UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
WASHINGTON, D.C. 20460

OFFICE OF CHEMICAL SAFETY
AND POLLUTION PREVENTION

DRAFT BIOLOGICAL OPINIONS ISSUED UNDER THE ENDANGERED SPECIES ACT, BY THE NATIONAL MARINE FISHERIES SERVICE, RELATED TO PESTICIDES AND PACIFIC SALMON AND STEELHEAD SPECIES

COMMENTS ACCEPTED THROUGH MAY 31, 2013

EPA has initiated formal consultation with the National Marine Fisheries Service on the potential effects of certain pesticides to Pacific salmon and steelhead listed under the Endangered Species Act as either threatened or endangered. As EPA receives draft Biological Opinions relative to these consultations, they will be posted to www.epa.gov/espp and included in a public docket EPA-HQ-OPP-2008-0654 so EPA may receive public input on any changes to a pesticide's registration recommended by the National Marine Fisheries Service.

BACKGROUND

The Endangered Species Act (ESA) requires that Federal Agencies assess their "actions" to determine whether species listed as threatened or endangered under the ESA may be affected by those actions, or whether critical habitat may be adversely modified. The registered uses of a pesticide constitute an EPA "action" under the ESA.

If EPA determines a pesticide's registered uses are likely to adversely affect a federally listed threatened or endangered species (listed species) or modify its critical habitat, EPA initiates "formal consultation" with the U.S. Fish and Wildlife Service or the National Marine Fisheries Service (the Service or Services), as appropriate. In response to a Federal Agency initiating formal consultation, the Service(s) develops a Biological Opinion (BO) in which it provides its opinion on whether the "action" is likely to jeopardize the continued existence of a listed species and/or is likely to adversely modify designated critical habitat and, if so, describes alternatives to avoid jeopardy.

PUBLIC INPUT

In 2005, EPA published in the Federal Register (FR 70 No. 211 pp. 66392-66402), a document titled *Endangered Species Protection Program Field Implementation*. That notice of how EPA intends to implement its responsibilities under the ESA, states (p 66401):

> "If EPA must formally consult with the Services, after the Services issue a draft Biological Opinion, EPA will welcome input from State, Tribal and local governments **on draft reasonable and prudent measures and alternatives**. The purpose of this review would be to determine whether the alternatives or measures can be reasonably implemented and whether there are different measures that may provide adequate protection but result in less impact to pesticide users. The Agency will consider this input in developing its

response to draft Biological Opinions."

EPA also is seeking input from the public in general, pesticide users, and public interest organizations on whether the alternatives or measures can be reasonably implemented and whether there are different measures that may provide adequate protection but result in less impact to pesticide users. The Agency will also consider this input in developing its response to draft Biological Opinions.

APPLICANT INPUT

Further, the Services' Consultation Handbook (pp. 2-13), supports their consultation regulations and states:

> "… the Service and the action agency meet their obligations to [the applicant or pesticide registrant] as outlined in 50 CFR section 402 through the following: The applicant is entitled to **review draft Biological Opinions obtained through the action agency and to provide comments through the action agency. The Service will discuss the basis of their biological determination with the applicant and seek the applicant's expertise in identifying reasonable and prudent alternatives** …".

COMMENTS

Draft Biological Opinions are being included in the docket (EPA-HQ-OPP-2008-0654) and posted to EPA's Web site (http://www.epa.gov/oppfead1/endanger/litstatus/effects) to seek input on the Service's recommended reasonable and prudent measures and alternatives, as noted above. Such input should be submitted by **May 31, 2013**. Comments received by EPA will be forwarded to the Service for their consideration.

As stated in the Services' regulations (50 CFR 402.14(g)(5)):

> "All comments on the draft Biological Opinion must be submitted to the Service through the Federal agency, although the applicant may send a copy of its comments directly to the Service."

SUBMITTING YOUR COMMENTS

You may submit your comments, identified by the docket identification (ID) number EPA-HQ-OPP-2008-0654 and the pesticide to which the Biological Opinion pertains, by one of the following methods:

Federal eRulemaking Portal: http://www.regulations.gov. Follow the on-line instructions for submitting comments.

Mail: Office of Pesticide Programs (OPP) Docket, Environmental Protection Agency, Mail Code 28221T, 1200 Pennsylvania Ave. NW., Washington, DC 20460.

Delivery: EPA Docket Center (EPA/DC), EPA West, Room 3334, 1301 Constitution Ave. NW., Washington, DC 20460. Such deliveries are only accepted during the Docket's normal hours of operation, and special arrangements should be made for deliveries of boxed information.

Instructions: EPA's policy is that all comments received will be included in the docket without change and may be made available on-line at http://www.regulations.gov, including any personal information provided, unless the comment includes information claimed to be Confidential Business Information (CBI) or other information whose disclosure is restricted by statute. Do not submit information that you consider to be CBI or otherwise protected through regulations.gov or e-mail. The regulations.gov website is an "anonymous access" system, which means EPA will not know your identity or contact information unless you provide it in the body of your comment. If you send an e-mail comment directly to EPA without going through regulations.gov, your e-mail address will be automatically captured and included as part of the comment that is placed in the docket and made available on the Internet. If you submit an electronic comment, EPA recommends that you include your name and other contact information in the body of your comment and with any disk or CD-ROM you submit. If EPA cannot read your comment due to technical difficulties and cannot contact you for clarification, EPA may not be able to consider your comment. Electronic files should avoid the use of special characters, any form of encryption, and be free of any defects or viruses.

Docket: All documents in the docket are listed in the docket index available at http://www.regulations.gov. To access the electronic docket, go to http://www.regulations.gov, select "Advanced Search," then "Docket Search." Insert the docket ID number where indicated and select the "Submit" button. Follow the instructions on the regulations.gov website to view the docket index or access available documents. Although listed in the index, some information is not publicly available, e.g., CBI or other information whose disclosure is restricted by statute. Certain other material, such as copyrighted material, is not placed on the Internet and will be publicly available only in hard copy form. Publicly available docket materials are available either in the electronic docket at http://www.regulations.gov, or, if only available in hard copy, at the OPP Docket located in the EPA/DC, Room 3334, EPA West, 1301 Constitution Ave. NW., Washington, DC. The EPA/DC Public Reading Room hours of operation are 8:30 a.m. to 4:30 p.m., Monday through Friday, excluding legal holidays. The telephone number of the Public Reading Room is (202) 566-1744, and the telephone number for the OPP Docket is (703) 305-5805. EPA/DC visitors are required to show photographic identification, pass through a metal detector, and sign the EPA visitor log. All visitor bags are processed through an X-ray machine and subject to search. Visitors will be provided an EPA/DC badge that must be visible at all times in the building and returned upon departure.

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

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

Dear Dr. Bradbury:

We are pleased to provide the National Oceanic and Atmospheric Administration, National Marine Fisheries Service's (NMFS) draft conference and biological opinion (Opinion), issued under the authority of section 7(a)(2) and 7(a)(3) of the Endangered Species Act (ESA) of 1973 as amended (16 U.S.C. 1536(a)(2)), on the effects of the U.S. Environmental Protection Agency's (EPA) registration of diflubenzuron, fenbutatin oxide, and propargite on threatened and endangered salmonids, and proposed and designated critical habitat.

Our agencies worked together and with applicants (pesticide registrants) and the U.S. Department of Agriculture (USDA) on up-front changes to the action (pesticide labels). Applicants proposed label changes specifying pesticide uses that reduce drift and loading in salmon habitats. Although we assessed all proposed label changes in our analyses, the proposed changes did not provide sufficient protections to avoid jeopardy or adverse modification of proposed and designated critical habitat for most listed salmonid species without Reasonable and Prudent Alternatives.

We discussed the Reasonable and Prudent Alternatives included in the draft Opinion with EPA and USDA on April 30, 2013. We continue to work with EPA, applicants, and USDA to refine our draft Reasonable and Prudent Alternatives for inclusion in our final biological opinion. We will consider alternative Reasonable and Prudent Alternatives or additional label changes that provide protections equivalent to Reasonable and Prudent Alternatives in the draft Opinion. If Reasonable and Prudent Alternatives are incorporated as changes to pesticide labeling for affected species, then jeopardy and adverse modification for those species would be avoided.

On April 26, EPA and NMFS received a prepublication copy of the National Academy of Sciences' National Research Council (NRC) report Assessing Risk to Endangered and Threatened Species from Pesticides, and were briefed on the report on April 29. The NRC panel studied on specific scientific and technical issues related to pesticides risk assessments for listed species, and the report contains multiple conclusions and recommendations. At this time, we do not know in what ways our approach to consultation might change in response to this report, or whether any changes to the consultation approach would alter any conclusions. Because we wished to provide time for your review, we submit the draft without consideration or discussion of the NRC report. However, our agencies along with the U.S. Fish and Wildlife Service and USDA have already begun working together to address the implications of the report for each agency's risk assessment methodology.

If you have questions regarding this Opinion please contact me or Mr. Stan Rogers, Chief (Acting), Endangered Species Act Interagency Cooperation Division, at (301) 427-8478.

Sincerely,

Perry Gayaldo

for Helen M. Golde
Director (Acting), Office of Protected Resources

Endangered Species Act Section 7 Consultation

Draft Conference and Biological Opinion

Environmental Protection Agency Registration of Pesticides

Containing

Diflubenzuron, Fenbutatin oxide, and Propargite

May 1, 2013 Draft

1 Contents

National Marine Fisheries Service

Endangered Species Act Section 7 Consultation

Draft Conference and Biological Opinion

Agency: United States Environmental Protection Agency

Activities Considered: Authorization of pesticide products (as described by product labels) containing the active ingredients diflubenzuron, fenbutatin oxide, and propargite, and their formulations in the United States and its affiliated territories.

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

Approved by: ____

Date: ____

Section 7(a)(2) of the Endangered Species Act of 1973, as amended (ESA; 16 U.S.C. §1531 *et seq.*) requires each federal agency to insure that any action they authorize, fund, or carry out is not likely to jeopardize the continued existence of any endangered or threatened species or result in the destruction or adverse modification of critical habitat of such species. When a federal agency's action "may affect" a protected species or designated critical habitat, that agency is required to consult formally with the National Marine Fisheries Service (NMFS) or the U.S. Fish and Wildlife Service (USFWS)[1], depending upon the endangered species, threatened species, or

[1] Generally, NMFS conducts consultation for marine and anadromous species, while FWS conducts consultations for freshwater and terrestrial species.

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) submitted consultation requests with NMFS on its proposals to authorize use, pursuant to the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA), 7 U.S.C. 136 *et seq.*, of pesticide products containing the active ingredients (a.i.) diflubenzuron, fenbutatin oxide, and propargite on August 1, 2002. EPA authorization of pesticide uses are categorized as FIFRA sections 3 (new product registrations), 4 (re-registrations and special review), 18 (emergency use), or 24(c) [Special Local Needs (SLN)]. At that time, EPA determined uses of pesticide products containing diflubenzuron would have no effect on 10 Evolutionarily Significant Units (ESU)/ Distinct Population Segments (DPS) and may affect but were not likely to adversely affect 16. EPA determined uses of pesticide products containing fenbutatin oxide would have no effect on 2 ESUs/DPSs, may affect but were not likely to adversely affect 1, and may adversely affect 23 of the 26 ESUs/DPSs. EPA determined uses of pesticide products containing propargite would have no effect on 7 ESUs/DPSs, may affect but were not likely to adversely affect 12, and may adversely affect 7 of the 26 ESUs/DPSs. Lower Columbia River coho and Puget Sound steelhead were listed later on June 28, 2005 and May 11, 2007 respectively and designated critical habitat was proposed for both species on January 14, 2013. EPA did not make adverse modification determinations for any of the a.i.s for any of the ESUs/DPSs which had proposed or designated critical habitat. This document represents NMFS' biological and conference opinion (Opinion) on the impacts of EPA's separate authorizations of pesticide products containing diflubenzuron, fenbutatin oxide, and propargite on the listed ESUs/DPSs. NMFS is considering these authorizations in one opinion because the chemicals are all insecticides. This is a partial consultation intended to comply with a court order[2] requiring EPA to make a determination on

[2] *Northwest Coalition for Alternatives to Pesticides vs. National Marine Fisheries Service*, Civ. NO 07-1791 (W.D. WA).

the effect of diflubenzuron, fenbutatin oxide, and propargite and 51 other active ingredients on listed Pacific Salmonids.[3] Consultation with NMFS will not be complete for registration of these a.i.s until EPA makes effect determinations on all other species and designated critical habitat under NMFS jurisdiction and consults with NMFS as necessary.

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

This Opinion is based on NMFS' review of the package of information the EPA submitted with its 2002 and 2004 requests for consultation on the proposed authorizations of the above a.i.s. It also includes our review of recovery plans for listed Pacific salmonids, past and current research, monitoring reports from prior research, previous Opinions, published and unpublished scientific information on the biology and ecology of threatened and endangered salmonids in the action area, and other sources of information gathered and evaluated during the consultation on the proposed authorization of products containing the a.i.s diflubenzuron, fenbutatin oxide, or propargite. NMFS also reviewed pesticide labels, available monitoring data and other local, county, and state information, online toxicity databases, incident reports, data generated by pesticide registrants, and exposure models run by NMFS. NMFS also considered information and comments provided by EPA, by the registrants identified as applicants by EPA, and information and comments submitted by others during EPA's public comment process.

[3] Two additional Pacific salmonids species have been listed since the court order. Although the court's order did not address these two species, NMFS analyzed the effects of EPA's action to them because they belong to the same taxon and require consideration of the same information.

2 Background

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

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

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

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

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

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

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

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

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

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

On July 2, 2012, NMFS issued its sixth Opinion. This sixth consultation evaluated the herbicide thiobencarb.

3 Consultation History

EPA requested consultation for the registration of diflubenzuron on July 29, 2004, fenbutatin oxide on November 29, 2002, and propargite on July 25, 2002. Since 2008, NMFS has completed six Biological Opinions covering other active ingredients.

In meetings held on September 20 (fenbutatin oxide), 21 (diflubenzuron), and 27 (propargite), 2011, NMFS, EPA, and registrants discussed the consultation process and the role of the applicant in the consultation process, and presented general information to assist with consultation.

Since the initial meetings, pesticide registrants and EPA provided information via regular mail, email, and telephone conversations for the purposes of consultation. The types of information exchanged included active pesticide labels, scientific studies, and pesticide use information. The exchange of studies and information continued throughout the consultation process as new studies were requested and submitted. Information exchange between applicants and NMFS occurred via EPA.

On July 31, 2012, NMFS and EPA discussed, via phone, holding additional applicant meetings with EPA, pesticide registrants, and USDA as we continued consultation on diflubenzuron, fenbutatin oxide, and propargite. We communicated that the purpose of these meetings would be to work more closely with EPA, registrants, and USDA earlier in the consultation process, to clarify the action (labels), and to identify up-front risk reduction measures.

NMFS also requested that EPA update the list of active pesticide registrations for these chemicals and submit any new labels registered since the original set of labels were sent to NMFS in 2011. Between August 28, 2012 and April 17, EPA sent updated label information to NMFS.

We scheduled a meeting with EPA, USDA, and diflubenzuron registrants on October 4, 2012, with EPA, USDA, and propargite registrants on October 25, 2012, and with EPA, USDA, and fenbutatin oxide registrants on November 8, 2012. NMFS communicated to EPA and registrants that the goals of the meeting were to (1) present and receive feedback from applicants on the action; (2) highlight to applicants and EPA likely risks to listed salmonids, e.g., effects on salmonid prey, (3) discuss and receive input from applicants and EPA on potential label changes that would reduce risk to listed salmonids.

Prior to each of the meetings we provided meeting participants with two files for use during the meeting: (1) a spreadsheet indicating pesticide products containing the pesticide under consultation that are currently registered with EPA, and (2) a summary label specifications including uses, formulation types, application types and other information used in the assessment. NMFS used these files to guide discussions during the meetings and all participants were able to ask for clarifications about label specifications, update the lists of active pesticide products as needed, and discuss risks posed by specific uses, application rates, and application types.

At all three meetings, pesticide registrants indicated their interest in making clarifications on labels where needed and their willingness to consider changes to the action (label changes) that would reduce risks to listed species. All parties agreed to organize a follow-up meeting to communicate label changes and clarifications.

The follow-up meeting for diflubenzuron occurred on November 5, 2012, for propargite on December 13, 2012, and for fenbutatin oxide on January 16, 2013. EPA and USDA were also in attendance at all meetings. At these meetings, pesticide registrants presented their proposed label changes. Registrants also asked for more detailed modeling results before considering more changes. NMFS responded that we were still in progress of conducting the analysis. We also noted that there would be time between the draft opinion and final opinion to consider further label changes if necessary and within the constraints of the court-approved schedule.

At the January 16, 2013 meeting with EPA, USDA, and fenbutatin oxide registrants, NMFS and EPA committed to a preliminary consideration of the registrant's proposed 25-ft buffer. NMFS and EPA held a meeting on February 4, 2013 to discuss preliminary modeling results of the 25-ft buffers. NMFS and EPA preliminary model results were similar and both agencies agreed that 25-ft buffers were not sufficient to avoid jeopardy for listed species. Both agencies agreed to reach back out to registrants with the information in case they would like to make further label changes. EPA coordinated a meeting with registrants to explain that 25-ft buffers were not sufficient to avoid jeopardy. The meeting was held on March 18, 2013. Prior to the meeting, NMFS provided an email with model results indicating that a 500-ft buffer, or risk reductions equivalent to a 500-ft buffer, would likely avoid jeopardy to listed salmonids in this consultation. NMFS and EPA provided examples of other measures, or combinations of measures, that could be implemented if they were equally as effective as a 500-ft buffer. Applicants asked for model inputs to run the models for their own analysis. NMFS explained and shared model inputs with EPA, USDA, and applicants.

USDA was engaged throughout the consultation process. USDA participated in meetings with EPA, NMFS, and applicants and acted as liaison to the agricultural community and some pesticide user groups. For example, in February USDA reached out to University of California Farm Advisors representing the four California counties with the majority of fenbutatin oxide use (according to the California PUR database). This outreach provided information on how fenbutatin oxide is used in these areas and has the potential to be useful for making decisions about label changes and/or other conservation measures as needed.

NMFS also met with other agencies to discuss use patterns for pesticides under consultation. On January 23, 2013, NMFS met with the Washington State Department of Agriculture to discuss the most up to date pesticide use summaries for fenbutatin oxide and propargite. There is currently no use information for diflubenzuron in Washington.

On March 29, 2013 NMFS shared the draft description of the action with EPA and USDA. We asked EPA to share the draft with the applicants. We shared the draft to give all parties an opportunity to review the description of the action before completing the draft opinion. There were a number of labels changes proposed by applicants of all three chemicals under consultation. Review of the draft opinion is an opportunity to ensure that NMFS captured those changes to the action accurately and represents agreement among NMFS, EPA, and the applicants on the action.

Propargite applicants indicated that we had not captured all of their proposed changes in the draft description of the action. Applicants provided comments on the document and requested a meeting to discuss changes. EPA coordinated a meeting with NMFS and propargite applicants on April 3, 2013. During the meeting registrants and NMFS discussed errors and edits noted by the applicants. NMFS made the edits to the draft description of the action communicated by the propargite applicants and emailed the edited document back to EPA and applicants for their review on April 4, 2013. On April 8, 2013, NMFS received final edits from propargite applicants on the draft description of the action.

On April 4, 2013, EPA provided comments from fenbutatin oxide applicants on the draft description of the action.

Diflubenzuron applicants also indicated that we had not captured all of their proposed changes in the draft description of the action. Registrants provided comments on the document and requested a meeting to discuss changes. EPA coordinated a meeting with NMFS and diflubenzuron applicants on April 15, 2013. During the meeting, registrants and NMFS discussed errors and edits noted by the applicants. NMFS made the edits to the draft description of the action communicated by the diflubenzuron applicants. During this meeting, diflubenzuron registrants also made several additional changes to the action and presented two new studies they prepared regarding use of diflubenzuron in livestock/poultry premises. The studies were transmitted to NMFS on April 10, 2013. NMFS indicated that given the late arrival of these studies, they may not be considered in the draft biological opinion. However, NMFS agreed to consider this information before the final opinion is released.

On April 19, 2013 EPA provided new PRZM-EXAMS estimates for diflubenzuron use on manure for indoor applications and subsequent applications of manure (spreading) to agricultural fields. NMFS included these exposure estimates in the analysis.

Resources Addressed in the BEs

EPA's BEs considered the effects of pesticides containing the 3 a.i.s to 26 species of listed Pacific salmonids and their designated critical habitat. Two species listed more recently, the Lower Columbia River coho and the Puget Sound steelhead, were not considered in the BEs. EPA determined that fenbutatin oxide and propargite may adversely affect some ESUs, may affect but are not likely to adversely affect other ESUs and have no effect on the rest of the

ESUs. EPA determined that its action of registering pesticide products containing diflubenzuron may affect but is not likely to adversely affect some endangered or threatened Pacific salmon and steelhead and have no effect on the rest of the ESUs. EPA made no determinations as to effects of its actions on designated critical habitat. When an action agency concludes its action will not affect any listed species or critical habitat, no section 7 consultation is necessary (USFWS, & NMFS 1998). Formal consultation on the proposed action is necessary because EPA concluded for two of the a.i.s that the proposed action may adversely affect some listed Pacific anadromous salmonids and their designated critical habitat, and NMFS did not concur with any of EPA's "NLAA" determinations for each of the three a.i.s. Once formal consultation is triggered, NMFS evaluate impacts to all affected species and all designated critical habitat. In this Opinion, NMFS will analyze the impacts to all ESUs/DPSs of Pacific salmonids and designated critical habitat 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.

4 Description of the Action

4.1 The Federal Action

EPA's registrations of all pesticides containing diflubenzuron, propargite, or fenbutatin oxide for use as described on product labels is the proposed action for this consultation.[4] The proposed action includes (1) approved product labels containing diflubenzuron, propargite, and fenbutatin oxide, (2) degradates and metabolites of diflubenzuron, propargite, and fenbutatin oxide, (3) formulations, including other ingredients within formulations, (4) adjuvants, and (5) tank mixtures (Figure 1). EPA registers pesticides under FIFRA to provide tools for pest control that do not cause unreasonable adverse effects to the environment throughout the U.S. and its affiliated territories. EPA's goal for reassessing currently registered pesticide active ingredients is every 15 years. Given EPA's timeframe for pesticide registration reviews, NMFS' evaluation of the proposed action is also 15 years. No interrelated and interdependent activities are associated with the proposed action. The purpose of EPA's proposed action is to provide pest control that does 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 registered with a label identifying approved uses by EPA's Office of Pesticide Programs (OPP). Once registered, a pesticide may not legally be used unless the use is consistent with directions on its approved label(s) (http:www.epa.gov/pesticides/regulating/registering/index.htm). EPA authorization of pesticide uses are categorized as FIFRA sections 3 (new product registrations), 4 (re-registrations and special review), 18 (emergency use), or 24(c) Special Local Needs (SLN).

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

[4] EPA's registrations are three separate actions that we have combined in one opinion. For convenience, we will refer to one action.

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."

EPA, NMFS, and FWS agreed on December 12, 2007 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 encompasses the impacts to listed Pacific salmonid ESUs/DPSs of all uses authorized by EPA, regardless of whether those uses have historically occurred.

EPA provided copies of active product labels for the three pesticides under consultation. After reviewing the labels provided, NMFS, EPA, USDA, and pesticide registrants (applicants) held meetings to clarify labels, provide updates to labels, and document any label cancellations. At these meetings, the agencies and pesticide registrants also discussed potential risks to salmonids and ways to reduce risk. In some cases, pesticide registrants proposed changes to their existing labels that reduce risks to salmonids. The following description of diflubenzuron, propargite, and fenbutatin oxide registrations (the action) represents information acquired from labels, including the label changes agreed upon by registrants and EPA, and from EPA's BEs, REDs, and other documents.

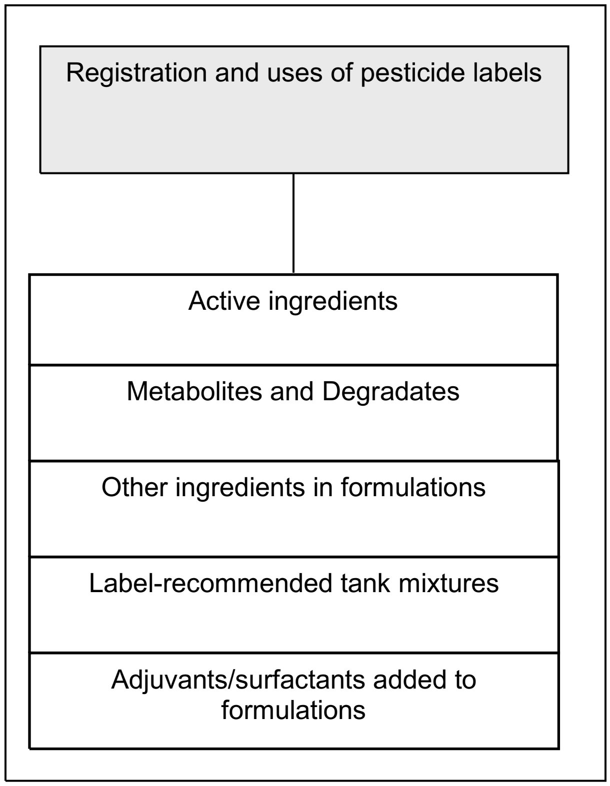

Figure 1. Stressors of the action

4.1.1 Pesticide Labels.

For this consultation, EPA's proposed action encompasses all approved product labels containing diflubenzuron, fenbutatin-oxide, or propargite, including degradates, metabolites, and formulations, other ingredients within the formulations, adjuvants, and tank mixtures. Combined these comprise the stressors of the action (Figure 1). These a.i.s combined are labeled for a variety of uses including applications to croplands, pastures, non-crop areas (field border, fence rows, roadsides, farmsteads, ditch banks), developed areas, recreational areas (e.g., campgrounds, golf courses, parks, parkways), shelterbelts, rights of way, easements, trees and shrubs in public and private forests, forest plantings and forest nurseries, Christmas tree and conifer nurseries, residential areas and landscape plantings.

4.1.2 Active and Other ingredients

Diflubenzuron, fenbutatin oxide, and propargite are the a.i.s that kill or otherwise affect targeted organisms (described on labels). Pesticide products contain these a.i.s and also contain other ingredients (referred to as "inerts" or "other" ingredients on the labels). Inert ingredients are ingredients which EPA defines as not "pesticidally" active. 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 intended 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 used in many common household products. Nonylphenol (NP), one of the APEs, has been linked to endocrine-disrupting effects in aquatic animals (described in the *Effects of the Proposed Action* section).

4.1.3 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.

4.1.4 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.

4.1.5 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 that mitigate risk to humans and the environment. Mitigations may include phase-out and cancellation of uses and pesticide products. The terms of EPA's regulatory decisions are summarized in RED documents (EPA 1997a, 1994b), (2001a). Registrants can submit applications for the registration of new products and new uses following reregistration of an active ingredient. Several types of products are registered, including the pure (or nearly pure) active ingredient, often referred to as technical grade active ingredient (TGAI), technical, or technical product. This is generally used in manufacturing and testing, and not applied directly to crops or other use sites. Products that are applied to crops or other use sites (e.g., rights of way, landscaping), 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 actions includes all authorizations for use of pesticide products including use of existing stocks, and active labels, of products containing the three a.i.s for the duration of the proposed action.

4.1.6 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 15 years.

4.1.7 Interrelated and Interdependent Activities

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

4.1.8 Registration Information of Pesticide a.i.s under Consultation

As discussed above, the proposed action encompasses EPA's registration of all the product uses described on labels (and proposed changes to those labels as described in the Consultation History and below) of all pesticides containing diflubenzuron, fenbutatin-oxide, and propargite. EPA and applicants provided copies of all active product labels for the three 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.

4.2 Diflubenzuron

Diflubenzuron was first registered in the United States in 1976 for use as an insecticide. The insecticide behaves as a chitin inhibitor to inhibit the growth of many leaf-eating larvae, mosquito larvae, aquatic midges, rust mite, boll weevil, and flies. Diflubenzuron is used primarily on citrus, cattle, cotton, forestry, mushrooms, ornamentals, pastures, soybeans, standing water, sewage systems, and wide-area general outdoor treatment sites (EPA 1997a, 2009c). It was first produced for use against gypsy moth larvae in forested areas, including Christmas tree plantations and nursery crops grown in proximity to gypsy moth infested areas. Cotton was added to the registration in 1979 to control certain lepidopterous larvae and boll weevils during the growing seasons and at the end of the growing season to reduce the size of the boll weevil population.

Soybeans were added to the registration in 1982 to control lepidopterous larvae such as green cloverworm and velvetbean caterpillar. According to EPA, diflubenzuron is used particularly when there is a surge in the population of these larvae or resistance precludes use of standard insecticides (EPA 1997a, 2009c). Mushrooms were added to the label in 1983 to control fungus gnats or mushroom flies. Diflubenzuron is incorporated into the compost when it is applied, or as a drench after the casing is spread, covering the compost containing the spawn.

Boluses for cattle were first registered in 1985. The bolus stays in the stomach of treated animals, slowly eroding and releasing diflubenzuron which prevents molting by maggots of flies breeding in manure. Also in 1985, mosquito control was added to the label controlling breeding in irrigation water tailings, and waste water drained from irrigated fields

4.2.1 Usage information

4.2.1.1 Agricultural Uses

Diflubenzuron is applied on a variety of crops. It is applied on alfalfa and clover grown for seed, artichokes, bell and non-bell peppers, citrus, oats and barley (except CA), pummelos, cotton, an assortment of brassicas, ornamentals, peanuts, pear, rice (CA only), soybeans (except CA), a variety of stonefruit trees and nut trees, triticale (except CA), turfgrass (sod farms) and wheat (Table 1). Diflubenzuron is also applied to feed for cattle and horses. This a.i. is also used in commercial fish production ponds and is applied to non-crop areas livestock premises.

4.2.1.2 Non-agricultural Uses.

Diflubenzuron is used on fence rows, roadsides, grasslands, subterranean and above ground termite bait stations, trees and shrubs in public and private forests, forest plantings and forest nurseries, Christmas tree and conifer nurseries, residential areas and landscape plantings, and recreational areas (campgrounds, golf courses, parks, parkways).

4.2.1.3 Registered Formulation Types.

Diflubenzuron products are formulated as pellets, aqueous flowables, wettable powders, suspension concentrates, liquid concentrates, water dispersible granules in water soluble packages, nutritional supplements for cattle, concentrates for cattle, and termite bait cartridges.

4.2.2 Methods and Rates of Application

4.2.2.1 Methods

Diflubenzuron can be applied using a variety of methods and equipment. It may be applied as baits, in feed for livestock or as a spot treatment to control flies in manure, broadcast using aircraft (fixed wing or helicopter), ground boom sprayers, and hand held nozzle sprayers.

4.2.2.2 Application Rates

Most single application rates of diflubenzuron are limited to less than a pound per acre on all crops (Table 1). Sites with the greatest application rates include pear (0.75 lb/A), and containerized ornamentals in greenhouses (up to 0.81 lb/A). Non-crop uses to control flies around livestock/poultry operations allow for application rates exceeding 8 lbs/A – however, the broadcast application that employs this high rate is limited to use in indoor poultry houses, and will be clarified in new proposed label wording. Spot treatment applications inside and outside of livestock / poultry premises will be restricted to rates of 0.117 lbs/A per application, and a total of 2 lbs/A per year in registrants new proposed label wording. Multiple applications are permitted on several use sites. Typically, either the maximum number of applications and/or maximum seasonal rate is specified.

4.2.3 Metabolites and Degradates

Several degradates of diflubenzuron have been identified in laboratory and field studies including CPU, 2,6-diflubenzoic acid (DFBA), 4-choroaniline (PCA), 2,6-diflubenzamide (DFBAM), and 2,6-difluorobenzene.

Table 1. Summary of diflubenzuron uses and use rates based on labels and meetings with EPA and applicants.

Label No.	Use	Use Site	Land Use Category	Max. Single App. Rate lbs a.i./A	Max. No. of App. Per Year	Annual App Rate lbs a.i./A	Min. App. Interval (days)	App. Method	Buffer[5]
OR-080032, EPA 400-461	Alfalfa grown for seed	Crop	Agriculture	0.03125	2	0.0625	10	Ground, aerial	YES
WA-000024, EPA 400-461	Alfalfa grown for seed	Crop	Agriculture	0.03125	2	0.0625	10	Ground, aerial	YES
ID-000013, EPA 400-461	Alfalfa/clover grown for seed	Crop	Agriculture	0.03125	2	0.0625	10	Ground, aerial	YES
CA-970009, EPA 400-461	Artichokes	Crop	Agriculture	0.25	NS	0.75	15	Ground, aerial	YES
400-	Barley (Except CA)	Crop	Agriculture	0.0625	1	0.0625	NA	Ground,	YES

[5] Do not apply within 25 feet by ground or 150 feet by air of bodies of water such as lakes, reservoirs, rivers, permanent streams, natural ponds, marshes or estuaries. All applications must include a 25 foot vegetative buffer strip within the buffer zone to decrease runoff.

Notes: bw=body weight.

Label No.	Use	Use Site	Land Use Category	Max. Single App. Rate lbs a.i./A	Max. No. of App. Per Year	Annual App Rate lbs a.i./A	Min. App. Interval (days)	App. Method	Buffer[5]
461								aerial	
400-465	Bell and Non-bell peppers	Crop	Agriculture	0.125	5	0.375	7	Ground, aerial	YES
400-461	Bell and Non-bell peppers	Crop	Agriculture	0.125	5	0.375	7	Ground, aerial	YES
400-461	Citrus (Oranges, Grapefruit, Tangerine, Pummelo, and their hybrids)	Crop	Agriculture	0.3125	3-full rate, 6-split rate	0.9375	90 (30 day-pending)	Ground, aerial	YES
400-476	Citrus (Oranges, Grapefruit, Tangerines)	Crop	Agriculture	0.3125	3-full rate, 6-split rate	0.9375	30	Ground, aerial	YES
400-487	Citrus (Oranges, grapefruits, Tangerine, Pummelos / Pomelos and their hybrids	Crop	Agriculture	0.3125	3-full rate, 6-split rate	0.9375	90 (30 day-pending)	Ground, aerial	YES
OR-080033, EPA 400-461	Commercial hybrid poplar/cottonwood plantations	Crop	Agriculture	0.25	2	0.025	NS	Ground, aerial	YES
WA-020008, EPA 400-	Commercial hybrid poplar/cottonwood plantations	Crop	Agriculture	0.25	2	0.25	NS	Ground, aerial	YES

Label No.	Use	Use Site	Land Use Category	Max. Single App. Rate lbs a.i./A	Max. No. of App. Per Year	Annual App Rate lbs a.i./A	Min. App. Interval (days)	App. Method	Buffer[5]
461									
400-461	Cotton	Crop	Agriculture	0.125	6	0.375	5, or NS	Ground, aerial	YES
400-465	Cotton	Crop	Agriculture	0.125	6	0.375	5, or NS	Ground, aerial	YES
400-461	Leafy brassica (Broccoli raab, Chinese cabbage (bok choy), Collards, Kale, Mizuna, Mustard greens, Mustard spinach, Rape greens, Turnip greens)	Crop	Agriculture	0.0625	4	0.25	7	Ground	YES
400-461	Oats (Except CA)	Crop	Agriculture	0.0625	1	0.0625	NA	Ground, aerial	YES
400-465	Ornamentals (greenhouse or field chrysanthemums)	Crop	Agriculture	0.125	26	3.25	14	Ground	YES
400-483	Ornamentals (greenhouse)	Crop	Agriculture	0.27 0.54 0.81	NS NS NS	NS NS NS	7 28 28	Foliar under bench soil drench	NO
400-477	Ornamentals (greenhouse)	Crop	Agriculture	0.27 0.54 0.81	NS NS NS	NS NS NS	7 28 28	Foliar under bench soil drench	NO

Label No.	Use	Use Site	Land Use Category	Max. Single App. Rate lbs a.i./A	Max. No. of App. Per Year	Annual App Rate lbs a.i./A	Min. App. Interval (days)	App. Method	Buffer[5]
400-461	Peanuts	Crop	Agriculture	0.125	3	0.375	14	Ground, aerial	YES
400-461	Pear	Crop	Agriculture	0.75	4	1	14	Ground	YES
400-465	Pear	Crop	Agriculture	0.75	4	1	14	Ground	YES
400-461	Rice (CA)	Crop	Agriculture	0.25	1	0.25	NA	Aerial, retain treated water for 14 days	YES
400-461	Soybean (Except CA)	Crop	Agriculture	0.0625	2	0.125	30	Ground, aerial	YES
400-465	Soybean (Except CA)	Crop	Agriculture	0.0625	2	0.125	30	NS	YES
400-461	Stonefruit (Apricot, Nectarine, Peach, Plum, Prune) does not include Cherries	Crop	Agriculture	0.25	2	0.5	21	Ground	YES
400-461	Tree nuts (Almond, Beech nut, Brazil nut, butternut, chestnut,	Crop	Agriculture	0.25	4	1	21	Ground	YES
400-461	Triticale (except CA)	Crop	Agriculture	0.0625	1	0.0625	NA	Ground, aerial	YES
400-461	Turfgrass (sod farms only)	Crop	Agriculture	0.03125	4	0.125	14	NS	YES

Label No.	Use	Use Site	Land Use Category	Max. Single App. Rate lbs a.i./A	Max. No. of App. Per Year	Annual App Rate lbs a.i./A	Min. App. Interval (days)	App. Method	Buffer[5]
400-461	Wheat	Crop	Agriculture	0.0625	1	0.0625	NA	Ground, aerial	YES
400-472	Cattle (beef and dairy)	Non-crop agriculture	Agriculture	4.75 g/bolus dose	1	NA	NA	Bolus orally administere d with balling gun	NO
400-523	Cattle	Non-crop agriculture	Agriculture	0.10 mg/kg bw	NS	NS	Daily	Cattle feed	NO
400-536	Cattle	Non-crop agriculture	Agriculture	0.10 mg/kg bw	NS	NS	Daily	Cattle feed	NO
400-537	Cattle	Non-crop agriculture	Agriculture	0.10 mg/kg bw	NS	NS	Daily	Cattle feed	NO
2724-795	Cattle	Non-crop Agriculture	Agriculture	0.1 mg a.i./kg bw	NS	NS	NA	Cattle feed	NO
2724-794	Cattle	Non-crop agriculture	Agriculture	0.1 mg a.i./kg bw	NS	NS	1	Cattle feed	NO
61483-91	Cattle and Horses	Non-crop agriculture	Agriculture	3ml of 5% formulation / 100 lbs bw	NA	NA	NA	Pour, wipe, or spray on animal	NO
2724-798	Cattle and Horses	Non-crop agriculture	Agriculture	0.15 mg a.i./kg bw	NS	NS	1	Feed	NO
270-339	Horses	Non-crop agriculture	Agriculture	0.15 mg a.i./kg bw	NS	NS	1	Feed	NO

<table>
<tr><th>Label No.</th><th>Use</th><th>Use Site</th><th>Land Use Category</th><th>Max. Single App. Rate lbs a.i./A</th><th>Max. No. of App. Per Year</th><th>Annual App Rate lbs a.i./A</th><th>Min. App. Interval (days)</th><th>App. Method</th><th>Buffer[5]</th></tr>
<tr><td>400-465</td><td>Commercial fish production ponds and tanks (ornamental fish and bait fish)</td><td>Non-crop Agriculture</td><td>Agriculture</td><td>67 µg/L in closed system</td><td>NS</td><td>NS</td><td>14</td><td>Uniform application to water in a contained system- 14 d hold before release</td><td>YES</td></tr>
<tr><td rowspan="3">400-461[6]</td><td rowspan="3">Livestock/Poultry Premises: litter, stale/waste feed, manure, manure/straw mixture, feed muck/spoilage, spoiled organic refuse, bedding material, floors/walls, posts, cage frames, ceilings</td><td rowspan="3">Non-crop Agriculture</td><td rowspan="3">Agriculture</td><td rowspan="2">8.2</td><td>NS, but 1 / productio n cycle</td><td>NS</td><td>NS</td><td>Broadcast[7] (indoor poultry use only)</td><td rowspan="3">YES</td></tr>
<tr><td>NS, but 1 / productio n cycle</td><td>NS</td><td>NS</td><td>Band (indoor poultry use only)</td></tr>
<tr><td>0.117</td><td>NS</td><td>2.0</td><td>21</td><td>Spot treatments</td></tr>
</table>

[6] To reduce runoff caused by this use, the label includes a required 100-ft setback or a 35-ft vegetative buffer from surface waters for manure application as described in the California EPA Dairy Program Regulations and Requirements, Waste Discharge Requirements General Order No. r5-2007-0035, amended by Order No. r5-2009-0029 on 23 April 2009: (http://www.waterboards.ca.gov/northcoast/water_issues/programs/dairies/pdf/120127/npdes/120127_12_0001_NPDES_CAFO.pdf For spot treatments,

[7] For livestock poultry use, broadcast application method is restricted to indoor poultry house use.

Label No.	Use	Use Site	Land Use Category	Max. Single App. Rate lbs a.i./A	Max. No. of App. Per Year	Annual App Rate lbs a.i./A	Min. App. Interval (days)	App. Method	Buffer[5]
400-474[3]	Livestock/Poultry Premises: litter, stale/waste feed, manure, manure/straw mixture, feed muck/spoilage, spoiled organic refuse, bedding material, floors/walls, posts, cage frames, ceilings	Non-crop Agriculture	Agriculture	8.2	NS, but 1 / productio n cycle	NS	NS	Broadcast[4] (indoor poultry use only)	YES
					NS, but 1 / productio n cycle	NS	NS	Band (indoor poultry use only)	
				0.117	NS	2.0	21	Spot treatments	
400-461	Non-crop areas (field border, fence rows, roadsides, farmsteads, ditch banks, wasteland, Conservation Reserve Program (CRP) land	Non-crop agriculture	Agriculture	0.03125	NS	0.0938	NS	Ground, aerial	YES
400-465	Non-crop areas (field border, fence	Non-crop agriculture	Agriculture	0.03125	NS	NS (0.0938*)	NS	Ground, aerial	YES

Label No.	Use	Use Site	Land Use Category	Max. Single App. Rate lbs a.i./A	Max. No. of App. Per Year	Annual App Rate lbs a.i./A	Min. App. Interval (days)	App. Method	Buffer[5]
	rows, roadsides, farmsteads, ditch banks, wasteland, Conservation Reserve Program (CRP) land								
400-474	Non-crop areas (field border, fence rows, roadsides, farmsteads, ditch banks, wasteland, Conservation Reserve Program (CRP) land	Non-crop agriculture	Agriculture	0.03125	NS	0.0938	NS	Ground, aerial	YES
400-461	Grasslands (rangelands, pastures, improved pastures)	Pasture / Hay Herbaceous and Shrub/scrub	Agriculture Undeveloped	0.03125	NS	0.0938	14 for grasshopper or NS	Ground, aerial	YES
400-465	Grassland and non-crop areas	Pasture / Hay Herbaceous and Shrub/scrub	Agriculture/ Undeveloped	0.03125	NS	0.0938	14 for grasshopper, or NS	Ground, aerial	YES
400-	Grasslands	Pasture	Agriculture/	0.03125		0.0938	14 for	Ground,	YES

Label No.	Use	Use Site	Land Use Category	Max. Single App. Rate lbs a.i./A	Max. No. of App. Per Year	Annual App Rate lbs a.i./A	Min. App. Interval (days)	App. Method	Buffer[5]
474	(rangelands, pastures, improved pastures)	Herbaceous and Shrub/scrub	Undeveloped		NS		grasshopper, or NS	aerial	
499-500	Subterranean and above ground termite bait stations	All Developed	Developed	NA	NA	NA	NA	Placement	NO
68850-2	Subterranean and above ground termite bait stations	All Developed	Developed	NA	NA	NA	NA	Placement	NO
75313-2	Subterranean and above ground termite bait stations	All Developed	Developed	NA	NA	NA	NA	Placement	NO
499-488	Subterranean termite bait stations	All Developed	Developed	NS	NS	NA	NA	Placement	NO
400-474	Trees and shrubs in public and private forests, forest plantings and forest nurseries, Christmas tree and conifer nurseries, residential areas and landscape	All	Agriculture/ Developed/ Undeveloped / Rights-of-way	0.25	NS	0.25	7 for quarantine uses, or NS	Ground, aerial	YES

Label No.	Use	Use Site	Land Use Category	Max. Single App. Rate lbs a.i./A	Max. No. of App. Per Year	Annual App Rate lbs a.i./A	Min. App. Interval (days)	App. Method	Buffer[5]
	plantings, recreational areas, shelterbelts, rights-of-way								
400-465	Trees and shrubs in public and private forests, forest plantings and forest nurseries, Christmas tree and conifer nurseries, residential and municipal shade tree areas and landscape plantings, recreational areas (e.g., campgrounds, golf courses, parks, parkways), shelterbelts, rights of way, easements.	All	Developed/ Undeveloped / Agriculture/ Rights-of-way	0.25	NS	0.25	7 for quarantine uses, or NS	Ground, aerial	YES
400-543	Water holding receptacles or standing water sites around the home	All Developed	Developed	1054 µg/L	NS	NS	7	Effervescent tablet	NO

Label No.	Use	Use Site	Land Use Category	Max. Single App. Rate lbs a.i./A	Max. No. of App. Per Year	Annual App Rate lbs a.i./A	Min. App. Interval (days)	App. Method	Buffer[5]
2724-801	Technical or Manufacturing Use Product	NA	NA	NA	NA	NA	NA	NA	NA
400-466	Technical or Manufacturing Use Product	NA	NA	NA	NA	NA	NA	NA	NA
400-467	Technical or Manufacturing Use Product	NA	NA	NA	NA	NA	NA	NA	NA
400-488	Technical or Manufacturing Use Product	NA	NA	NA	NA	NA	NA	NA	NA
68850-1	Technical or Manufacturing Use Product	NA	NA	NA	NA	NA	NA	NA	NA
75313-1	Technical or Manufacturing Use Product	NA	NA	NA	NA	NA	NA	NA	NA

4.3 Fenbutatin oxide

Fenbutatin oxide was initially registered in the United States in 1974 for use as a miticide / acaricide. Fenbutatin oxide is a non-systemic organotin compound. Target pests include mites, aphids, mealybugs, white flies and scales. The first end-use product was registered in 1975 for use on apples, pears, and some citrus crops. Since that time, several other food crops and outdoor and greenhouse ornamentals have been added to the label. Fenbutatin oxide is persistent in the environment.

4.3.1 Usage Information

Fenbutatin oxide is primarily used in agriculture with key markets in Florida and California. However, fenbutatin-oxide is also used in Idaho, Oregon, and Washington. The agricultural use of fenbutatin-oxide is classified as restricted use due to high acute toxicity to aquatic organisms (EPA 2002a).

4.3.1.1 Agricultural Uses

Fenbutatin oxide is registered for use against mites on almonds, apples, cherries, citrus fruits, cucumbers, eggplant, grapes, papayas[8], peaches, pears, pecans, plums, raspberries, strawberries, and walnuts, greenhouse crops.

4.3.1.2 Non-agricultural Uses

Currently nurseries located outside of agricultural settings both in developed and undeveloped areas may use fenbutatin oxide on ornamentals. According to the applicant, United Phosphorus, Inc.(UPI), use of fenbutatin oxide on ornamentals in established landscapes will be canceled (Appendix 8 – 1/16/2013 UPI PowerPoint presentation).

4.3.1.3 Registered Formulation Types

There are two end use products of fenbutatin oxide. These are Promite 50WP (Sepro Corporation), and Vendex 50WP (UPI.). Each contain 50 percent active ingredient, and each are formulated as wettable powders, premeasured in 1 pound soluble packets.

[8] Papayas are not grown in WA, OR, or ID, and Fenbutatin-oxide is not labeled for use on papayas in CA.

4.3.2 Methods and Rates of Application

4.3.2.1 Methods

Ground applications are allowed, airblast and boom spray methods. Applications of fenbutatin oxide need to provide "thorough and complete" coverage of infested foliage and fruit. Agricultural applications will typically be made to fruit trees with air blast sprayers. Applications may not be made through irrigation systems. In addition, per discussions on January 16, 2013 with the applicant United Phosphorus, Inc., aerial application within California, Oregon, Idaho, and Washington will be prohibited (Appendix 8).

4.3.2.2 Application Rates

Use is 1.5 lbs per acre or less per application for agricultural products. Annually, fenbutatin-oxide use on eggplant and strawberries grown in California may be up to 4.5 lbs a.i. per acre. Citrus in California annually may have up to 3 lbs. per acre fenbutatin-oxide a.i. applied, and single application rates may be up to 2 lbs. per acre. Generally for most crops, the cumulative annual maximum use per acre is 2 lbs, the number of times it may be applied per year is two times (at a lesser rate of 1.0 to 1.5 lbs per acre) and the interval between uses is 21 days (Table 2). Cherry growers may use up to 2.25 lbs annually per year, single application rates may be up to 1.5 lbs per acre. Commercial operations growing ornamentals may apply a maximum rate of 1.0 lb a.i. per acre, and with 21 day intervals, fenbutatin-oxide may be applied 4 different times with a maximum annual application of 4 lbs a.i. per acre (See Appendix 8).

