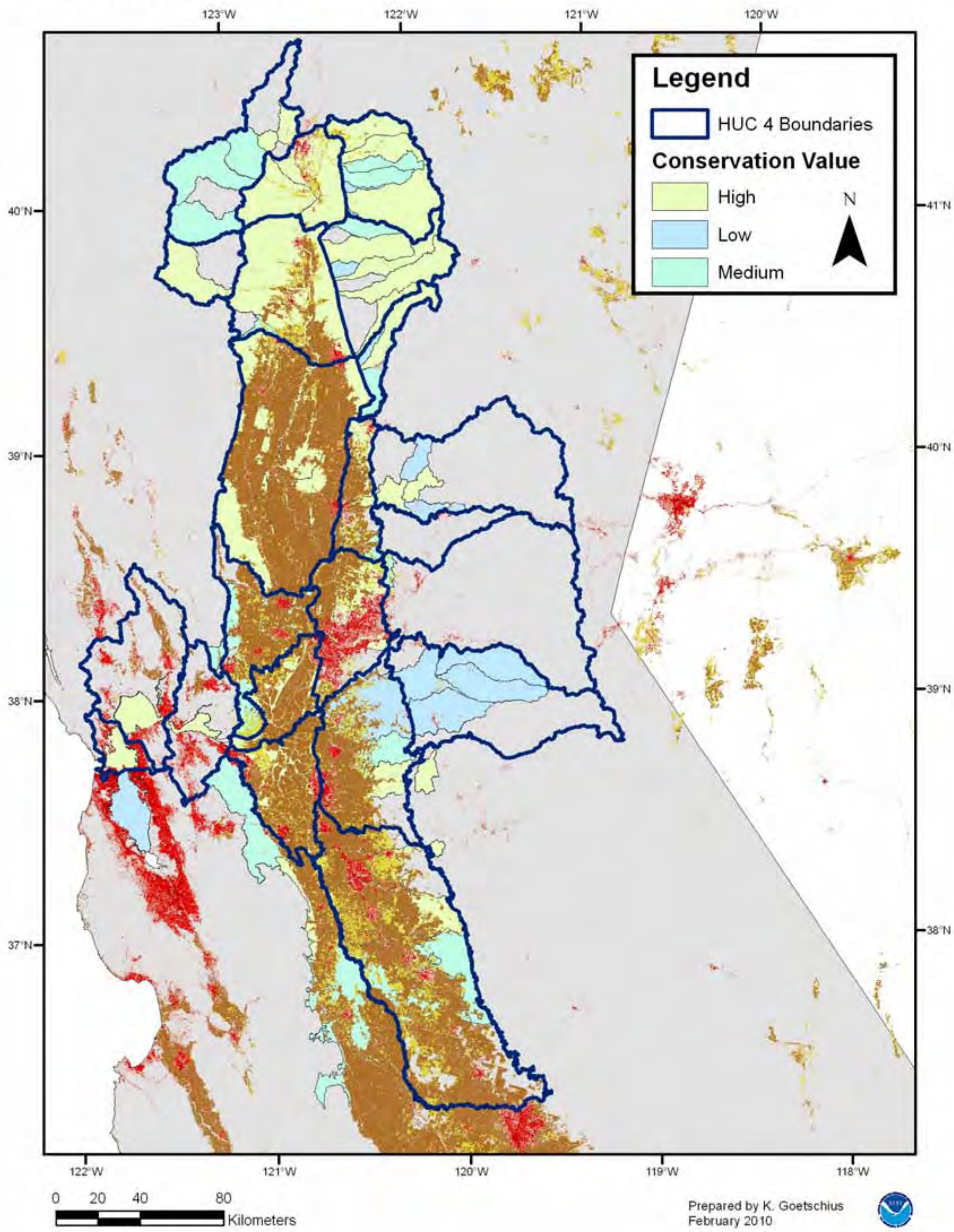


Central California Coast Steelhead DPS
Critical Habitat
123°W
122°W
39°N
38°N
37°N
121°W
Legend
HUC 4 Boundaries
San Francisco Bay
Conservation Value
High
Low
Medium
N
0 10 20 40
Kilometers
Prepared by K. Goetschius
February 2010