4.3.3 Metabolites and Degradates.

Fenbutatin oxide is persistent in the environment. In field dissipation studies, fenbutatin oxide was found to have a half-life of typically greater than one year (range 271-1370 days). Residues in the soil tend to accumulate from year to year (EPA 1994b, 2002a, 2009a). There is no evidence of hydrolysis although slow degradation through aqueous photolysis is expected to occur in clear shallow waters based on laboratory tests. One major degradate was identified in the aqueous photolysis study: 1,3-dihydroxy-1,1,3,3-tetrakis (2-methyl-2-phenyl propyl)-distannoxane (SD31723). Two degradates were identified in field dissipation trials: SD31723 and 2-methyl-2- phenylpropyl stanonic acid (SD 33608) (EPA 2009b).

Table 2. Summary of fenbutatin oxide labels, uses, and use sites

Label No.	Use	Use Site	Land Use Category	Max. Single App. Rate (lbs a.i./A)	Max. No./Yr	Annual App Rate (lbs a.i./A)	App. Interval (days)	App. Method	Buffer[9] Y/N
70506-210	Technical	NA[10]	NA	NA	NA	NA	NA	NA	Y
70506-211	Apple	Crop	Agriculture	1	2	2	21	Ground (including airblast)	Y
70506-211	Pear	Crop	Agriculture	1	2	2	21	Ground (including airblast)	Y
70506-211	Grape	Crop	Agriculture	1.25	2	2	21	Ground (including airblast)	Y
70506-211	Citrus (CA and AZ)	Crop	Agriculture	2	2	3	30	Ground (including airblast)	Y

[9] 25-ft buffer to salmon-bearing waters

[10] NA: Not Applicable

Label No.	Use	Use Site	Land Use Category	Max. Single App. Rate (lbs a.i./A)	Max. No./Yr	Annual App Rate (lbs a.i./A)	App. Interval (days)	App. Method	Buffer[9] Y/N
70506-211	Peach, Plum, Prune, Nectarine	Crop	Agriculture	1	2	1.5	21	Ground (including airblast)	Y
70506-211	Cherry (including sweet and sour)	Crop	Agriculture	1.5	2	2.25	21	Ground (including airblast)	Y
70506-211	Papaya, U.S. (Except CA, not grown in WA, OR, ID)	Crop	Agriculture	1	4	4	NS	Ground (including airblast)	Y
70506-211	Almond (CA only), Pecan (CA only), Pistachio (CA only), Walnut (CA, WA, OR, ID)	Crop	Agriculture	1.25	2	2	21	Ground (including airblast)	Y
70506-211	Strawberry (CA only)	Crop	Agriculture	1.5	3	4.5	NS	Ground	Y
70506-211	Strawberry (except	Crop	Agricultur	1	2	2	21	Ground	Y

Label No.	Use	Use Site	Land Use Category	Max. Single App. Rate (lbs a.i./A)	Max. No./Yr	Annual App Rate (lbs a.i./A)	App. Interval (days)	App. Method	Buffer[9] Y/N
	CA)		e						
70506-211	Eggplant (CA only)	Crop	Agriculture	1.5	3	4.5	21	Ground	Y
70506-211	Raspberry (Black, Red) in Washington and Oregon	Crop	Agriculture	1	1	1	NA	Ground (including airblast)	Y
70506-211; 67690-40	Christmas Trees in Washington and Oregon	Crop	Agriculture	1	1	1	NA	Ground (including airblast)	Y
70506-211; 67690-40	Greenhouse (indoor) and outdoor ornamentals	Crop/ Developed (Nurseries)	Agriculture/ Developed	1	4	4	21	NS	Y

4.4 Propargite

Propargite is an organosulfur miticide/acaricide used on a variety of bearing and non-bearing agricultural food crops, as well as non-food agricultural sites. It was first registered in 1969. Sites on which propargite has the highest percent of crop treated include grapes, walnuts, almonds, nectarines, and mint.

4.4.1 Usage Information

EPA provided information that indicated approximately 2 million lbs of propargite active ingredient were applied annually, primarily in California (EPA 2002c). In 2002, 980,441 lbs of propargite were applied in California (CDPR PUR). In more recent years, use of propargite in California declined to 389,492 lbs in 2008, 380,650 lbs in 2009, and 295,162 lbs in 2010 (CDPR PUR).

4.4.1.1 Agricultural Uses

In California, Oregon, Washington, and Idaho, registered food-use crops include grapes, citrus, nectarines, almonds, beans, corn, hops, potatoes, and mint. Non-bearing crops include berries and various stone and pome fruits. Other non-food agricultural sites include cotton, alfalfa for seed, clover for seed, carrots for seed, roses and conifers grown as Christmas trees or nursery stock.

4.4.1.2 Non-agricultural Uses.

Propargite can be used in nurseries and Christmas tree & conifer plantations. There are no other non-agricultural areas where propargite may be applied.

4.4.1.3 Registered Formulation Types.

Propargite end-use products are formulated as liquids (emulsifiable concentrates) and as a solid (wettable powder).

4.4.2 *Methods and Rate of Application.*

4.4.2.1 Methods

Propargite may be spray applied using ground application or aerial methods. In some cases chemigation may be used to apply propargite to fields. Also a “sticker” is recommended to enhance leaf surface retention.

4.4.2.2 Application Rates

Active labels allow a maximum single application rate of 3.2 lbs propargite active ingredient per acre to walnuts and citrus crops (oranges and grapefruit) (Table 3). Nectarines may have up to 2.88 lbs a.i. applied per acre. All other crops are at or below 2.5 lbs a.i. applied per acre. Table 3 provides maximum annual application rates for each use. Walnuts and almonds have the highest annual rate of up to 6.4 lbs per acre. No more than two applications are allowed per year for all uses.

4.4.3 *Metabolites and Degradates*

EPA indicated that the main transformation products of propargite are bis-[2,-(4-(1,1-dimethyl-ethyl)-phenoxy)cyclohexyl] sulfite (BGES); 2,2-dimethyl-2-(4’-(2-hydroxy-cyclohexoxy)phenyl)ethanol (OMT-G); p-tertiarybutylphenol (PTBP); propargite glycol ether-2-[4-(1,1-dimethylethyl)phenoxy]-cyclohexane-1-ol (TBPC); and 2-[4-(2-hydroxycyclohexoxy)phenyl]-2,2-dimethyl acetic acid (TBPC-acid), and a sulfate derivative of TBPC (EPA 2008).

Table 3. Summary of propargite labels including uses and use rates

Label No.	Use	Use Site	Land Use Category	Max. Single App. Rate (lbs a.i./A)	Max. No. of App. Per Year	Annual App Rate (lbs a.i./A)	Min. App. Interval (days)	App. Method	Buffer[11] Y/N
400-104; ID960016	Alfalfa (grown for seed only)	Crop	Agriculture	2.05	2	4.1	14	Ground, aerial	Y
400-104; WA040019	Alfalfa (grown for seed only)	Crop	Agriculture	2.05	2	4.1	14	Ground, aerial	Y
400-104-2A; CA-830024	Alfalfa (grown for seed only)	Crop	Agriculture	2.5	2	5	14	Ground, aerial	Y
400-104; OR-080016	Alfalfa (grown for seed only)	Crop	Agriculture	1.6	2	3.2	14	Ground aerial	Y

[11] **Label statements from EPA Reg. No. 400-89:** Do not apply by ground within 50 feet or by air within 75 feet of lakes, reservoirs, rivers, permanent streams, marshes or natural ponds; estuaries and commercial fish ponds. The above excludes irrigation canals and waterways as well as man-made irrigation conveyance structures and impoundments, unless an exclusion contains water year-round.

Other conservation measures on labels: All labels include the following conservation measure: Only apply this product when wind speed is between 2-10 mph at the application site; for ground and aerial applications apply the coarsest droplet size spectrum that provides sufficient coverage and mite control. In CA, ID, OR, and WA labels to include the following statements: (1) Choose only nozzles and pressures that produce a medium or coarse droplet size (>250 microns volume median diameter), and (2) In CA, ID, OR, and WA,for aerial applications, volume applied must be greater than or equal to 10 gallons per acre. There will also be a statement suggesting inclusion of a drift control agent in the spray tank..

Label No.	Use	Use Site	Land Use Category	Max. Single App. Rate (lbs a.i./A)	Max. No. of App. Per Year	Annual App Rate (lbs a.i./A)	Min. App. Interval (days)	App. Method	Buffer[11] Y/N
400-427	Almonds	Crop	Agriculture	3.20	2	6.4	21	Ground (including airblast)	Y
400-89	Almonds	Crop	Agriculture	3.00	2	6.0	21	Ground (including airblast), aerial	Y
400-104	Almonds (CA and AZ only)	Crop	Agriculture	3.1	2 per season	6.0	21	Ground (including airblast), aerial	Y
400-104	Beans, dry	Crop	Agriculture	2.46	2	3.7	21	Ground, aerial	Y
400-89	Berries: Aronia berry; Bearberry; Bilberry; Blackberry; Blueberry; Boysenberry;	Crop	Agriculture	1.50	2	3.0	21	Ground (including airblast)	Y

Label No.	Use	Use Site	Land Use Category	Max. Single App. Rate (lbs a.i./A)	Max. No. of App. Per Year	Annual App Rate (lbs a.i./A)	Min. App. Interval (days)	App. Method	Buffer[11] Y/N
	Caneberries; Cloudberry; Curran, black; Currant, red; Dewberry; Elderberry; Gooseberry; Highbush cranberry; Huckleberry; Jostaberry; Juneberry; Lingonberry; Mulberry; Partiridgeberry; Raspberry; Salal; Seagrape; Serviceberry; Strawberry								
400-104	Carrots grown for	Crop	Agriculture	NS	NS	NS	NS	NS	Y

Label No.	Use	Use Site	Land Use Category	Max. Single App. Rate (lbs a.i./A)	Max. No. of App. Per Year	Annual App Rate (lbs a.i./A)	Min. App. Interval (days)	App. Method	Buffer[11] Y/N
	seed								
400-104; ID770005	Carrot grown for seed	Crop	Agriculture	2.46	NS	NS	NS	Ground, aerial	Y
400-104; WA040019	Carrot grown for seed	Crop	Agriculture	2.46	2	4.9	14	Ground, aerial	Y
400-89	Citrus: Calamondin; Citron, Citrus; Citrus hybrids; Grapefruit; Kumquat; Lemon; Lime; Lime, sweet; Orange, sour; Orange, sweet; Orange, trifoliate; Pummelo; Sapote, white; Tangelo; Tangerine	Crop	Agriculture	1.50	2	3.0	21	Ground (including airblast)	Y
400-104; OR-080017	Clover and carrot grown for seed	Crop	Agriculture	2.46	2	2.92	NS	Ground, aerial	Y

Label No.	Use	Use Site	Land Use Category	Max. Single App. Rate (lbs a.i./A)	Max. No. of App. Per Year	Annual App Rate (lbs a.i./A)	Min. App. Interval (days)	App. Method	Buffer[11] Y/N
400-104	Clover grown for seed	Crop	Agriculture	NS	NS	NS	NS	NS	Y
400-104; CA-040013	Clover grown for seed	Crop	Agriculture	1.6	NS	NS	NS	Ground, aerial	Y
400-104; ID770005	Clover grown for seed	Crop	Agriculture	2.46	NS	NS	NS	Ground, aerial	Y
400-104; WA040019	Clover grown for seed	Crop	Agriculture	2.46	2	4.9	14	Aerial	Y
400-104	Conifers (in plantations, nurseries, shade houses and containers only; Pacific Northwest only, excluding CA)	Crop	Agriculture	2.25	2	4.5	28	Ground, aerial	Y
400-104	Cotton	Crop	Agriculture	1.6	2	3.3	21	Ground, aerial	Y

Label No.	Use	Use Site	Land Use Category	Max. Single App. Rate (lbs a.i./A)	Max. No. of App. Per Year	Annual App Rate (lbs a.i./A)	Min. App. Interval (days)	App. Method	Buffer[11] Y/N
400-104; CA-820083	Cotton (Imperial and Riverside counties only)	Crop	Agriculture	1.6	2	3.3		Ground, aerial	Y
400-89	Currants, Dates, Figs	Crop	Agriculture	1.50	2	3.0	21	Ground (including airblast)	Y
400-89	Field Corn (CA only)	Crop	Agriculture	2.53	1	2.53	NA	Ground, aerial	Y
400-104	Field Corn, Popcorn, Seed Corn	Crop	Agriculture	2.46	1	2.46	NS	Ground, aerial, chemigation	Y
400-427	Grapefruit, navel oranges (post-harvest use)	Crop	Agriculture	3.2	1	NS	NA	Ground (including airblast)	Y
400-427	Grapes	Crop	Agriculture	2.88	2	5.76	21	Ground (including airblast)	Y

Label No.	Use	Use Site	Land Use Category	Max. Single App. Rate (lbs a.i./A)	Max. No. of App. Per Year	Annual App Rate (lbs a.i./A)	Min. App. Interval (days)	App. Method	Buffer[11] Y/N
400-104	Hops	Crop	Agriculture	1.5	2	3.0	21	Ground	Y
400-89	Hops	Crop	Agriculture	1.50	2	3.0	21	Ground (including airblast)	Y
400-104	Jojoba	Crop	Agriculture	1.64	2	3.28	10	Ground, aerial	Y
400-89	Mint	Crop	Agriculture	2.06	2	4.1	14	Ground, aerial	Y
400-104	Mint (except CA)	Crop	Agriculture	2.05	2	4.1	14	Ground, aerial	Y
400-427	Nectarines	Crop	Agriculture	2.88	2	5.76	21	Ground (including airblast), aerial	Y
400-104-2A; CA-940031	Non-bearing almonds and walnuts interplanted with	Crop	Agriculture	2.46	2	3.7	21	Ground, aerial	Y

Label No.	Use	Use Site	Land Use Category	Max. Single App. Rate (lbs a.i./A)	Max. No. of App. Per Year	Annual App Rate (lbs a.i./A)	Min. App. Interval (days)	App. Method	Buffer[11] Y/N
	beans								
400-89	Nut trees: almond, hazelnut, macadamia, pecan pistachio, walnut	Crop	Agriculture	1.50	2	3.0	21	Ground (including airblast)	Y
400-427	Oranges (CA); Grapefruit (CA) (in-season use)	Crop	Agriculture	3.20	2	5.80	28	Ground (including airblast)	Y
400-427	Peanuts	Crop	Agriculture	1.60	2	3.2	14	Ground	Y
400-104	Peanuts (except CA)	Crop	Agriculture	1.6	2	3.2	14	Ground, aerial	Y
400-89	Persimmons	Crop	Agriculture	1.50	2	3.0	21	Ground (including airblast)	Y
400-104; OR-080019	Potato	Crop	Agriculture	2.05	2	4.1	14	Ground, aerial	Y
400-104;	Potato	Crop	Agriculture	2.05	2	4.1	14	Chemigation	Y

Label No.	Use	Use Site	Land Use Category	Max. Single App. Rate (lbs a.i./A)	Max. No. of App. Per Year	Annual App Rate (lbs a.i./A)	Min. App. Interval (days)	App. Method	Buffer[11] Y/N
OR-080018									
400-104	Potato (Pacific Northwest only)	Crop	Agriculture	2.05	2	4.1	21; 14 in WA	Ground, aerial, chemigation (not in CA)	Y
400-104; ID040010	Potato	Crop	Agriculture	2.05	2	4.1	14	Ground, aerial	Y
400-89	Potato (Pacific Northwest only)	Crop	Agriculture	2.06	2	4.1	21 (Washington State only: 14 days)	Ground, chemigation (not in CA), aerial	Y
400-104	Sorghum (only CA)	Crop	Agriculture	1.64	1	1.64	NS	Aerial	Y
400-427	Stonefruit includes: apricots, cherries, peaches, plums/prunes	Crop	Agriculture	1.92	2	3.84	21	Ground (including airblast)	Y
400-104	Sugar beets for seed	Crop	Agriculture	NS	NS	NS	NS	NS	Y

Label No.	Use	Use Site	Land Use Category	Max. Single App. Rate (lbs a.i./A)	Max. No. of App. Per Year	Annual App Rate (lbs a.i./A)	Min. App. Interval (days)	App. Method	Buffer[11] Y/N
400-104; OR-080014	Sugar beets for seed	Crop	Agriculture	2.46	2	4.92	NS	Aerial	Y
400-104	Sweet corn (fresh, processing and for seed)	Crop	Agriculture	2.46	1	2.46	NS	Ground, aerial, chemigation	Y
400-89	Tree Fruit: apples, apricots, cherries, nectarines, peaches, plums/prunes, quince	Crop	Agriculture	1.50	2	3.0	21	Ground (including airblast)	Y
400-104	Walnuts	Crop	Agriculture	3.2	2	6.4	21	Ground, aerial	Y
400-89	Walnuts	Crop	Agriculture	3.2	2	6.4	21	Ground (including airblast), aerial	Y
400-427	Walnuts (CA only)	Crop	Agriculture	3.2	2	6.4	21	Ground (including	Y

Label No.	Use	Use Site	Land Use Category	Max. Single App. Rate (lbs a.i./A)	Max. No. of App. Per Year	Annual App Rate (lbs a.i./A)	Min. App. Interval (days)	App. Method	Buffer[11] Y/N
								airblast)	
400-104	Aronia berry, bearbearry, bilberry, blackberry, blueberry, lowbush blueberry, boysenberry, caneberries, cloudberry, black currant, red currant, dewberry, elderberry, gooseberry, highbush cranberry, huckleberry, jostaberry, juneberry, lingonberry, mulberry,	Crop	Agriculture	1.5	2	3.0	21; 28 for citrus	Ground (including airblast), aerial	Y

Label No.	Use	Use Site	Land Use Category	Max. Single App. Rate (lbs a.i./A)	Max. No. of App. Per Year	Annual App Rate (lbs a.i./A)	Min. App. Interval (days)	App. Method	Buffer[11] Y/N
	partridgeberry, raspberry, salal, seagrape, serviceberry, strawberry; calamondin, citrus citron, citrus hybrids, grapefruit, kumquat, lemon, lime, sweet lime, sour orange, sweet orange, trifoliate orange, pummelo, white sapote, tangelo, tangerine; currants, dates, figs; almond trees, hazelnut trees, macadamia trees,								

Label No.	Use	Use Site	Land Use Category	Max. Single App. Rate (lbs a.i./A)	Max. No. of App. Per Year	Annual App Rate (lbs a.i./A)	Min. App. Interval (days)	App. Method	Buffer[11] Y/N
	pecan trees, pistachio trees, walnut trees; persimmons; apples, apricots, cherries, nectarines, peaches, plums/prunes, quince								
400-427	Citrus: Calamondin; Citron, Citrus; Citrus hybrids; Grapefruit; Kumquat; Lemon; Lime; Lime, sweet; Orange, sour; Orange, sweet; Orange, trifoliate; Pummelo; Sapote, white; Tangelo; Tangerine	Crop / Developed (Nurseries)	Agriculture/ Developed	1.92	2	3.84	28	Ground (including airblast)	Y
400-427	Berries; Aronia	Crop /	Agriculture	1.92	2	3.84	21	Ground	Y

Label No.	Use	Use Site	Land Use Category	Max. Single App. Rate (lbs a.i./A)	Max. No. of App. Per Year	Annual App Rate (lbs a.i./A)	Min. App. Interval (days)	App. Method	Buffer[11] Y/N
	berry; Bearberry; Bilberry; Blackberry; Blueberry (including low bush); Boysenberry; Caneberries; Cloudberry; Currant, both black and red; Dewberry; Elderberry; Gooseberry; Highbush cranberry; Huckleberry; Jostaberry; Juneberry; Lingonberry; Mulberry; Partridgeberry;	Developed (Nurseries)	/ Developed					(including airblast)	

Label No.	Use	Use Site	Land Use Category	Max. Single App. Rate (lbs a.i./A)	Max. No. of App. Per Year	Annual App Rate (lbs a.i./A)	Min. App. Interval (days)	App. Method	Buffer[11] Y/N
	Raspberry; Salal; Seagrape; Serviceberry; Strawberry								
400-427	Christmas trees & Conifers (for plantations & nursery use only)	Crop / Developed (Nurseries)	Agriculture / Developed	2.4	2	4.8	28	Ground (including airblast), aerial	Y
400-89	Conifers in plantations, nurseries, shade houses & containers (Pacific Northwest only, except CA	Crop / Developed (Nurseries)	Agriculture / Developed	2.25	2	4.45	28	Ground (including airblast), aerial	Y
400-427	Currants, Dates, Figs, Persimmons; Nut trees: almond, hazelnut,	Crop / Developed (Nurseries)	Agriculture / Developed	1.92	2	3.84	21	Ground (including airblast)	Y

Label No.	Use	Use Site	Land Use Category	Max. Single App. Rate (lbs a.i./A)	Max. No. of App. Per Year	Annual App Rate (lbs a.i./A)	Min. App. Interval (days)	App. Method	Buffer[11] Y/N
	macadamia, pecan, pistachio, walnut; Fruit trees: apples, apricots, cherries, nectarines, peaches, pears, plums/prunes, quince								
400-427	Ornamentals: Carnations, Chrysanthemums	Crop / Developed (Nurseries)	Agriculture / Developed	0.48	2	0.96	14	Ground (including airblast)	Y
400-427	Other ornamentals (field and nursery)	Crop / Developed (Nurseries)	Agriculture / Developed	1.6	2	3.2	14	Ground (including airblast)	Y
400-427	Roses (field grown)	Crop / Developed (Nurseries)	Agriculture / Developed	1.6	2	3.2	14	Ground (including airblast), aerial	Y
400-83	Roses (field grown)	Crop /	Agriculture	1.56	2	3.12	14	Ground	Y

Label No.	Use	Use Site	Land Use Category	Max. Single App. Rate (lbs a.i./A)	Max. No. of App. Per Year	Annual App Rate (lbs a.i./A)	Min. App. Interval (days)	App. Method	Buffer[11] Y/N
		Developed (Nurseries)	/ Developed					(including airblast), aerial	
400-565	Technical Manufacturing Use Product	NA	NA	NA	NA	NA	NA	NA	NA
400-95	Technical, only for formulation into a miticide for uses on specified crops	NA	NA	NA	NA	NA	NA	NA	NA

5 Action Area

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

In this instance, as a result of the 2002 order in Washington Toxics Coalition v. EPA, EPA initiated consultation on its authorization of 37 pesticide a.i.s and their effects on listed Pacific salmonids under NMFS' jurisdiction and associated designated critical habitat in the states of California, Idaho, Oregon, and Washington. Consequently, for purposes of this Opinion, the action area consists of the entire range and most life history stages of listed salmon and steelhead and their designated critical habitat in California, Idaho, Oregon, and Washington. The action area encompasses all freshwater, estuarine, marsh, swamps, nearshore, and offshore marine surface waters of California, Oregon, and Washington. The action area also includes all freshwater surface waters in Idaho (Figure 2).

Diflubenzuron, fenbutatin-oxide, and propargite are the seventh set of pesticides identified in the consultation schedule established in the settlement agreement and are analyzed in this Opinion. NMFS' analysis focuses only on the effects of EPA's action on listed Pacific salmonids in the above-mentioned states. It includes the effects of these pesticides on the recently listed Lower Columbia River coho salmon, Puget Sound steelhead, and Oregon Coast coho salmon. The Lower Columbia River coho salmon was listed as endangered in 2005. The Puget Sound steelhead and the Oregon Coast coho salmon were listed as threatened in 2007 and 2008, respectively. This Opinion also analyzes the effects of EPA's proposed action on recently proposed designated critical habitats for Puget Sound steelhead and Lower Columbia River coho salmon (January 14, 2013, 50 CFR Part 226).

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

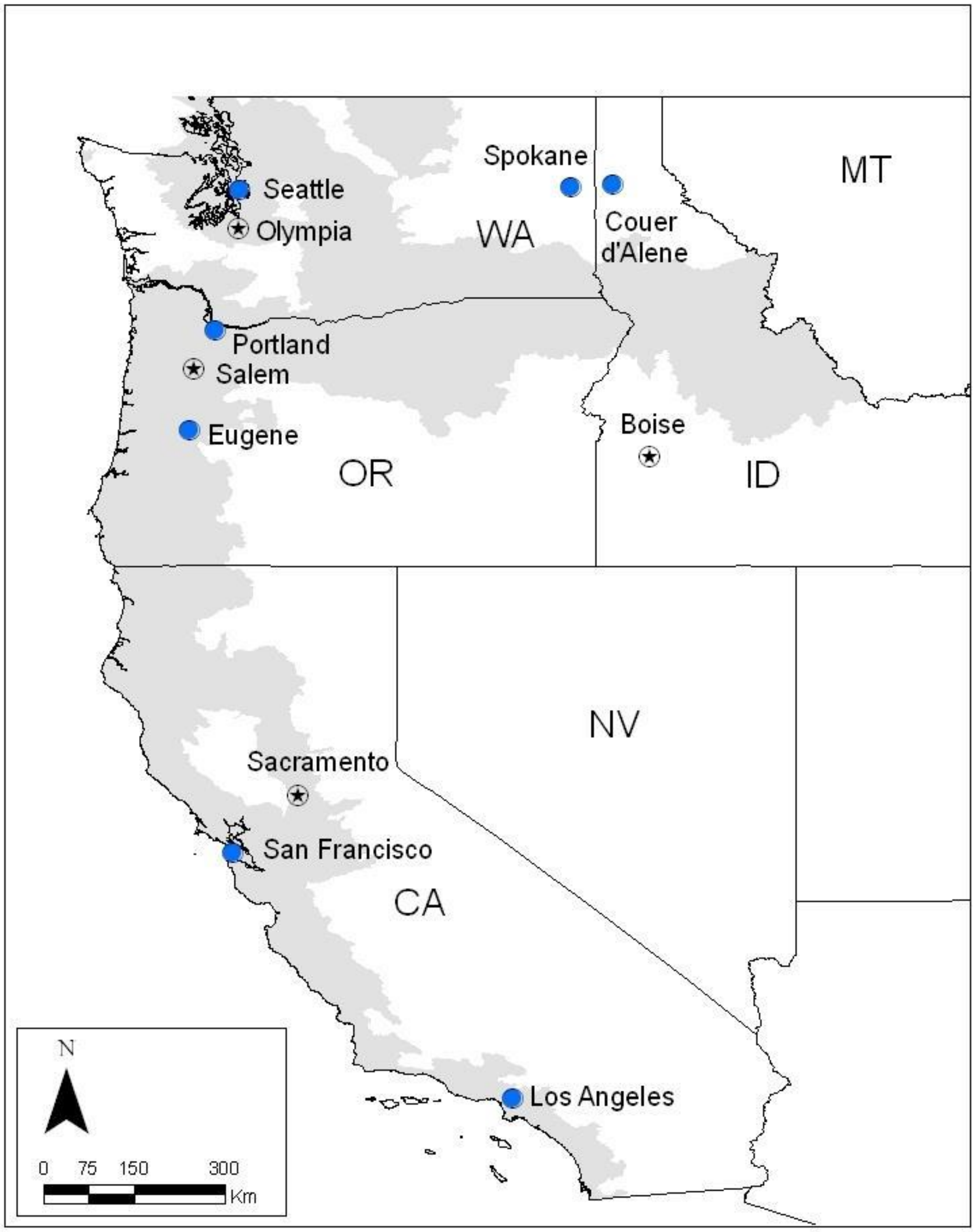

Figure 2. **Map showing extent of inland action area with the range of all ESU and DPS boundaries for ESA listed salmonids highlighted in gray.**

6 Approach to the Assessment

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

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

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

In the *Risk Characterization Section*, we determine whether population level effects are anticipated by evaluating the evidence to support or refute risk hypotheses. Our analysis is ultimately a qualitative assessment that draws on a variety of quantitative and qualitative tools and measures to address risk to listed resources.

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

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

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

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

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

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

6.1 Evidence Available for the Consultation

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

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

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

6.2 Application of Approach in this Consultation

For this consultation on diflubenzuron, fenbutatin oxide, and propargite we adapt the general approach to incorporate elements of EPA's ecological risk assessment (ERA) framework (EPA 1998). Figure 3 shows the overall risk framework used in this Opinion. This risk assessment framework organizes the available information in three phases: problem formulation, analysis of exposure and response, and risk characterization (EPA 1998). We adapted the EPA framework to incorporate ESA section 7 consultation requirements. We organize, evaluate, and synthesize the available information on listed resources and the stressors of the action. We evaluate the risk to listed species and the risk to designated critical habitat from the stressors of the action as a separate and distinct analysis from the Jeopardy analysis (See *Effects of the Proposed Action to Threatened and Endangered Pacific Salmonids* and *Effects of the Proposed Action to Designated Critical habitat* sections).

6.2.1 Problem Formulation

Problem formulation includes conceptual models based on our initial evaluation of the relationships between stressors of the action (pesticides and other identified chemical stressors) and receptors (listed species and habitat). Unlike ERAs[12] conducted by EPA pursuant FIFRA, which begin with the use, fate, and toxicity properties of the three active ingredients, and evaluate risk based on standard toxicity test organisms, NMFS begins with the species' range and life history to determine relevant assessment endpoints, identifies if those endpoints are likely to be affected by the stressors of the action, and seeks data with which to evaluate those effects. We employ a species-centric approach, rather than a chemical-centric approach. Assessment endpoints and measures may vary by life stage and are presented in Table 4. Many of the relevant assessment endpoints

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

and measures are not ones typically considered or used in EPA's registration of pesticide active ingredients.

Table 4. Examples of assessment endpoints and measures

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

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

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

6.2.2 Species Risk Hypotheses

1. Exposure to diflubenzuron, fenbutatin oxide, or propargite via drift or runoff is sufficient to:

a. kill salmonids from direct exposure;

b. reduce salmonid survival through impacts to growth;

c. reduce salmonid survival through impacts to reproduction;

d. reduce salmonid growth through impacts on the availability and quantity of prey;

e. impair swimming;

f. accumulate in salmonids thus impairing fitness

2. Exposure to degradates of diflubenzuron, fenbutatin oxide, or propargite will cause adverse effects to salmonids and their habitats.
3. Exposure to adjuvants, tank mixtures and other chemicals within pesticide products containing diflubenzuron, fenbutatin oxide, or propargite cause adverse effects to salmonids and their habitats.
4. Exposure to other pesticides present in the action area can act in combination with the three insecticides to increase effects to salmonids and their habitats.
5. Exposure to elevated temperatures enhances the toxicity of the stressors of the action.

6.2.3 Designated Critical Habitat

When designated critical habitat for the species is identified, primary constituent elements (PCEs) of that habitat are also identified (Table 5) To determine potential effects to designated critical habitat, we evaluate the effects of the action by first looking at whether PCEs of critical habitat are potentially affected by the stressors of the action. Effects to PCEs include changes to the functional condition of salmonid habitat caused by the action in the action area. Properly functioning salmonid PCEs are essential to the conservation of the ESU/DPS. Watersheds (HUC 5) within each ESU/DPS have been ranked on their significance to the conservation value of the species. NMFS assigned each HUC 5 watershed as high, medium, or low in respect to their conservation value to the species. The stressors of the action for this Opinion are chemicals introduced into the environment by application of pesticide products containing diflubenzuron, fenbutatin oxide, or propargite. PCEs potentially affected by the stressors of the action include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, estuarine areas, and nearshore marine environments. Based on the action we do not anticipate offshore marine areas to be exposed.

Table 5. Essential physical and biological attributes of PCEs in salmonid critical habitat designations

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

[a] Natural cover examples include shade, large wood, riparian systems, log jams, beaver dams, aquatic vegetation, large rocks and boulders, side channels, and undercut banks.
[b] Forage includes aquatic and terrestrial invertebrates and fish and shellfish species that support growth and maturation of salmonids.

Based on the PCEs and life stage potentially affected (Table 5), we developed risk hypotheses for critical habitat. Properly functioning salmonid PCEs are important to the conservation of the ESU/DPS. The stressors of the action for this Opinion are chemicals introduced into the environment by application of pesticide products.

6.2.4 Critical Habitat Risk Hypotheses

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 resources in freshwater rearing sites.
3. Exposure to the stressors of the action is sufficient to degrade water quality, natural cover, and/or reduce prey resources in freshwater migratory corridors.
4. Exposure to the stressors of the action is sufficient to degrade water quality and/or reduce prey resources in estuarine areas.

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

6.3 Evaluating Exposure and Response

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

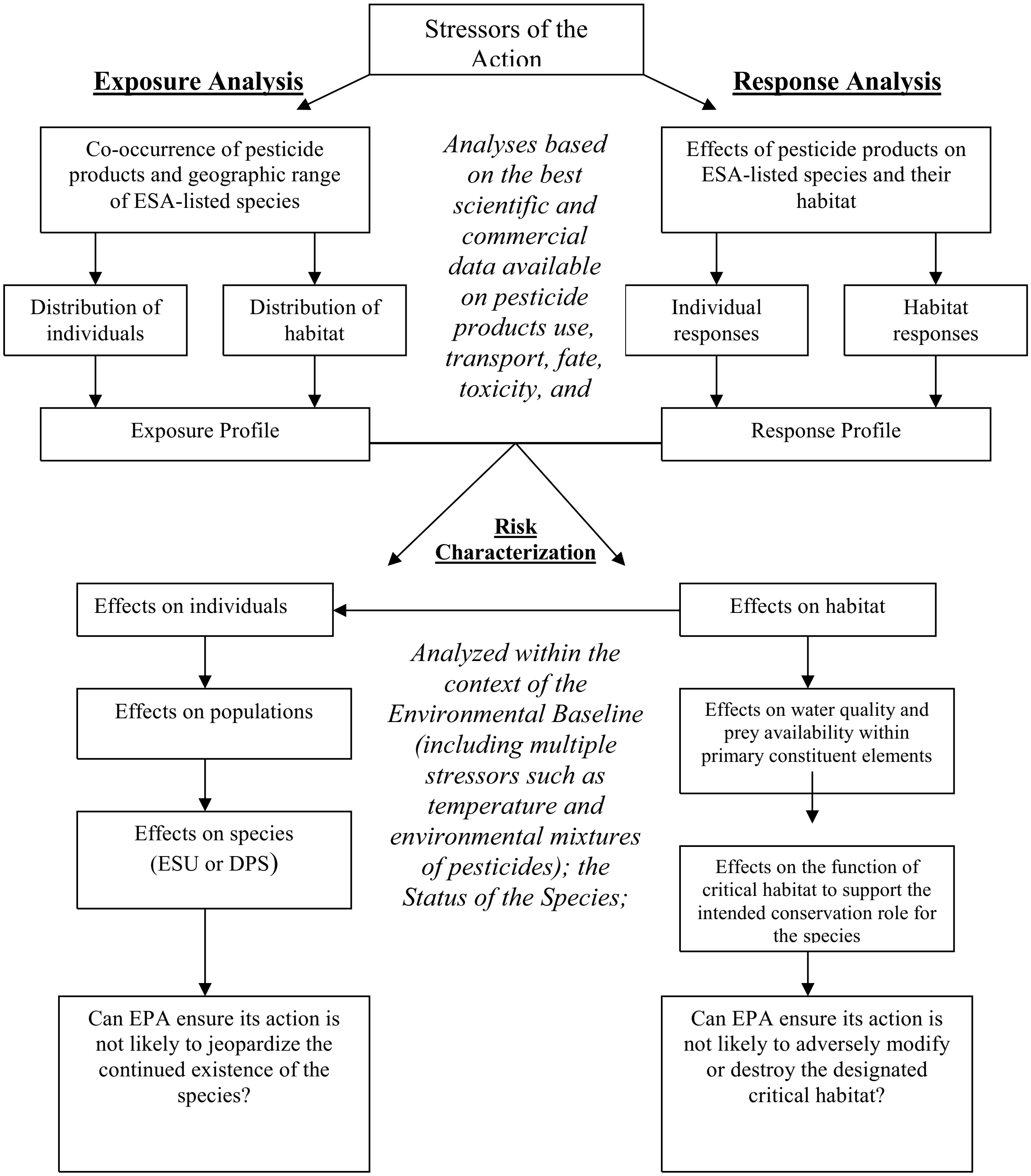

Figure 3. Conceptual framework for assessing risks of EPA's action to ESA listed resources

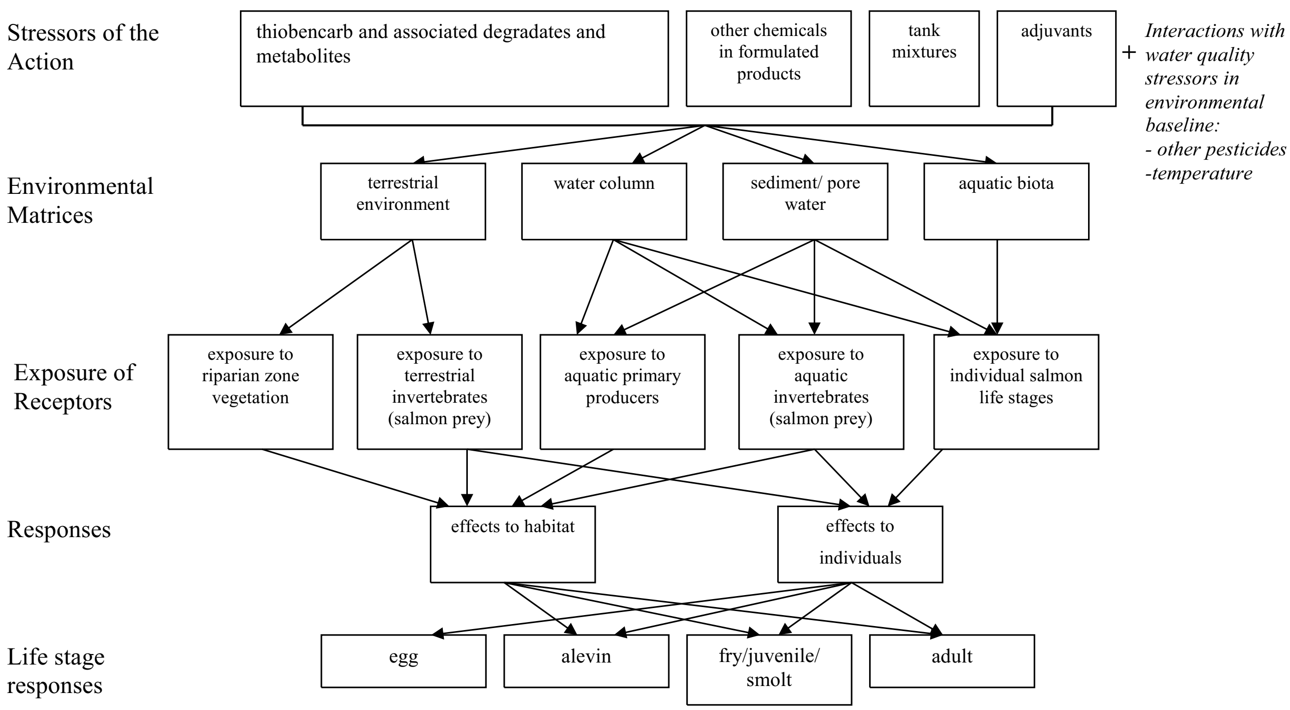

Figure 4. Exposure pathways for stressors of the action, and general response of Pacific salmonids and habitat

6.4 Analysis Plan

6.4.1 Status of the Species

In this section, we present information regarding each of the ESUs and DPSs considered in this Opinion. We discuss life history, population abundance and trends and overall viability of the species. We also present information on designated critical habitat for each species including the conservation value for watersheds that comprise the ESU/DPS. This provides part of the context in which we evaluate the effect of the proposed action.

6.4.2 Environmental Baseline

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

6.4.3 Effects of the Proposed Action to Threatened and Endangered Pacific Salmonids

In the *Exposure* section we discuss life histories of the various species which may make them more or less likely to be exposed to stressors of the actions. Then we evaluate measured and estimated environmental concentrations of the stressors from various sources. In this section we also evaluate spatial and temporal co-occurrences of land types associated with pesticide use sites and overlay them with salmonid ranges. The *Response* section presents 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 individual fitness effects are sufficiently large to affect population viability metrics such as abundance and productivity.

6.4.4 Integration and Synthesis

We begin with a summary of risk as described/identified in the *Risk Characterization*. In separate sections for listed species and critical habitat, we combine these risk conclusions regarding the effects of the proposed action with information in the *Status of the Species* and *Environmental Baseline* to determine potential effects on populations and species.

6.4.5 Conclusion

For each of the three insecticides, we present a summary of the lines of evidence showing the level of confidence we ascribe to each line of evidence evaluated (Figure 5). For each line of evidence, we indicate the strength of the relationship i.e., our confidence, by showing one of three types of arrows. A bolded arrow indicates a high level of confidence that the best available information supports findings with a low level of uncertainty. A non-bolded arrow indicates a medium level of confidence where the best available information supports findings with a moderate level of uncertainty. A dashed arrow shows a low level of confidence where the best available information suggests findings, however a high level of uncertainty. When no information is available to either refute or support a line of evidence, a question mark appears instead of a yes or no.

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

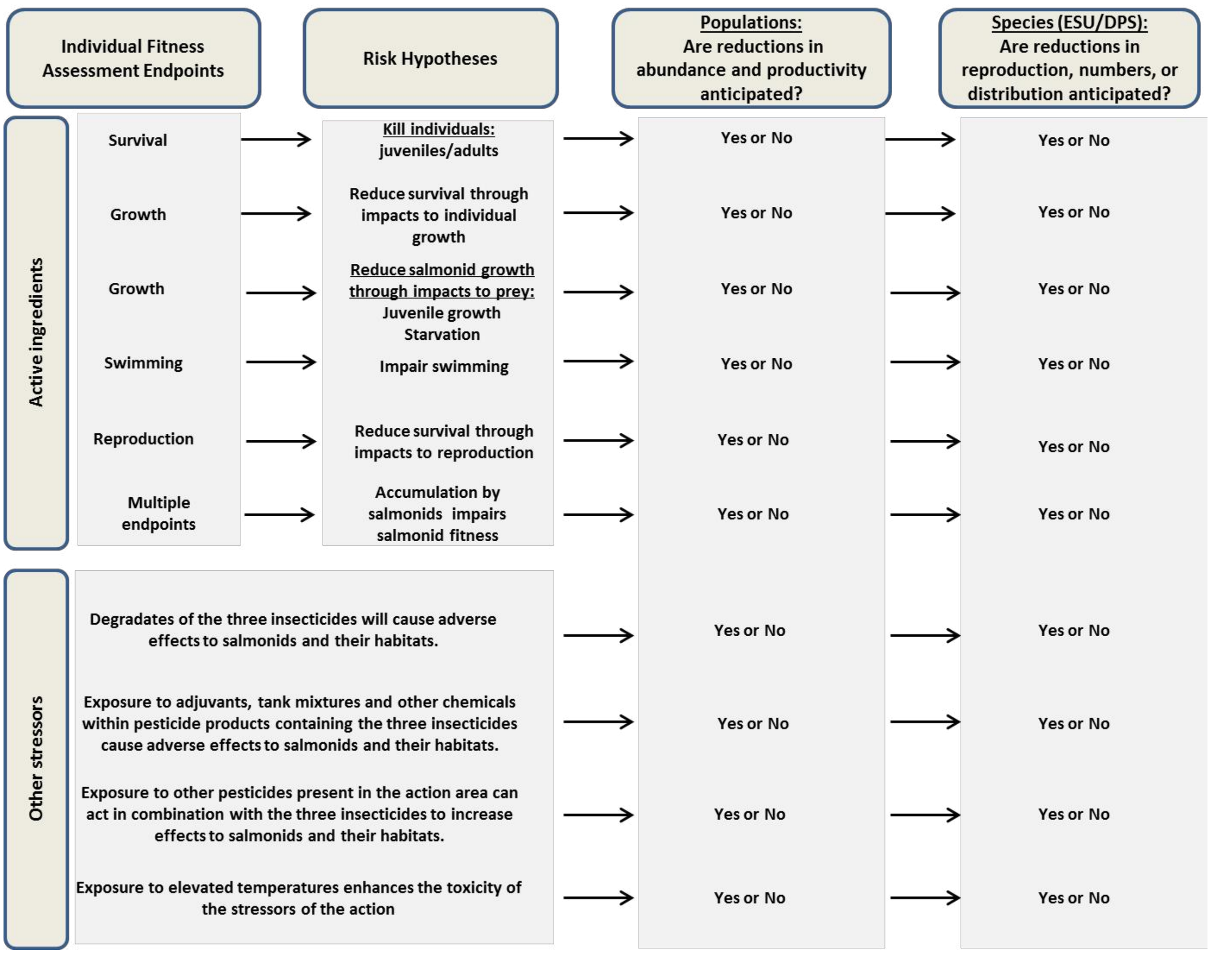

Figure 5. Lines of evidence

6.5 Other Considerations

We evaluated lines of evidence constructed as 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, when appropriate, to provide examples of potential risks to listed salmonids and their habitat. Multiple methods and tools currently exist for addressing contaminant-induced risk to the environment. Hazard-based assessments, probabilistic risk assessment techniques, combinations of the two, and deterministic approaches such as screening level assessments have been applied to questions of risk related to human health and the environment.

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

In some recent pesticide risk assessments, probabilistic techniques have been used to evaluate the probability of exceeding a "toxic" threshold for aquatic organisms by combining pesticide monitoring data with species sensitivity distributions (Giddings 2009, Geisy et al. 1999). There is utility in information generated by probabilistic approaches if supported by robust data. We compared the species sensitivity distributions presented by the pesticide applicant, Chemtura Corporation (Gagne et al. 2013) with the probability distributions of salmonid prey acute lethality values that we developed (presented in the *Risk Characterization* section).

NMFS considered the use of probabilistic risk assessment techniques for addressing risk of diflubenzuron, fenbutatin oxide, and propargite use at population and species (ESU and DPS) scales for the stressors of the action. However, we encountered significant limitations in available data that suggested the information was not sufficient to define exposure and/or response probabilities necessary to determine the probability of risk. 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 each a.i., we selected the lowest reported salmonid LC_{50} from the available information to ensure risk was not underestimated. When we consider the data limitations coupled with the inherent complexity of EPA's proposed action, 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 did not factor the status of the species 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.

7 Status of Listed Resources

The purpose of this section is to characterize the condition of the 28 salmonid species[13] 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 diflubenzuron, fenbutatin-oxide, and propargite - containing products (Table 6). More detailed information on the status of these species and critical habitat are found in a number of published documents including recent recovery plans, status reviews, stock assessment reports, and technical memorandums. Many are available on the Internet at http://www.nmfs.noaa.gov/pr/species/.

Table 6. 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

[13] 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 (DPS) of any species of vertebrate fish or wildlife which interbreeds when mature (16 U.S. C 1533)." Pacific salmon other than steelhead that have been listed as endangered or threatened were listed as "evolutionarily significant units (ESU), which NMFS uses to identify distinct population segments of Pacific salmon. Any ESU or DPS is a "species" for the purposes of the ESA.

**NOAA proposed designated critical habitat for Low Columbia River coho salmon and Puget Sound steelhead in January 14, 2013 (50 CFR Part 226). Final designation for these ESU/DPSs is expected later in 2013 or early 2014.

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.

7.1 Species Status

The status of an ESU or DPS is determined by the degree that it (1) maintains sufficient genetic and phenotypic diversity to ensure continued fitness in the face of environmental change, (2) maintains spatial distribution of populations so that not all populations would be affected by a catastrophic event, and (3) maintains sufficient connectivity among populations within the ESU or DPS to maintain long-term demographic and evolutionary processes (ICTRT 2007, McElhany et al. 2000, Spence et al. 2008a). 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 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[14] 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; (2) abundance; (3) annual growth rate, including trends and variability of annual growth rates; and (4) diversity (McElhany et al. 2000).

A population's tendency to increase in abundance 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).

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

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

7.2 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.

7.2.1 Primary Constituent Elements

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 features (*e.g.,* water temperature, water quality, forage, natural cover, etc.) were identified for each life stage. Specifically, during all freshwater life stages, salmonids require cool water that is free of contaminants. During the juvenile life stage, salmonids also require stream habitat that provides excess forage (*i.e.*, prey abundance). Besides potential toxicity, water free of contaminants is important as contaminants can disrupt normal behavior necessary for successful migration, spawning, and juvenile rearing. Sufficient forage is necessary for juveniles to maintain growth that reduces freshwater predation mortality, increases overwintering success, initiates smoltification, and increases ocean survival. Natural cover such as submerged and overhanging large wood and aquatic vegetation provides shelter from predators, shades freshwater to prevent increase in water temperature, and creates important side channels. 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, Sept. 2, 2005), 12 ESUs of Oregon, Washington, and Idaho salmon (chum, sockeye, Chinook) and steelhead (70 FR 52630, Sept. 2, 2005), and for the Oregon Coast coho salmon (73 FR 7816, Feb. 11, 2008). They are:

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

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

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

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

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

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

NMFS similarly developed the following list of species habitat requirements and PCEs for coho salmon ESUs (64 FR 24049, May 5, 1999). 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.

7.2.2 Conservation Values

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) for many of the ESU/DPSs. 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.

7.3 Chinook Salmon

7.3.1 Description of the Species

Chinook salmon are the largest of the Pacific salmon and historically ranged from the Ventura River in California to Point Hope, Alaska in North America, and in northeastern Asia from Hokkaido, Japan to the Anadyr River in Russia (Healey 1991). Chinook salmon prefer streams that are deeper and larger than those used by other Pacific salmon species. We discuss the distribution, life history, status, and critical habitat of 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, Groot and Margolis 1991). One race, the "stream-type," resides in fresh water for a year or more following emergence from gravel nests. Juveniles migrate to sea as yearlings. Stream-type Chinook salmon normally return in late winter and early spring (spring-run) as immature adults and reside in deep pools during summer before spawning in fall. The other race, the "ocean-type," migrate to the ocean within their first year (sub-yearlings) and usually return as full mature adults in fall (fall-run). Fall-run adults spawn soon after river entry.

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

Successful incubation depends on several factors including dissolved oxygen (DO) levels, temperature, substrate size, amount of fine sediment, and water velocity. Chinook salmon egg incubation time is highly correlated with water temperature (McCullough 1999). Spawning sites have larger gravel and more water flow up through the gravel than the sites used by other Pacific salmon. Maximum survival of incubating eggs and the pre-emergent alevins occurs at water temperatures between about 5.5° and 13.5°C. Development time is influenced by degree days with fertilization to emergence taking up to 325 days at 2°C and about 50 days at 16°C (McCullough 1999). Fry emergence commonly begins in December and continues into mid-April (Leidy 1984). When emerging from the redd, fry move through the interstitial spaces in the redd substrate to escape the gravel. However, a high content of fines and sand in the redd substrate can severely hinder fry emergence and cause high mortality (Bjornn and Reiser 1991).

Optimal temperatures for both Chinook salmon fry and fingerlings range from 12° to 14°C (Boles 1988). Temperatures above 15°C increase the risk of diseases and lower the tolerance to other stressors (McCullough 1999). At about 19°C, Chinook salmon cease to eat. In the laboratory, 50% mortality during a 24 hour period is observed at 24° to 25°C (Brett 1952, Hanson 1997) the exact lethal temperature being somewhat dependent on the temperature that the fish has been acclimated to.

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

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

Chinook salmon fry may also move into non-natal tributaries (*i.e.,* streams other than those where they incubated) to rear (Limm and Marchetti 2009, Teel et al. 2009). In 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 2003, Sommer et al. 2001). The increased growth rate is likely caused by the higher water temperatures as well as the higher prey densities in these habitats. Having sufficient growth during the juvenile stage is critical as some studies indicate that size at smolting influence survival during the first year in the ocean. As flow decreases and water temperature increases in summer, juveniles move out of the inundated floodplain habitat or succumb to lethal temperatures and stranding.

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

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

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

7.3.2 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. Hazards from climate change include elevated water temperature, earlier spring runoff and lower summer flows, and winter flooding.

7.4 Puget Sound Chinook Salmon

The Puget Sound ESU (Figure 6) 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 7). 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 7. 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%

Independent Populations	Historical Abundance	Mean Number of Spawners	Hatchery Abundance Contributions
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

7.4.1 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 (Hayman et al. 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).

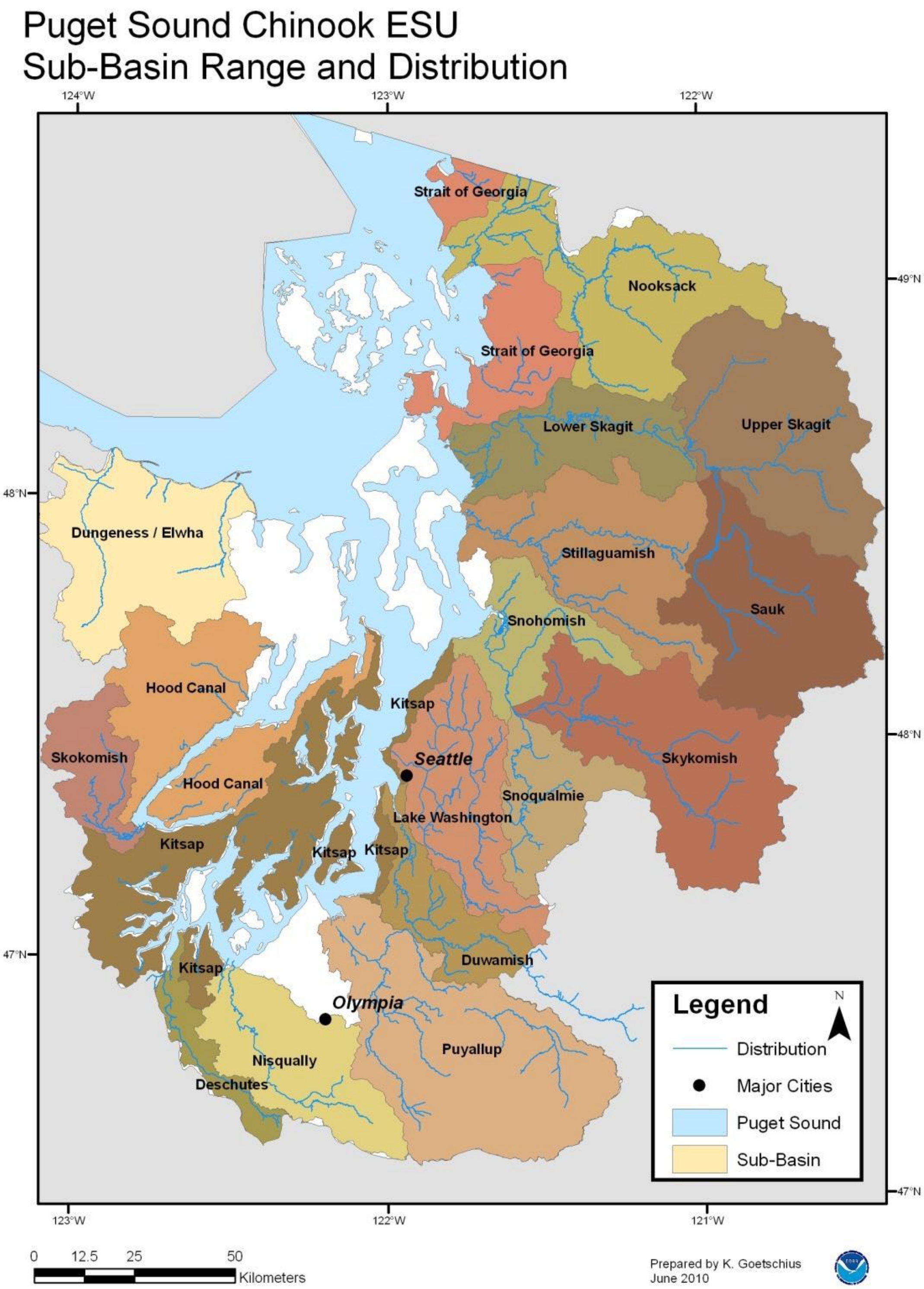

Figure 6. Puget Sound Chinook salmon distribution

7.4.2 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 7). 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.

7.4.3 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 (Figure 7). The 19 nearshore marine areas were all given a high conservation value rating. (Table 8).

Table 8. 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.

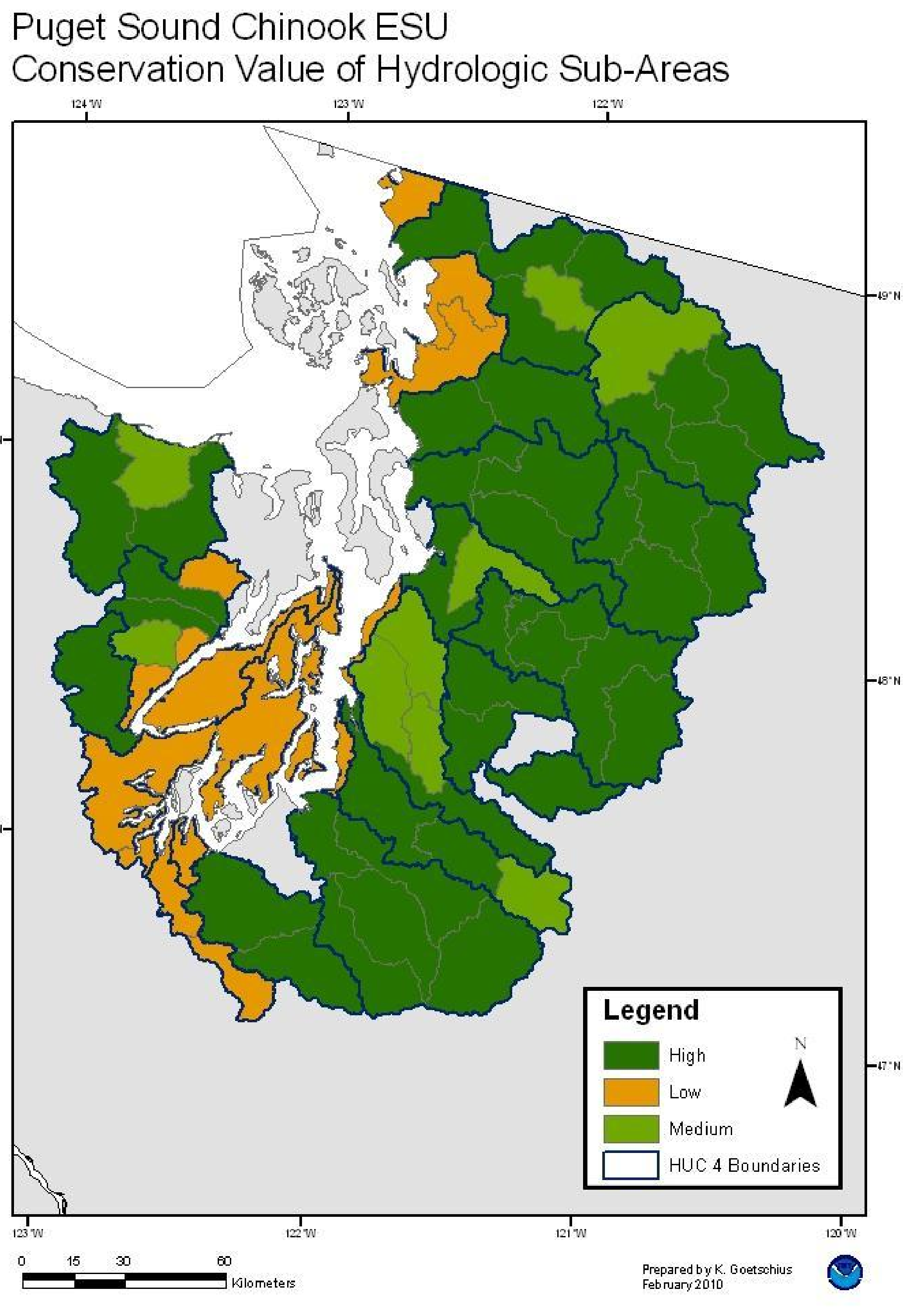

Figure 7. Puget Sound Chinook salmon Conservation Values per Sub-watershed

7.5 Lower Columbia River Chinook Salmon

The Lower Columbia River (LCR) Chinook salmon ESU (Figure 8) 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.

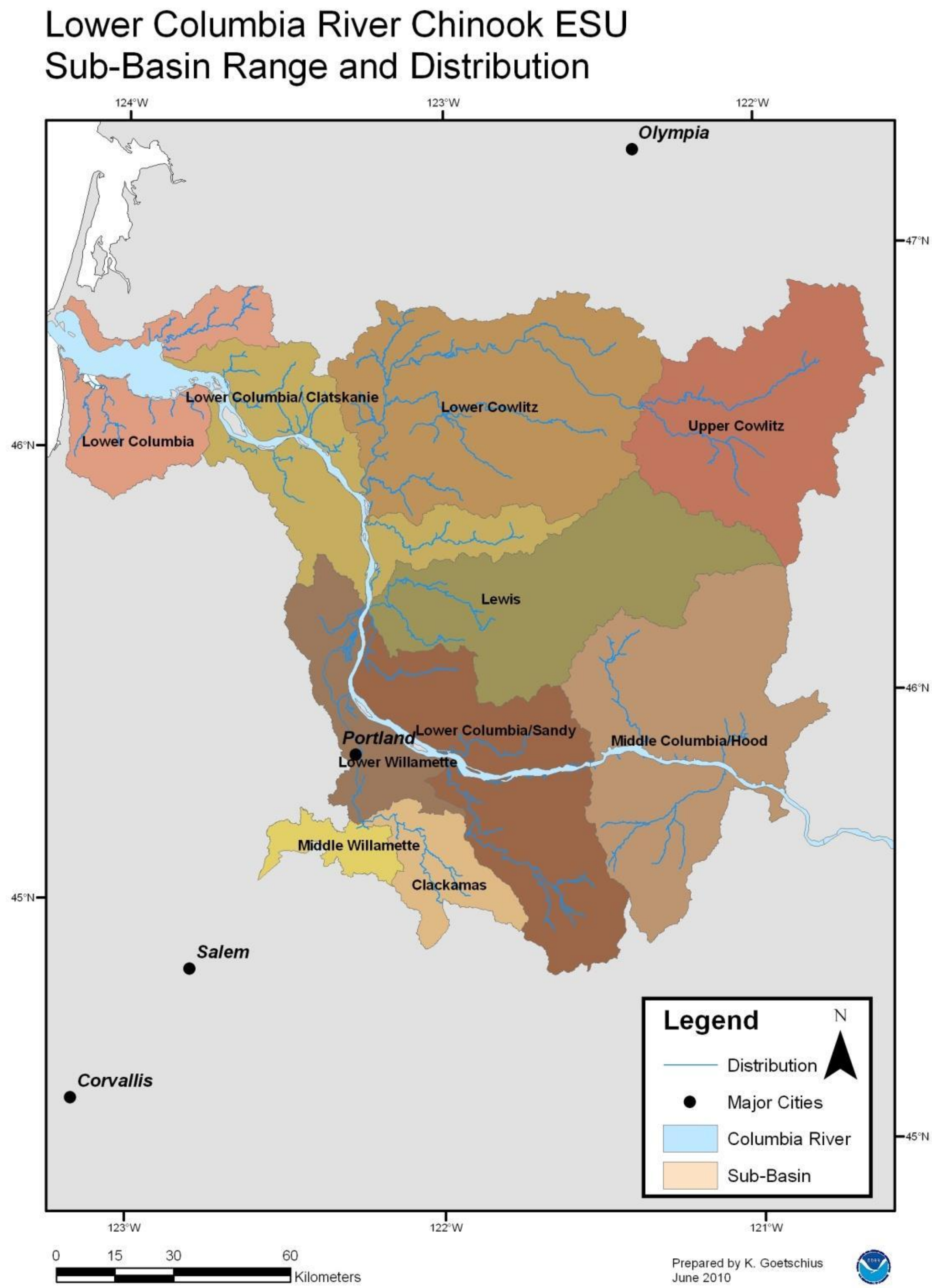

Figure 8. Lower Columbia River Chinook salmon distribution.

7.5.1 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.

7.5.2 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 self-sustaining (Good et al. 2005). Table **9** identifies populations within the LCR Chinook salmon ESU, their abundances, and hatchery input.

Table 9. **Lower Columbia River Chinook salmon - population structure, abundances, and hatchery contributions (Good et al. 2005, Myers et al. 2006).**

Run	Population	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
	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

Run	Population	Historical Abundance	Mean Number of Spawners	Hatchery Abundance Contributions
	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.

7.5.3 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.

As shown in Figure 9, 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 10). 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 10. 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.

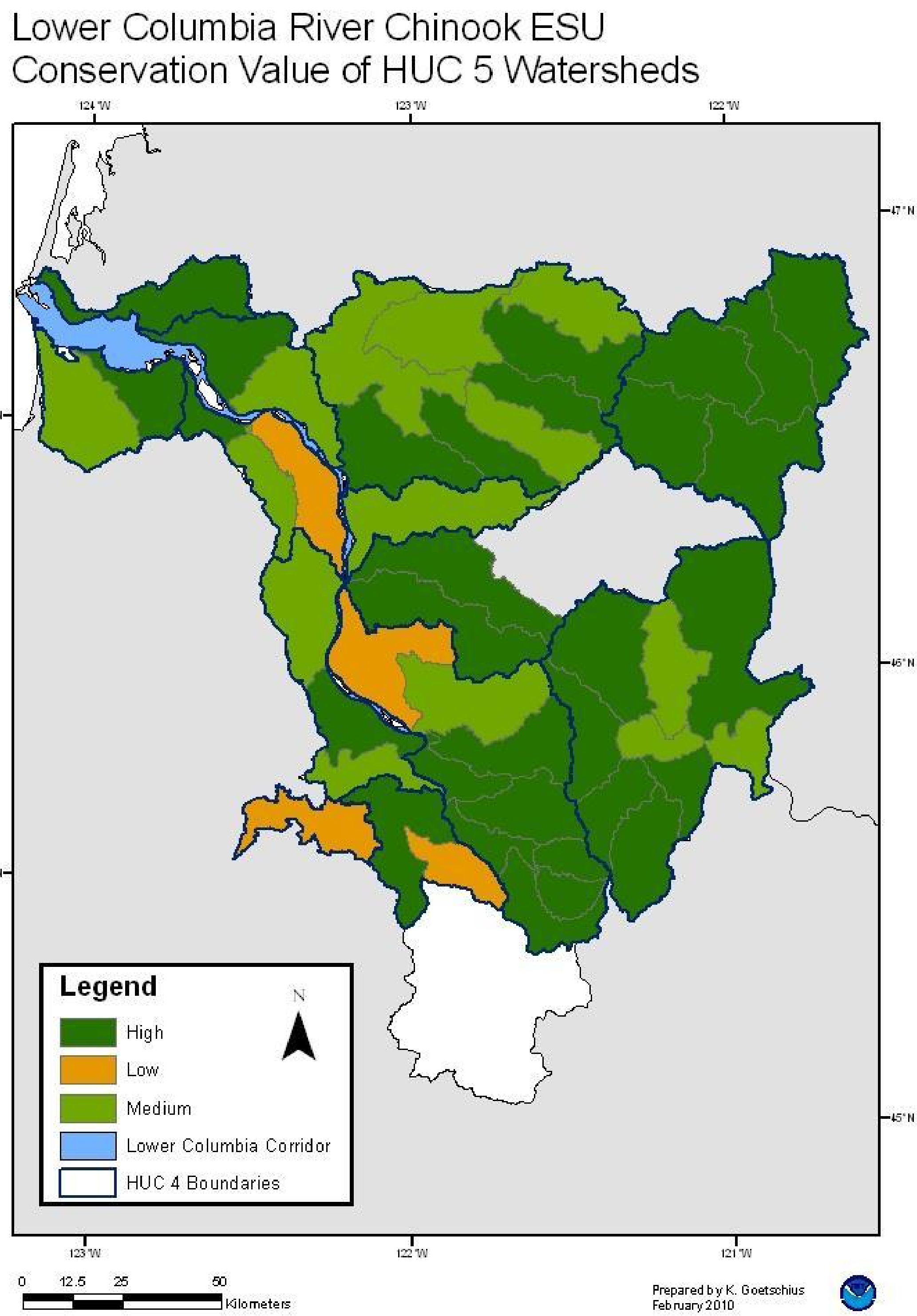

Figure 9. Lower Columbia River Chinook salmon Conservation Values per Sub-Area

7.6 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 (Figure 10).

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.

7.6.1 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.

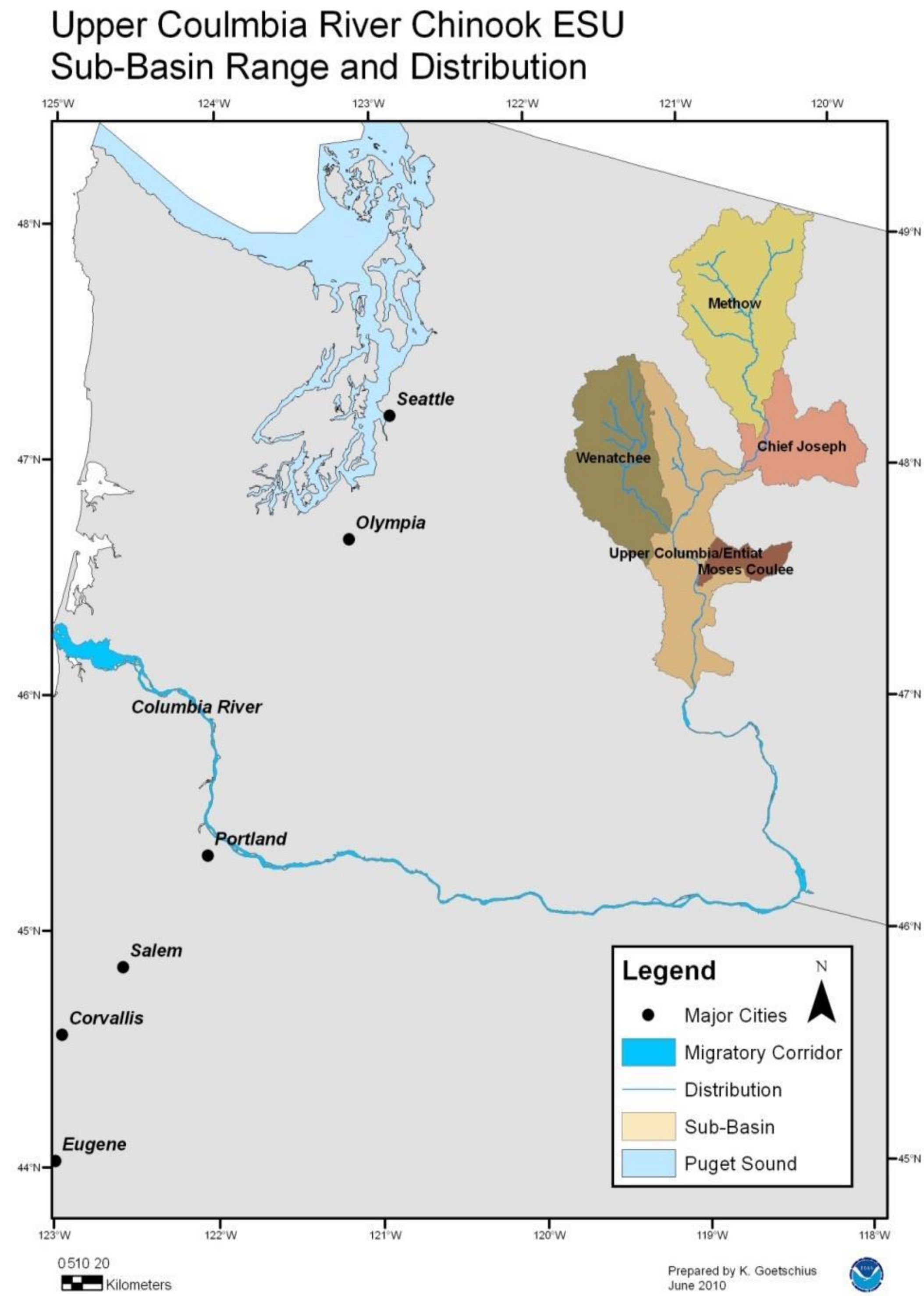

Figure 10. Upper Columbia River Chinook salmon distribution

7.6.2 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 11 identifies populations within the UCR Spring-run Chinook salmon ESU, their abundances, and hatchery input.

Table 11. Upper Columbia River Spring-run Chinook salmon - preliminary population structure, abundances, and hatchery contributions

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	Unknown	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. 2005).

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 2008b, ICTRT 2008a, 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 2008b, ICTRT 2008a, 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.

7.6.3 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 12). The Columbia River rearing/migration corridor downstream of the spawning range was rated as having a high conservation value (Figure 11).

Spawning and rearing PCEs are somewhat degraded in tributary systems by urbanization in lower reaches, grazing in the middle reaches, and irrigation and diversion in the major upper drainages. 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. Moreover siltation further affects critical habitat by reducing water quality through contaminated agricultural runoff; and removing 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.

Table 12. 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.

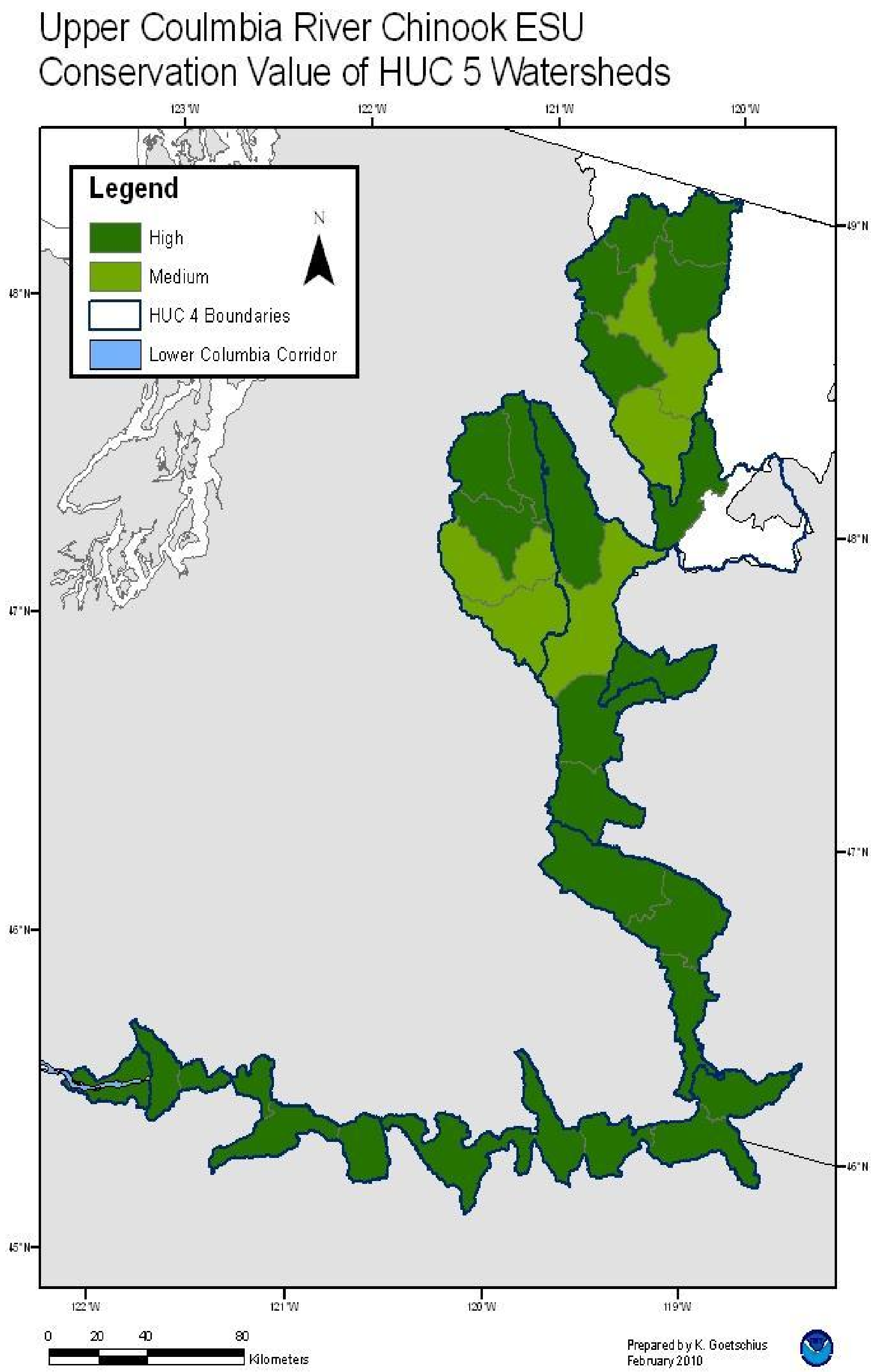

Figure 11. Upper Columbia River Spring-run Chinook salmon Conservation Values per Sub-Area

7.7 Snake River Fall-run Chinook Salmon

The Snake River (SR) Fall-run Chinook salmon ESU (Figure **12**) 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.

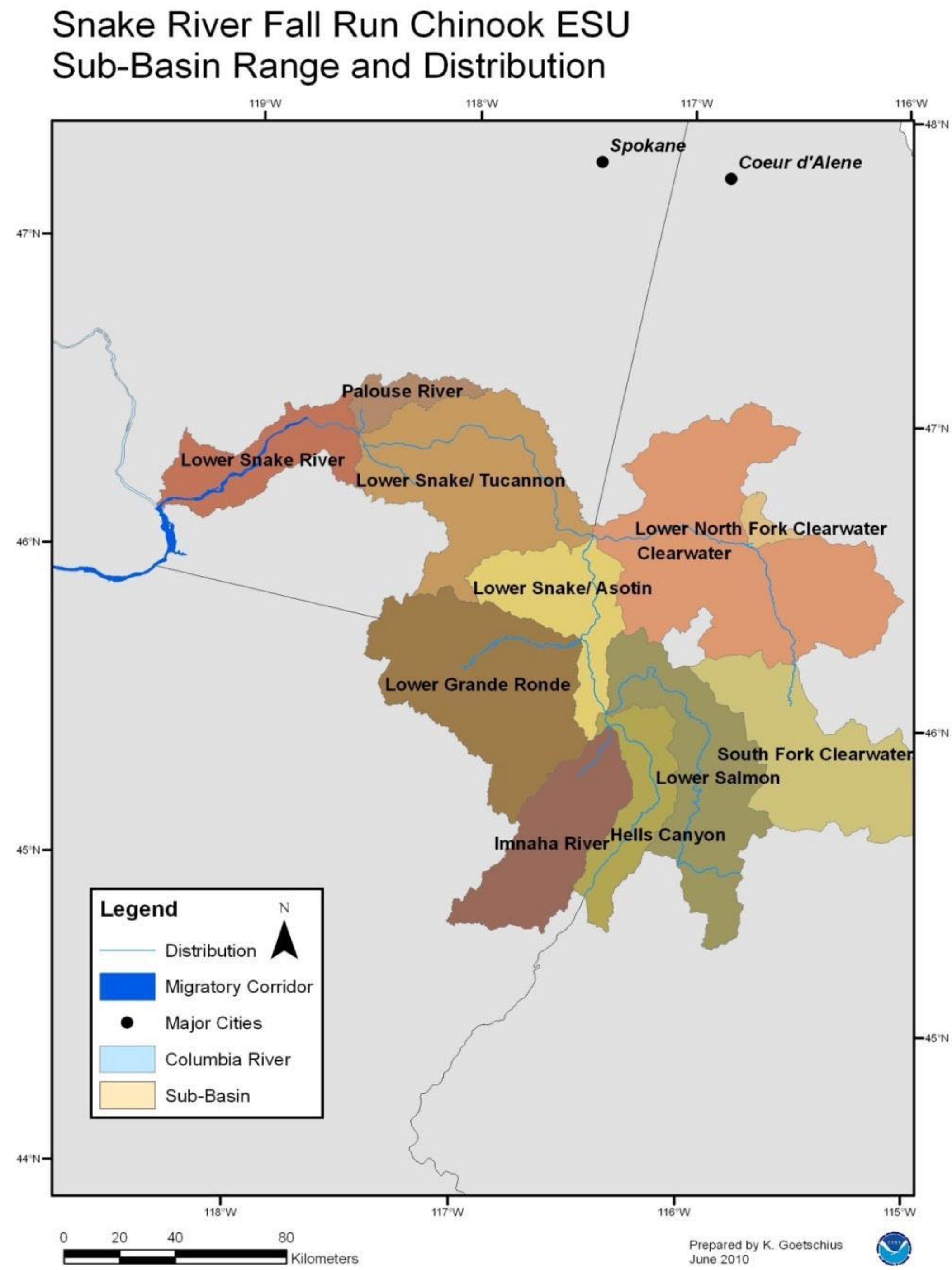

Figure 12. Snake River Fall-run Chinook salmon distribution

7.7.1 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 Horner 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.

7.7.2 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 juvenile migration which is reduced by the presence of the Snake and Columbia River hydropower system within the lower mainstem; (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 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; traveling along with 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.

7.8 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 **13**). 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.

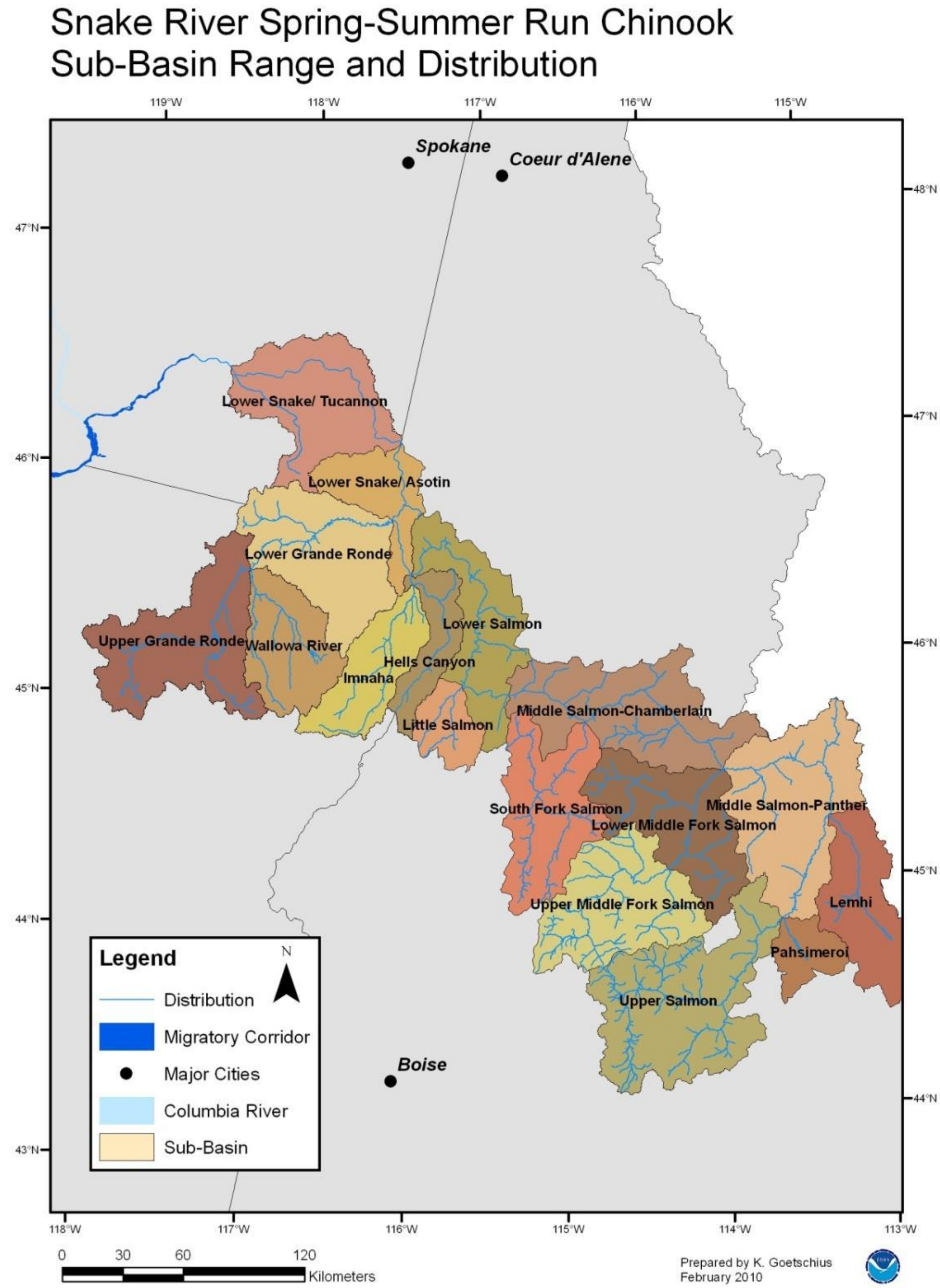

Figure 13. Snake River Spring/Summer-run Chinook salmon distribution.

7.8.1 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.

7.8.2 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 13). 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 13 identifies populations within the Snake River Spring/Summer-run Chinook salmon ESU, their abundances, and hatchery input.

Table 13. Snake 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 (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).

7.8.3 Critical Habitat

NMFS designated critical habitat for the Snake River (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 Julius 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.

7.9 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 (**Figure 14**). Seven artificial propagation programs are included in the ESU (70 FR 37160, June 28, 2005). 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.

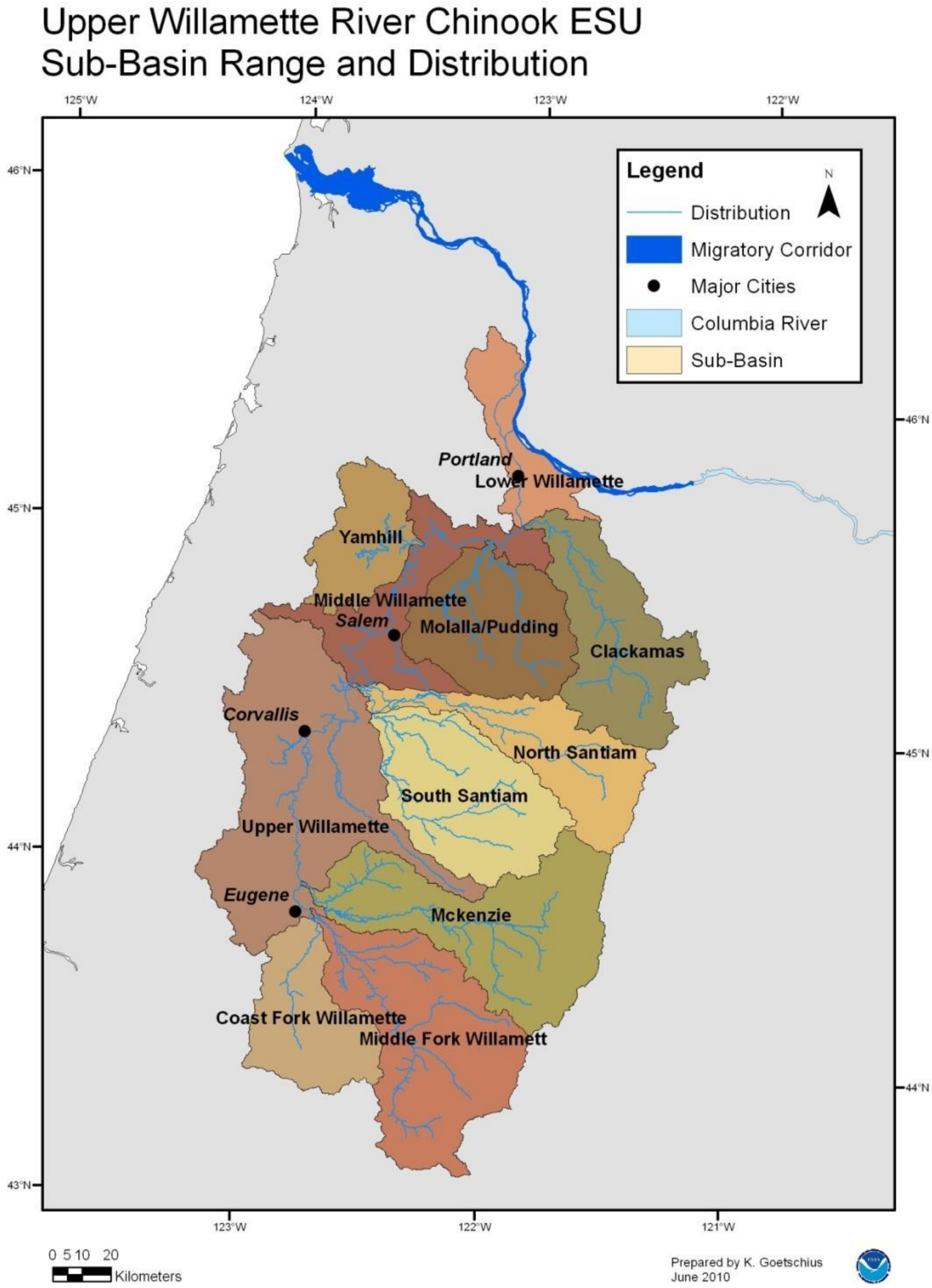

Figure 14. Upper Willamette River Chinook salmon distribution

7.9.1 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.

7.9.2 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 14 identifies populations within the UWR Chinook salmon ESU, their abundances, and hatchery input.

The W/LCRTRT identified seven historical independent populations (Myers et al. 2006) (Table 14). 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 14. 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).

7.9.3 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 15). Nineteen watersheds received a low rating, 18 received a medium rating, and 22 received a high rating of conservation value to the ESU (NMFS 2005b). 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.

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.

Table 15. 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

7.10 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 15**. 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. The Russian River is of particular importance for preventing the extinction and contributing to the recovery of CC Chinook (NOAA 2013).

7.10.1 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.

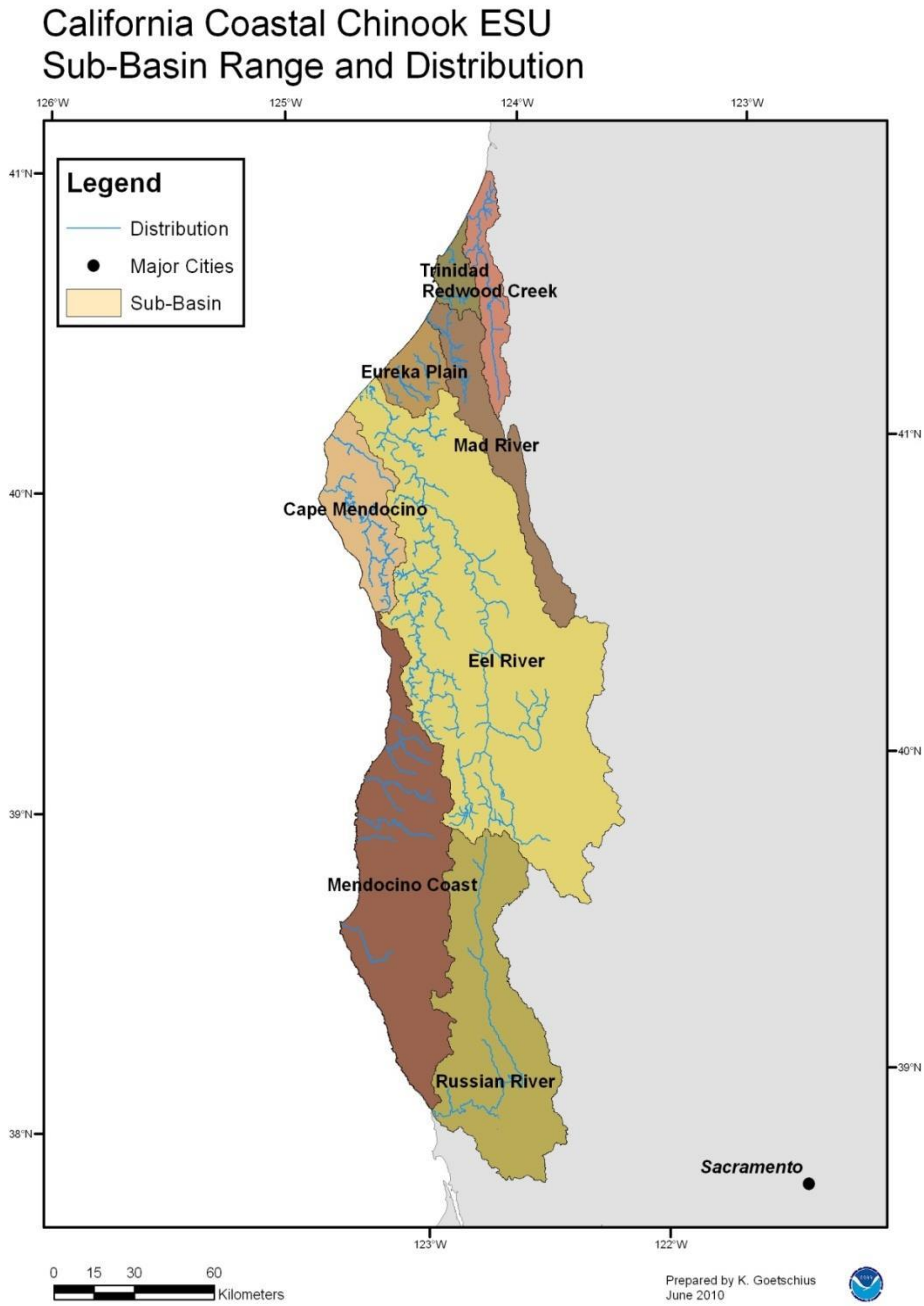

Figure 15. California Coastal Chinook salmon distribution

Table 16. **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
Humboldt 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	

7.10.2 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. 2008a). The number of extant populations is uncertain. The Russian River and the Eel River have the largest populations of CC Chinook salmon of the ESU and due to their size are critically important to rebuilding the smaller extirpated populations and preserving genetic diversity. Chinook adult escapement has been steadily increasing and recent Russian River population estimates are at, or near, recovery targets (NOAA 2013).

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 16** 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. 2008a). The Eel River interior fall-run populations are severely depressed (Spence et al. 2008a). 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 (SCWA) 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.

7.10.3 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 17). 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, Sept. 2, 2005).

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 (Figure 16).

Table 17. 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.

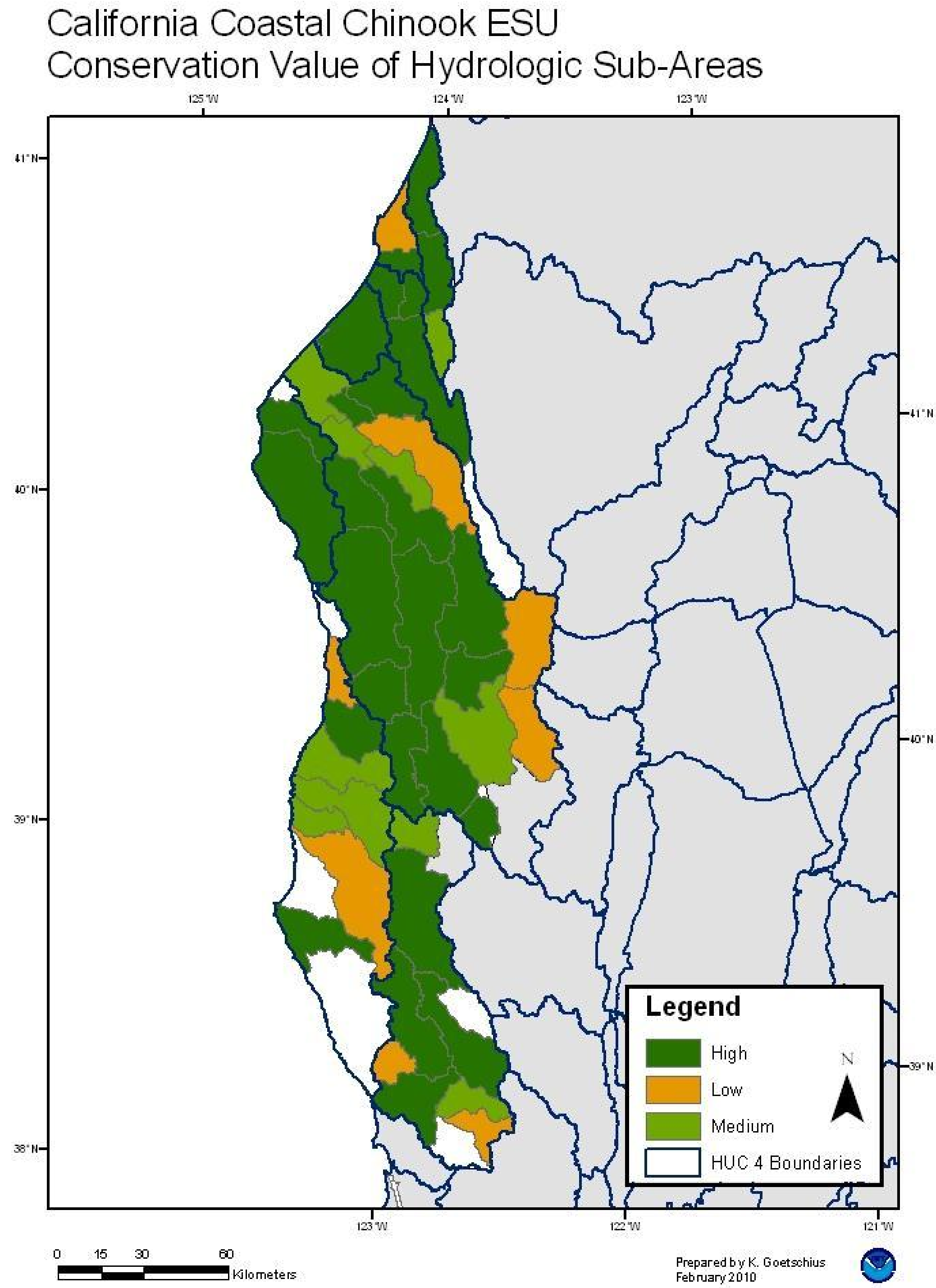

Figure 16. California Coastal Chinook salmon Conservation Values per Sub-Area

7.11 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 17**). 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 18** identifies populations within the CV Spring-run Chinook salmon ESU, their abundances, and hatchery input.

7.11.1 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.