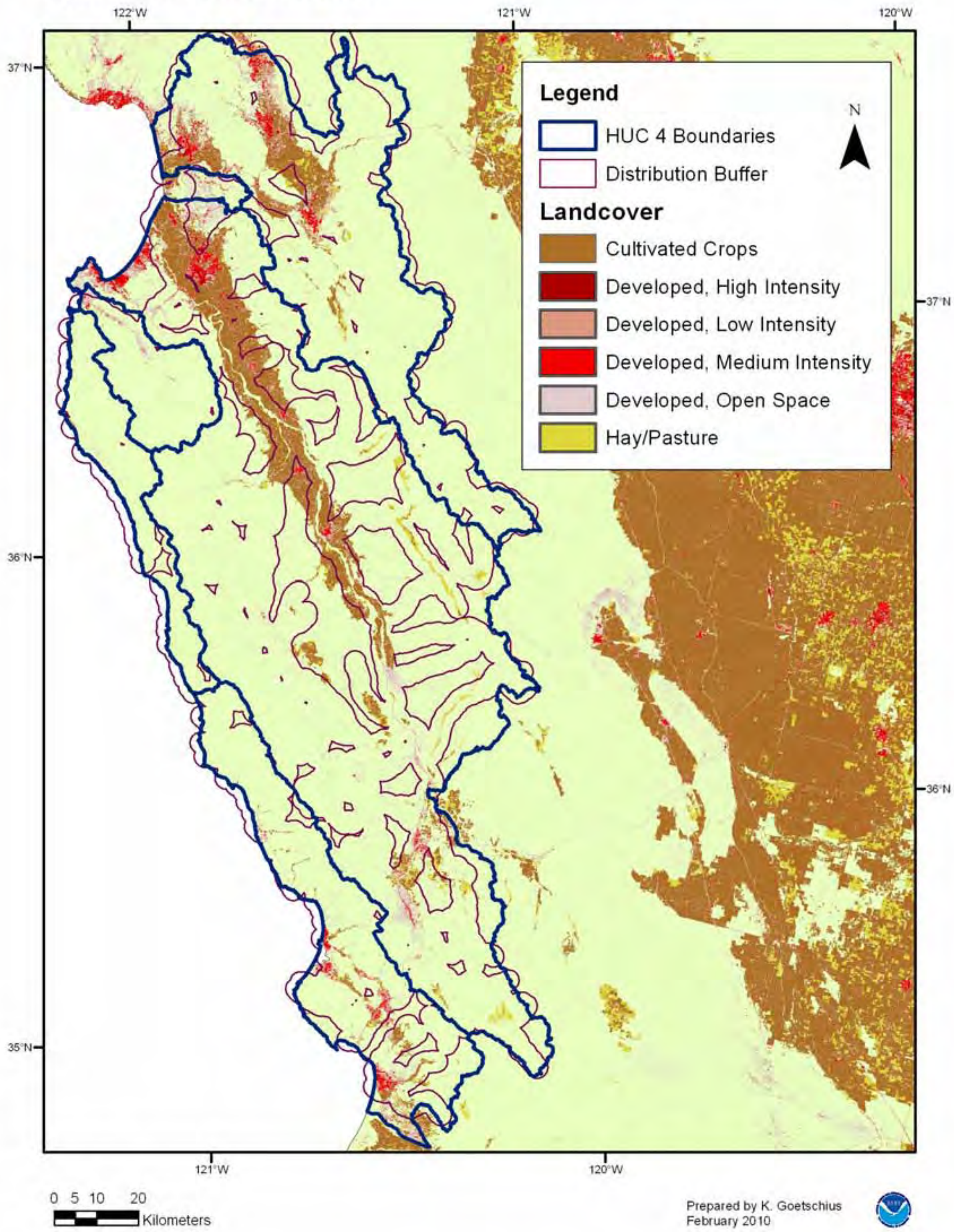
California Central Valley Steelhead DPS Species Distribution