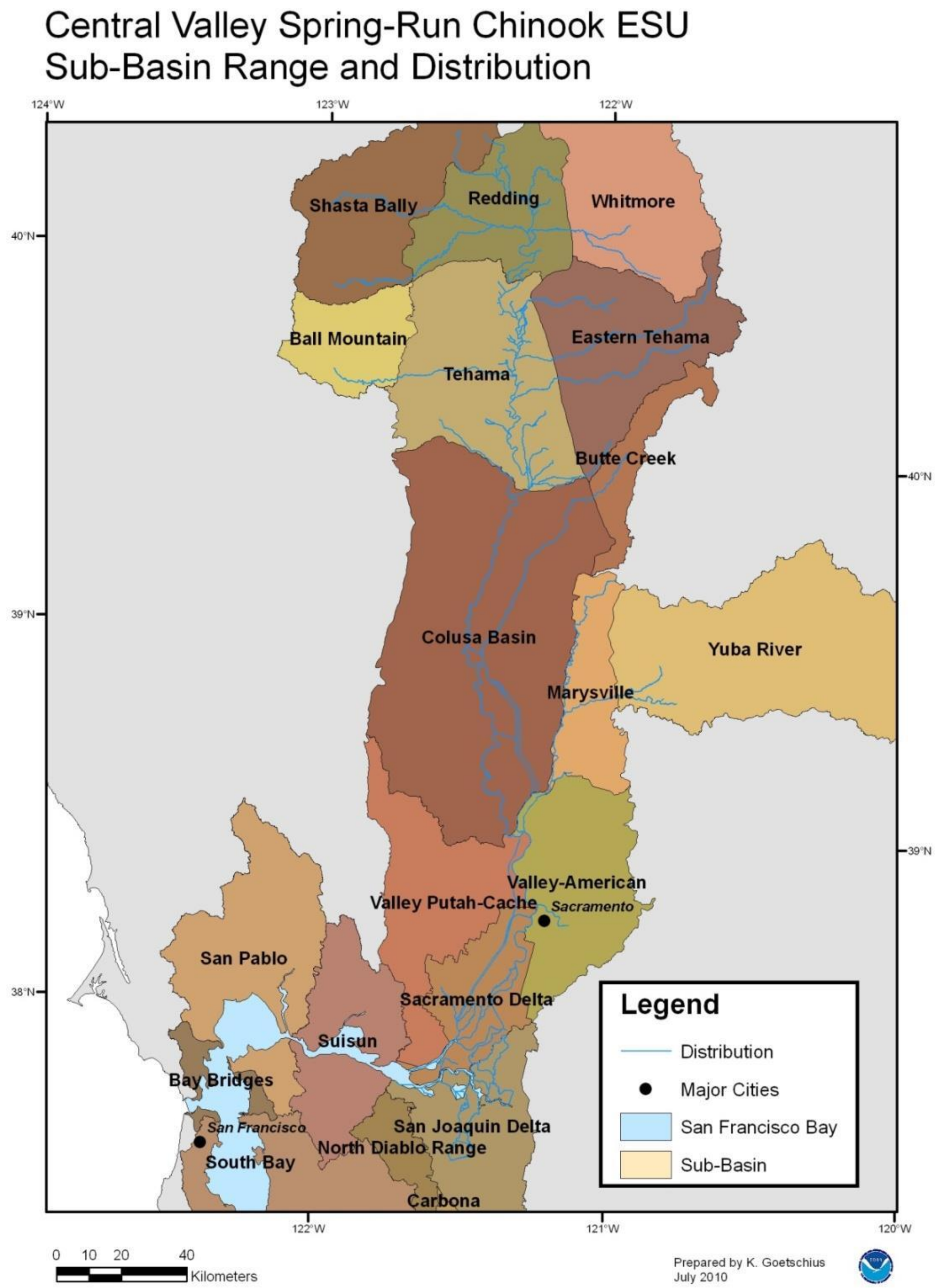

Figure 17. **Central Valley Spring-run Chinook salmon distribution**

7.11.2 Status and Trends

NMFS originally listed CV Spring-run Chinook salmon as threatened on September 16, 1999 (64 FR 50393), and reaffirmed their threatened status on June 28, 2005 (70 FR 37160). 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 18. Central Valley Spring-run Chinook salmon--preliminary population structure, historic and most recent natural production, spawner abundance, and hatchery contributions (Good et al. 2005, USFWS and Reclamation 2007)

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 and Reclamation 2007). Numbers of naturally produced fish increased significantly in

1998 to an estimated 48,755 adults and estimated natural production has remained above 10,000 fish since then (USFWS and Reclamation 2007).

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 and 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.

7.11.3 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 19).

There are 38 occupied HSA watersheds within the freshwater and estuarine range of this ESU. As shown in Figure 18, 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.

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

Table 19. CV Spring-run Chinook salmon CALWATER HSA watersheds with conservation values

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

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

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

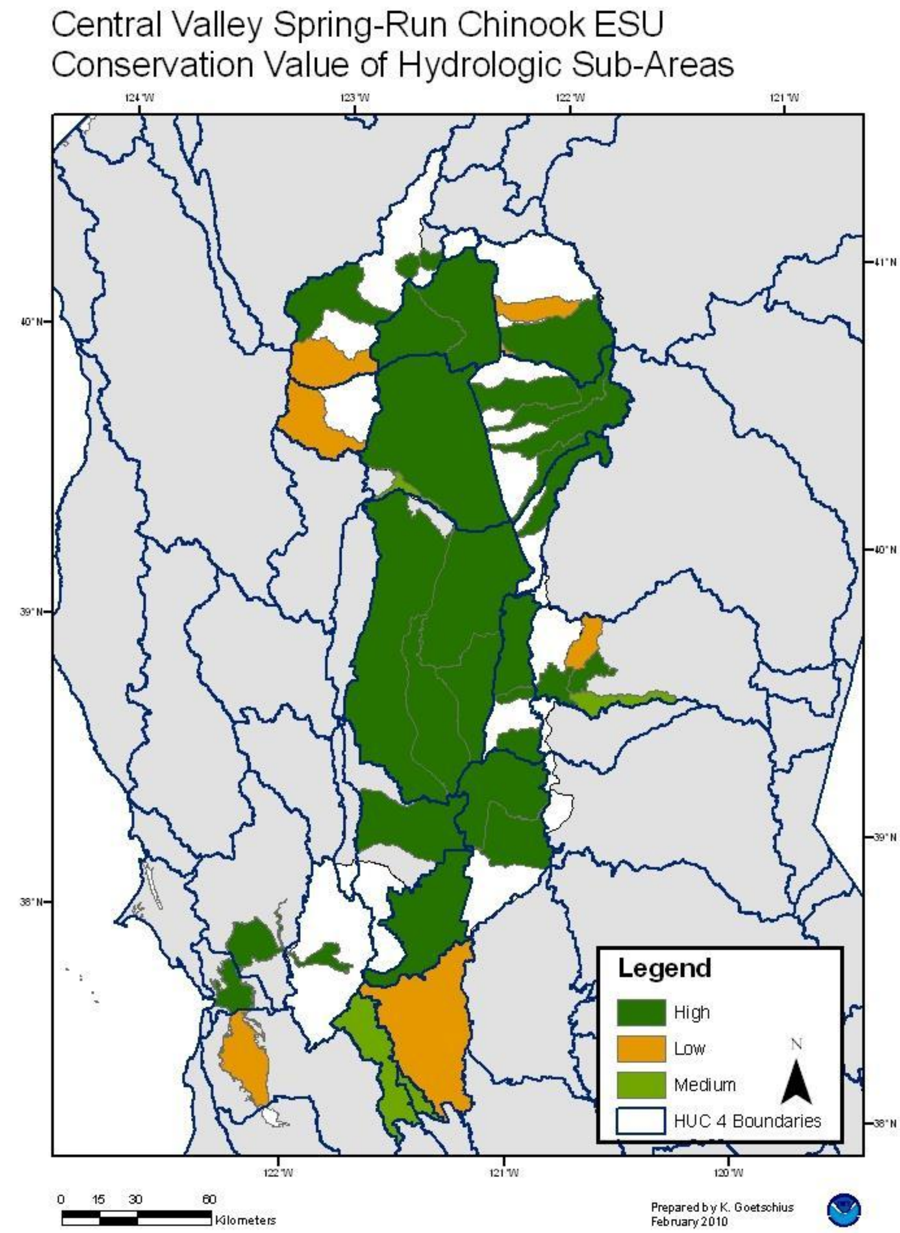

Figure 18. Central Valley Spring-run Chinook salmon Conservation Values per Sub-Area

7.12 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 19**). The ESU now consists of a single spawning population.

7.12.1 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.

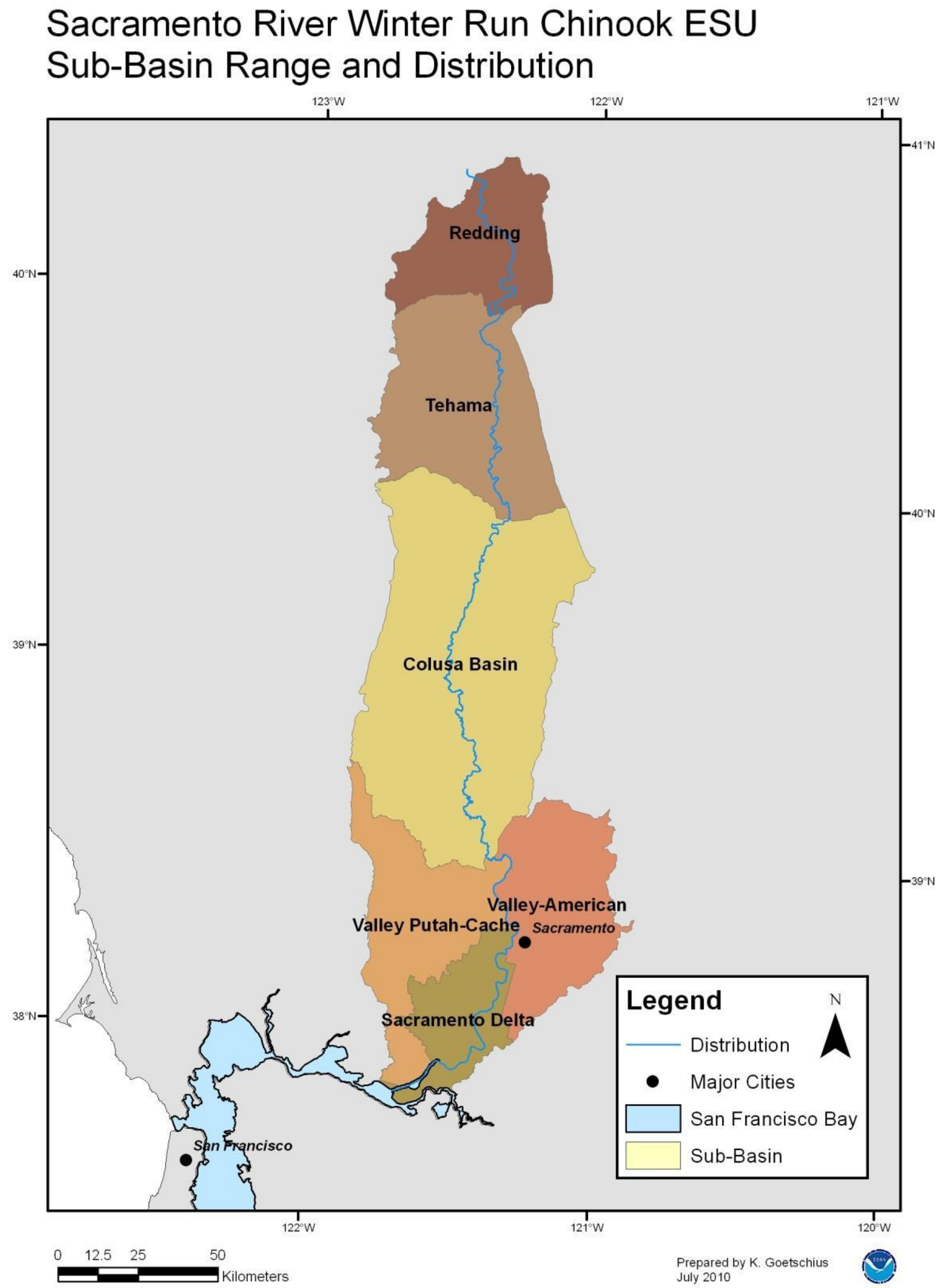

Figure 19. **Sacramento River Winter-run Chinook salmon distribution**

7.12.2 Status and Trends

NMFS listed Sacramento River Winter-run Chinook salmon as endangered on January 4, 1994 (59 FR 440), and reaffirmed their endangered status on June 28, 2005 (70 FR 37160). The winter-run Chinook salmon spawned and reared in the upper Sacramento River and its tributaries (Slater 1963, Yoshiyama et al. 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 (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 (USFWS and Reclamation 2007). However, the natural produced winter-run Chinook salmon plunged in 2007 and 2008, with 4,461 adults estimated for 2007 and a preliminary estimate between of 2,600-2,950 adults for 2008 (USFWS 2008).

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

7.12.3 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.

7.13 *Chum Salmon*

7.13.1 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 (Johnson et al. 1997). Chum salmon juveniles use shallow, low flow habitats for rearing that include inundated mudflats, tidal wetlands and their channels, and sloughs.

7.13.2 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). Climate change also poses significant hazards to the survival and recovery of salmonids. Hazards from climate change include elevated water temperature, earlier spring runoff and lower summer flows, and winter flooding.

7.14 Hood Canal Summer-run Chum Salmon

The Hood Canal (HC) Summer-run chum salmon ESU (Figure 20) 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 (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 20 identifies populations within the HC Summer-run chum salmon ESU, their abundances, and hatchery input.

Table 20. 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.

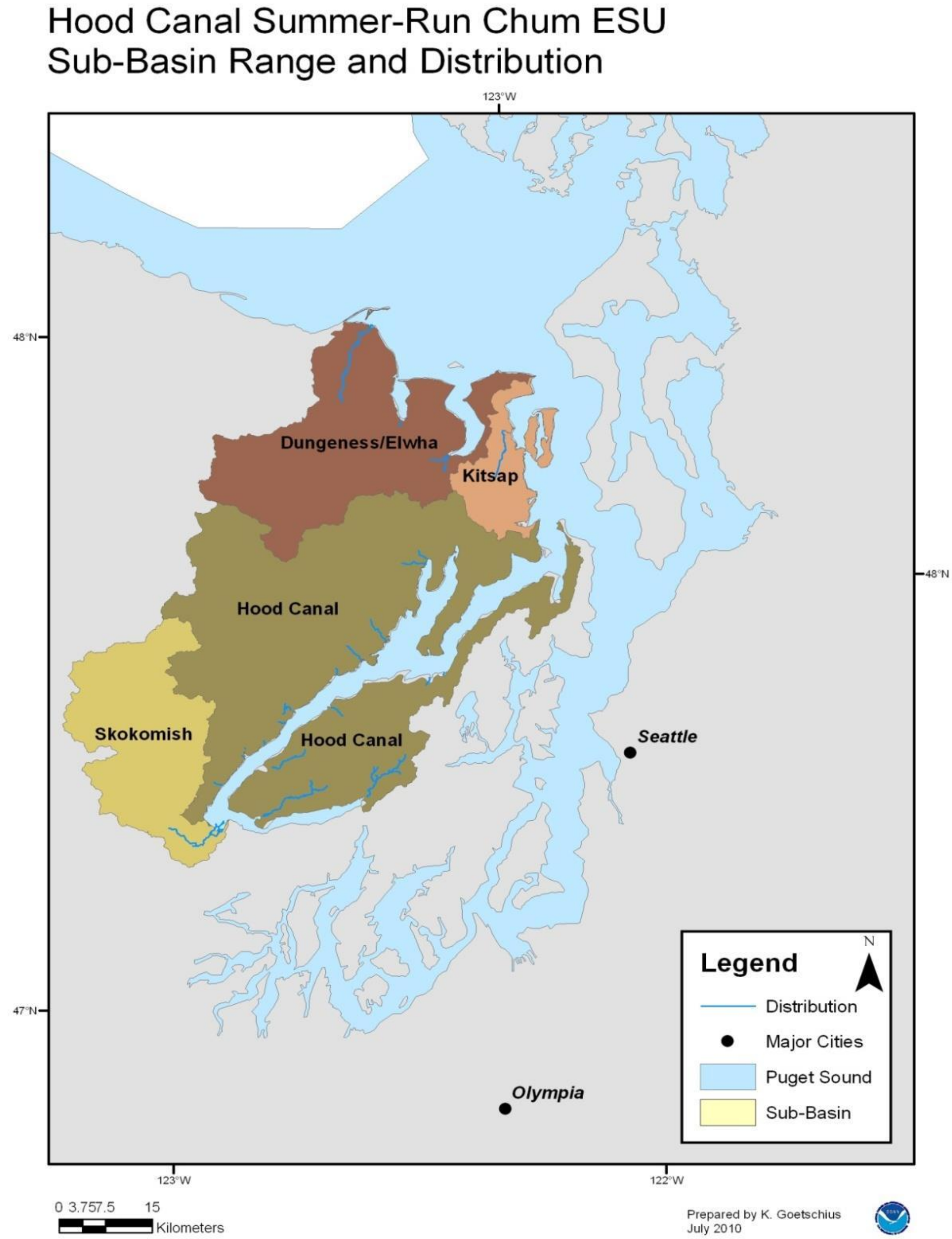

Figure 20. Hood Canal Summer-run Chum salmon distribution

7.14.1 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 (Washington Department of Fish and Wildlife (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.

7.14.2 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.

7.14.3 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 (Figure 21), nine watersheds were rated as having a high conservation value while three were rated as having a medium value for conservation (Table 21). 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 21. 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.

The spawning PCE is degraded by excessive fine sediment in the gravel, and the 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.

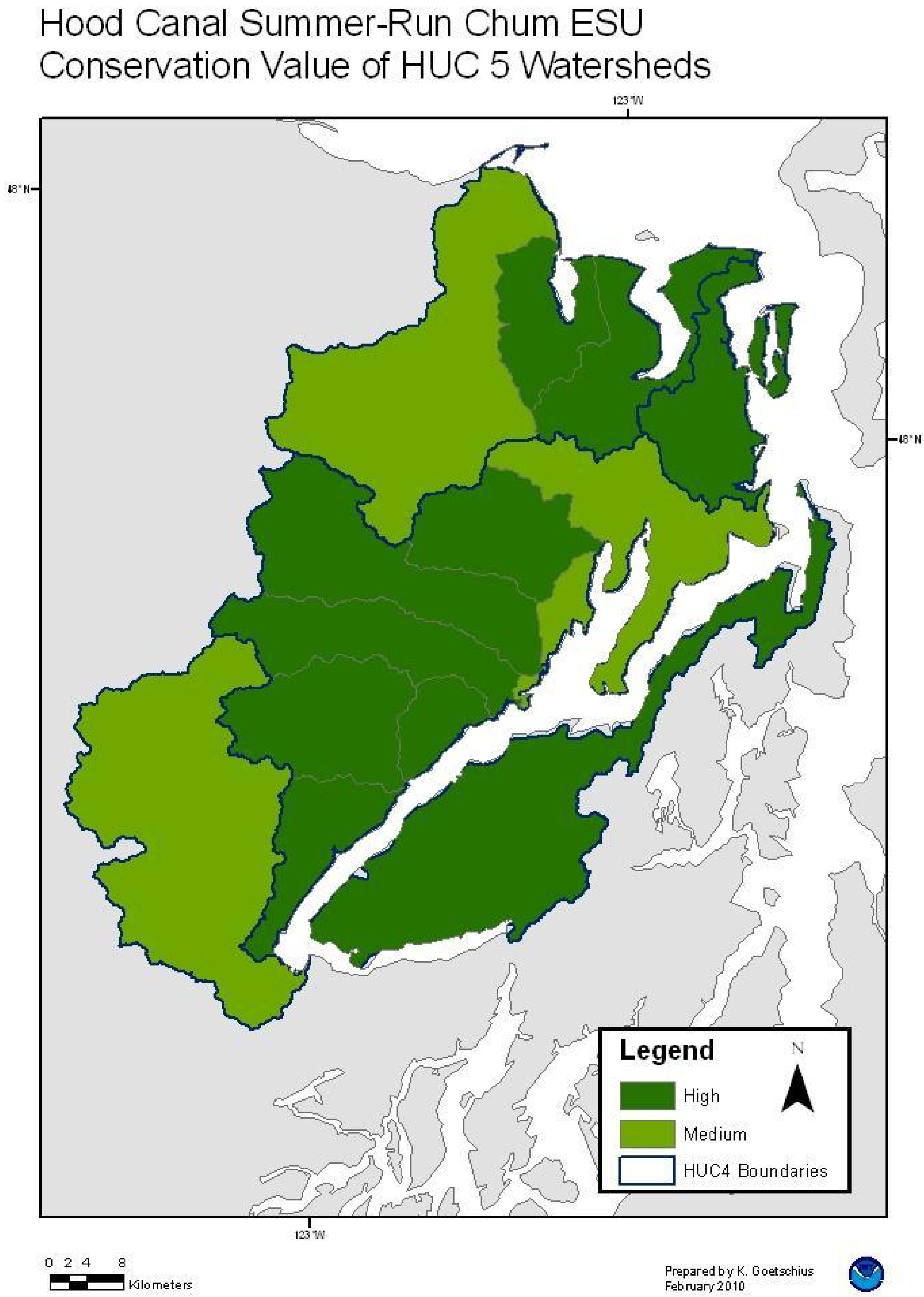

Figure 21. Hood Canal Summer-run Conservation Values per Sub-area

7.15 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 **22**). 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.

Table 22. 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.

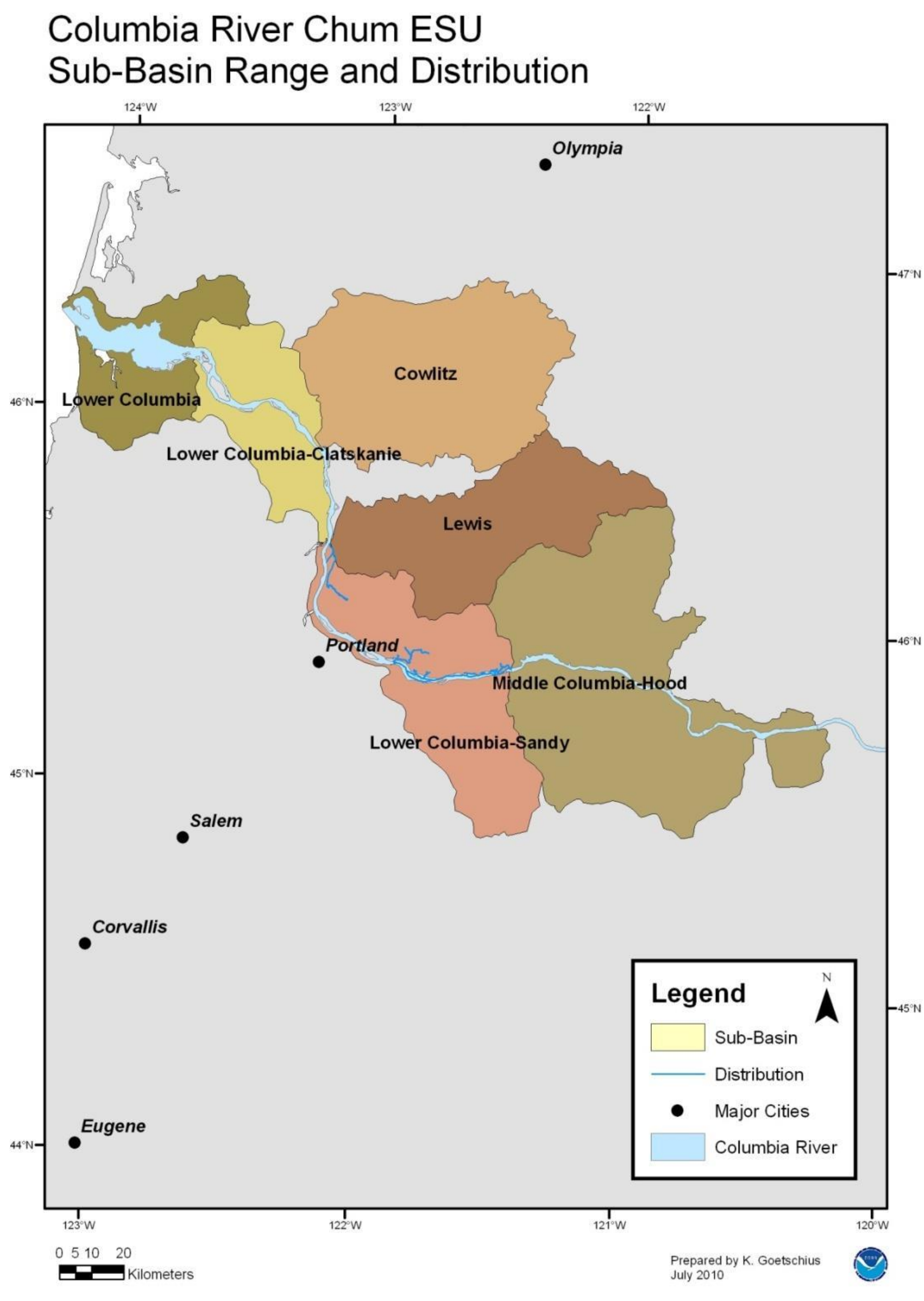

Figure 22. **Columbia River Chum salmon distribution**

7.15.1 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.

7.15.2 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 22). 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, Washington Department of Fish and Wildlife (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 (Washington Department of Fish and Wildlife (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.

7.15.3 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 23). The remaining three subbasins were given a medium conservation value (Figure 23). Washington's federal lands were rated as having high conservation value to the species.

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.

Table 23. 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.

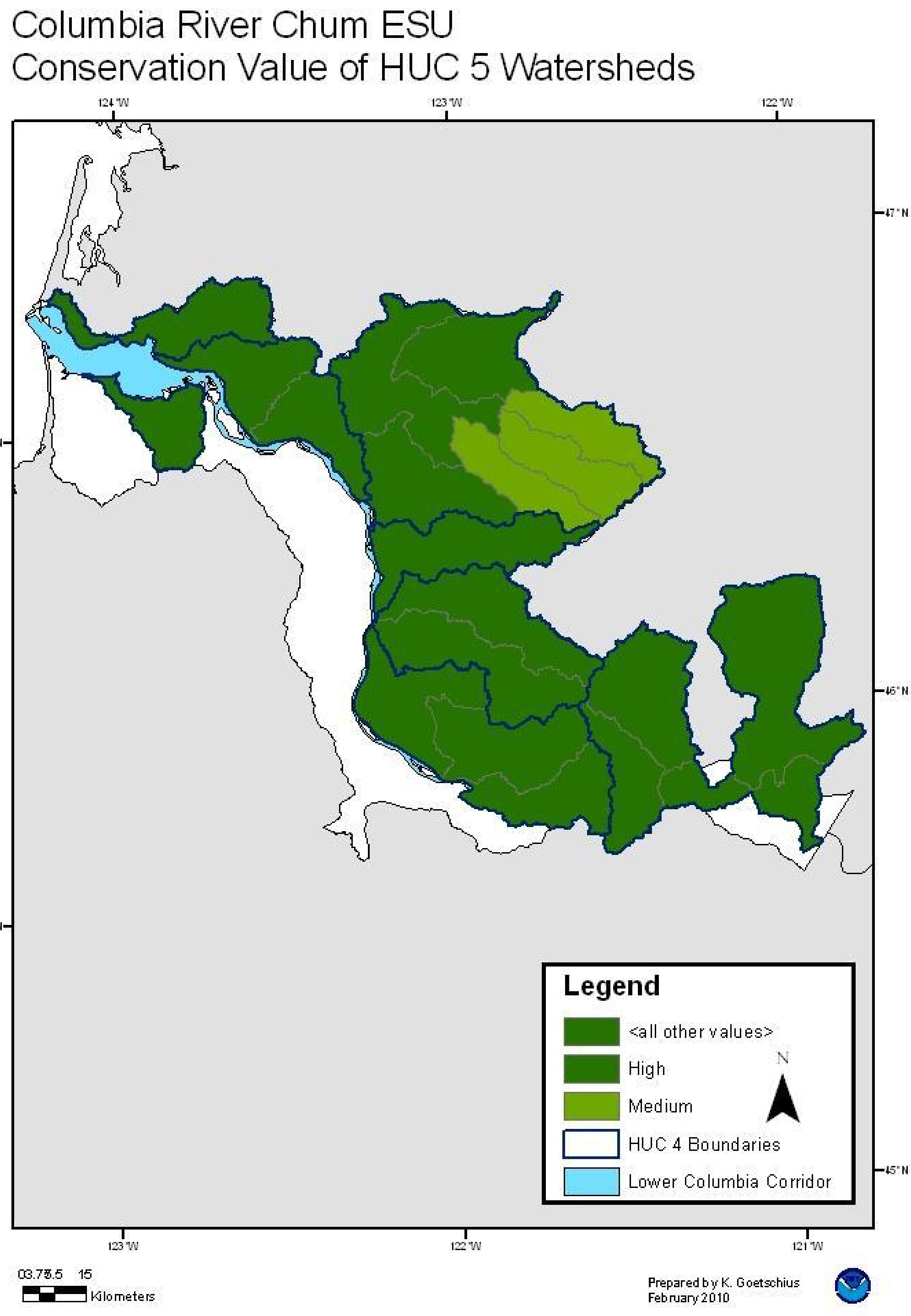

Figure 23. Columbia River Chum salmon Conservation Values per Sub-area

7.16 *Coho Salmon*

7.16.1 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, Rosenfeld 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, Tschaplinski and Hartman 1983). High densities of juvenile coho salmon also occur in log jams (Brown 1985, Tschaplinski and Hartman 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 1975a, Nickelson et al. 1992b, 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, 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 1975b, Nickelson et al. 1992b). 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.

7.16.2 Status and Trends

Coho salmon depend on the quantity and quality of the freshwater aquatic systems for spawning, rearing, and on the ocean conditions where they grow to maturity. Coho salmon have declined from overharvests, hatchery supplementation, 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). Climate change also poses significant hazards to the survival and recovery of salmonids. Hazards from climate change include elevated water temperature, earlier spring runoff and lower summer flows, and winter flooding.

7.17 Lower Columbia River (LCR) 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, Washington, and the Willamette River to Willamette Falls, Oregon (Figure **24**). This ESU also includes 25 artificial propagation programs (70FR 37160, June 28, 2005).

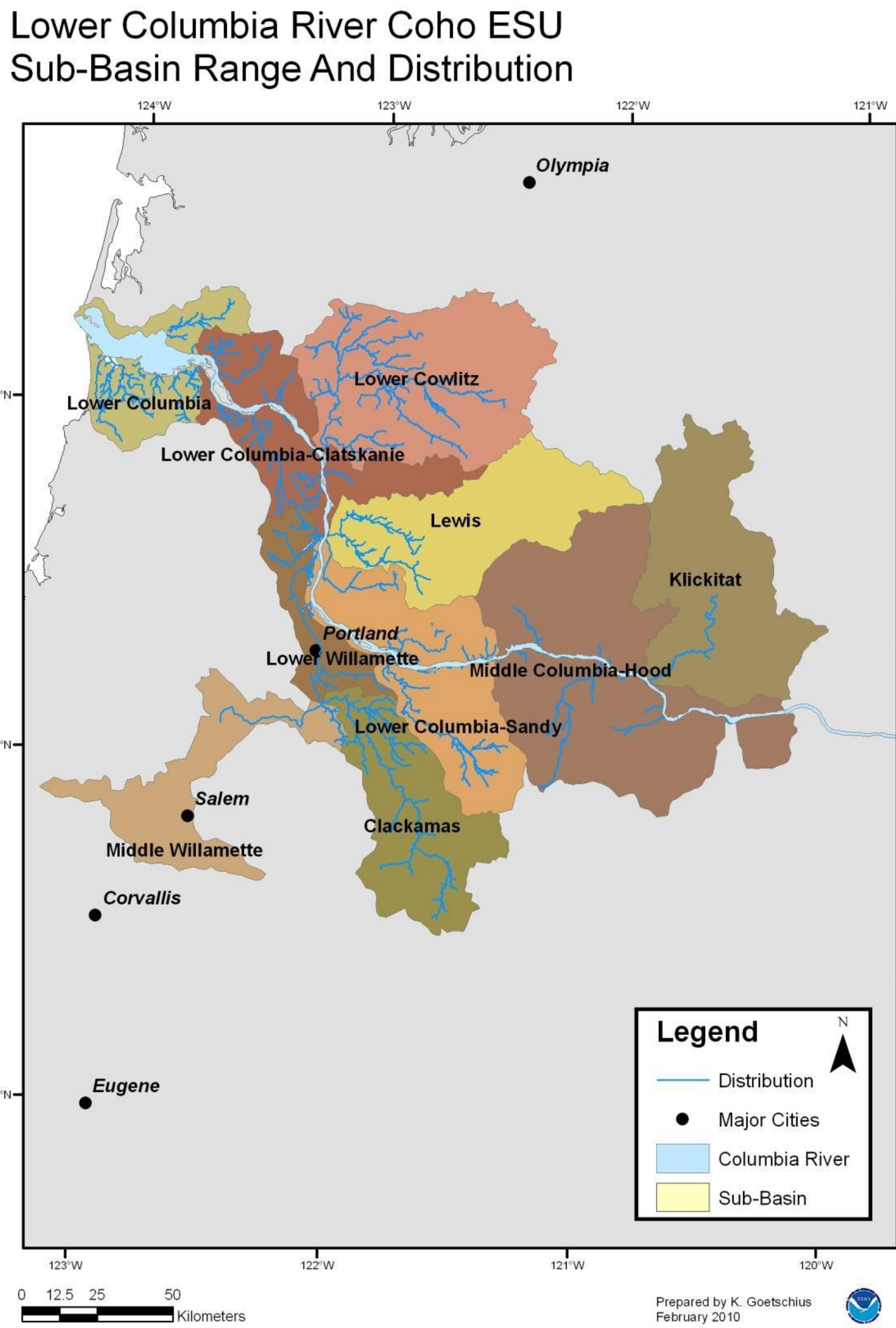

Figure 24. LCR coho salmon distribution

7.17.1 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).

7.17.2 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 24). 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 24. 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 spawners. 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).

7.17.3 Critical Habitat

NMFS proposed critical habitat for Lower Columbia River coho salmon on January 14, 2013 (50 CFR Part 226). Final designations are expected late in 2013, or by early in 2014. 33 of the 54 subbasins reviewed in NMFS' assessment of the LCR coho salmon ESU's proposed designation was rated as having a high conservation value. 18 were given a medium conservation value, and three were of low conservation value (Figure 25).

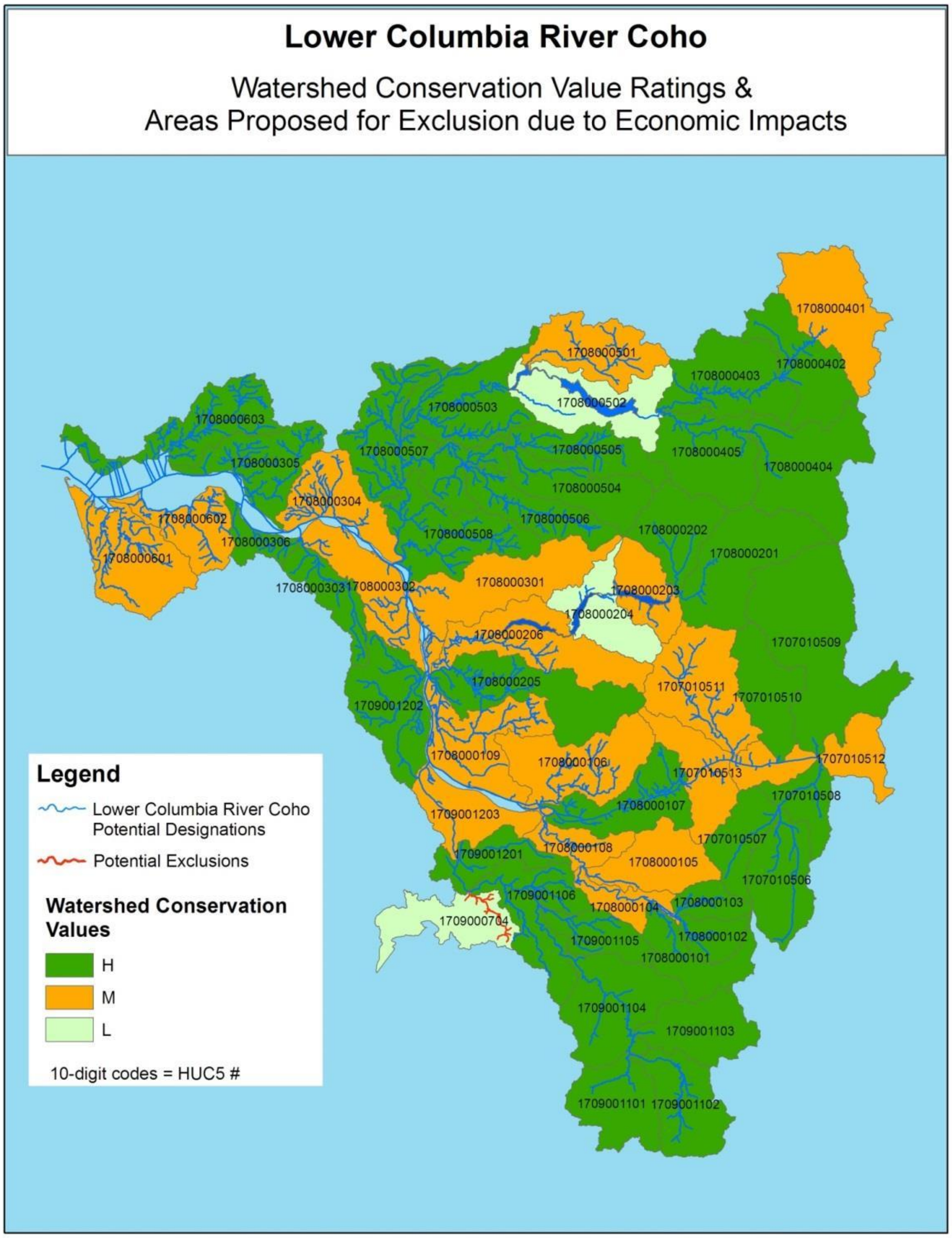

Figure 25. Proposed designated critical habitat for the Lower Columbia River coho salmon ESU

7.18 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 26**). 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.

7.18.1 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.

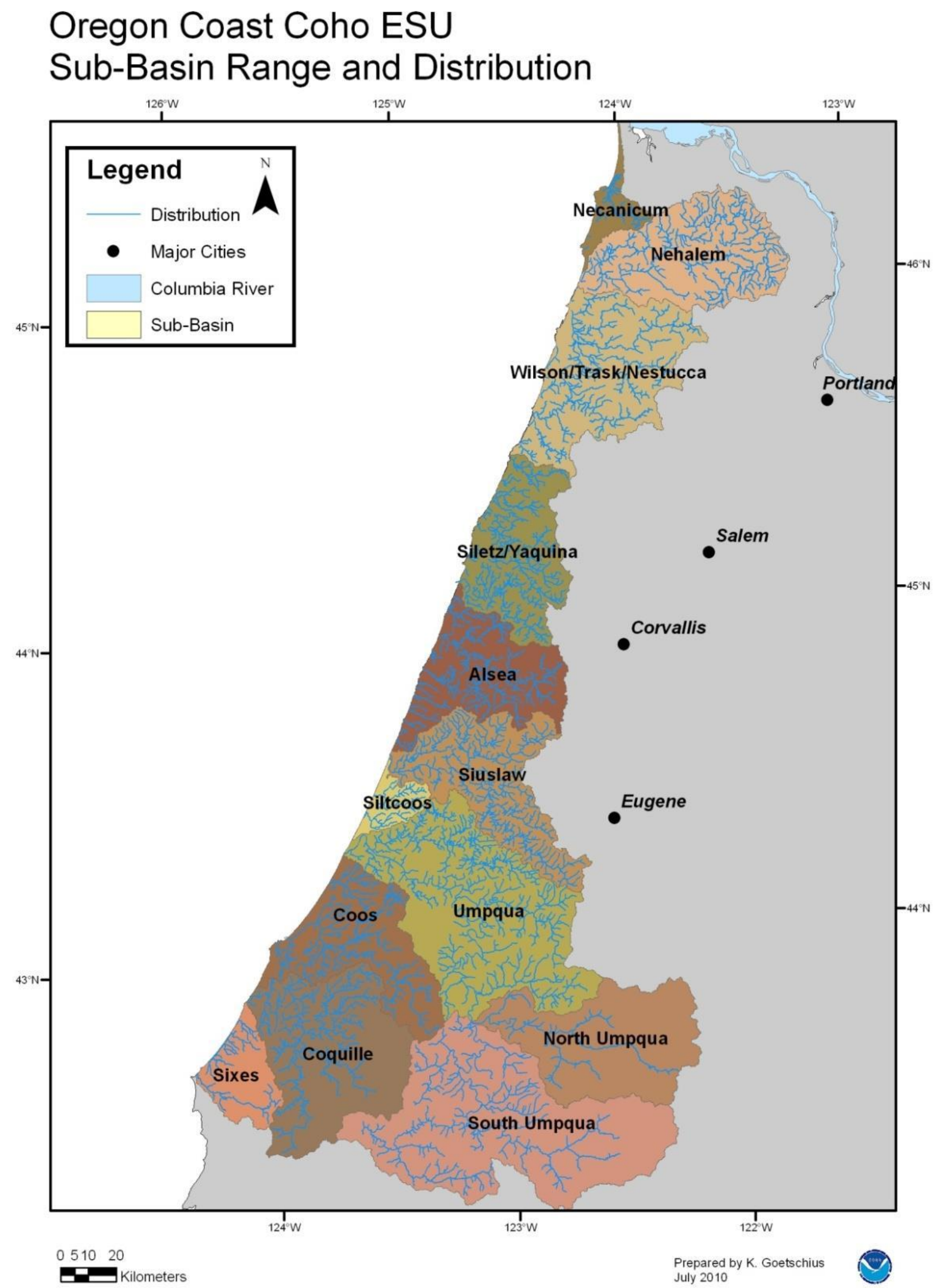

Figure 26. Oregon Coast Coho salmon distribution

7.18.2 Status and Trends

NMFS listed the OC coho salmon as a threatened species on February 11, 2008 (73 FR 7816). Lawson *et al.* (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

Table 25. 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 River 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.

period 2000 through 2008 with a range from a low of 66,169 to a high of 260,000 naturally produced spawners. Table 25 identifies independent populations within the OC coho salmon ESU, historic and recent abundances, and hatchery input.

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 (ODFW 2009).

7.18.3 Critical Habitat

NMFS designated critical habitat for Oregon Coast coho salmon on February 11, 2008 (73 FR 7816). The designation includes 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 26, and Figure 27).

Table 26. 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.

The spawning PCE has been impacted in many watersheds from the inclusion of fine sediment into spawning gravel 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. The 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 lying areas in the Umpqua subbasins, and in coastal watersheds within the Siletz/Yaquina, Siltcoos, and Coos subbasins. Reductions in water quality have been observed in 12 watersheds due to contaminants and excessive nutrition. The migration PCE has been impacted throughout the ESU by culverts and road crossings that restrict passage. As described above the PCEs vary widely throughout the critical habitat area designated for OC coho salmon, with many watersheds heavily impacted with low quality PCEs while habitat in other coho salmon bearing watersheds having sufficient quality for supporting the conservation purpose of designated critical habitat.

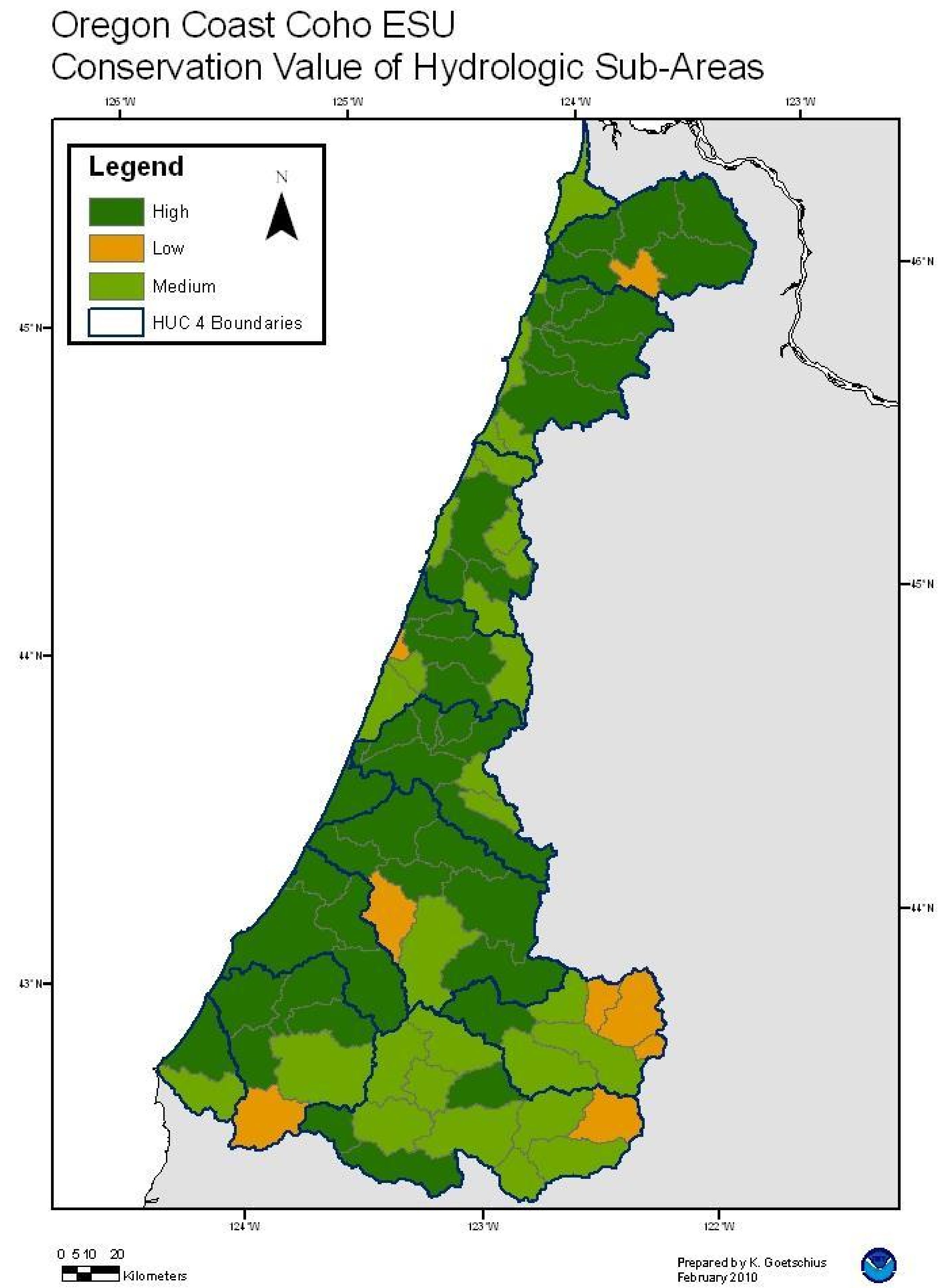

Figure 27. Oregon Coast Coho salmon conservation values per sub-area

7.19 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 28). 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.

7.19.1 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.

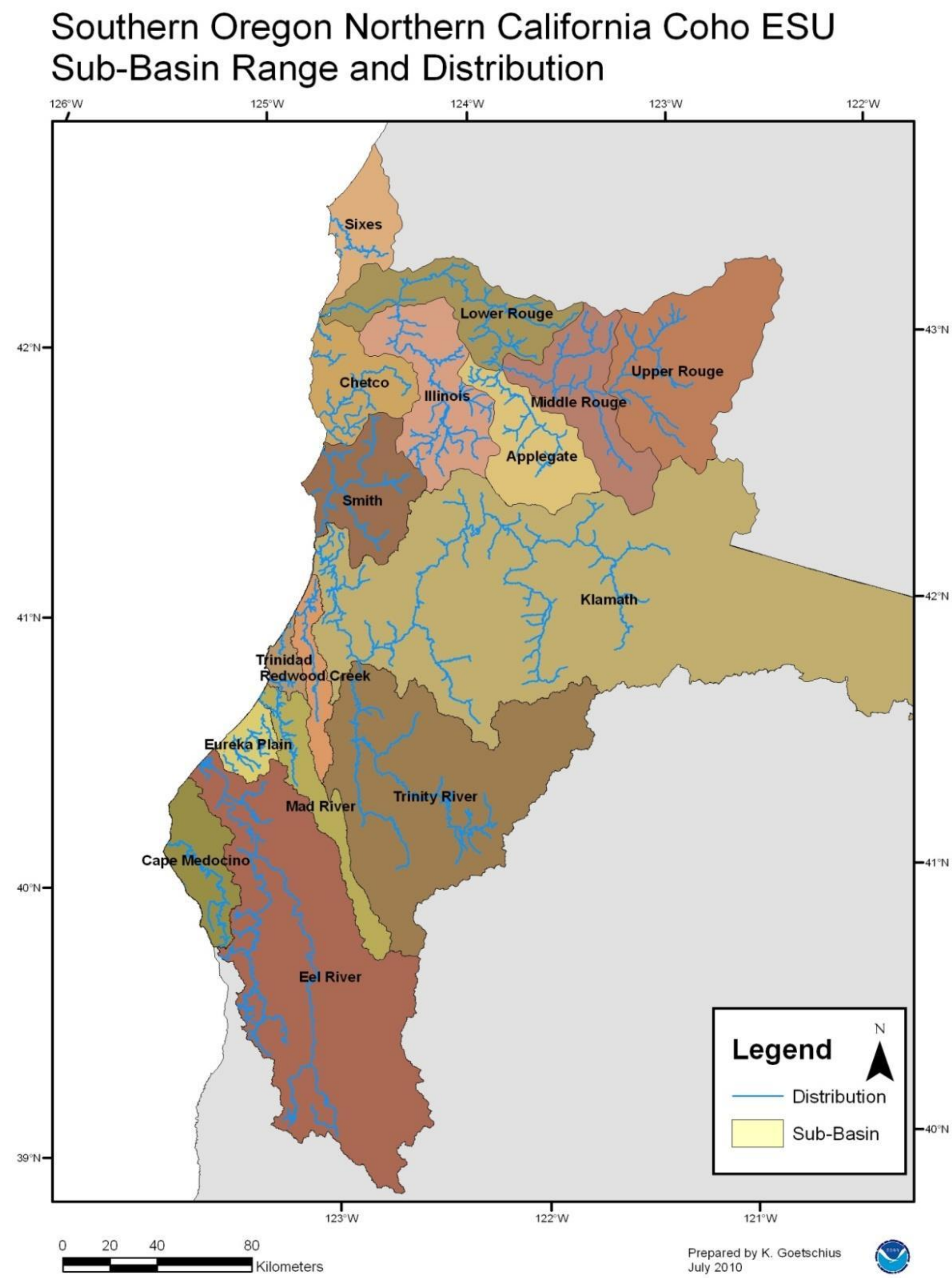

Figure 28. SONCC coho salmon distribution

7.19.2 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. 2006). 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.* (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.

7.19.3 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 from the loss of riparian vegetation resulting in unsuitably high water temperatures. Rearing and juvenile migration PCEs have been reduced from the disconnection of floodplains and off-channel habitat in low gradient reaches of streams, consequently reducing winter rearing capacity.

7.20 *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 Aptos Creek, 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 29)
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.

7.20.1 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).

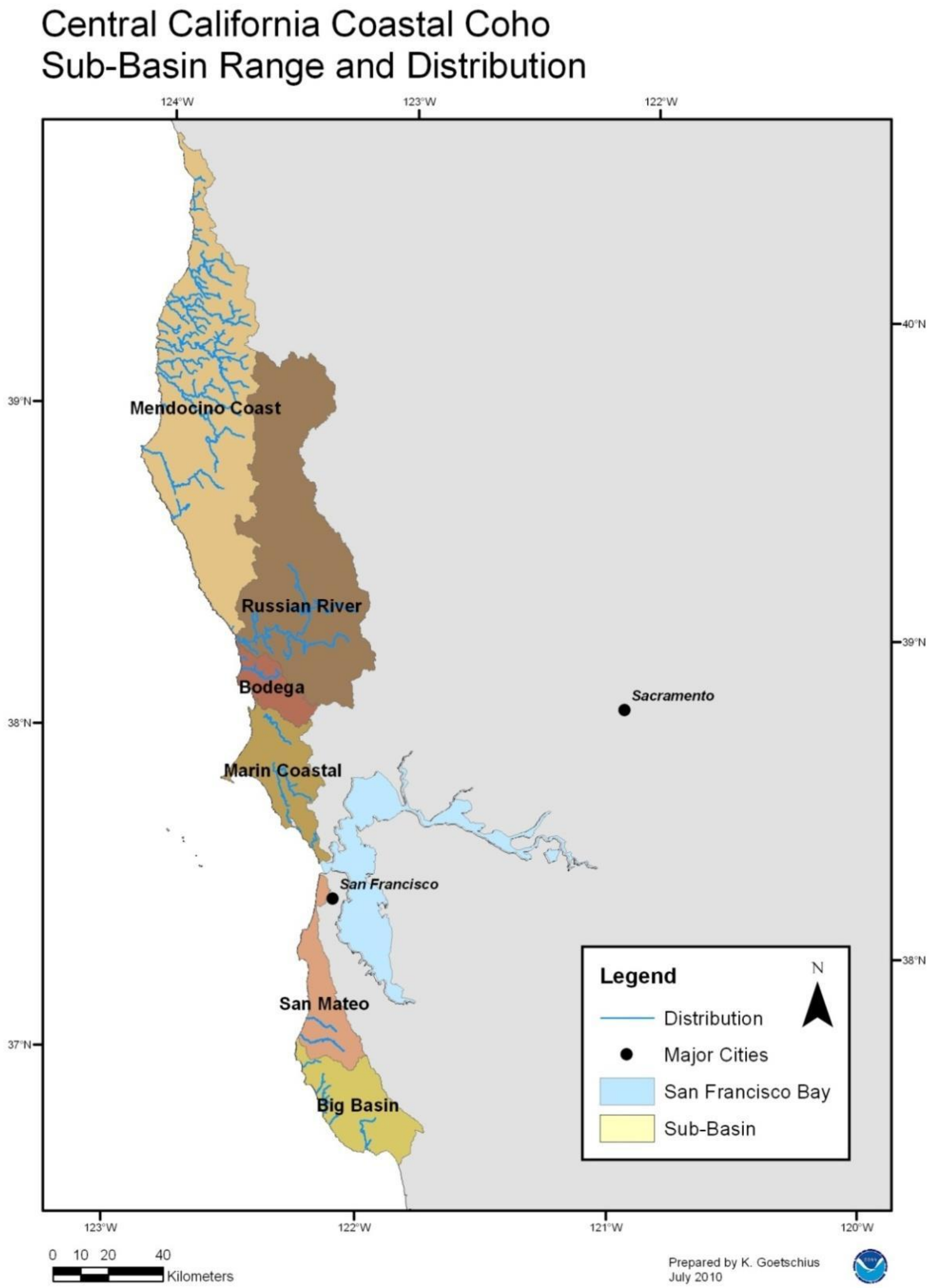

Figure 29. CCC Coho salmon distribution

7.20.2 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. 2008a). 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. 2008a). Coho salmon are considered effectively extirpated from the San Francisco Bay (NMFS 2001, Spence et al. 2008a). The Russian River is of particular importance for preventing the extinction and contributing to the recovery of CCC coho salmon (NOAA 2013). 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. 2008a). 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. 2008a). 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 27. 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 (Table 27). 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 (Sonoma County Water Agency (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.

7.20.3 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.

7.21 *Sockeye Salmon*

7.21.1 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.

7.21.2 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). Climate change also poses significant hazards to the survival and recovery of salmonids. Hazards from climate change include elevated water temperature, earlier spring runoff and lower summer flows, and winter flooding.

7.22 Ozette Lake Sockeye Salmon

7.22.1 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 **30**). 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, June 28, 2005).

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.

7.22.2 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).

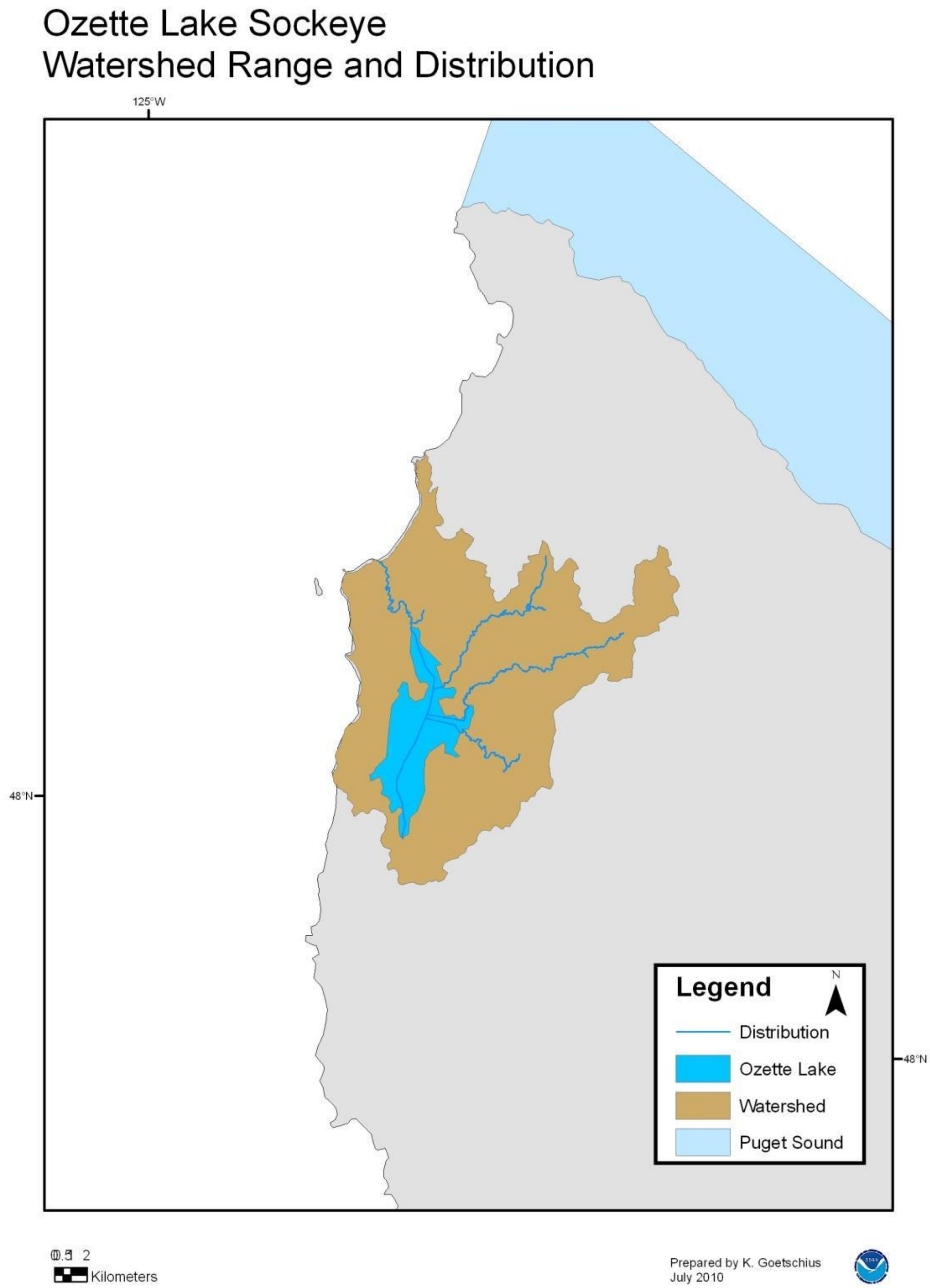

Figure 30. **Ozette Lake Sockeye salmon distribution**

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 2008f). 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).

7.22.3 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 (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. 1992).

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.

7.22.4 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).

7.23 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 (70 FR 37160, June 28, 2005). The Redfish Lake is located in the Salmon River basin, a subbasin within the larger Snake River basin (Figure **31**).

7.23.1 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.

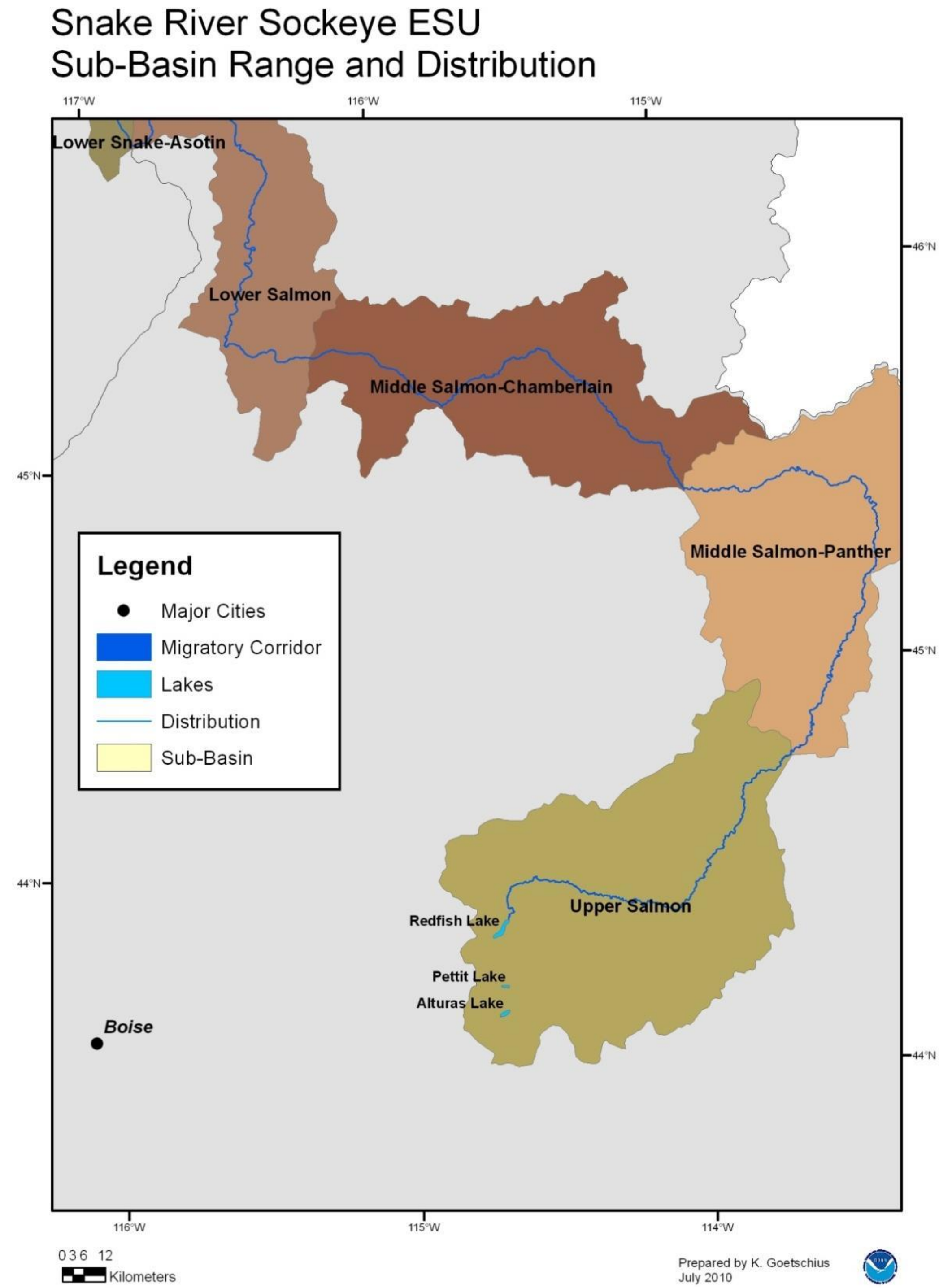

Figure 31. SR Sockeye Salmon distribution

Historically, sockeye salmon entered the Columbia River system in June and July, and arrived at Redfish Lake between August and September (NMFS 2008d). 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.

7.23.2 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. 1997). 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).

7.23.3 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 from activities 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.

7.24 Steelhead

7.24.1 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. 1992a). 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. 1992a). 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. 1982, 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 (Shapovalov 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 et al. 1992a). 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.

7.24.2 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). Climate change also poses significant hazards to the survival and recovery of salmonids. Hazards from climate change include elevated water temperature, earlier spring runoff and lower summer flows, and winter flooding.

7.25 *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 32). The remaining hatchery programs are not considered part of the DPS because they are more than moderately diverged from the local native populations.

7.25.1 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 2005d). 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 2005d).

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.

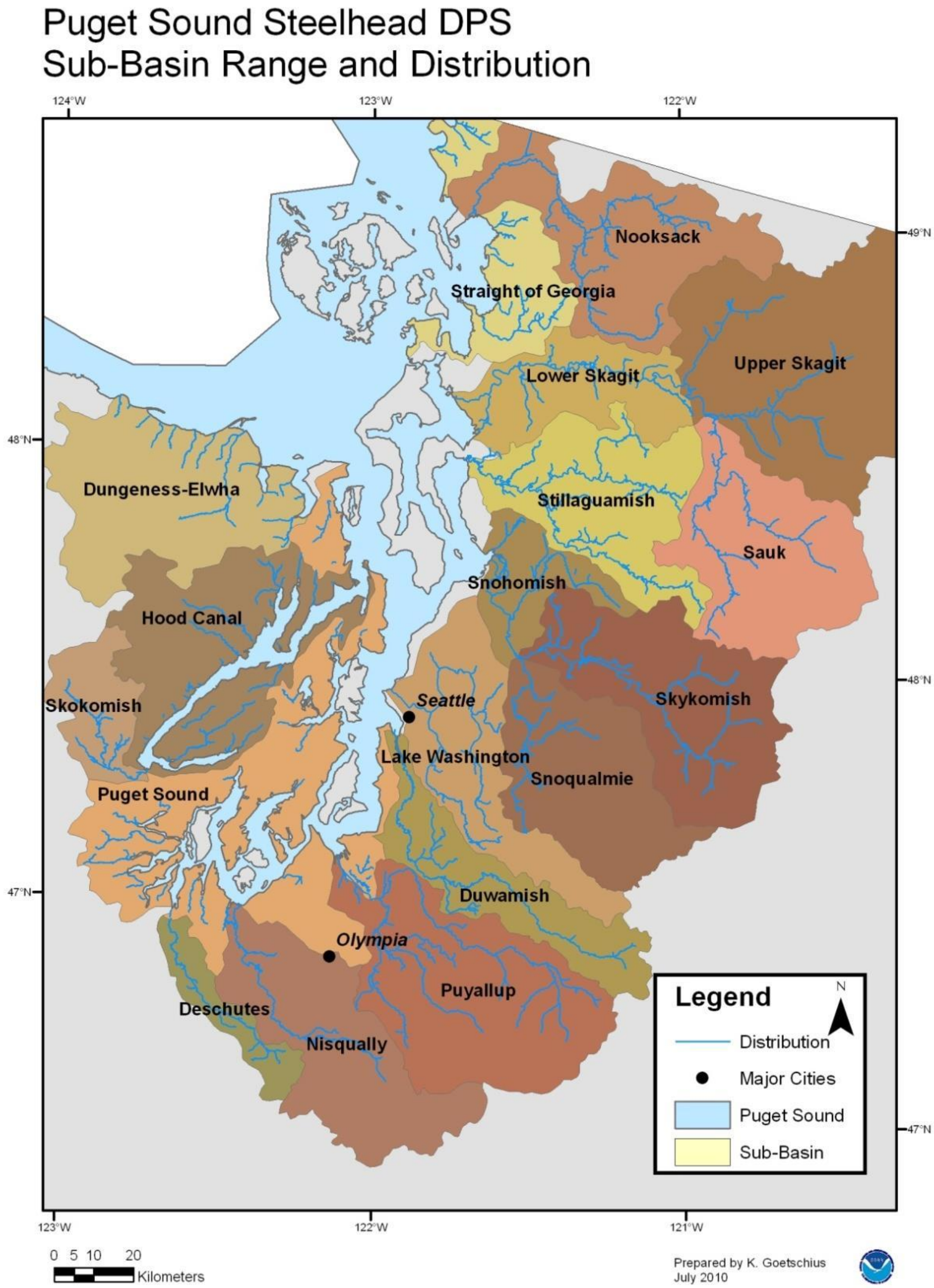

Figure 32. Puget Sound steelhead distribution

7.25.2 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. Table 28 provides the geometric mean estimates of escapement of natural spawners for Puget Sound steelhead.

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 (Washington Department of Fish and Wildlife (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 (2008) data show escapement of natural spawners for the period 1980 to 2004 and the period 2000 to 2004 (Washington Department of Fish and Wildlife (WDFW) 2008).

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 2005d)). 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 (Washington Department of Fish and Wildlife (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 (Washington Department of Fish and Wildlife (WDFW) 2008). The stock is now virtually extirpated with only eight and four returning fish in 2007 and 2008, respectively (Washington Department of Fish and Wildlife

(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.

Table 28. Geometric mean estimates of escapement of natural spawners for Puget Sound steelhead

Population	Run type	Long Term	5-Year
Canyon	SSH	N/A	N/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
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 (2005e) status review updated for Puget Sound steelhead, http://www.nwr.noaa.gov/ESA-Salmon-Listings/Salmon-Populations/Steelhead/STPUG.cfm)

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 2005d). 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 2005d).

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 2005d). 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 2005d). 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 estimated that 31 of the 53 stocks were of native origin and predominantly natural production (Washington Department of Fish and Wildlife (WDFW) 1993).

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*. (Phelps et al. 1997), indicated reproductive isolation of and/or poor spawning success by hatchery-origin fish. This was shown in a later study on the Clackamas River in Oregon (kostow et al. 3003). 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.

7.25.3 Critical Habitat

NMFS proposed designated critical habitat for the Puget Sound steelhead DPS on January 14, 2013 (50 CFR Part 226). Of 70 assessed watersheds (HUC 5), 41 were assigned a high and 18 were assigned a medium conservation value (Figure 33). The remaining watersheds were either of low conservation value, or have been proposed to be excluded for economic considerations.

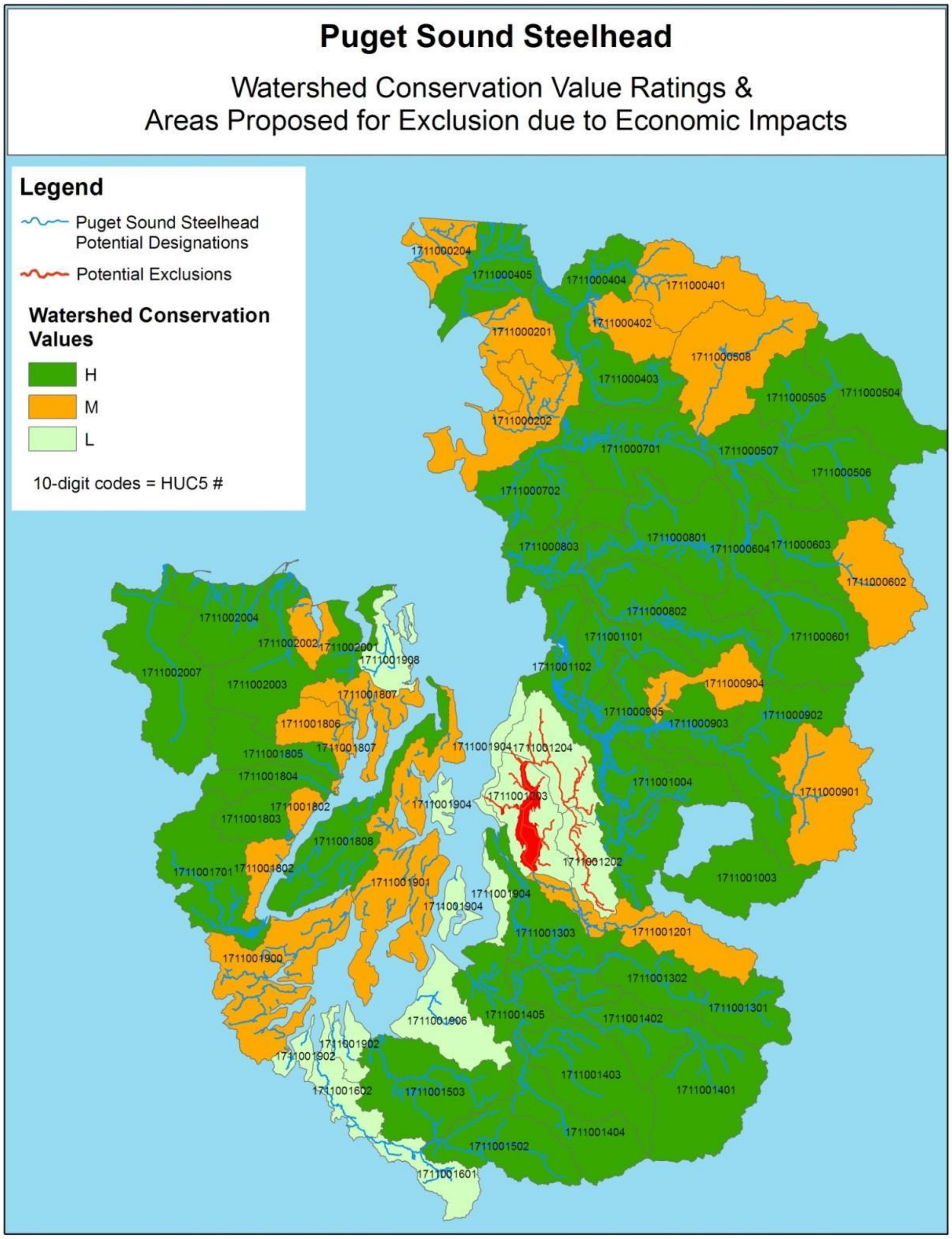

Figure 33. Puget Sound steelhead DPS proposed designated critical habitat

7.26 *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 34**). 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.

7.26.1 Life History

The LCR steelhead DPS includes both summer- and winter-run stocks (Table 29). 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.

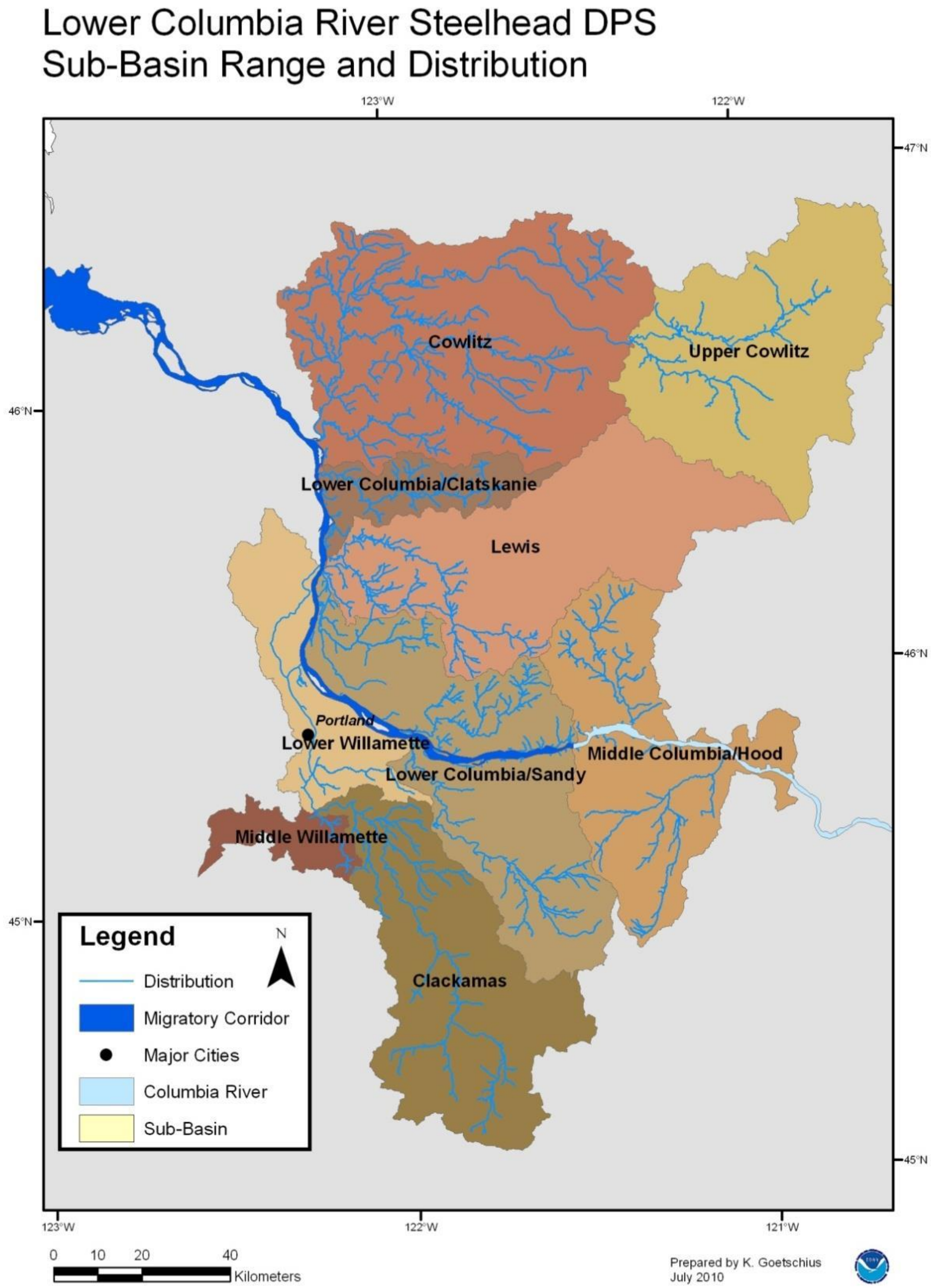

Figure 34 Lower Columbia River steelhead distribution

7.26.2 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, Myers 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.

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)

Table 29. 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%
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		Unknown	Unknown	Unknown
EF Lewis River		Unknown	--/474	
Washougal River		Unknown	--/668	
Hood River		Unknown	931/--	~83%
Wind River		2,288	--/627	~5%

7.26.3 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 30).

Table 30. 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 migration PCEs 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 (Figure 35).

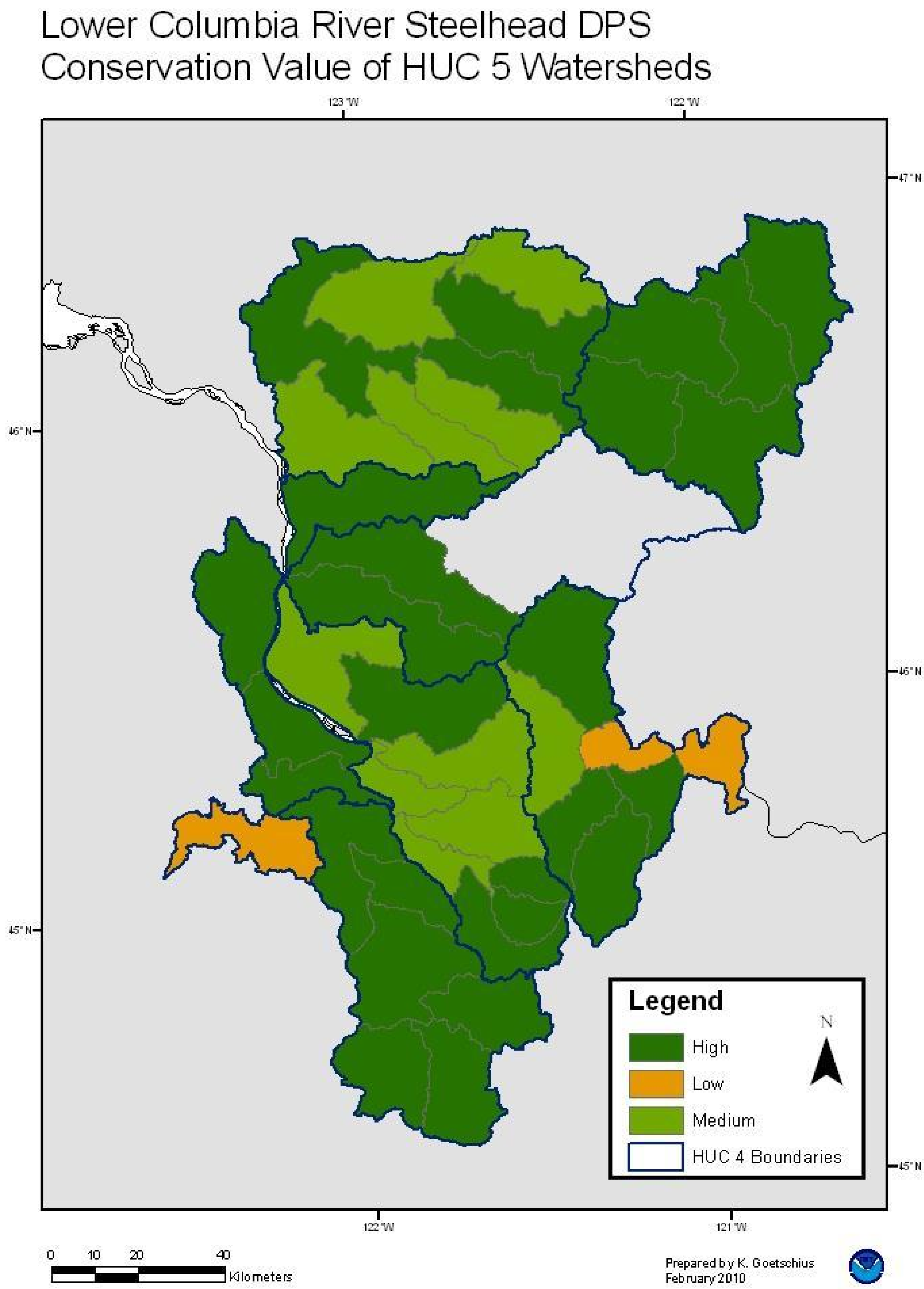

Figure 35 Lower Columbia River Steelhead conservation values per sub-area

7.27 *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 36**). 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.

7.27.1 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 from 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).

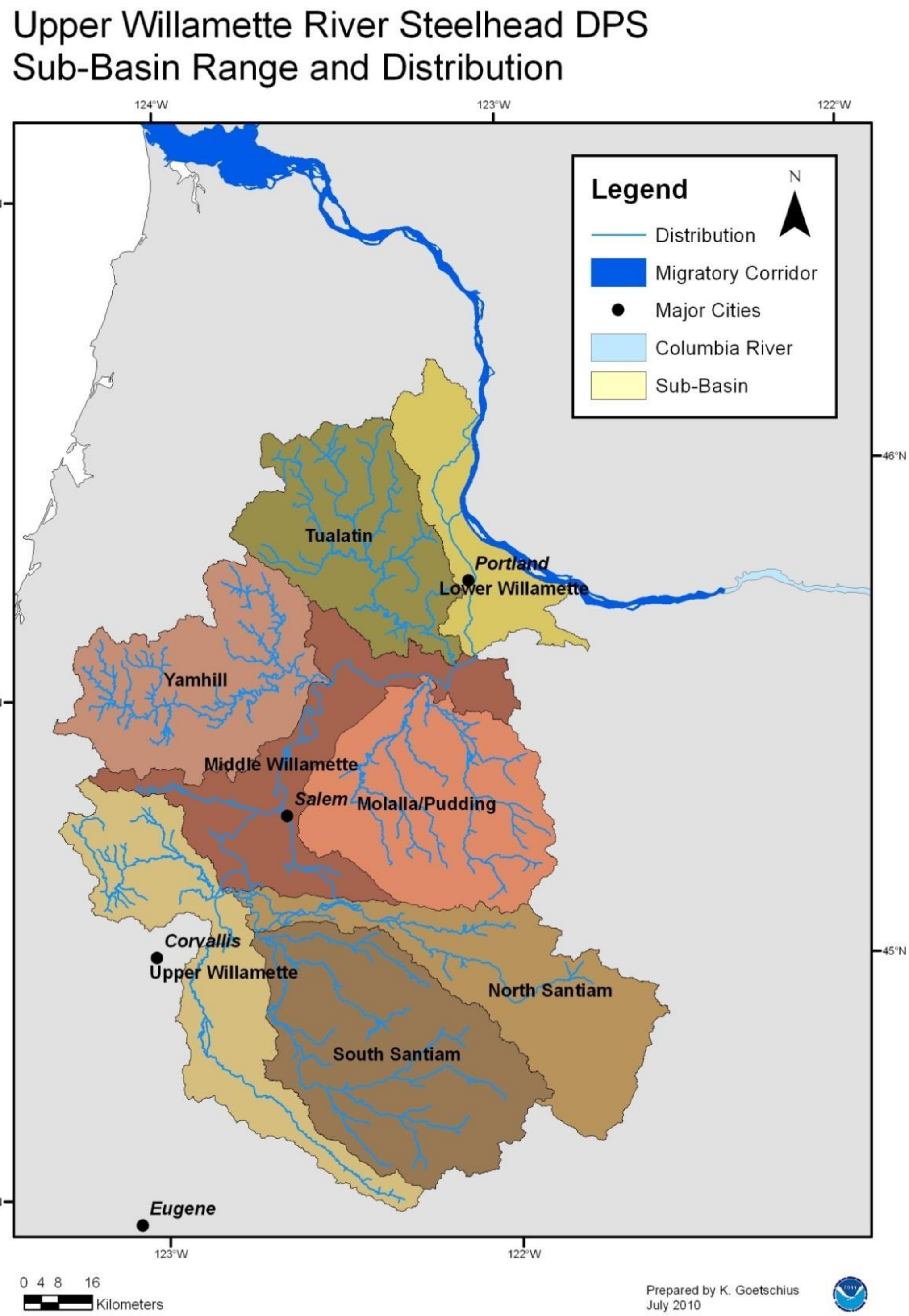

Figure 36. UWR Steelhead distribution

7.27.2 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. Mean 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 31. 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 (McElhany et al. 2007).

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 (Table 31). Nevertheless, remains of the early run still exist and the release of hatchery fish has been discontinued.

7.27.3 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 2005c).

Table 32. 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.

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 32).

The current condition of critical habitat designated for the UWR steelhead is degraded (Figure 37), and provides a reduced the conservation value necessary for species recovery. Critical habitat is affected by reduced quality of juvenile rearing and migration 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.

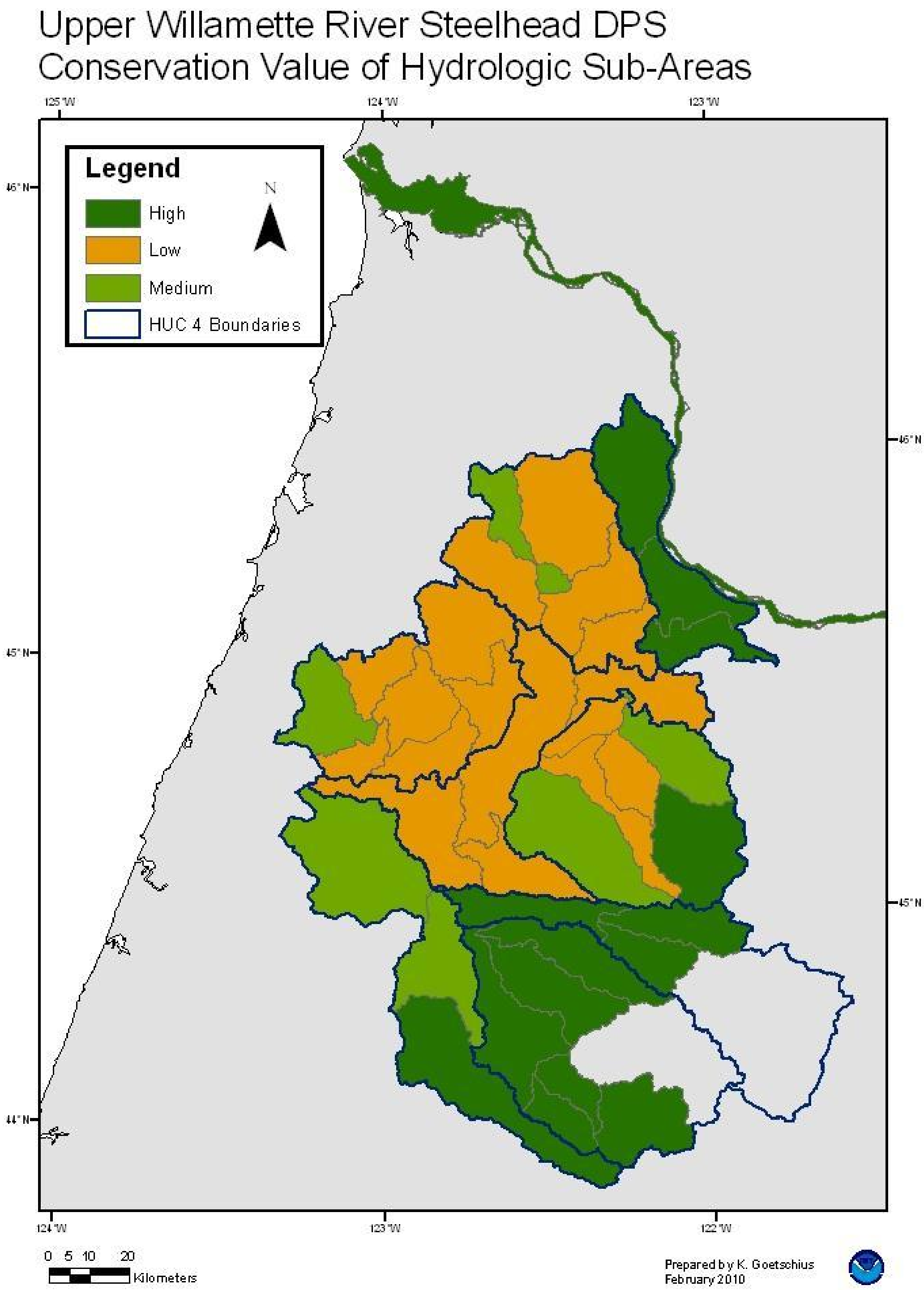

Figure 37. Upper Willamette River Steelhead conservation values per sub-area

7.28 *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 38). 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 (ICTRT 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).

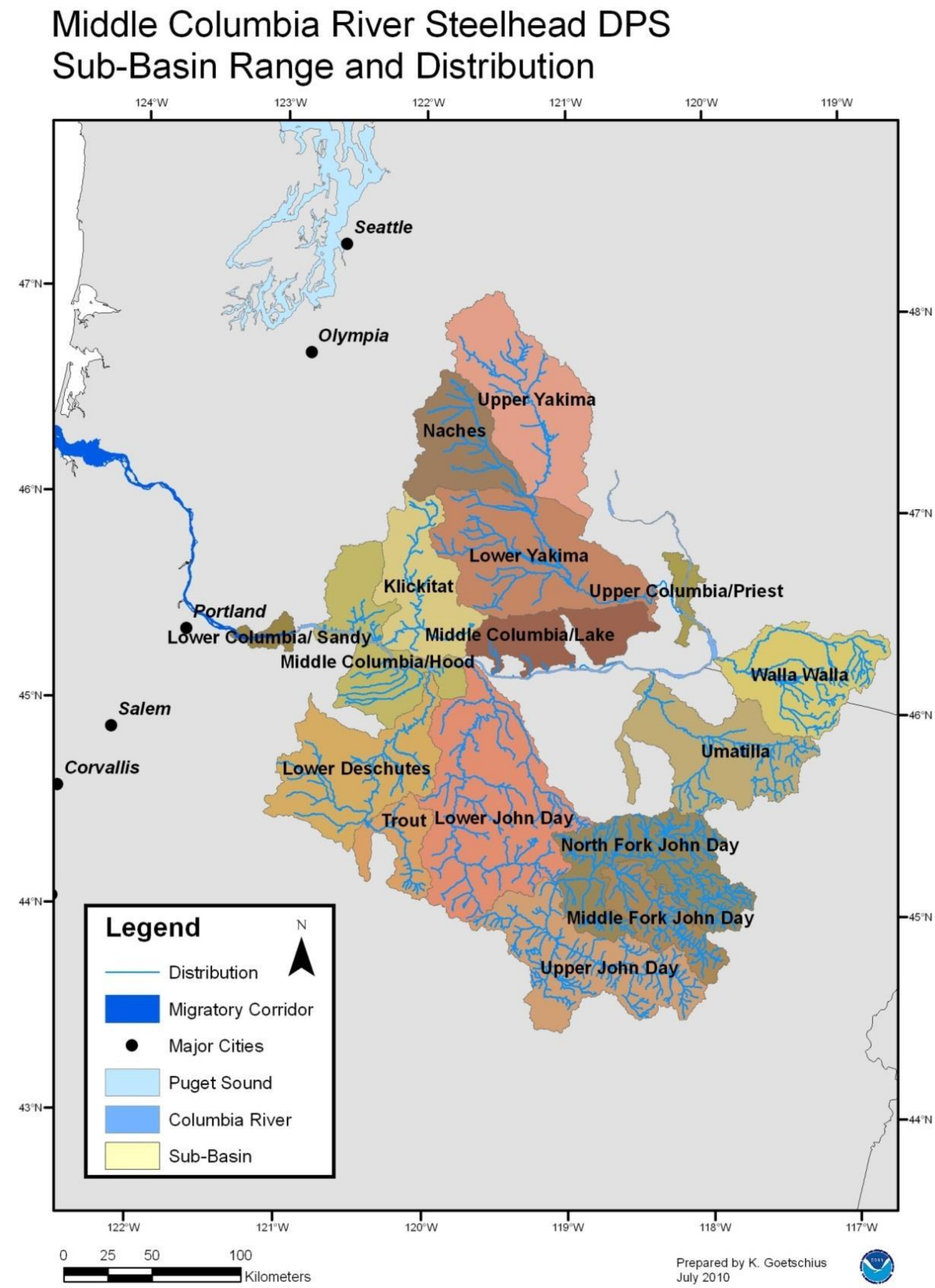

Figure 38. MCR Steelhead distribution

7.28.1 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.

7.28.2 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 ICTRT 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.

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).

Table 33. 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

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 (Good et al. 2005). The median long-term λ was 0.98, assuming that hatchery spawners do not contribute to production, and 0.97 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).

Table 33 summarizes MCR steelhead independent populations, abundances and hatchery contributions(ICTRT 2003, 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).

7.28.3 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 34). The lower Columbia River rearing/migration corridor downstream of the spawning range is also considered to have a high conservation value.

Table 34. 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 (Figure 39). Critical habitat is affected by reduced quality of juvenile rearing and migration 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.

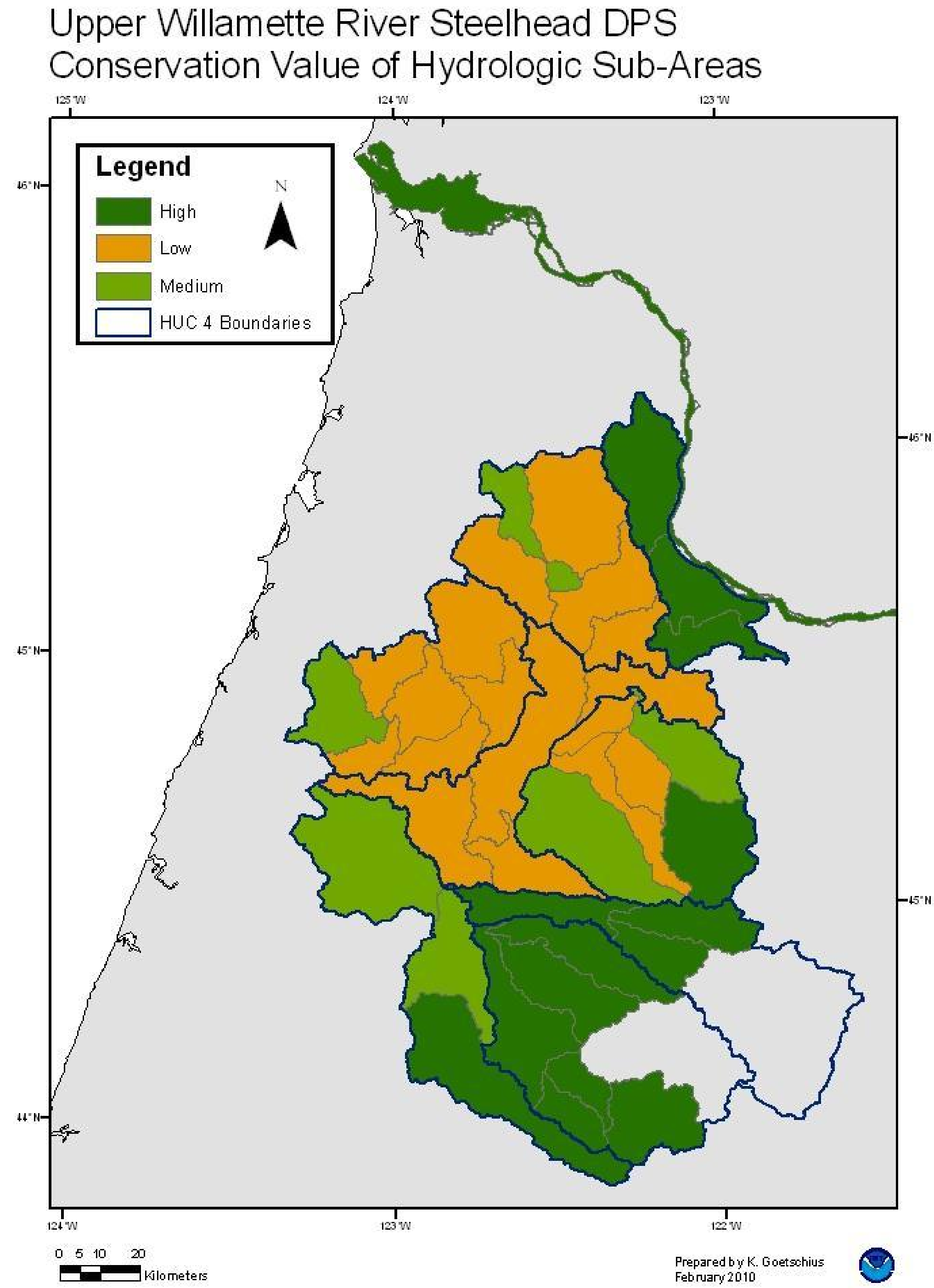

Figure 39. Upper Willamette River Steelhead conservation values per sub-area

7.29 *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 40). 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.

7.29.1 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 one or two years at sea.