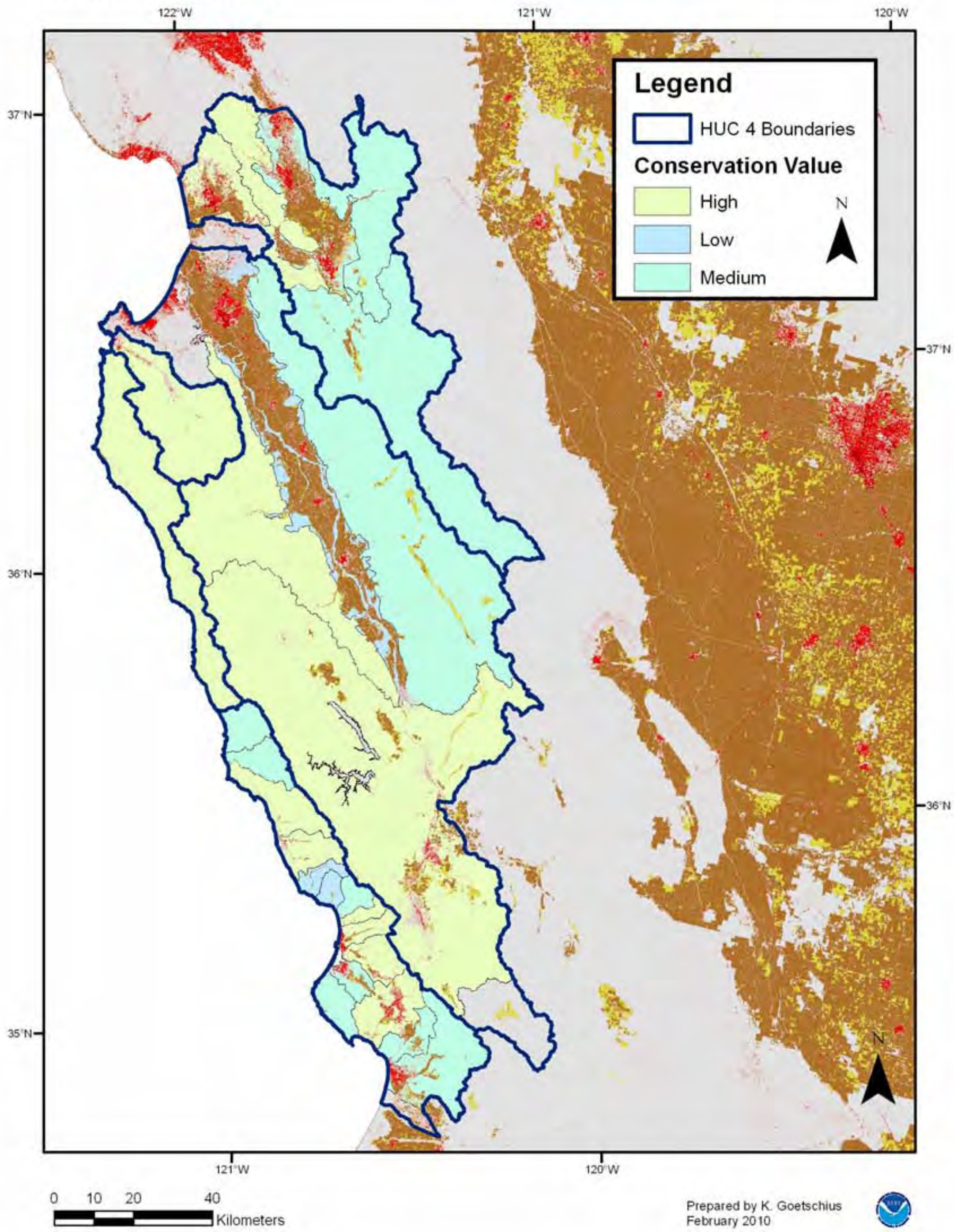
California Central Valley Steelhead DPS Critical Habitat