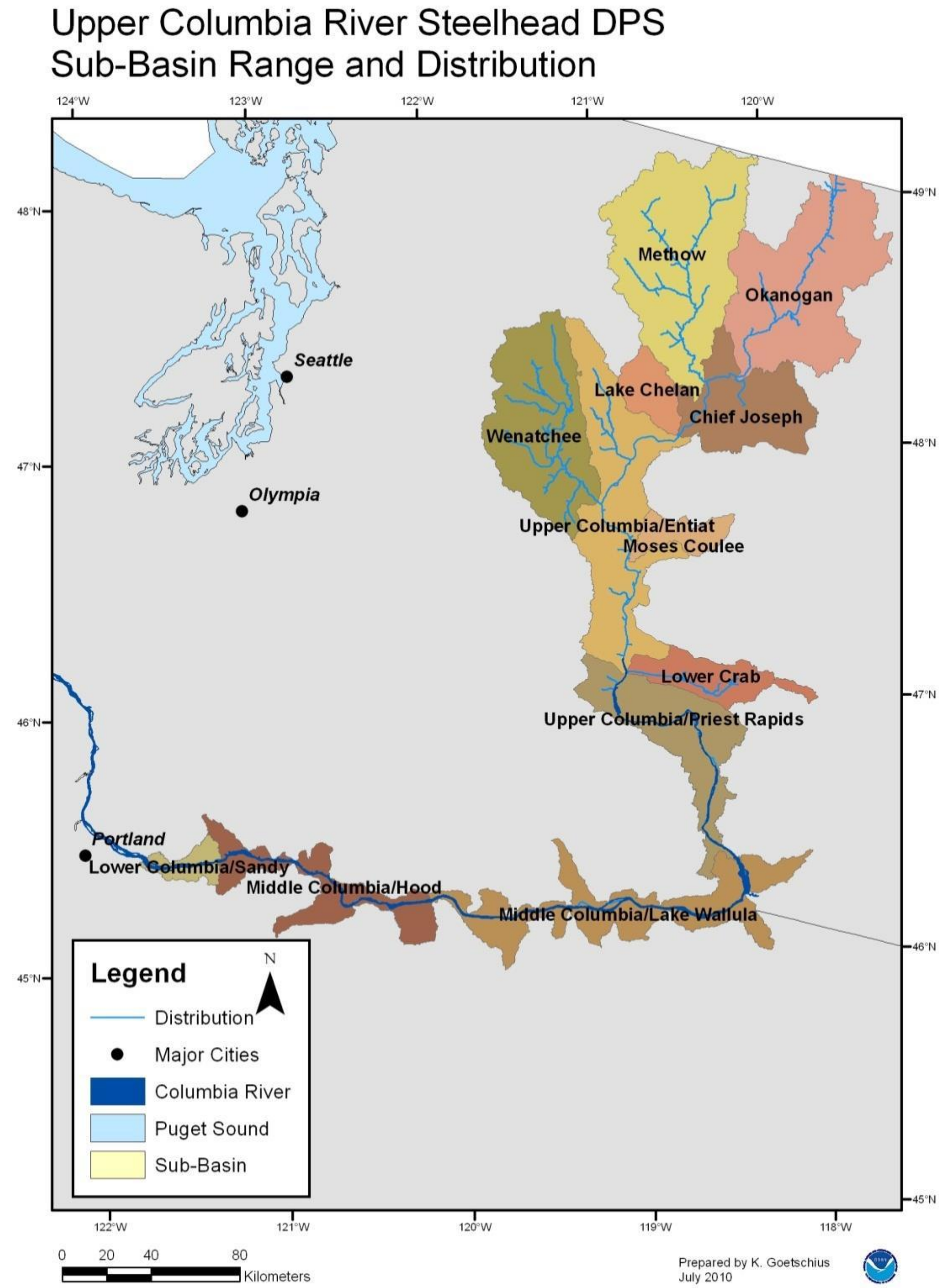

Figure 40. UCR Steelhead distribution

7.29.2 Status and Trends

NMFS originally listed UCR steelhead as endangered on August 19, 1997 (62 FR 43937). NMFS changed the listing to threatened on January 5, 2006 (71 FR 834). After litigation resulting in a change in the DPS' status to endangered and then again as threatened, on August 24, 2009, NMFS reaffirmed the species' status as 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 35. 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 (Table 35), 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).

7.29.3 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 36). The lower Columbia River rearing/migration corridor downstream of the spawning range is of high conservation value.

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 (Figure 41). Critical habitat is affected by reduced quality of juvenile rearing and migration 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.

Table 36. 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.

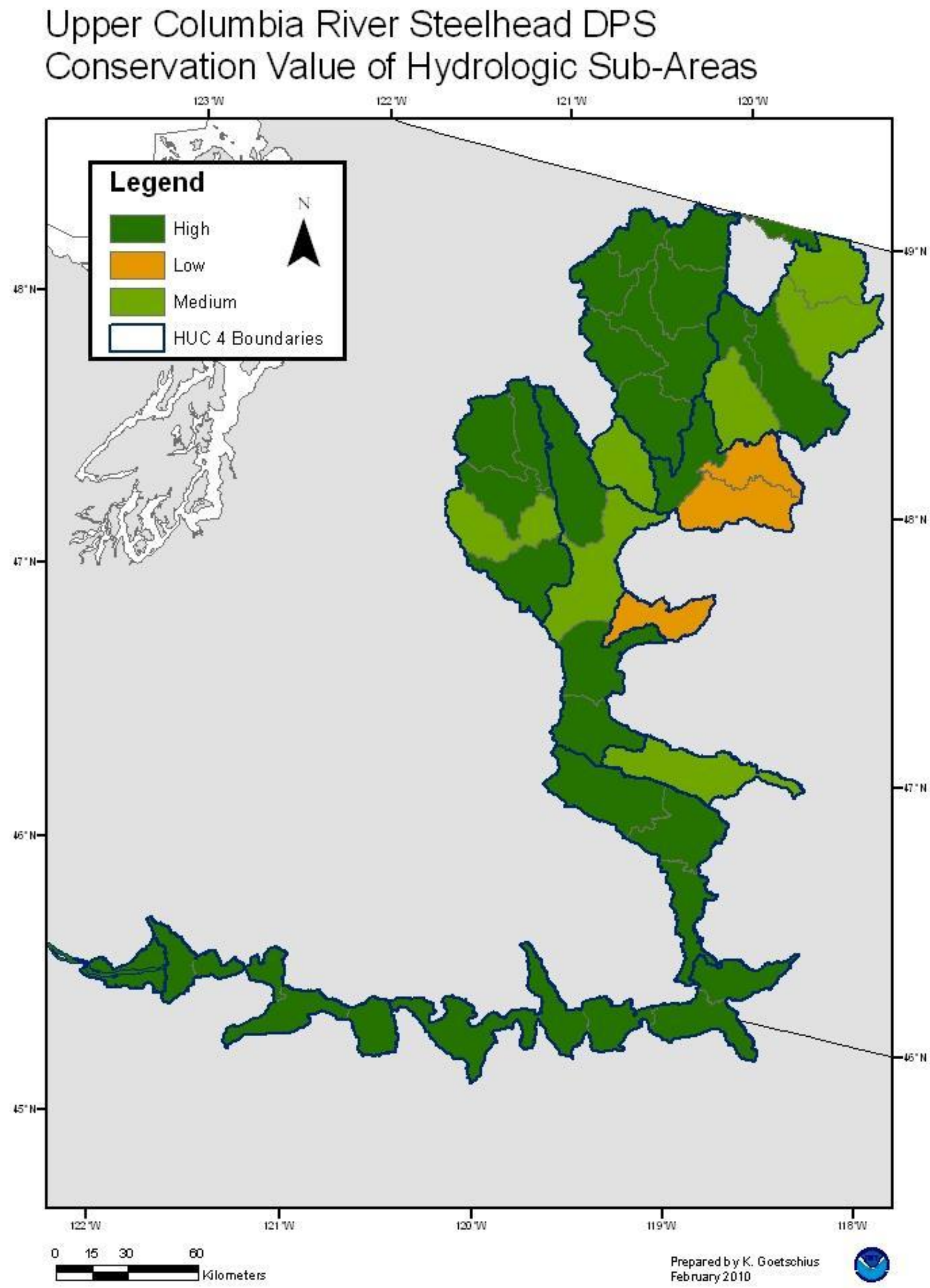

Figure 41. Upper Columbia River Steelhead conservation values per sub-area.

7.30 *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 confluence of the Snake River with the Columbia River in Washington, into Idaho's Salmon River Basin (Figure **42**). 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.

7.30.1 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.

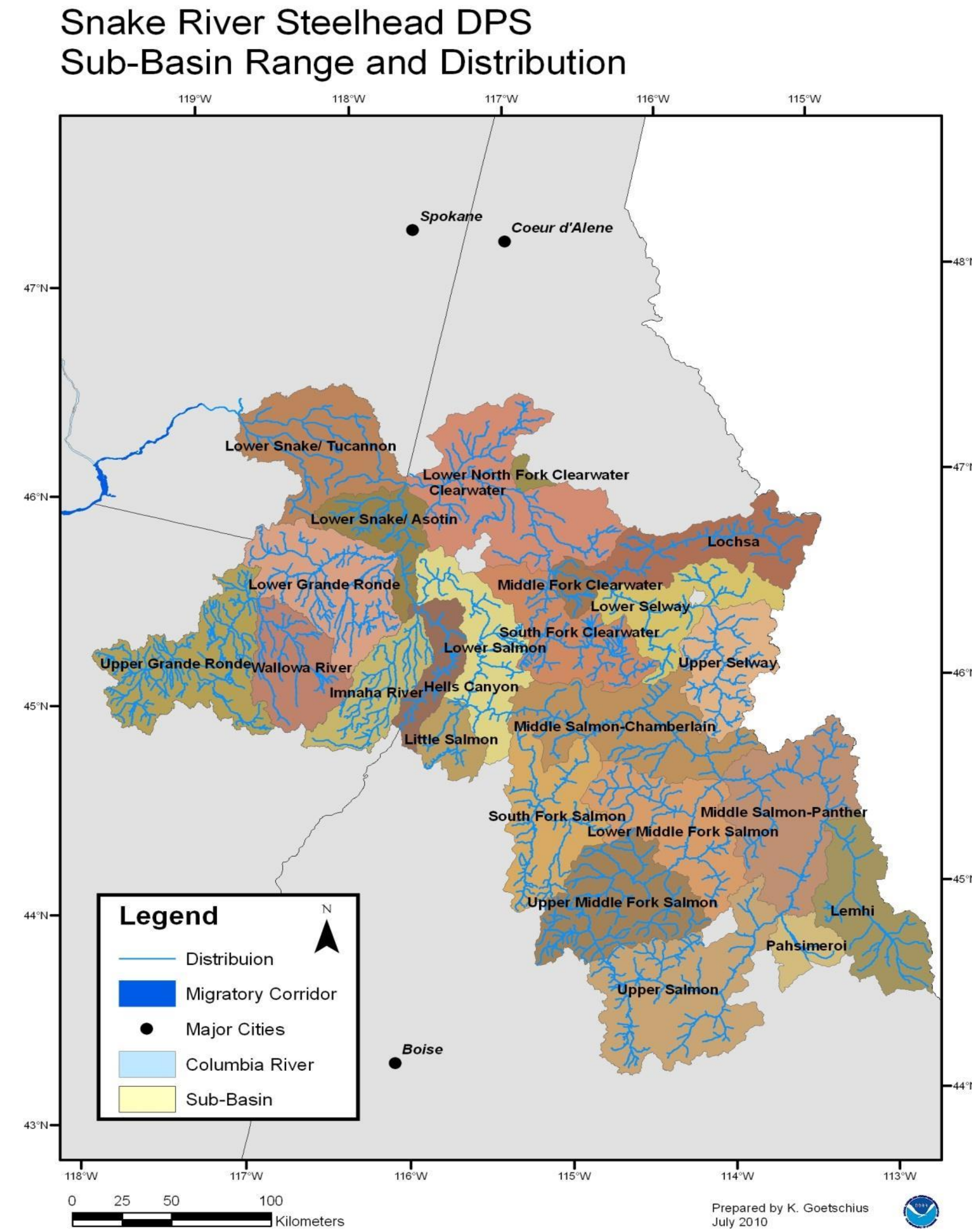

Figure 42 SR Basin Steelhead distribution

7.30.2 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 ICTRT (ICTRT 2003) 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 37 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 37). 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.

7.30.3 Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52630). Figure 43 shows the conservation rankings per sub-area. Of the watersheds assessed, 229 were rated as having a high conservation value, 42 as medium value, and 12 as low value (Table 38). The Columbia River migration corridor was also given a high conservation value rating (NMFS 2005a).

The current condition of critical habitat designated for SR basin steelhead is moderately degraded. Critical habitat is affected by reduced quality of juvenile rearing and migration 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.

Table 38 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)
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.

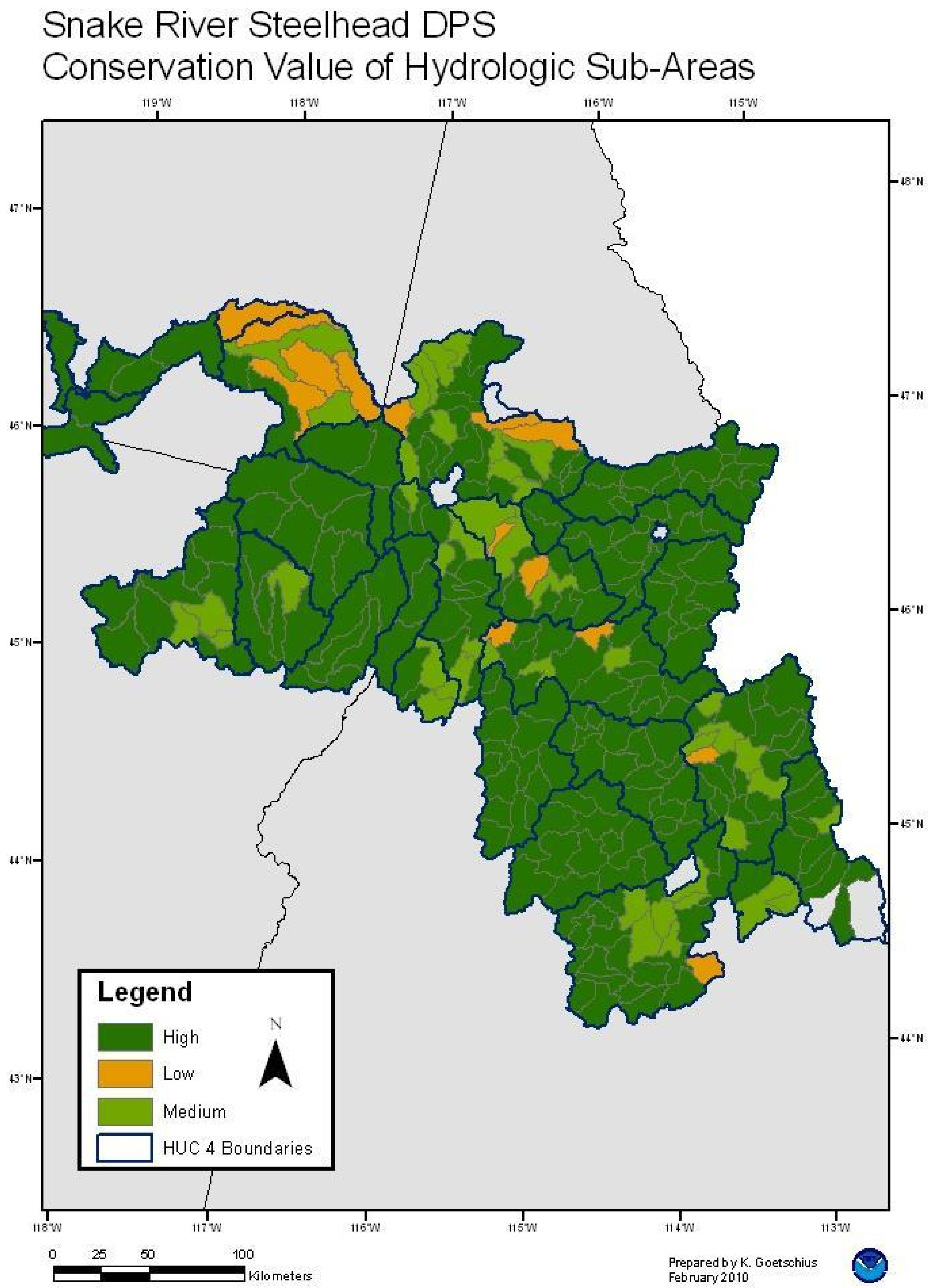

Figure 43. Snake River Steelhead conservation values per sub-area

7.31 *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 44**).

7.31.1 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 (Busby 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).

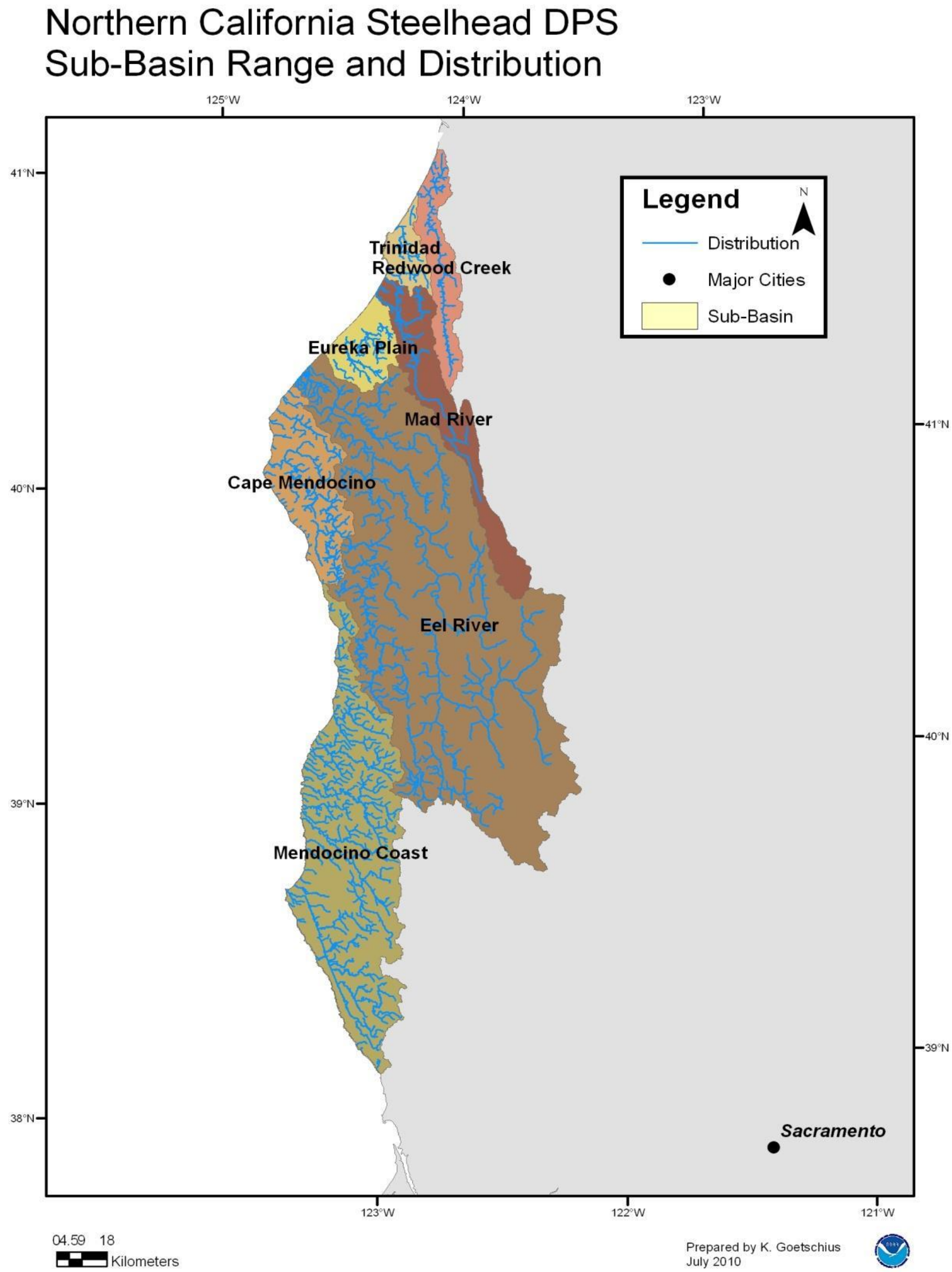

Figure 44. Northern California Steelhead distribution

7.31.2 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. 2008a). 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 39).

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).

Table 39. 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)	Unknown	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
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	

7.31.2.1.1.1 *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)

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. 2008a). Hatchery influence has likely been limited.

7.31.3 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 2005a) (Table 40, and Figure 45). Two estuarine habitat areas used for rearing and migration (Humboldt Bay and the Eel River Estuary) also received a high conservation value rating.

Table 40. 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.

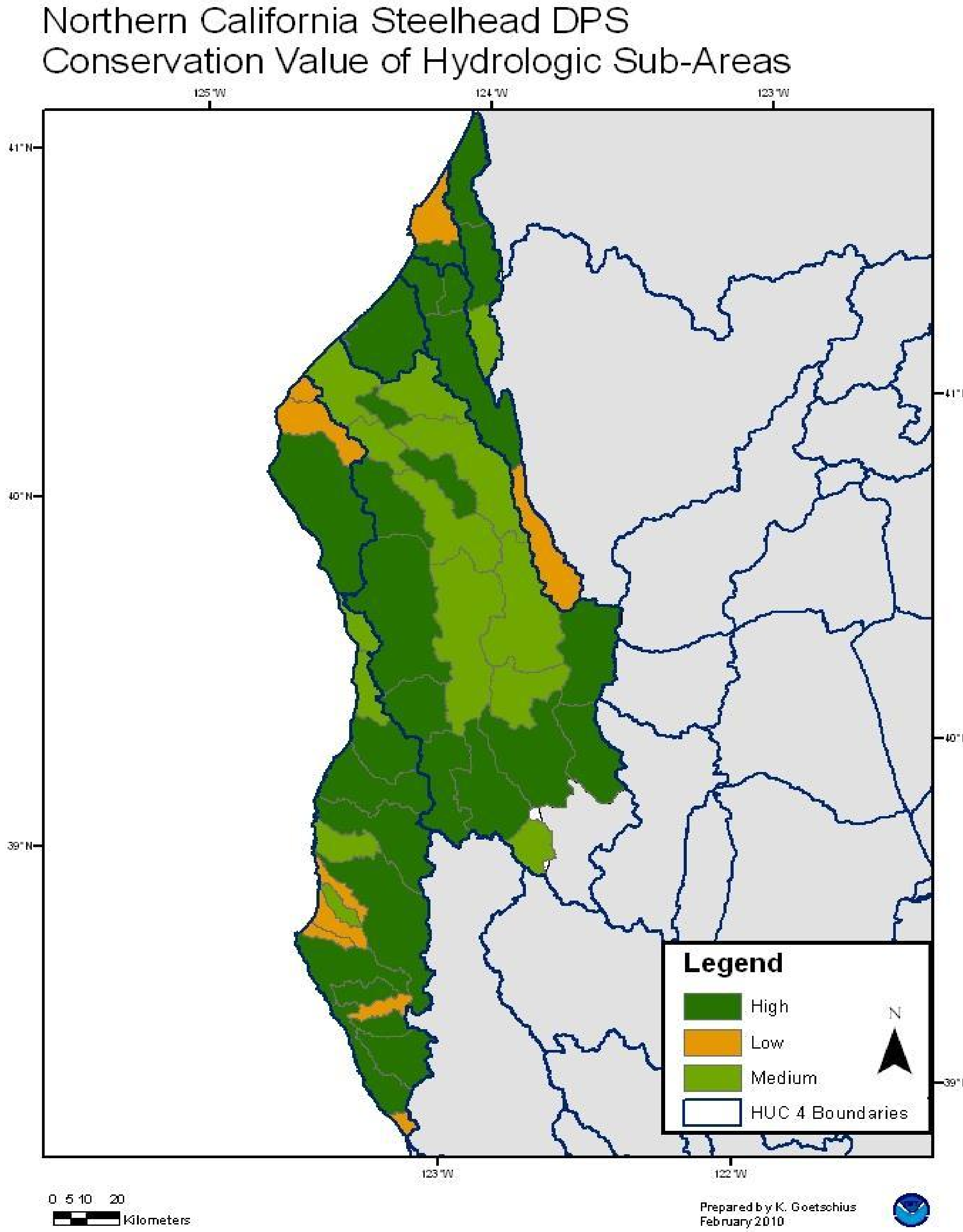

Figure 45. Northern California Steelhead conservation values per sub-area

7.32 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 46).

7.32.1 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 et al. 2004). Most 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).

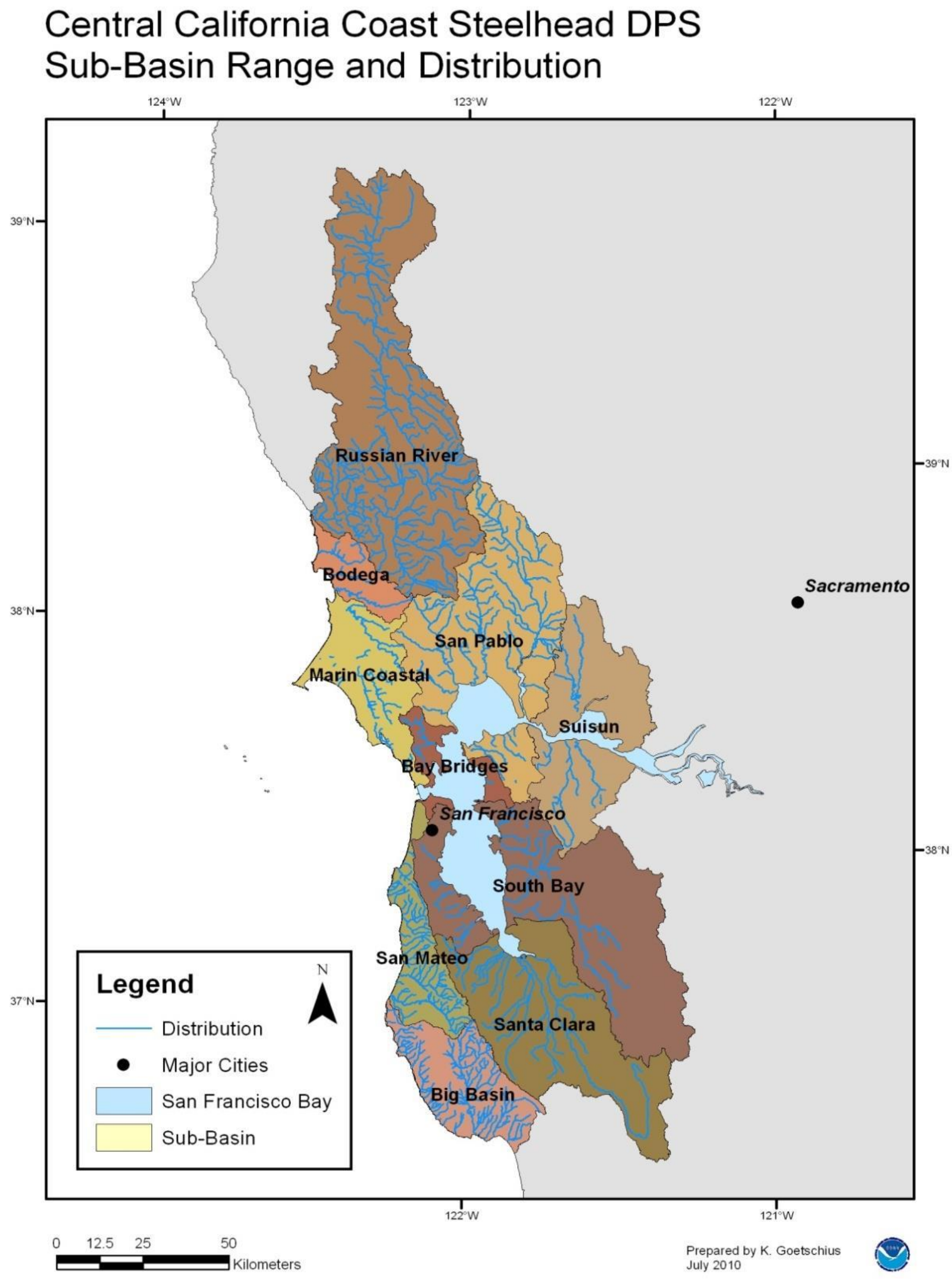

Figure 46. CCC steelhead distribution

7.32.2 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 et al. 2005). 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 (Table 41), both have been estimated at less than 15% of their abundances just 30 years earlier (Good et al. 2005). The Russian River is of particular importance for preventing the extinction and contributing to the recovery of CCC steelhead (NOAA 2013). Steelhead access to significant portions of the upper Russian River has also been blocked (Busby et al. 1996, NMFS 2008a).

Table 41. CCC Steelhead populations, historic population type, abundances, and hatchery contributions (Good et al. 2005, NMFS 2008a).

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 et al. 2005)
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 et al. 2005). 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 2008a). Declines in juvenile southern populations are consistent with the more general estimates of declining abundance in the region (Good et al. 2005). 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 et al. 2005, NMFS 2008a).

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.

7.32.3 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. As shown in Figure 47, 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 2005c) (

Table 42). 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 42. 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.

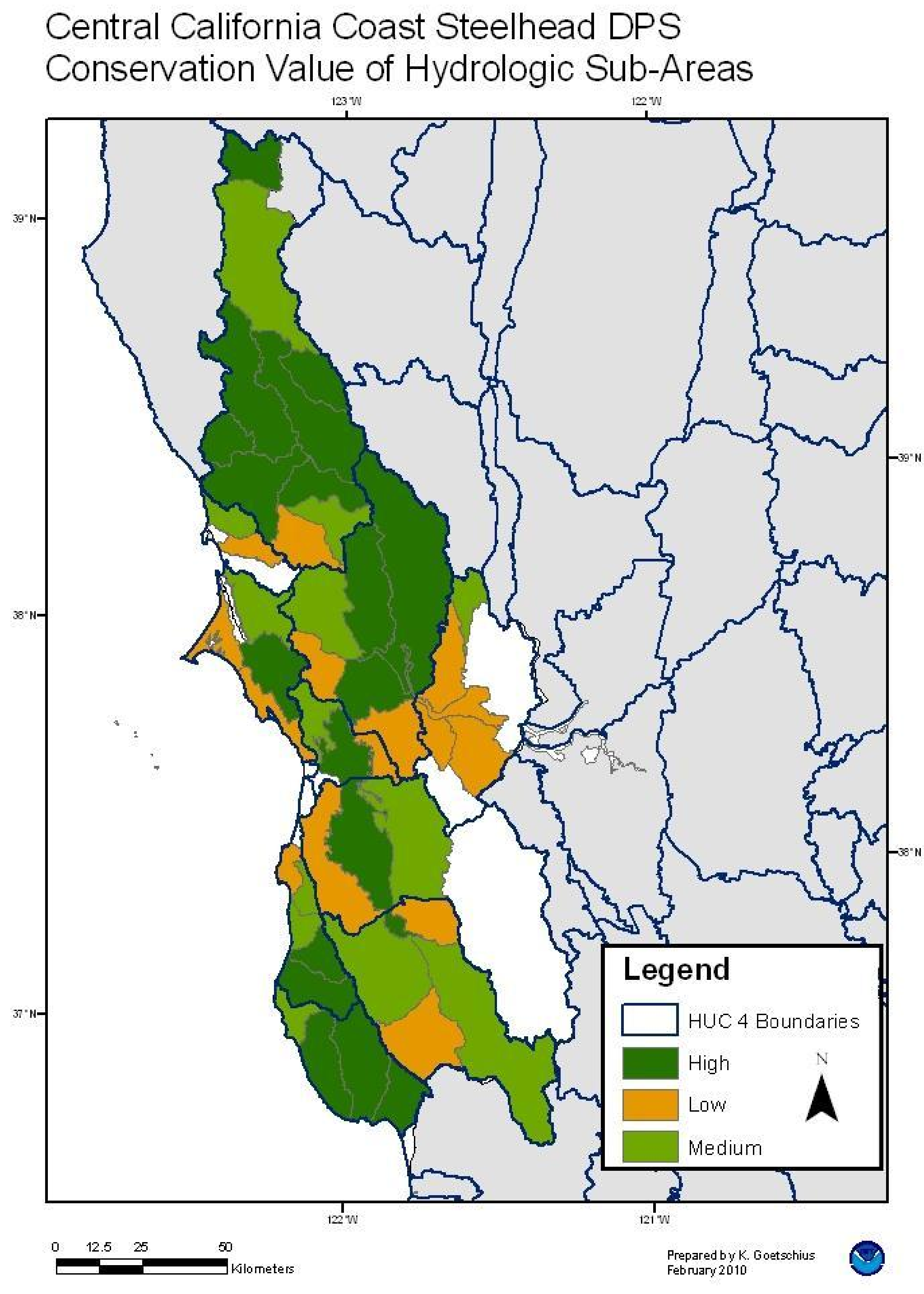

Figure 47. Central California Coast Steelhead conservation values per sub-area.

7.33 *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 48).**

7.33.1 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 et al. 1996), and spawn from December through April, with peaks from January through March, in small streams and tributaries where cool, well oxygenated water is available year-round (Hallock et al. 1961, McEwan and Jackson 1996). 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 and Jackson 1996). 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) 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 (Hallock et al. 1961).

7.33.2 Status and Trends

NMFS originally listed CCV steelhead as threatened on March 19, 1998, and reaffirmed their threatened status on January 5, 2006 (71 FR 834). The CCV steelhead DPS may have consisted of 81 historical and independent populations (Lindley et al. 2006). 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 et al. 2006). Existing wild steelhead stocks in the Central Valley are mostly confined to the upper Sacramento River and its tributaries, including Antelope, Deer, and Mill Creeks and the Yuba River. Populations may exist in Big Chico and Butte Creeks. A few wild steelhead are produced in the American and Feather Rivers (Good et al. 2005). Steelhead have also been observed in Clear Creek and Stanislaus River (Good et al. 2005, Demko and Cramer 2000). Until recently, steelhead were considered extirpated from the San Joaquin River system. Recent monitoring has detected small self-sustaining populations of steelhead in the Stanislaus, Mokelumne, Calaveras, and other streams previously thought to be void of steelhead (Good et al. 2005). In 2004, a total of 12 steelhead smolts were collected in monitoring trawls at the Mossdale station in the lower San Joaquin River (CDFG unpublished data).

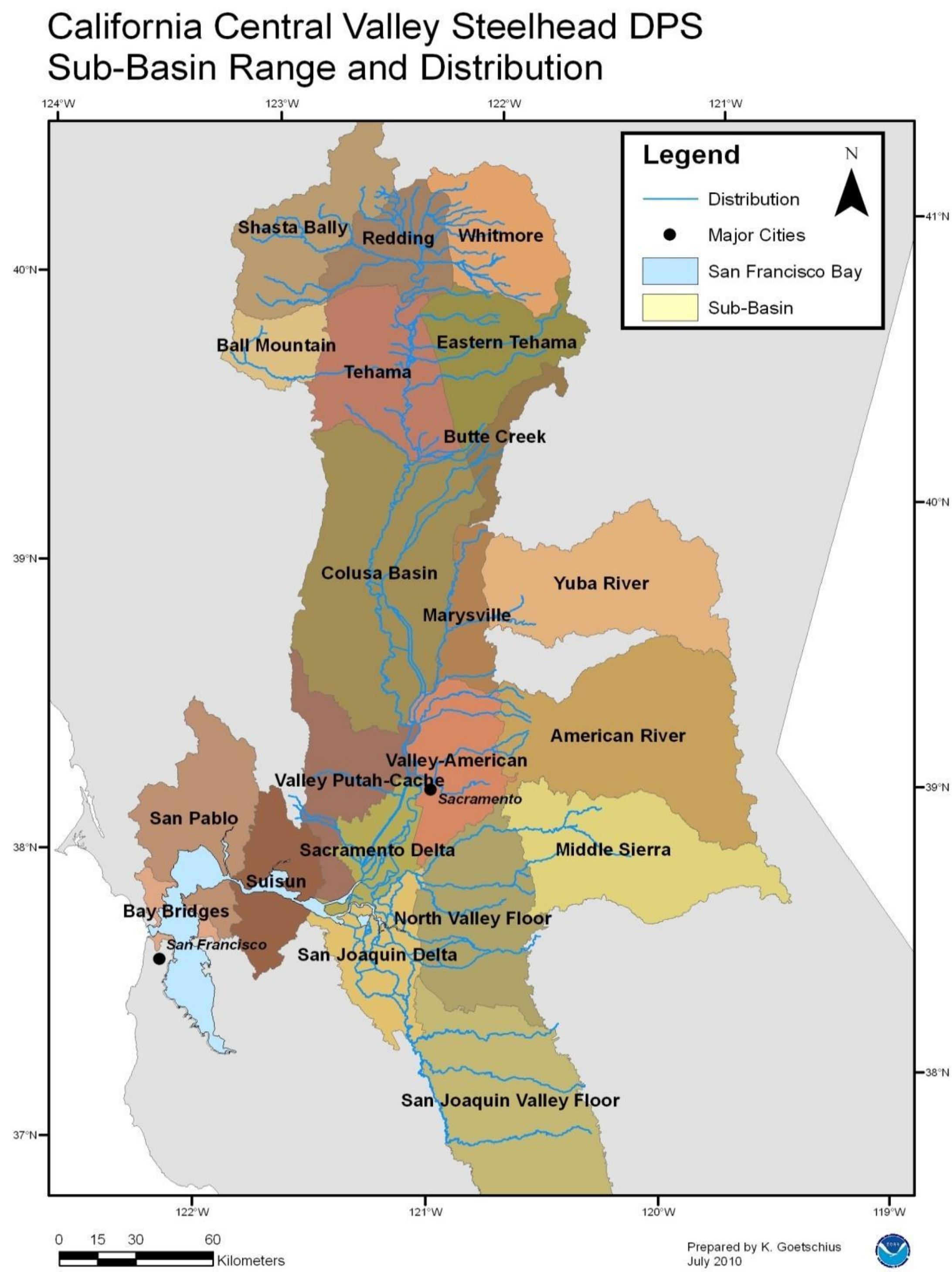

Figure 48. CCV steelhead distribution

Historic CCV steelhead run size may have approached one to two million adults annually (McEwan 2001). By the early 1960s, the steelhead run size had declined to about 40,000 adults (McEwan 2001). Over the past 30 years, the naturally spawned steelhead populations in the upper Sacramento River have declined substantially. Hallock *et al.* (1961) estimated an average of 20,540 adult steelhead 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 and Jackson 1996, McEwan 2001). Based on catch ratios at Chipps Island in the Delta and using some generous assumptions regarding survival, the average number of CV steelhead females spawning naturally in the entire Central Valley during the years 1980 to 2000 was estimated at about 3,600 (Good et al. 2005).

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

The CCV steelhead distribution ranged over a wide variety of environmental conditions and likely contained biologically significant amounts of spatially structured genetic diversity (Lindley et al. 2006). 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.

7.33.3 Critical Habitat

NMFS designated critical habitat for CCV steelhead on September 2, 2005 (70 FR 52488). Critical habitat includes stream reaches such as those of the Sacramento, Feather, and Yuba Rivers, and Deer, Mill, Battle, and Antelope creeks in the Sacramento River basin; the lower San Joaquin River to the confluence with the Merced River, including its tributaries, and the waterways of the Delta (Figure 49). 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 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 43. 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 (Table 43). 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.

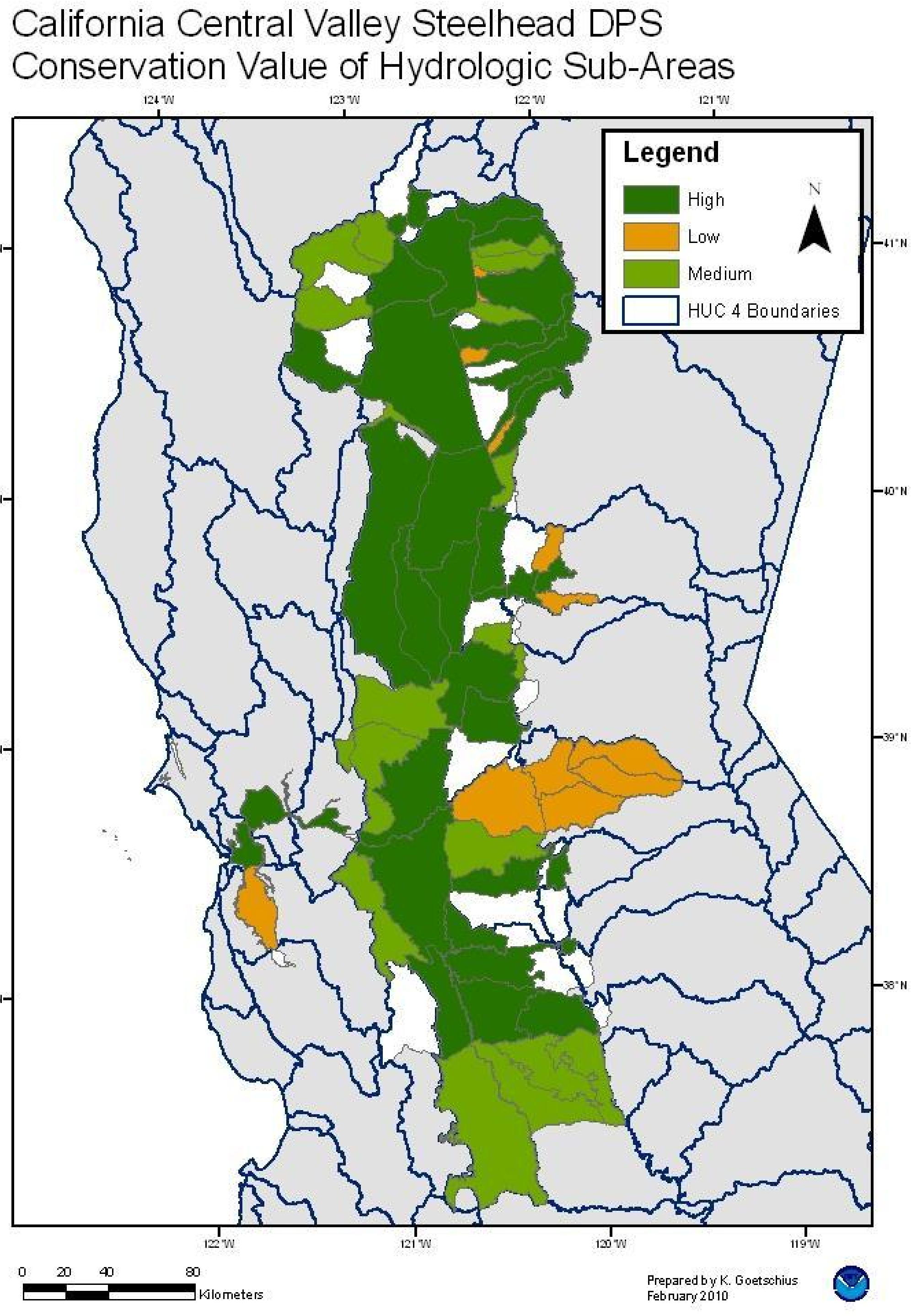

Figure 49. California Central Valley Steelhead conservation value per sub-area

7.34 *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 **51**).

7.34.1 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.

7.34.2 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. Figure 50 depicts the conservation values for this DPS. 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 (

Table 44)(NMFS 2005c). 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.

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.

Table 44. 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.

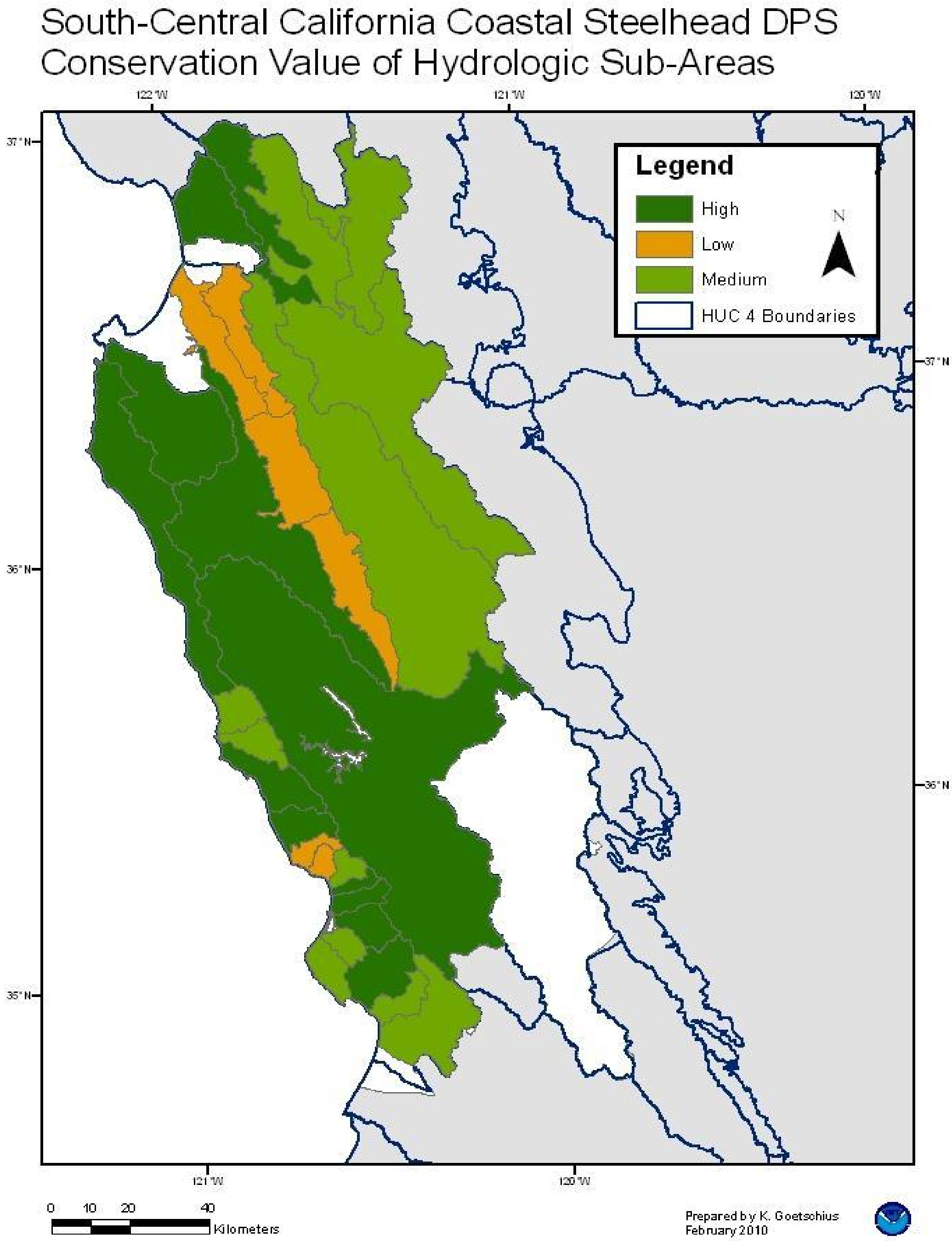

Figure 50. South-Central California Coast Steelhead conservation values per sub-area

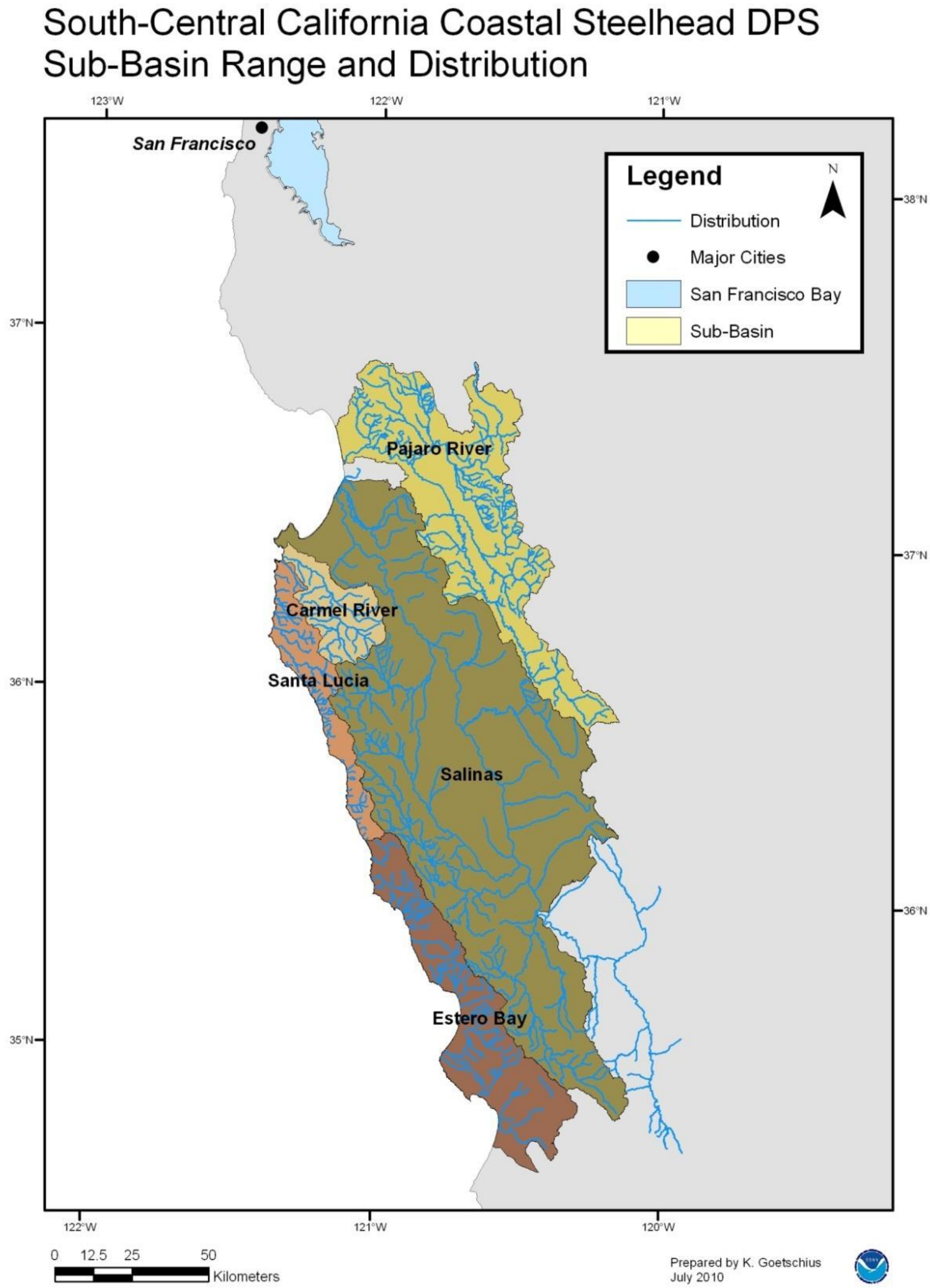

Figure 51. **S-CCC steelhead distribution**

7.35 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 **52**). Artificially propagated steelhead that reside within the historical geographic range of this DPS are not included in the listing.

7.35.1 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). 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).

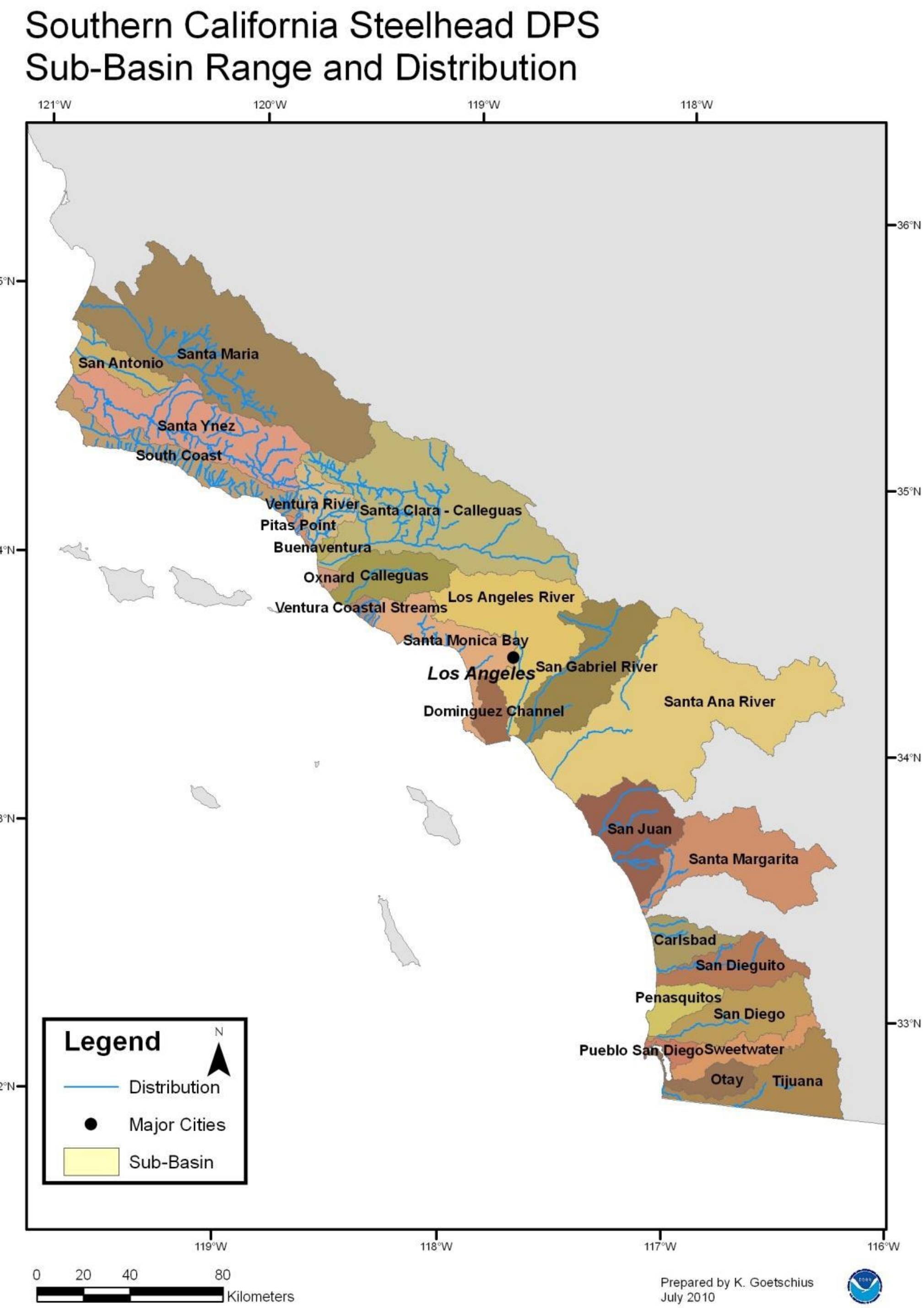

Figure 52 Southern California steelhead distribution

7.35.2 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 45).

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

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

Construction of dams and corresponding increase in 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 et al. 2005). 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 et al. 1996). Table 45 identifies populations within the SC Steelhead salmon ESU, their abundances, and hatchery input.

7.35.3 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 46). Figure 53 provides conservation values for this DPS per sub-area. Three watersheds received a low, five received a medium, and 21 received a high conservation value rating for the ESU (NMFS 2005c).

Table 46. 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.

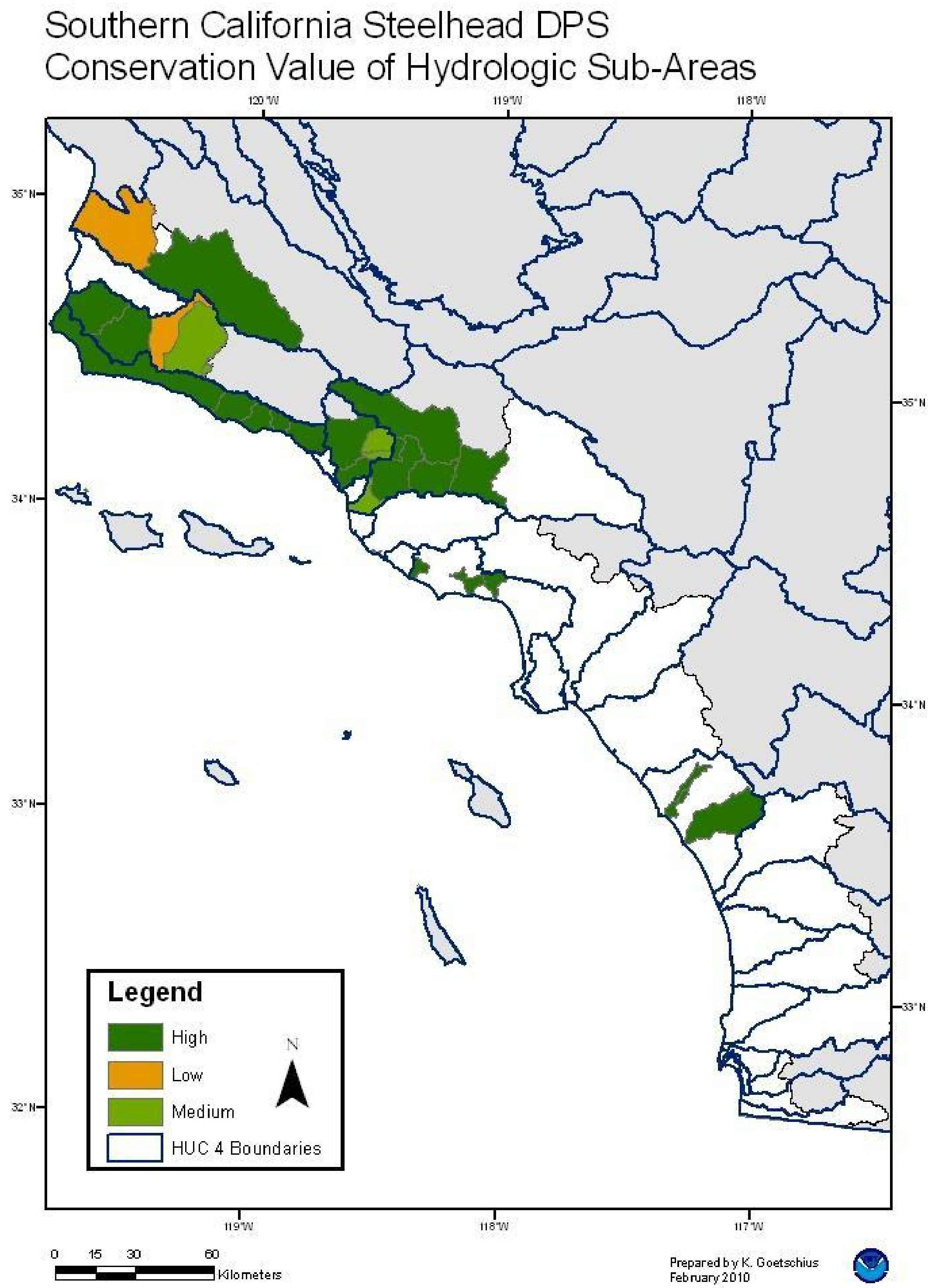

Figure 53. Southern California Steelhead Conservation Values per Sub-area

8 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 EPA's actions in combination with the status of the species, environmental baseline and the cumulative effects to determine whether EPA can insure that the likelihood of jeopardy or adverse modification of designated critical habitat will be avoided.

The proposed action under consultation is focused geographically 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 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.

8.1 Natural Mortality Factors

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

8.1.1 Parasites and/or Disease

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

(Flavobacterium columnare).

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

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

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, CDFG 2003).

8.1.2 Predation

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

8.1.3 *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. As indicated above, southern resident killer whales have a strong preference for Chinook salmon (up to 78% of identified prey) during late spring to fall (Hanson et al. 2005, Hard et al. 1992, Ford and Ellis 2006). Generally, harbor seals do not feed on salmonids as frequently as California sea lions (Pearcy 1997). California sea lions from the Ballard Locks in Seattle, Washington have been estimated to consume about 40% of the steelhead runs since 1985/1986 (Gustafson et al. 1997). In the Columbia River, salmonids may contribute substantially to sea lion diet at specific times and locations (Pearcy 1997). Spring Chinook salmon and steelhead are subject to pinniped predation when they return to the estuary as adults (NMFS 2006). Adult Chinook salmon in the Columbia River immediately downstream of Bonneville Dam have also experienced increased predation by California sea lions. In recent years, sea lion predation of adult Lower Columbia River winter steelhead in the Bonneville tailrace has increased. This prompted ongoing actions to reduce predation effects. They include the exclusion, hazing, and in some cases, lethal take of marine mammals near Bonneville Dam (NMFS 2008d).

8.1.4 *Avian Predation*

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

Antolos *et al.* (2005) quantified predation on juvenile salmonids by Caspian terns nesting on Crescent Island in the mid-Columbia reach. Between 1,000 and 1,300 adult terns were associated with the colony during 2000 and 2001, respectively. These birds consumed about 465,000 juvenile salmonids in the first and approximately 679,000 salmonids in the second year. However, caspian tern predation in the estuary was reduced from a total of 13,790,000 smolts to 8,201,000 smolts after relocation of the colony from Rice to East Sand Island in 1999. Based on PIT-tag recoveries at the colony, these were primarily steelhead for Upper Columbia River stocks. Less than 0.1% of the inriver migrating yearling Chinook salmon from the Snake River and less than 1% of the yearling Chinook salmon from the Upper Columbia were consumed. PIT-tagged coho smolts (originating above Bonneville Dam) were second only to steelhead in predation rates at the East Sand Island colony in 2007 (Roby et al. 2008). There are few quantitative data on avian predation rates on Snake River sockeye salmon. Based on the above, avian predators are assumed to have a minimal effect on the long-term survival of Pacific salmon[15].

8.1.5 Fish Predation

Pikeminnows (*Ptychocheilus oregonensis*) are significant predators of yearling juvenile migrants (Friesen and Ward 1999). Chinook salmon were 29% of the prey of northern

[15] On March 15, NMFS authorized lethal removal of up to 460 sea lions over the next five years. The Humane Society of the U.S. sued to stop the killing and sought injunctive relief. The court denied emergency injunctive relief but will consider additional injunctive relief most likely by the end of May, and will consider the merits of the case later this year or early next year. Since the court's denial of an emergency injunction, several sea lions have been euthanized.

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) 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 and Ward 1999).

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, Beamish and Neville 1995, Beamish et al. 1992).

8.1.6 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 loss of large woody debris (Rinne 2004, Buchwalter et al. 2004). Nevertheless, fire is also one of the dominant habitat-forming processes in mountain streams (Bisson et al. 2003). As a result, many large fires burning near streams can result in fish kills with the survivors actively moving downstream to avoid poor water quality conditions (Rinne 2004, Greswell 1999). The patchy, mosaic pattern burned by fires provides a refuge for those fish and invertebrates that leave a burning area or simply spares some fish that were in a different location at the time of the fire (USFS 2000). Small fires or fires that burn entirely in upland areas also cause ash to enter rivers and increase smoke in the atmosphere, contributing to ammonia concentrations in rivers as the smoke adsorbs into the water (Greswell 1999).

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

8.1.7 Oceanographic Features, Climatic Variability and Climate Change

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

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

Over the past century, regime shifts have occurred as a result of the North Pacific's natural climate regime. Reversals in the prevailing polarity of the PDO occurred around 1925, 1947, 1977, and 1989 (Hare and Mantua. 2000, Mantua et al. 1997). The reversals in 1947 and 1977 correspond to dramatic shifts in salmon production regimes in the North Pacific Ocean (Mantua et al. 1997). During the pre-1977 climate regime, the productivity of salmon populations from the Snake River exceeded expectations (residuals were positive) when values of the PDO were negative (Levin 2003). During the post-1977 regime when ocean productivity was generally lower (residuals were negative), the PDO was negative (Levin 2003).

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

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

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

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

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

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

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

8.2 *Anthropogenic Mortality Factors*

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

Prior to discussion of each geographic region, three major issues are highlighted: pesticide contamination, elevated water temperature (due to a number of causes), 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 diflubenzuron, fenbutatin-oxide, and propargite in the U.S. and its territories. These and other chemicals have been in use for multiple decades, they have documented presence in our nation's rivers, and thus over the years have contributing effects to the environmental baseline. As water temperature plays such a strong role in salmonid distribution, we also provide a general discussion of anthropogenic temperature impacts. Next, we discuss the health of riparian systems and floodplain connectivity, as this habitat is vital to salmonid survival. Finally, we provide a brief overview of the results of section 7 consultations relevant to this analysis.

8.3 Baseline Pesticide Detections in Aquatic Environments

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

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

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

8.3.1 National Water-Quality Assessment Program

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

Twenty-four pesticides and one degradate were each detected in over 10% of streams in agricultural, urban, or mixed land use areas. These 25 compounds include 11 agriculture-use herbicides and the atrazine degradate deethylatrazine; 7 urban-use herbicides; and 6 insecticides used in both agricultural and urban areas. Five of the insecticides were carbaryl, carbofuran, chlorpyrifos, diazinon, and malathion. NMFS assessed the effects of these five insecticides on listed salmonids in its 2008 and 2009 Opinions (NMFS 2008e, NMFS 2009d). The contributing adverse effects of these chemicals on the environmental baseline and salmonids continue today as the EPA has yet to implement the conservation measures (reasonable and prudent alternatives and measures) identified in the previous Opinions.

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

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

8.3.2 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, sedimentation, and manure.

According to EPA's database of NPDES permits, about 243 NPDES individual permits are co-located with listed Pacific salmonids in California. 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 maps 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.

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. In response, EPA has developed a Pesticide General Permit through the NPDES permitting program to regulate such discharges. The permit is currently undergoing Section 7 consultation.

8.4 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 are impaired by pollutants including drinking, recreation, aquatic habitat, and industrial uses. This process is in accordance with section 303(d) of the CWA. Estuaries, lakes, and streams listed under 303(d) are those that are considered impaired or threatened by pollution. They are water quality limited, do not meet state surface water quality standards, and are not expected to improve within the next two years.

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

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

Table 47. Washington State water temperature thresholds for salmonid habitat. These temperatures are representative of limits set by California, Idaho, and Oregon (WSDE 2006).

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 48). Because the 303(d) list is limited to the subset of rivers tested, the chart values should be regarded as lower-end estimates. Each of the four states are in the process of finalizing their 2010 Water Quality Integrated Reports, complete with 303(d) list.

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

Table 48. 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	Puget Sound	–	–	373.7	–	373.7
	Lower Columbia River	–	147.0	218.6	–	365.6
	Upper Columbia River Spring - Run	–	–	19.3	–	19.3
	Snake River Fall - Run	–	-	113.4	160.2	273.6
	Snake River Spring/Summer - Run	–	121.1	111.7	275.9	508.7
	Upper Willamette River	–	533.0	–	–	533.0
	California Coastal	9,623.5	–	–	–	9,623.5
	Central Valley Spring - Run	29.9	–	–	–	29.9
	Sacramento River Winter - Run	29.9	–	–	–	29.9
Chum Salmon	Hood Canal Summer - Run	–	–	47.7	–	47.7
	Columbia River	–	95.0	216.2	–	311.2
Coho Salmon	Lower Columbia River	–	99.2	221.5	–	320.7
	Oregon Coast	–	920.4	–	–	920.4
	Southern Oregon and Northern California Coast	11,044.5	694.5	–	–	11,739.0
	Central California Coast	4,731.7	–	–	–	4,731.7
Sockeye Salmon	Ozette Lake	–	–	22.5	–	22.5
	Snake River	–	–	–	0.0	0.0
Steelhead	Puget Sound	–	–	373.7	–	373.7
	Lower Columbia River	–	147.0	140.3	–	287.3
	Upper Willamette River	–	299.0	–	–	299.0
	Middle Columbia River	–	1498.5	209.4	–	1707.9
	Upper Columbia River	–	–	33.5	–	33.5

	Snake River	–	121.1	111.7	739.8	972.6
	Northern California	6,7920.0	–	–	–	6,7920.0
	Central California Coast	2,948.8	–	–	–	2,948.8
	California Central Valley	367.8	–	–	–	367.8
	South-Central California Coast	282.8	–	–	–	282.8
	Southern California	151.5	–	–	–	151.5

*CA 2010, Oregon 2004-06, Washington 2008, and Idaho 2008. (California EPA TMDL Program 2011, Oregon Department of Environmental Quality 2008, Washington State Department of Ecology 2009, Idaho Department of Environmental Quality 2009).

8.5 *Baseline Habitat Condition*

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

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

Floodplains are relatively flat areas adjacent to streams and rivers that stretch from the banks of the channel to the base of the enclosing valley walls. They allow for the lateral movement of the main channel and provide storage for floodwaters during periods of high flow. The floodplain includes the floodway, which consists of the stream channel, and adjacent areas that actively carry flood flows downstream; and the flood fringe, which are areas that are inundated, but which do not experience a strong current. 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).

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

Healthy riparian habitat and floodplain connectivity are vital for supporting a salmonid population. Chinook salmon and steelhead have life history strategies that rely on floodplains during their juvenile life stages. 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). 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).

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.

8.6 Baseline Pesticide Consultations

NMFS has consulted with EPA on the registration of several pesticides. NMFS (NMFS 2008c) determined that current use of chlorpyrifos, diazinon, and malathion is likely to jeopardize the continued existence of 27 listed salmonid ESUs/DPSs. NMFS (NMFS 2009b) 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. NMFS also published conclusions regarding the registration of 12 different a.i.s (NMFS 2010b). NMFS concluded that pesticide products containing azinphos methyl, disulfoton, fenamiphos, methamidophos, or methyl parathion are not likely to jeopardize the continuing existence of any listed Pacific Salmon or destroy or adversely modify designated critical habitat. NMFS also concluded that the effects of products containing bensulide, dimethoate, ethoprop, methidathion, naled, phorate, or phosmet are likely to jeopardize the continued existence of some listed Pacific Salmonids and to destroy or adversely modify designated habitat of some listed salmonids. NMFS issued a biological opinion on the effects of four herbicides and two fungicides (NMFS 2011). NMFS concluded that products containing 2,4-D are likely to jeopardize the existence of all listed salmonids, and adversely modify or destroy the critical habitat of dome ESU / DPSs. Products containing chlorothalonil or diuron were also likely to adversely modify or destroy critical habitat, but not likely to jeopardize listed salmonids. NMFS also concluded that products containing captan, linuron, or triclopyr BEE do not jeopardize the continued existence of any ESUs/DPSs of listed Pacific salmonids or adversely modify designated critical habitat. Most recently in 2012, NMFS completed two additional biological opinions covering four more pesticides. In May, 2012 NMFS issued an opinion on oryzalin, pendimethalin, and trifluralin concluding each of these chemicals are likely to jeopardize the continued existence of some listed Pacific salmonids, and adversely modify designated critical habitat of some listed salmonids (NMFS 2012a). In July 2012, NMFS issued an opinion on thiobencarb, an herbicide authorized for use only on rice. California is the only state within the range of listed Pacific salmonids that has approved the use of thiobencarb and is the only state among the action area states that grows rice. The thiobencarb opinion focused on three listed Pacific salmon ESUs/DPSs in California's Central Valley where rice is grown. NMFS concluded EPAs registration of thiobencarb would harm listed species, but not jeopardize the continued existence of these three species and would not adversely modify their designated critical habitat.

8.7 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 2007). We summarize the principal anthropogenic factors occurring in the environment that influence the current status of listed species within each region. Table 49 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. A graphic depiction of most of the information in this table can be found at: http://www.nwr.noaa.gov/publications/protected_species/salmon_steelhead/status_of_esa_salmon_listings_and_ch_designations_map.pdf

Table 49. USGS Subregions and accounting units within the Northwest and Southwest Regions, along with ESUs/DPSs present within the area (Seaber et al. 1987)

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
	Lower Columbia River Basin	—	OR, WA	170800	Lower Columbia Chinook; Columbia Chum; Lower Columbia Steelhead; Lower Columbia Coho

Region	USGS Subregion	Accounting Unit	State	HUC no.	ESU/DPS
	Willamette River Basin	—	OR	170900	Upper Willamette Chinook; Upper Willamette Steelhead; Lower Columbia Chinook; Lower Columbia Steelhead; Lower Columbia Coho
Oregon-Washington-Northern California Coastal Drainages	Oregon-Washington Coastal Basin	Washington Coastal	WA	171001	Ozette Lake Sockeye
		Northern Oregon Coastal	OR	171002	Oregon Coast Coho
		Southern Oregon-Northern California Coastal	OR, CA	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

8.7.1 *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 49). Table 50 and Table 51 show land area in km² for each ESU/DPS located in the Southwest Coast Region.

Table 50. Area of land use categories within the range Chinook and Coho Salmon ESUs in km². The total area for each category is given in bold. Land cover was determined via the National Land Cover Database 2006, developed by the Multi-Resolution Land Characteristics (MRLC) Consortium, a consortium of nine federal agencies (USGS, EPA, USFS, NOAA, NASA, BLM, NPS, NRCS, and USFWS). Land cover class definitions are available at: http://www.mrlc.gov/nlcd_definitions.php

Land Cover		Chinook Salmon			Coho Salmon	
sub category	code	CA Coastal	Central Valley	Sacramento River	S Oregon and N CA	Central CA Coast
Water		**128**	**367**	**367**	**205**	**158**
Open Water	11	128	367	367	194	158
Perennial Snow/Ice	12	0	0	0	11	0
Developed Land		**1,139**	**2,755**	**2,755**	**1,979**	**996**
Open Space	21	828	1,174	1,174	1,390	630
Low Intensity	22	140	635	635	238	173
Medium Intensity	23	98	616	567	97	141
High Intensity	24	11	153	153	24	32
Barren Land	31	62	178	178	230	21
Undeveloped Land		**19,067**	**15,063**	**15,063**	**43,324**	**9,169**
Deciduous Forest	41	838	657	657	1,041	208
Evergreen Forest	42	10,642	3,707	3,707	27,253	4,744
Mixed Forest	43	1,547	476	476	2,394	921
Shrub/Scrub	52	3,858	3,245	3,245	9,652	1,630
Herbaceous	71	2,118	6,261	6,261	2,798	1,628
Woody Wetlands	90	43	189	189	130	25
Emergent Wetlands	95	20	527	527	56	13
Agriculture		**406**	**5,796**	**5,796**	**1,189**	**249**
Hay/Pasture	81	182	754	754	719	6
Cultivated Crops	82	224	5,043	5,043	470	243
TOTAL (inc. open water)		20,740	23,982	23,982	46,697	10,572
TOTAL (w/o open water)		20,612	23,615	23,615	46,503	10,414

Table 51. Area of Land Use Categories within the Range of Steelhead Trout DPSs (km²). The total area for each category is given in bold. Land cover was determined via the National Land Cover Database 2006, developed by the Multi-Resolution Land Characteristics (MRLC) Consortium, a consortium of nine federal agencies (USGS, EPA, USFS, NOAA, NASA, BLM, NPS, NRCS, and USFWS). Land cover class definitions are available at: http://www.mrlc.gov/nlcd_definitions.php

Land Cover		Steelhead				
sub category	code	Northern CA	Central CA Coast	CA Central Valley	South-Central CA coast	Southern CA
Water		**92**	**1,426**	**422**	**114**	**161**
Open Water	11	92	1,426	422	114	161
Perennial Snow/Ice	12	0	0	0	0	0
Developed Land		**748**	**3,725**	**3,534**	**1,765**	**7,517**
Open Space	21	612	1,234	1,472	1,019	2,013
Low Intensity	22	50	890	792	249	1,825
Medium Intensity	23	32	1,244	837	173	2,800
High Intensity	24	3	333	211	23	780
Barren Land	31	53	24	222	300	99
Undeveloped Land		**16,139**	**10,949**	**19,138**	**14,968**	**12,911**
Deciduous Forest	41	752	179	744	2	1
Evergreen Forest	42	9,751	2,501	3,942	1,730	932
Mixed Forest	43	1,154	2,092	593	1,924	989
Shrub/Scrub	52	2,936	2,262	3,786	4,957	8,265
Herbaceous	71	1,495	3,509	9,396	6,193	2,594
Woody Wetlands	90	33	37	245	93	79
Emergent Wetlands	95	19	369	431	69	51
Agriculture		**194**	**545**	**10,507**	**1,497**	**1,016**
Hay/Pasture	81	178	35	1,640	196	161
Cultivated Crops	82	15	511	8,867	1,301	855
TOTAL (inc. open water)		17,173	16,645	33,601	18,344	21,604
TOTAL (w/o open water)		17,081	15,220	33,179	18,230	21,443

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 52).

Table 52. Select rivers in the southwest coast region (Carter and Resh 2005).

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

Table 58 displays major landuse categories in California. Within the Southwest Coast Region, forest and vacant land are the dominant land uses. Grass, shrubland, and urban uses are the dominant land uses in the southern basins (Table 53). 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 et al. 2004, Burton et al. 1998). The basin is home to nearly 5 million people and this population is projected to increase two-fold in the next 50 years (Belitz et al. 2004, Burton et al. 1998).

Table 53. Land uses and population density in several southwest coast watersheds (Carter and Resh 2005).

Watershed	Land Use Categories (Percent)				Density (people/mi^2)
	Agriculture	Forest	Urban	Other	
Rogue River	6	83	<1	9 grass & shrub	32
Klamath River	6	66	<1	24 grass, shrub, wetland	5
Eel River	2	65	<1	31 grass & shrub	9
Russian River	14	50	3	31 (23 grassland)	162
Sacramento River	15	49	2	30 grass & shrub	61
San Joaquin River	30	27	2	36 grass & shrub	76
Salinas River	13	17	1	65 (49 grassland)	26
Santa Ana River	11	57	32	---	865
Santa Margarita River	12	11	3	71 grass & shrub	135

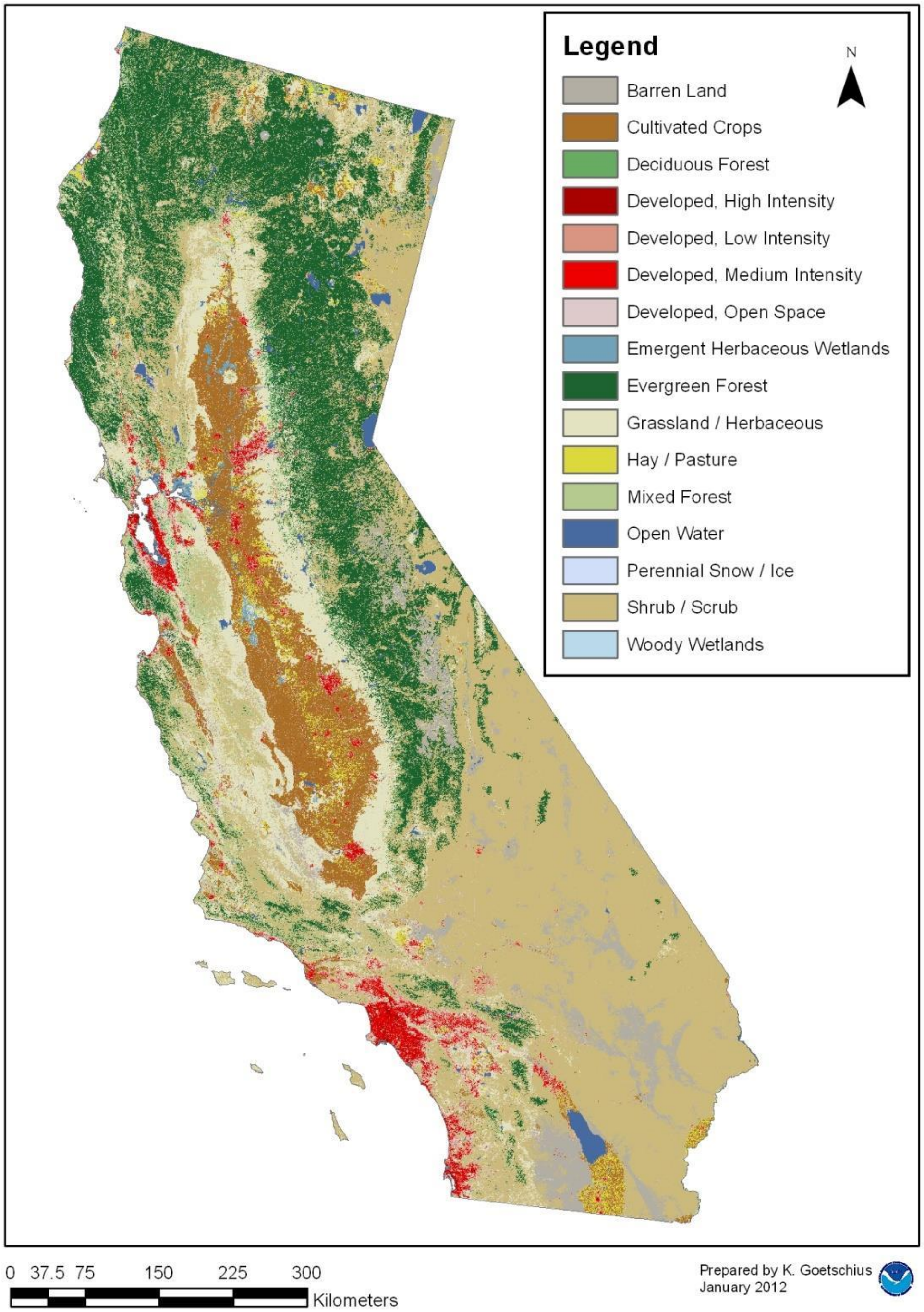

Figure 54. Landuse in Southwest Region. Using the National Land Cover Database 2006.

As a watershed becomes urbanized, human 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). 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. Agricultural uses of the a.i.s are described in the *Description of the Proposed Action*. Pesticide detection data for these same a.i.s are reported in the Monitoring subsection of the *Effects* chapter.

Pesticide Reduction Programs in the Southwest Coast Region

When using these three 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.

California State Code does not include specific limitations on pesticide application aside from human health protections. It only includes statements advising that applicators are required to follow all federal, state, and local regulations.

Additionally, pesticide reduction programs already exist in California to minimize levels of the above a.i.s into the aquatic environment. 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 et al. 2005). 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 et al. 2005). 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. 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 pesticide 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 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 spray 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 exceedances, 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 diflubenzuron, fenbutatin-oxide, and propargite use in salmonid habitat are:

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

- Provide a 20-foot 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 run-off could occur. Prepare land around fields to contain run-off 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.

- Conduct irrigations efficiently to prevent excessive loss of irrigation waters through run-off. 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 application of pesticides listed in this bulletin and any irrigation that results in surface run-off 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 run-off is contained for 72 hours following the application.

- 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.

Water Diversions for Agriculture in the Southwest Coast Region

Agricultural land use further impacts salmonid aquatic habitats through water diversions or withdrawals from rivers and tributaries. 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 and Resh 2005). 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.

Surface and Ground Water Contaminants

California's most recent 303(d) list is the 2010 Integrated Report, which was approved by EPA on October 11, 2011. The 2010 list includes 3,489 stream segments, rivers, lakes, and estuaries and 13 pollutant categories (Figure 55).

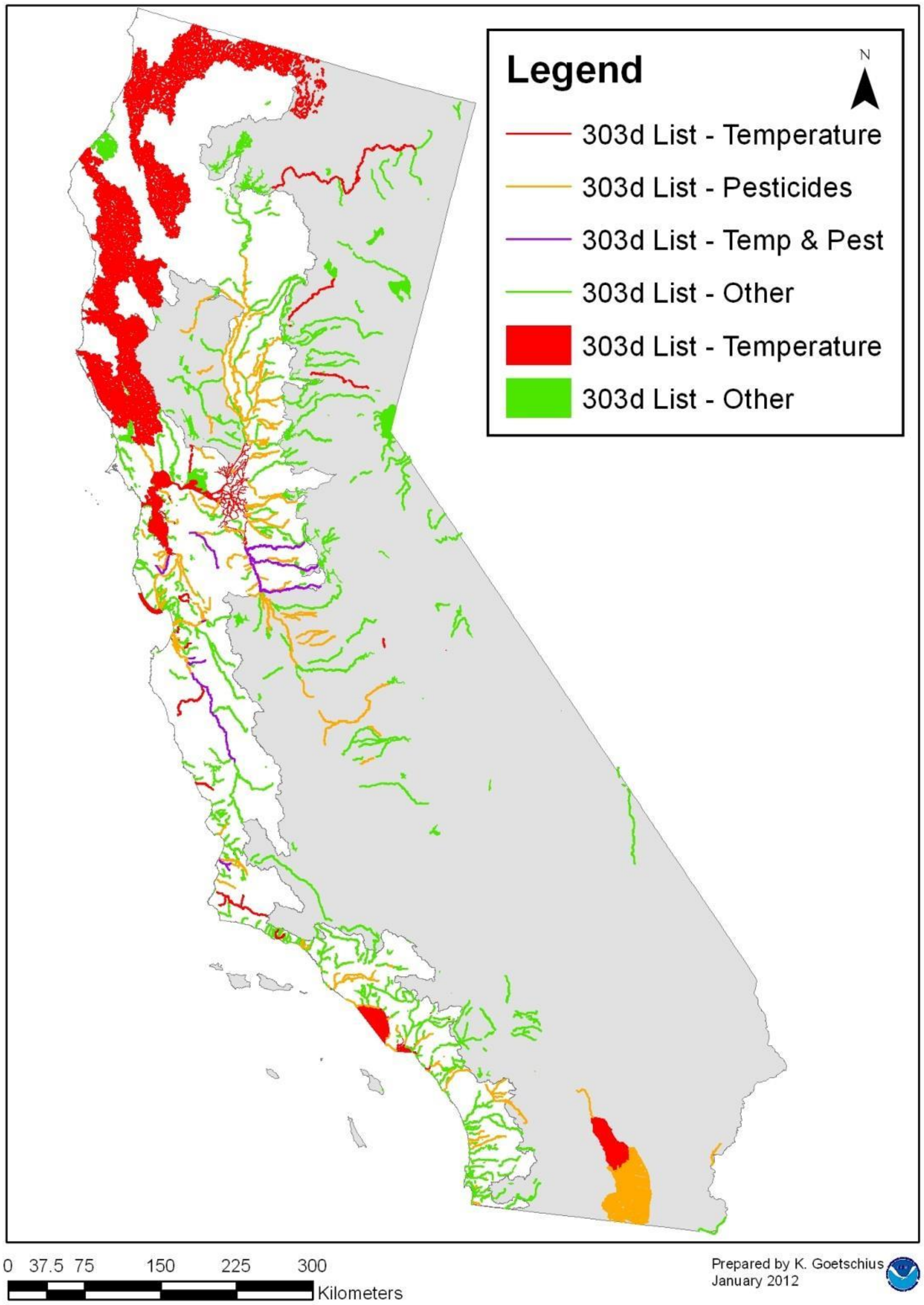

Figure 55. California 303(d) List: Water bodies and stream segments included in the 2010 Integrated Report.

Pollutants represented on the list include pesticides, metals, sediments, nutrients or low dissolved oxygen, temperature, bacteria and pathogens, and trash or debris. The 2010 303(d) list identifies water bodies listed due to elevated temperature (Table 54).

Table 54. California's 2010 Integrated Report, Section 303(d) List of Water Quality Limited Segments: segments listed for exceeding temperature limits which require the development of a TMDL

Pollutant	Estuary Acres Affected	River / Stream Km Affected	# Water Bodies
Temperature	-	18,332.0	69

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. 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), 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 et al. 2004). 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 et al. 2004). Additionally, Belitz *et al.* (2004) 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. Diuron was detected in 92% of urban samples – a rate much higher than the national frequency of 25 % (Belitz et al. 2004). 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), 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 over 20% of the samples (Dubrovsky et al. 1998). Further, many samples contained mixtures of at least 7 pesticides, with a maximum of 22 different compounds. Diuron was detected in all three subbasins, despite land use differences.

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 et al. 1998).

Sacramento River Basin: NAWQA Analysis

Another study conducted by the USGS from 1996 - 1998 within the Sacramento River Basin compared the pesticides in surface waters at four specific sites – urban, agricultural, and two integration sites (Domagalski 2000). Pesticides included thiobencarb, carbofuran, molinate, simazine, metolachlor, dacthal, chlorpyrifos, carbaryl, and diazinon – as well as the three herbicides assessed in this Opinion. Land use differences between sites are reflected in pesticide detections. Thiobencarb was detected in 90.5 % of agricultural samples, but only 3.3% of urban samples (Domagalski 2000). This finding is unsurprising as rice is the dominant crop within the agricultural basin. 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). In the San Joaquin Basin alone, the historic floodplain covered 1.5 million acres with 2 million acres of riparian vegetation (CDFG 1993). 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). 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).

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). 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).

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 and Resh 2005). 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 et al. 2003).

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 (Figure 56). There are about 1,400 dams within the State of California, more than 5,000 miles of levees, and more than 140 aqueducts (Mount 1995). In 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 and Resh 2005).

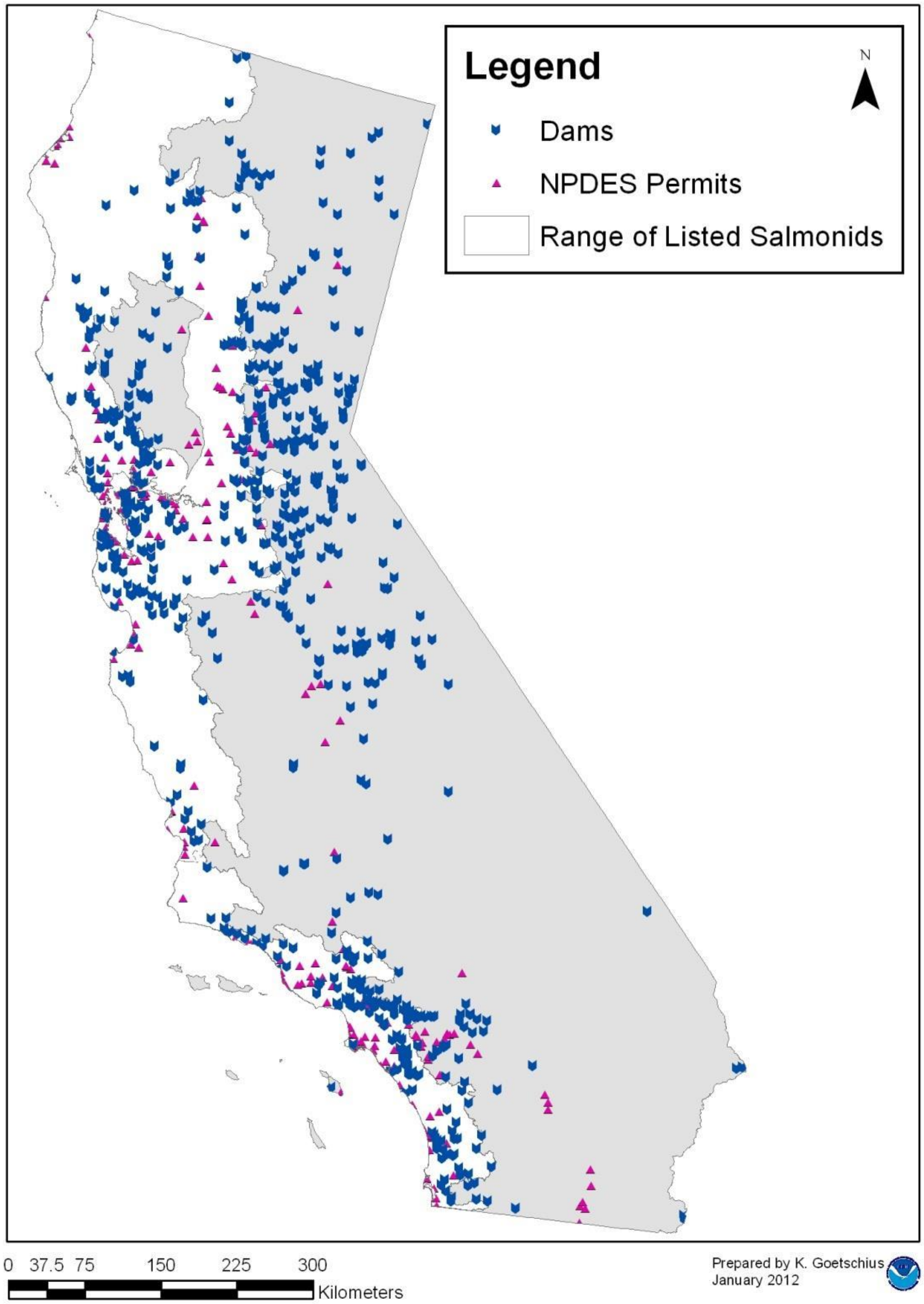

Figure 56. Southwest Coast dams and NPDES permit sites.

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). 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) establish 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-run 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 et al. 1989). Wilcove, Rothstein *et al.* (1998) 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 et al. 2007). 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.

8.7.2 Pacific Northwest: Columbia River Basin 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 (Figure 57).

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 49). Table 55, Table 56, and Table 57 show the types and areas of land use within each salmonid ESU/DPS.

Table 55. Area of land use categories within Chinook Salmon ESUs in km². The total area for each category is given in bold. Land cover was determined via the National Land Cover Database 2006, developed by the Multi-Resolution Land Characteristics (MRLC) Consortium, a consortium of nine federal agencies (USGS, EPA, USFS, NOAA, NASA, BLM, NPS, NRCS, and USFWS). Land cover class definitions are available at: http://www.mrlc.gov/nlcd_definitions.php

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		**6,447**	**662**	**200**	**219**	**283**	**127**
Open Water	11	6,147	651	186	219	252	122
Perennial Snow/Ice	12	300	11	14	0	32	6
Developed Land		**5,311**	**1,949**	**875**	**484**	**981**	**2,008**
Open Space	21	1,624	708	205	350	329	646
Low Intensity	22	1,734	571	234	70	114	750
Medium Intensity	23	405	310	61	19	31	333
High Intensity	24	277	126	12	2	2	117
Barren Land	31	971	234	362	43	506	162
Undeveloped Land		**22,502**	**13,005**	**16,123**	**21,437**	**52,608**	**14,251**
Deciduous Forest	41	987	553	21	57	10	239
Evergreen Forest	42	13,983	8,006	7,589	10,704	27,215	9,046
Mixed Forest	43	2,532	933	7	5	4	1,068
Shrub/Scrub	52	2,896	2,298	6,539	5,063	14,208	2,350
Herbaceous	71	956	570	1,818	5,583	10,933	1,032
Woody Wetlands	90	651	395	91	29	99	439
Emergent Wetlands	95	496	250	59	28	102	76
Agriculture		**1,404**	**944**	**952**	**5,179**	**4,288**	**5,883**
Hay/Pasture	81	1,152	636	317	57	444	3,585
Cultivated Crops	82	251	308	635	5,122	3,843	2,298
TOTAL (inc. open water)		35,663	16,560	18,150	27,319	58,160	22,269
TOTAL (w/o open water)		29,516	15,910	17,964	27,100	57,908	22,148

Table 56. Area of land cover types within chum, coho, and sockeye ESUs in km². The total area for each category is given in bold. Land cover was determined via the National Land Cover Database 2006, developed by the Multi-Resolution Land Characteristics (MRLC) Consortium, a consortium of nine federal agencies (USGS, EPA, USFS, NOAA, NASA, BLM, NPS, NRCS, and USFWS). Land cover class definitions are available at: http://www.mrlc.gov/nlcd_definitions.php

Land Cover Category		Chum Salmon		Coho Salmon		Sockeye Salmon	
sub category	code	Hood Canal	Columbia River	Lower Columbia River	Oregon Coast	Ozette Lake	Snake River
Water		**750**	**652**	**681**	**203**	**30**	**33**
Open Water	11	703	651	670	203	30	19
Perennial Snow/Ice	12	47	1	11	0	0	15
Developed Land		**392**	**1,658**	**1,977**	**1,577**	**1**	**15**
Open Space	21	135	614	719	1,113	1	3
Low Intensity	22	79	476	583	168	0	2
Medium Intensity	23	20	265	314	51	0	0
High Intensity	24	6	112	127	20	0	0
Barren Land	31	152	191	235	225	0	10
Undeveloped Land		**3,343**	**8,284**	**13,345**	**24,832**	**197**	**1,262**
Deciduous Forest	41	97	537	564	414	3	0
Evergreen Forest	42	2,371	4,008	8,157	14,133	148	741
Mixed Forest	43	197	844	948	3,898	2	0
Shrub/Scrub	52	425	1,759	2,417	4,065	27	198
Herbaceous	71	134	515	612	1,822	7	271
Woody Wetlands	90	62	373	396	26	8	16
Emergent Wetlands	95	57	248	251	235	1	35
Agriculture		**64**	**690**	**956**	**908**	**0**	**12**
Hay/Pasture	81	62	505	644	846	0	12
Cultivated Crops	82	2	185	312	62	0	0
TOTAL (inc. open water)		4,548	11,284	16,959	27,520	228	1,323
TOTAL (w/o open water)		3,845	10,633	16,289	27,320	199	1,304

Table 57. Area of land use categories within steelhead DPSs in km². The total area for each category is given in bold. Land cover was determined via the National Land Cover Database 2006, developed by the Multi-Resolution Land Characteristics (MRLC) Consortium, a consortium of nine federal agencies (USGS, EPA, USFS, NOAA, NASA, BLM, NPS, NRCS, and USFWS). Land cover class definitions are available at: http://www.mrlc.gov/nlcd_definitions.php

Land Cover Category		Steelhead					
sub category	code	Puget Sound	Lower Columbia River	Upper Willamette River	Middle Columbia River	Upper Columbia River	Snake River
Water		**6,444**	**256**	**61**	**585**	**371**	**315**
Open Water	11	6,144	245	61	574	357	283
Perennial Snow/Ice	12	300	12	0	12	14	33
Developed Land		**5,314**	**1,621**	**1,269**	**2,354**	**1,127**	**1,209**
Open Space	21	1,624	529	393	1,289	348	514
Low Intensity	22	1,734	522	533	655	315	144
Medium Intensity	23	705	295	239	204	90	40
High Intensity	24	277	118	79	27	15	3
Barren Land	31	974	158	25	180	359	508
Undeveloped Land		**22,504**	**10,390**	**7,026**	**53,559**	**19,590**	**67,891**
Deciduous Forest	41	987	379	164	53	25	35
Evergreen Forest	42	13,983	6,839	3,837	17,923	7,668	39,965
Mixed Forest	43	2,532	581	743	39	8	18
Shrub/Scrub	52	2,897	1,835	1,282	32,161	9,794	16,335
Herbaceous	71	957	401	655	2,869	1,906	12,298
Woody Wetlands	90	651	247	298	229	107	119
Emergent Wetlands	95	497	109	46	285	82	121
Agriculture		**1,405**	**862**	**4,299**	**12,953**	**3,663**	**6,643**
Hay/Pasture	81	1,153	66	2,501	854	437	449
Cultivated Crops	82	252	295	1,798	12,099	3,226	6,194
TOTAL (inc. open water)		35,667	13,128	12,655	69,451	24,750	76,059
TOTAL (w/o open water)		29,522	12,884	12,593	68,877	24,394	75,776

Pesticide Reduction Programs in the Pacific Northwest Region

When using any of the a.i.s addressed in this Opinion, 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. Additionally, pesticide reduction programs exist in Idaho, Oregon, and Washington to minimize levels of pesticides in the aquatic environment. Washington's Department of Transportation also has limitations on the use of pesticides on rights-of way. Diflubenzuron is approved for use in rights-of-way. The shoulder is typically treated once annually. Diflubenzuron is also approved for use on ornamental planting beds, residential and municipal shade trees, landscape plantings, and recreational areas (e.g., campgrounds, golf courses, parks, and parkways). Fenbutatin-oxide and propargite may be used on ornamentals in nurseries.

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 (www.salmonsafe.org). Salmon-Safe is an independent eco-label recognizing organizations who have adopted conservation practices that help restore native salmon habitat in Pacific Northwest, California, and British Columbia. 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. Diflubenzuron, fenbutatin-oxide, and propargite are on the high risk list. Over 300 farms, 250 vineyards, and 240 parks currently have the Salmon-Safe certification. Salmon-Safe has also worked with over 20 corporate / industrial sites and is beginning programs that focus on golf courses and nurseries.

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.