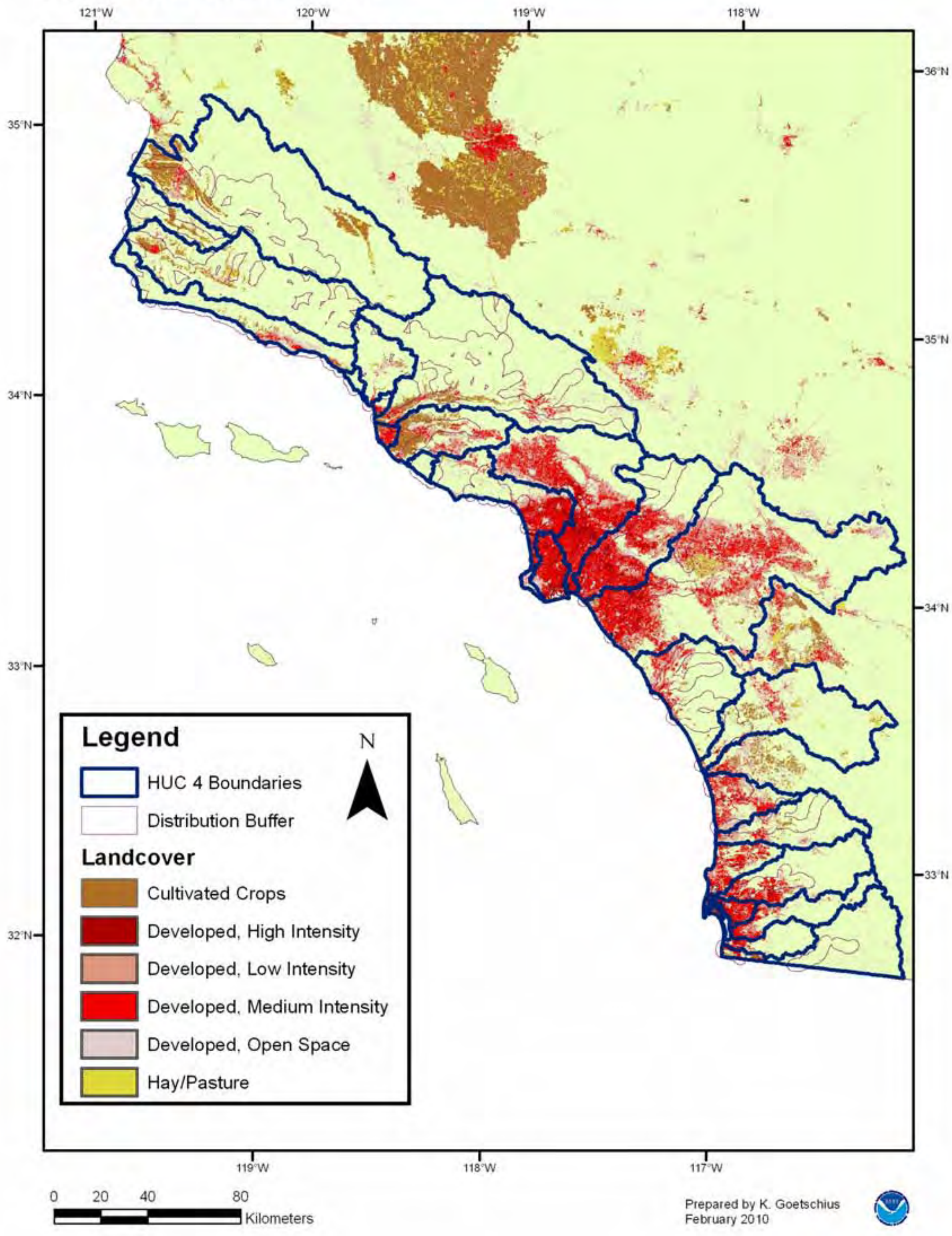
South-Central California Coastal Steelhead DPS Species Distribution



South-Central California Coastal Steelhead DPS Critical Habitat



Southern California Steelhead DPS Species Distribution



Southern California Steelhead DPS Critical Habitat