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 58 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 et al. 2004, Kammerer 1990). The Willamette River is also the 19th largest river in the nation in terms of average annual discharge (Kammerer 1990). The basins drain portions of the Rocky Mountains, Bitteroot Range, and the Cascade Range.

Table 58. Select tributaries of the Columbia River (Carter and Resh 2005).

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 et al. 2005). 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 et al. 2004). See Table 59 for a summary of land uses and population densities in several subbasins within the Columbia River watershed [data from (Stanford et al. 2005)].

Table 59. Land use and population density in select tributaries of the Columbia River (Stanford et al. 2005).

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 et al. 2005). 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.

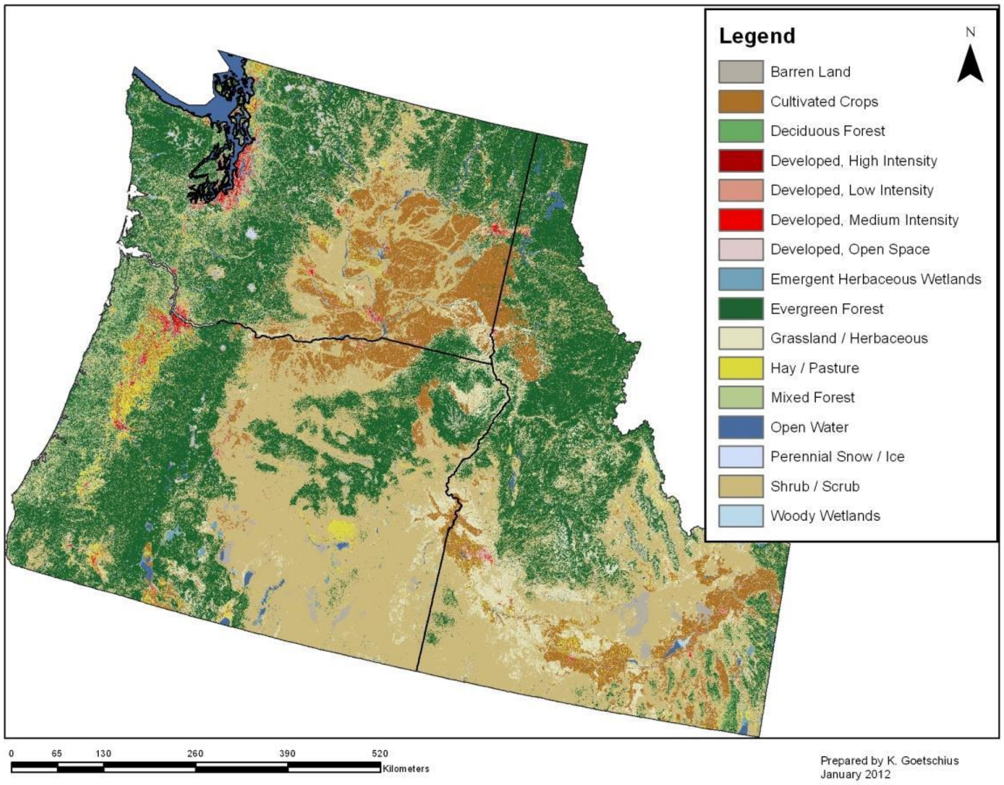

Figure 57. Pacific Northwest: National Land Cover Database 2006.

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)]. Ranching practices have led to increased soil erosion and sediment loads within adjacent tributaries. The worst of these effects may have occurred in the late 1800s and early 1900s from deliberate burning to increase grass production (NRC 2004). Several measures are 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. Agricultural uses of the a.i.s assessed in this Opinion are discussed in the *Description of the Proposed Action*, while detection data is discussed in the Monitoring subsection of the *Effects of the Proposed Action* chapter.

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). 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 et al. 2004).

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.

Surface and Ground Water Contaminants

NAWQA analyses were conducted for five basins within the Pacific Northwest Region. 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. 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 et al. 2004).

Fish and macroinvertebrate communities exhibit an almost linear decline in condition as the level of agriculture intensity increases within a basin (Cuffney et al. 1997, Fuhrer et al. 2004). A study conducted in the late 1990s examined 11 species of fish, including anadromous and resident fish collected throughout the 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 et al. 2004). 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 et al. 2004). From 1999 to 2000, the USGS conducted a NAWQA study in the Yakima River Basin. Fuhrer *et al.* (2004) 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.

The USGS also detected 76 pesticide compounds in the Yakima River Basin. They include 38 herbicides, 17 insecticides (such as carbaryl, diazinon, and malathion), 15 breakdown products, and 6 others (Fuhrer et al. 2004). In agricultural drainages, insecticides were detected in 80% of samples and herbicides were present in 91%. They were also detected in mixed landuse streams – 71% and 90 %, respectively. 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.

Ninety-one percent of the samples collected from the small agricultural watersheds contained at least two pesticides or pesticide breakdown products. Samples contained a median of 8 and a maximum of 26 chemicals (Fuhrer et al. 2004). The herbicide 2,4-D, occurred most often in the mixtures, along with azinphos methyl, the most heavily applied pesticide, and atrazine, one of the most aquatic mobile pesticides (Fuhrer et al. 2004). The most frequently detected pesticides in the Yakima River Basin are total DDTs, dichloro-diphenyl-dichloroethane (DDD), and dieldrin (Johnson and Newman 1983, Joy 2002, Joy and Madrone 2002, Fuhrer et al. 2004). Nevertheless, concentrations of total DDT in water have decreased since 1991. These reductions are attributed to erosion-controlling best management practices (BMPs).

Another study conducted by the USGS between May 1999 and January 2000 in the surface waters of Yakima Basin detected 25 pesticide compounds (Ebbert and Embry 2001). 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.

Central Columbia Plateau: NAWQA Analysis

The Central Columbia Plateau is a prominent apple growing region. 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 et al. 1998). 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).

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) reported that 50 pesticides and pesticide degradates of the 86 were detected in streams. Atrazine, simazine, metolachlor, deethylatrazine, diuron, and diazinon were detected in more than one-half of stream samples (Wentz et al. 1998). The highest pesticide concentrations generally occurred in streams draining predominately agricultural land. Forty-nine pesticides were detected in streams draining predominantly agricultural land. About 25 pesticides were detected in streams draining mostly urban areas.

Lower Clackamas River Basin: NAWQA Analysis

Carpenter *et al.* (2008) summarized four different studies that monitored pesticide levels in the lower Clackamas River from 2000 to 2005. Water samples were collected from sites in the lower mainstem Clackamas River, its tributaries, and in pre- and post-treatment drinking-water. In all, 63 pesticide compounds (33 herbicides, 15 insecticides, 6 fungicides, and 9 degradates) were detected in samples collected during storm and nonstorm conditions. 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 about one- half of samples. The a.i. in common household herbicides Roundup (glyphosate) and Cross bow (triclopyr and 2,4-D) were frequently detected together.

Upper 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 et al. 1998). This basin does not overlap with any of the 28 ESU/DPSs, though it does feed into the migratory corridor of all Snake River species, and eventually into the Columbia River. In basin wide stream sampling in May and June 1994, Eptam, atrazine (and desethylatrazine), metolachlor, and alachlor were the most commonly detected pesticides. These compounds accounted for 75% of all detections. Seventeen different pesticides were detected downstream from American Falls Reservoir.

Hood River Basin

The Hood River Basin ranks fourth in the state of Oregon in total agricultural pesticide usage (Jenkins et al. 2004). 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 60).

Table 60. Summarized detection information from (Carpenter et al. 2008). Note that percentages aren't comparable because results were pooled from multiple sources.

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	Organophosphate	22,294
Metam-Sodium	Carbamate	17,114
Phosmet	Organophosphate	15,919
Chlorpyrifos	Organophosphate	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 et al. 2004). 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 et al. 2004).

Discharges from sewage treatment plants, paper manufacturing, and chemical and metal production represent the top three permitted sources of contaminants within the lower basin according to discharge volumes and concentrations (Rosetta and Borys 1996). Rosetta and Borys (1996) 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) 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). Construction of the Grand Coulee Dam blocked 1,000 miles (1,609 km) of habitat from migrating salmon and steelhead (Wydoski and Whitney 1979). Similarly, over one third (2,000 km) of coho salmon habitat is no longer accessible (Good et al. 2005). 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 et al. 2007). 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 et al. 1998). In the most recent NAWQA survey, a total of 16 sites were evaluated - all of which showed signs of degradation (Williamson et al. 1998). 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 et al. 1998).

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).

In 2000, 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 (NMFS 2008d).

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 et al. 2005, Anderson et al. 2007). 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 *in* (Hinck 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 (Figure 58). 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 et al. 2007).

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). 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* (NMFS 2008d)]; 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 et al. 2007), *Appendix 1 in* (NMFS 2008d)]. 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 et al. 2007).

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) 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.

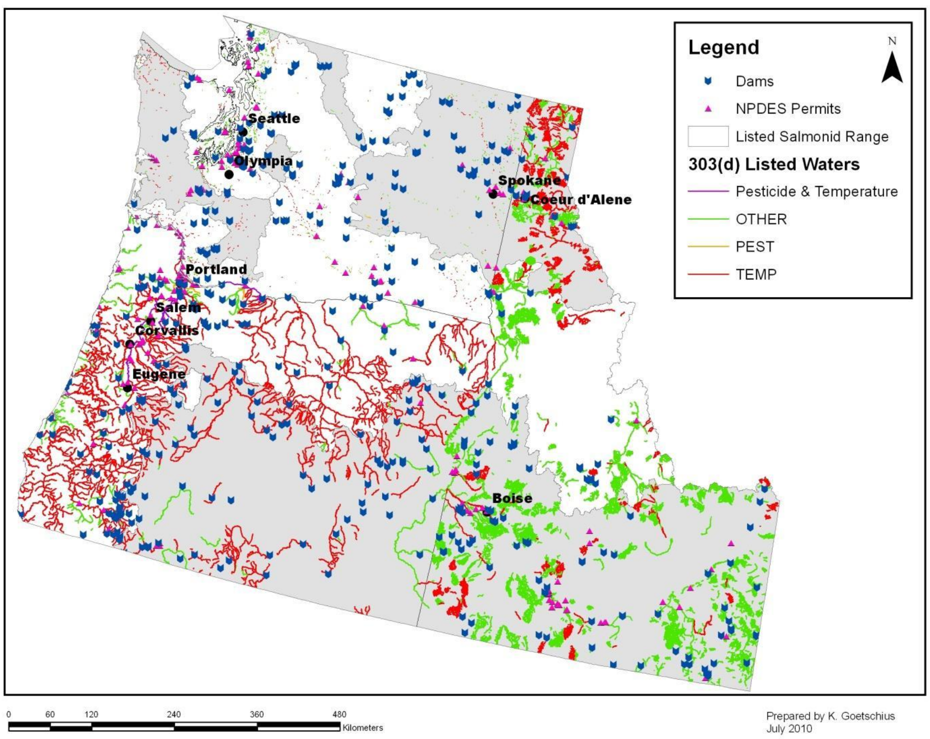

Figure 58. Pacific Northwest 303(d) waters, dams, and NPDES permit sites.

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). More recent estimates suggest that almost half of the total number of smolts produced in the basin come from hatcheries (Beechie et al. 2005).

The impact of artificial propagation on the total production of Pacific salmon and steelhead has been extensive (Hard et al. 1992). Hatchery practices, among other factors, are a contributing factor to the 90% reduction in natural coho salmon runs in the lower Columbia River over the past 30 years (Flagg et al. 1995). Past hatchery and stocking practices have resulted in the transplantation of salmon and steelhead from 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, Hard et al. 1992, Riggs 1990, Reisenbichler 1997), disease transfer, increased competition for food, habitat, or mates, increased predation, altered migration, and the displacement of natural fish (Steward and Bjornn 1990, Hard et al. 1992, Fresh 1997). 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 et al. 1992). 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 et al. 1992).

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 (NMFS 2008d). Oregon and Washington initiated a comprehensive program of hatchery and associated harvest reforms (Washington Department of Fish and Wildlife (WDFW) 2005, ODFW 2007). 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 these 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 et al. 2005).

During the mid-1800s, an estimated 10 to 16 million adult salmon of all species entered the Columbia River each year. Large 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 et al. 2005). The largest known harvest of Chinook salmon occurred in 1883 when Columbia River canneries processed 43 million lbs of salmon (Lichatowich 1999). 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 et al. 2005).

Harvested and spawning adults reached 2.8 million in the early 2000s, of which almost half are hatchery produced (Beechie et al. 2005). 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 et al. 2005). Recreational catch in both ocean and in-river fisheries varies from 140,000 to 150,000 individuals (Beechie et al. 2005).

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* (NMFS 2008d)].

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). 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 (NMFS 2008d).

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 et al. 2004). 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 et al. 2004).

8.7.3 Pacific Northwest: 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 and Whitney 1979, Kruckeberg 1991). Important commercial fishes include the five Pacific salmon and several rockfish species. A number of introduced species occur within the region, including brown and brook trout, 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 and McClure 2007). 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 et al. 2006). 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 et al. 2006).

An indication of this sensitivity occurs in Pacific herring, one of Puget Sound's keystone forage fish species (Collier et al. 2006). 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 et al. 2006). 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 et al. 2006). 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 and McClure 2007). 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). 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 and McClure 2007). Projected land cover changes indicate that trends are likely to continue over the next several decades with population changes (Ruckelshaus and McClure 2007). 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), 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 61). 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 and Meyer 2001). The combined effect of increased concentrations of ions in streams is the elevated conductivity observed in most urban streams.

Table 61. 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 et al. 2005). 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 et al. 1996). O'Neill *et al.* (2006) 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) 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). 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) 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 et al. 2005). 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 et al. 2004). Lake Washington, located within a highly urban area, has 15 non-native species identified (Ajawani 1956).

PAH compounds also have distinct and specific effects on fish at early life history stages (Incardona et al. 2004). PAHs tend to adsorb to organic or inorganic matter in sediments, where they can be trapped in long-term reservoirs (Johnson et al. 2002). 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 et al. 1992, Varanasi et al. 1989, Meador et al. 1995).

PAHs and their metabolites in invertebrate prey can be passed on to consuming fish species, PAHs are metabolized extensively in vertebrates, including fishes (Johnson et al. 2002). 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 et al. 2002).

Habitat Modification

Much of the estuarine wetlands in Puget Sound have been heavily modified, primarily from agricultural land conversion and urban development (NRC 1996). Although most estuarine wetland losses result from conversions to agricultural land by ditching, draining, or diking, these wetlands 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 et al. 1980). Tidal wetlands in Puget Sound amount to roughly 18% of their historical extent (Collins and Sheikh 2005). Coastal marshes close to seaports and population centers have been especially vulnerable to conversion with losses of 50 - 90%. By 1980, an estimated 27,180 acres of intertidal or shore wetlands had been lost at eleven deltas in Puget Sound (Bortleson et al. 1980). More recently, tidal wetlands in Puget Sound amount to about 17 - 19% of their historical extent (Collins and Sheikh 2005). Coastal marshes close to seaports and population centers have been especially vulnerable to conversion with losses of 50 - 90% common for individual estuaries. 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 et al. 2004).

About 800 miles of Puget Sound's shorelines are hardened or dredged (PSAT 2004, Ruckelshaus and McClure 2007). 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 and McClure 2007). Levee development within the rivers and their deltas has isolated significant portions of former floodplain habitat that was historically used by salmon and trout during rising flood waters.

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 2008b)). 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 2008b).

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) 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 62.

Table 62. Pollutants of Concern in Puget Sound (PSAT 2005).

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 et al. (2000) 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 and Ebbert 2000). 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 and Ebbert 2000). 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 and Ebbert 2000). 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 and Ebbert 2000). Sampled urban streams showed the highest detection rate for the three insecticides: carbaryl, diazinon, and malathion. No insecticides were found in shallow ground water below urban residential land (Bortleson and Ebbert 2000).

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 and McClure 2007). Even so, as the population and industry has risen in the region a number of new and legacy pollutants are of concern.

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 et al. 1993, NMA 2007). Metal mining for all metals (zinc, copper, lead, silver, and gold) peaked between 1940 and 1970 (Palmisano et al. 1993). Today, construction sand and gravel, Portland cement, and crushed stone are the predominant materials mined. Where sand and gravel is mined from riverbeds (gravel bars and floodplains) it may result in changes in channel elevations and patterns, instream sediment loads, and 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 et al. 2005). 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 et al. 1995).

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 et al. 2005).

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 and McClure 2007), other basins like the Snohomish River are diked and have active drainage systems to drain water after high flows that top the dikes. Dams were also built on the Cedar, Nisqually, White, Elwha, Skokomish, Skagit, and several other rivers in the early 1900s to supply urban areas with water, prevent downstream flooding, allow for floodplain activities (like agriculture or development), and to power local timber mills (Ruckelshaus and McClure 2007).

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 and McClure 2007, Wunderlich et al. 1994). Estimates suggest that nearly 400,000 salmon could begin using the basin within 30 years after the dams are removed (PSAT 2007).

In 1990, only one-third of the water withdrawn in the Pacific Northwest was returned to the streams and lakes (NRC 1996). 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). 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).

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 these 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 et al. 2005). 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.

8.7.4 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 et al. 2004, Kagan et al. 1999, Carter and Resh 2005).

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 et al. 1993). Roughly 80% of the Oregon Coastal Range is forested as well (Gregory 2000). Approximately 92% of the Nehalem River basin is forested, with only 4% considered agricultural (Belitz et al. 2004). 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 and Resh 2005).

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). 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). 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 et al. 2005). Nicholas *et al.*

(2005) 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 63 summarizes the change in area of tidal wetlands for several Oregon estuaries (Good 2000).

Table 63. Change in total area (acres2) of tidal wetlands in Oregon (tidal marshes and swamps) due to filling and diking between 1870 and 1970 (Good 2000).

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 Salmon Recovery Division 2008). LWD along the shore offered both shelter from predators and a barrier to encroaching vegetation (NMFS Salmon Recovery Division 2008).

Aerial photograph analysis shows near-shore vegetation has increased significantly over the past 50 years (Ritchie 2005). 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 Salmon Recovery Division 2008)]. 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 et al. 1993, NMA 2007). Metal mining for all metals (*e.g.*, zinc, copper, lead, silver, and gold) peaked in Washington between 1940 and 1970 (Palmisano et al. 1993). Today, construction sand, 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 and Resh 2005). According to Palmisano *et al.* (1993) dams in the coastal streams of Washington permanently block only about 30 miles of salmon habitat (Figure 58). 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).

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 these 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.

8.8 Integration of Environmental Baseline Effects 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. Increases in temperature, carbon dioxide 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), and ocean acidification. 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 diflubenzuron, fenbutatin-oxide, and propargite 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.

9 Effects of the Proposed Action to Threatened and Endangered Pacific Salmonids

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

9.1 Exposure Analysis

In this section, we identify and evaluate potential exposure of salmonids to the stressors of the action (Figure 59). We begin by presenting general life history information of vulnerable life stages of Pacific salmon and steelhead, referred to as salmonids. Next, we discuss the physical and chemical properties of the active ingredients and their degradation products that influence exposure of listed species and designated critical habitat to these stressors of the action. We then evaluate co-occurrence of salmon habitat with the stressors of the action by comparing the distribution of sites authorized for use of the pesticide products to the distribution of each species and their designated critical habitat.

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

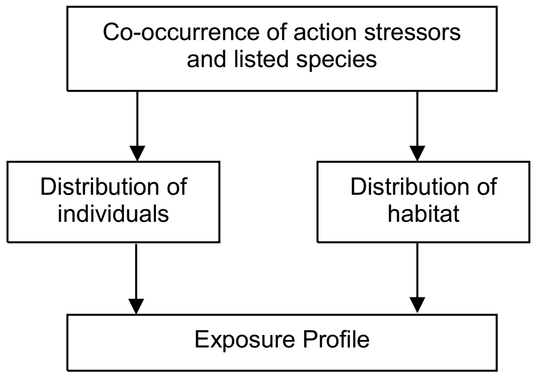

Figure 59. Exposure analysis

9.1.1 Threatened and Endangered Pacific Salmonids use of Aquatic Habitats

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

In this section we describe the habitats used by listed Pacific salmonids. General life history descriptions are provided below in Table 64. Additionally, temporal use of aquatic habitats for the 28 EUS/DPS is provided in Appendix five.

Table 64. 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 three to five years old) enter rivers (spring through fall, depending on run). Adults migrate and spawn in river reaches extending from above the tidewater inland hundreds of miles from the Pacific. Migrating adults typically follow the thalweg. Chinook salmon migrate and spawn in four distinct runs (spring, fall, summer, and winter). Chinook salmon are semelparous[1].	Generally spawn in the middle and upper reaches of main stem rivers and larger tributary streams.	The alevin life stage primarily resides just below the gravel surface until they approach or reach the fry stage. Immediately after leaving the gravel, fry distribute to habitats that provide refuge from fast currents and predators. Juveniles exhibit two general life history types: Ocean-type fish migrate to sea in their first year, usually within six months of hatching. Ocean-type juveniles may rear in the estuary for extended periods. Stream-type fish migrate to the sea in the spring of their second year.
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. Migrating adults typically follow the thalweg. Steelhead are iteroparous[2].	Usually spawn in fine gravel in a riffle above a pool.	The alevin life stage primarily resides just below the gravel surface until they approach or reach the fry stage. Immediately after leaving the gravel, swim-up fry usually inhabit shallow water along banks of stream or aquatic habitats on streams margins. Steelhead rear in a wide variety of freshwater habitats, generally for two to three years, but up to six or seven years is possible. They smolt and migrate to sea in the spring.

1 spawn only once
2 may spawn more than once

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

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

9.1.2 Chemical Exposure Pathways to Salmonids Habitats

Pesticides can contaminate salmonid habitats through direct application to control aquatic plants, mosquitoes, and other aquatic pests. Other common pathways of exposure in aquatic habitats include runoff, erosion, leaching, and spray drift from pesticide applications to terrestrial sites. The pesticide application method can influence the route of exposure. For example, spray-applied pesticides may result in off-target deposition of droplets at the time of application(Bird et al. 2002). The likelihood of spray drift to an aquatic habitat is determined by the application specifications, the proximity to the habitat, and meteorological conditions at the time of application.

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

9.1.3 Summary of Chemical Fate of the Three Active Ingredients

When the three a.i.s are present in the water column, exposure to salmonids can occur as chemicals contact and cross gill surfaces during respiration, or when they contact fish sensory systems (*i.e.,* olfactory sensory neurons). Exposure through these routes is most likely when the chemicals are in the dissolved phase. Other routes may contribute to overall exposure including incidental ingestion of the chemical sorbed to sediment or ingestion of the chemical in food items. Below we summarize chemical fate properties of the three a.i.s reported by EPA. Where discrepancies existed, we deferred to the most recent document.

9.1.3.1 Diflubenzuron

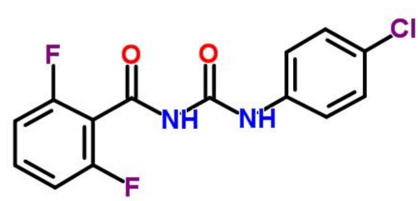

Figure 60. Chemical structure of diflubenzuron

Diflubenzuron (Figure 60) is a benzoylphenyl urea insecticide that acts as a stomach and contact poison to control pests (extoxnet.orst.edu). It is generally stable to abiotic degradation but it has relatively low persistence in soils and surface water due to rapid degradation by microbial processes (Table 65). Considering label information and environmental fate characteristics, relevant exposure pathways include spray drift from outdoor applications (e.g. to agricultural lands, rights-of-ways, forests, urban and residential areas, etc.) and discharge from treated rice fields. Although diflubenzuron has low persistence and is relatively immobile in soils, runoff is a main transport mechanism due to likely movement of eroded soil in runoff (EPA 2009c). Runoff may be of greater importance for the transport of the degradate 4-chlorophenylurea (CPU), which is more mobile in soil than diflubenzuron (EPA 2009c). The available data suggest that food chain transfer and long range deposition due to volatilization are less likely pathways of exposure to aquatic organisms.

Table 65. Environmental fate characteristics of diflubenzuron[1]

Parameter	Value
Water solubility	0.2 mg/L at 20 °C
Vapor pressure	9.00×10^{-10} mm Hg
Henry's law constant	1.87×10^{-09} atm m^3 mol^{-1}
Octanol/Water partition coefficient	Log K_{ow} = NR[2] 7.38 ± 0.35*
Hydrolysis (t½) pH 5, pH 7, & pH 9	187-stable, 117-158 d, 32-44 d
Aqueous photolysis (t½)	80 d
Soil photolysis (t½)	11.3 - 144 d
Aerobic soil metabolism (t½)	2 - 14 d
Anaerobic soil metabolism (t½)	2 - 14 d
Aerobic aquatic metabolism (t½)	3.7 - 26 d
Anaerobic aquatic metabolism (t½)	34 d
Soil partition coefficient	K_{oc} = 1938-6918 L/kg_{soil}
Fish Bioconcentration Factor (BCF)	34-200x (fillet) 78-360x (whole fish) 100-550x (viscera)

1- (EPA 2009c)
2- NR=Not reported
*- calculated (Virtual Computational Chemistry Laboratory, http://www.vcclab.org, 2005)

Several degradates of diflubenzuron have been identified in laboratory and field studies including CPU, 2,6-diflubenzoic acid (DFBA), 4-choroaniline (PCA), 2,6-diflubenzamide (DFBAM), and 2,6-difluorobenzene.

Table 66. Degradates of diflubenzuron[1]

Study	Degradate identified– percent of diflubenzuron applied
Hydrolysis	CPU- not quantified DFBA – not quantified
Aqueous photolysis	CPU- 8% DFBA- 4% DFBAM- 1% 2,6-difluorobenzene- not quantified
Soil photolysis	CPU- 12.9% DFBA- 3.0%
Aerobic soil metabolism	CPU- 37% DFBA- <10% DFBAM- <10% PCA- <10%
Anaerobic aquatic metabolism	CPU- 31% DFBA- 31% PCA- 0.4%
Aerobic aquatic metabolism	DFBA- 17% PCA- 48%
Terrestrial field dissipation	DFBA CPU
Aquatic field dissipation	CPU
Fish accumulation	DFBAM

1- (EPA 2009c)

9.1.3.2 Fenbutatin oxide

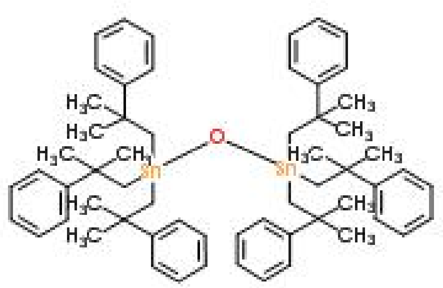

Figure 61. Chemical structure of fenbutatin oxide

Fenbutatin oxide (Figure 61) is a non-systemic organotin acaricide. Studies suggest that fenbutatin oxide is persistent and does not readily degrade in aquatic or terrestrial environments by either biotic or abiotic routes (EPA 2009a). There is no evidence of hydrolysis although slow degradation through aqueous photolysis is expected to occur in clear shallow waters based on laboratory tests (Table 67). In terrestrial field studies dissipation half-lives were estimated at 3.75 years and multiple applications resulted in accumulation of fenbutatin oxide in the soil (EPA 2009a). The persistence and high octanol water partition coefficient suggest the consumption of fenbutatin oxide-contaminated aquatic prey may be an important pathway of exposure in salmon. Organotins are well known for their ability to accumulate in food chains (Horiguchi et al. 2012, Harino et al. 2008, Ohji et al. 2007). EPA concluded that studies to determine bioconcentration factors in fish where not acceptable because the various tissue fractions did not reach equilibrium during the 28-d exposure period (EPA 2009a). Therefore, actual bioconcentration factors may be significantly higher than those reported in Table 67 below.

Drift is a likely pathway of exposure given that fenbutatin oxide pesticides may be spray applied in close proximity to salmonid habitats. Fenbutatin oxide is relatively immobile in soil but may be transported to aquatic habitats in runoff through erosion and sediment bound residues. Once in the aquatic system, fenbutatin oxide will partition from water and sorb to dissolved and particulate organic matter including biota.

Table 67. Environmental fate characteristics of fenbutatin oxide[1]

Parameter	Value
Water solubility	0.0127 mg/L at 20 °C
Vapor pressure	1.8×10^{-11} mm Hg
Henry's law constant	NR[2]
Octanol/Water partition coefficient	Log K_{ow} = 5.14
Hydrolysis ($t_{½}$) pH 5, pH 7, & pH 9	Stable, stable, stable
Aqueous photolysis ($t_{½}$)	110 d
Soil photolysis ($t_{½}$)	128 d
Aerobic soil metabolism ($t_{½}$)	944 - 2951 d
Anaerobic soil metabolism ($t_{½}$)	60 d
Aerobic aquatic metabolism ($t_{½}$)	NR
Anaerobic aquatic metabolism ($t_{½}$)	NR
Soil partition coefficient	K_{oc} = 74,450 – 320,677 L/kg_{soil}
Fish Bioconcentration Factor (BCF)	340-500x (fillet) 490-730x (whole fish) 1100-1600x (viscera)

1- (EPA 2009a)
2- NR=Not reported

One major degradate was identified in the aqueous photolysis study: 1,3-dihydroxy-1,1,3,3-tetrakis (2-methyl-2-phenyl propyl)-distannoxane. Two degradates were identified in field dissipation trials: dihydroxybis(2-methyl-2-phenylpropyl)stannane and 2-methyl-2- phenylpropyl stanonic acid (EPA 2009a). We found no information in the BE or from other sources on anticipated exposure values for fenbutatin oxide's degradates.

9.1.3.3 Propargite

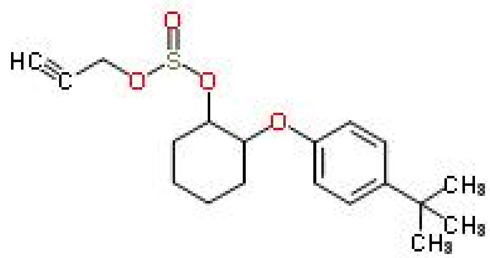

Figure 62. Chemical structure of propargite

Propargite (Figure 62) is an organosulfite acaricide with contact activity to larval and adult mites. Terrestrial field studies suggest propargite is moderately persistent with half-lives of 67 - 99 d (EPA 2008). Most laboratory tests evaluating degradation also suggest moderate persistence. However, relatively rapid degradation through hydrolysis occurs under alkaline conditions (Table 68). Drift is a likely transport pathway for propargite because it is applied as a liquid spray by ground and aerial application methods. Propargite is only slightly mobile in soil although transport of propargite to aquatic habitats can occur through erosion and runoff of soil particles to which propargite is adsorbed. Transport through volatilization is unlikely given propargite's vapor pressure. The high octanol/water partition coefficient suggests likely partitioning to organic compartments and the potential for dietary exposure. Bioconcentration studies indicate relatively high concentrations are achieved in fish tissues, although propargite is readily metabolized and excreted which limits the potential for biomagnification (EPA 2008).

Table 68. Environmental fate characteristics of propargite[1]

Parameter	Value
Water solubility	0.63 mg/L at 20 °C
Vapor pressure	4.49×10^{-8} mm Hg
Henry's law constant	3.28×10^{-8} atm m^3 mol^{-1}
Octanol/Water partition coefficient	Log $K_{ow} = 5.8$
Hydrolysis (t½) pH 5, pH 7, & pH 9	120 d, 75 d, 2.2 d
Aqueous photolysis (t½)	134 -140 d
Soil photolysis (t½)	63 - 113 d
Aerobic soil metabolism (t½)	168 d
Anaerobic soil metabolism (t½)	64.4 d
Aerobic aquatic metabolism (t½)	38 d
Anaerobic aquatic metabolism (t½)	47 d
Soil partition coefficient	K_{oc} = 2,963 – 57,966 L/kg_{soil} K_d = 60 - 218 L/kg_{soil}
Fish Bioconcentration Factor (BCF)	260x (fillet) 775x (whole fish) 1550x (viscera)

1- (EPA 2008)

The molecular structure of propargite may be modified by biotic (*e.g.,* microbial metabolism) or abiotic (*e.g.,* photolysis and hydrolysis) processes. The products of these processes may have different toxicities, environmental fate characteristics, and risks compared to the parent pesticide. EPA indicated that the main transformation products of propargite are bis-[2,-(4-(1,1-dimethyl-ethyl)-phenoxy)cyclohexyl] sulfite (BGES); 2,2-dimethyl-2-(4'-(2-hydroxy-cyclohexoxy)phenyl)ethanol (OMT-G); p-tertiarybutylphenol (PTBP); propargite glycol ether-2-[4-(1,1-dimethylethyl)phenoxy]-cyclohexane-1-ol (TBPC); and 2-[4-(2-hydroxycyclohexoxy)phenyl]-2,2-dimethyl acetic acid (TBPC-acid), and a sulfate derivative of TBPC (EPA 2008).

9.1.4 Exposure of salmonid habitats to the stressors of the action

9.1.4.1 Co-occurrence associated with pesticide uses

We evaluated co-occurrence of listed salmonids with the stressors of the action by comparing the spatial and temporal distribution of salmonids with labeled use of the three a.i.s. First, product labels were evaluated to determine which land use categories the three active ingredients may be applied according to product labeling (Table 69).

Table 69. Use sites for pesticides based on product labeling and corresponding land use categories of the National Land Cover Database

ACTIVE INGREDIENT	LAND USE CATEGORY				
	Agricultural	Undeveloped	Developed	Water	Rights of Way[1]
	Use Site – Land Use Sub Category				
Diflubenzuron	Crop Non-crop[2]	All[3]	All[4]	No use permitted	All[5]
Fenbutatin oxide	Crop	No use permitted	All[6]	No use permitted	No use permitted
Propargite	Crop	No use permitted	All[7]	No use permitted	No use permitted

1- The National Land Cover Database does not include a data layer for rights of way. Rights of way occur within the distribution of all listed Pacific salmonid ESU/DPSs.

Use specified on pesticide product label(s):

2- Livestock, aquaculture, livestock/poultry holding areas, field borders, fencerows, farmsteads, ditch banks, pastures, CRP lands

3- Grasslands, rangelands, public and private forests

4- Residential and municipal shade tree areas and landscape plantings, recreational areas, campgrounds, golf courses, parks, parkways, shelterbelts, standing water around home, subterranean and above ground termite bait stations

5- Rights of way and easements

6- Use on developed lands is limited to greenhouse and outdoor ornamentals at nurseries. UPI has indicated they will remove established ornamentals from labels which removes potential use from a broad range of other developed sites (urban, residential, industrial, etc.).

7- Use on developed lands is limited to ornamentals at nurseries.

Next, we determine the areal extent of label-authorized application sites within the distribution of each listed Pacific salmonid.

Table 70 indicates that the three a.i.s may be applied to lands within the freshwater distribution of all listed Pacific salmonids based on product labeling. This analysis was accomplished using GIS overlays containing land use classifications and salmon distributions[16] to determine the extent of overlap. The aerial coverage of the various land use categories and sub categories are reported in the *Environmental Baseline*.

Classification of land use categories in the National Land Cover Database (NLCD) are based on image data collected from orbiting Landsat satellites and have a spatial resolution of 30 m. The 2006 NLCD data layers used in this assessment are the most current data layers available and provide a consistent data for evaluating the distribution of pesticide use sites across the four states. The results of this data set have improved accuracy over previous layers. Changes were made to include improvements from the Multi-Resolution Land Characteristics consortium, NOAA Coastal-Chana Analysis Program, and two prototype zones that were published early in project evolution. Smaller scale land cover refinements were made throughout NOAA stewardship areas (Fry et al. 2011). However, misclassifications of land use category still exist that may affect the accuracy of our co-occurrence evaluation. For example, information from the Washington State Department of Agriculture demonstrates the misclassification of cultivated cropland as hay/pastures occurs within the distribution of listed salmonids (Figure 63). Future refinements to this NLCD database are planned to address this misclassification and others (Fry et al. 2011). In the interim, we recognize that our analysis may underestimate the co-occurrence of the listed salmonids with cultivated crop land and overestimates the co-occurrence with hay/pasture in some areas.

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

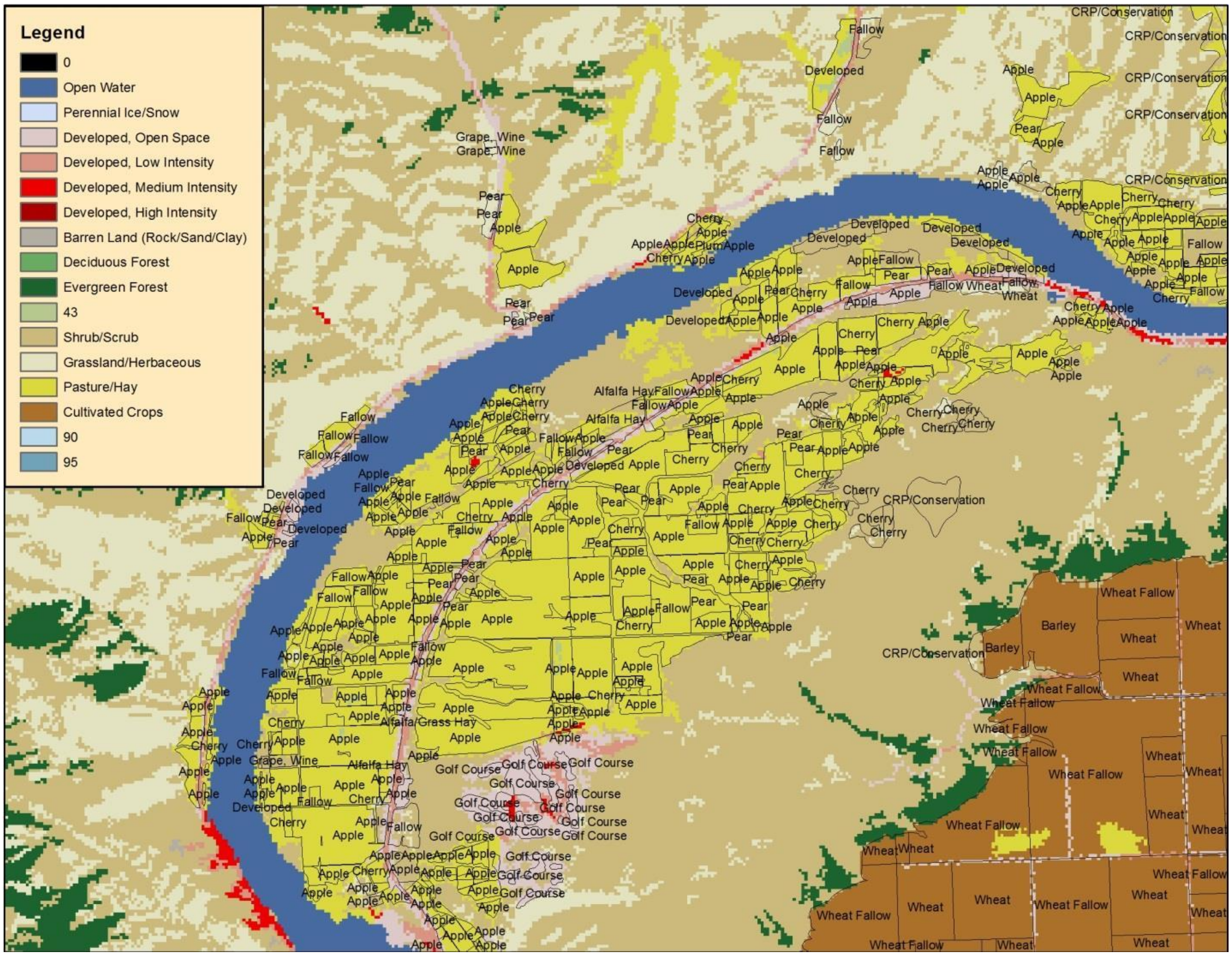

Figure 63. Comparison of NLCD land use categories with data collected by WSDA indicate cultivated crops are frequently misclassified as hay/pasture.

Table 70 does not include authorized use of diflubenzuron to rights-of-way which are present within the distributions of all listed salmonids. While fenbutatin oxide and propargite may be applied to developed lands, pesticide use to these sites is exclusively for indoor greenhouse and outdoor nurseries. Therefore, we recognize that potential treatment sites are only expected for a very small portion of the Developed Land Use Category. Most of the ESUs have crop land within their distribution suggesting potential exposure to authorized uses of the three a.i.s on crops. The only exceptions include the Ozette Lake sockeye and the Snake River sockeye which have no crop land within their ESU boundaries. We do not expect agricultural uses of the three a.i.s will result in any significant exposure to Ozette Lake sockeye. However, co-occurrence of the Snake River sockeye with applications of the three a.i.s to crops is expected because this species migrates through several agricultural watersheds during passage to and from the Pacific Ocean.

Table 70. Occurrence of pesticide use sites within the salmonids' freshwater spawning and rearing distribution

Species	ESU	Land Use Category	% of Range[1]	Are there labeled uses for this category?		
				Diflubenzuron	Fenbutatin oxide	Propargite
Chinook	Puget Sound	Ag –Crop	1	Yes	Yes	Yes
		Ag –noncrop	3	Yes	No	No
		Developed-All	15	Yes	Yes	Yes
		Undeveloped-All	63	Yes	No	No
	Lower Columbia River	Ag –Crop	2	Yes	Yes	Yes
		Ag –noncrop	4	Yes	No	No
		Developed-All	12	Yes	Yes	Yes
		Undeveloped-All	79	Yes	No	No
	Upper Columbia River Spring Run	Ag –Crop	3	Yes	Yes	Yes
		Ag –noncrop	2	Yes	No	No
		Developed-All	5	Yes	Yes	Yes
		Undeveloped-All	89	Yes	No	No
	Snake River Fall Run	Ag –Crop	19	Yes	Yes	Yes
		Ag –noncrop	<1	Yes	No	No
		Developed-All	2	Yes	Yes	Yes
		Undeveloped-All	78	Yes	No	No
	Snake River Spring/ Summer Run	Ag –Crop	7	Yes	Yes	Yes
		Ag –noncrop	1	Yes	No	No
		Developed-All	2	Yes	Yes	Yes
		Undeveloped-All	90	Yes	No	No
	Upper Willamette River	Ag –Crop	10	Yes	Yes	Yes
		Ag –noncrop	16	Yes	No	No
		Developed-All	9	Yes	Yes	Yes
		Undeveloped-All	64	Yes	No	No
	California Coastal	Ag –Crop	1	Yes	Yes	Yes
		Ag –noncrop	1	Yes	No	No
		Developed-All	5	Yes	Yes	Yes
		Undeveloped-All	92	Yes	No	No
	Central Valley Spring Run	Ag –Crop	21	Yes	Yes	Yes
		Ag –noncrop	3	Yes	No	No
		Developed-All	11	Yes	Yes	Yes
		Undeveloped-All	63	Yes	No	No
	Sacramento River Winter Run	Ag –Crop	21	Yes	Yes	Yes
		Ag –noncrop	3	Yes	No	No
		Developed-All	11	Yes	Yes	Yes
		Undeveloped-All	63	Yes	No	No

Species	ESU	Land Use Category	% of Range[1]	Are there labeled uses for this category?		
				Diflubenzuron	Fenbutatin oxide	Propargite
Chum	Hood Canal Summer Run	Ag –Crop	<1	Yes	Yes	Yes
		Ag -noncrop	1	Yes	No	No
		Developed-All	9	Yes	Yes	Yes
		Undeveloped-All	75	Yes	No	No
	Columbia River	Ag –Crop	2	Yes	Yes	Yes
		Ag -noncrop	4	Yes	No	No
		Developed-All	15	Yes	Yes	Yes
		Undeveloped-All	73	Yes	No	No
Coho	Lower Columbia River	Ag –Crop	2	Yes	Yes	Yes
		Ag -noncrop	4	Yes	No	No
		Developed-All	12	Yes	Yes	Yes
		Undeveloped-All	79	Yes	No	No
	Oregon Coast	Ag –Crop	<1	Yes	Yes	Yes
		Ag -noncrop	3	Yes	No	No
		Developed-All	6	Yes	Yes	Yes
		Undeveloped-All	90	Yes	No	No
	Southern Oregon and Northern California Coast	Ag –Crop	1	Yes	Yes	Yes
		Ag -noncrop	2	Yes	No	No
		Developed-All	4	Yes	Yes	Yes
		Undeveloped-All	93	Yes	No	No
	Central California Coast	Ag –Crop	2	Yes	Yes	Yes
		Ag -noncrop	<1	Yes	No	No
		Developed-All	9	Yes	Yes	Yes
		Undeveloped-All	87	Yes	No	No
Sockeye	Ozette Lake	Ag –Crop	0	Yes	Yes	Yes
		Ag -noncrop	0	Yes	No	No
		Developed-All	<1	Yes	Yes	Yes
		Undeveloped-All	86	Yes	No	No
	Snake River	Ag –Crop	0	Yes	Yes	Yes
		Ag -noncrop	1	Yes	No	No
		Developed-All	1	Yes	Yes	Yes
		Undeveloped-All	95	Yes	No	No

Species	ESU	Land Use Category	% of Range[1]	Are there labeled uses for this category?		
				Diflubenzuron	Fenbutatin oxide	Propargite
Steelhead	Puget Sound	Ag –Crop	1	Yes	Yes	Yes
		Ag -noncrop	3	Yes	No	No
		Developed-All	15	Yes	Yes	Yes
		Undeveloped-All	63	Yes	No	No
	Lower Columbia River	Ag –Crop	2	Yes	Yes	Yes
		Ag -noncrop	1	Yes	No	No
		Developed-All	12	Yes	Yes	Yes
		Undeveloped-All	79	Yes	No	No
	Upper Willamette River	Ag –Crop	14	Yes	Yes	Yes
		Ag -noncrop	20	Yes	No	No
		Developed-All	10	Yes	Yes	Yes
		Undeveloped-All	56	Yes	No	No
	Middle Columbia River	Ag –Crop	17	Yes	Yes	Yes
		Ag -noncrop	1	Yes	No	No
		Developed-All	3	Yes	Yes	Yes
		Undeveloped-All	77	Yes	No	No
	Upper Columbia River	Ag –Crop	13	Yes	Yes	Yes
		Ag -noncrop	2	Yes	No	No
		Developed-All	5	Yes	Yes	Yes
		Undeveloped-All	79	Yes	No	No
	Snake River	Ag –Crop	8	Yes	Yes	Yes
		Ag -noncrop	1	Yes	No	No
		Developed-All	2	Yes	Yes	Yes
		Undeveloped-All	89	Yes	No	No
	Northern California	Ag –Crop	<1	Yes	Yes	Yes
		Ag -noncrop	1	Yes	No	No
		Developed-All	4	Yes	Yes	Yes
		Undeveloped-All	94	Yes	No	No
	Central California Coast	Ag –Crop	3	Yes	Yes	Yes
		Ag -noncrop	<1	Yes	No	No
		Developed-All	22	Yes	Yes	Yes
		Undeveloped-All	66	Yes	No	No
	California Central	Ag –Crop	26	Yes	Yes	Yes
		Ag -noncrop	5	Yes	No	No

Species	ESU	Land Use Category	% of Range[1]	Are there labeled uses for this category?		
				Diflubenzuron	Fenbutatin oxide	Propargite
	Valley	Developed-All	11	Yes	Yes	Yes
		Undeveloped-All	57	Yes	No	No
	South-Central California Coast	Ag –Crop	7	Yes	Yes	Yes
		Ag -noncrop	1	Yes	No	No
		Developed-All	10	Yes	Yes	Yes
		Undeveloped-All	82	Yes	No	No
	Southern California	Ag –Crop	4	Yes	Yes	Yes
		Ag -noncrop	1	Yes	No	No
		Developed-All	35	Yes	Yes	Yes
		Undeveloped-All	60	Yes	No	No

1- Spatial coverage within the species' freshwater distribution. These statistics may not account for land use categories in watersheds that are connected to the migration corridor (e.g. the Columbia River watersheds along the migration corridor of Snake River Sockeye).

We also evaluated the temporal aspect of pesticide use and overlap with species presence. Listed Pacific salmonids are most likely to be exposed to pesticides when they are in close proximity to pesticide application sites (e.g. during their freshwater residence in streams and rivers, estuaries, and nearshore marine habitats). Appendix five indicates that all of the listed species are present in freshwater habitats at some life stage, and most are present year round. Current product labels do not restrict when the three a.i.s can be applied. Although frequency of use is expected to vary seasonally with pest pressure, historical use information from California DPR's pesticide use database indicates all three a.i.s are applied throughout the year (http://www.cdpr.ca.gov/docs/pur/purmain.htm). Given the spatial and temporal overlap, we expect some individuals within each of the listed ESUs/DPSs will be exposed to the three a.i.s over the 15-year duration of the action. Exposure will be variable among individuals and populations. Temporal and spatial relationships influencing the likely extent of exposure for each species are further discussed in the *Integration and Synthesis* section below.

9.1.5 Modeling: Estimates of exposure to the a.i.s

9.1.5.1 EPA exposure estimates

The EPA salmonid BEs report estimates of aquatic concentrations of the three active ingredients derived with the PRZM-EXAMS model. However, more recent assessments by EPA provide estimates of the three active ingredients that are more comprehensive and more representative of current label authorizations. The results of this modeling effort are provided below. Table 71 summarizes estimates derived by EPA to evaluate diflubenzuron uses (EPA 2009c). Model inputs frequently did not match current label restrictions for either the number of applications or the maximum application rate. Thus some of EPA's previous estimates may be of limited relevance in evaluating exposure to salmonids. For each land use category (e.g. Agricultural crops) we omitted older, EPA-derived, exposure estimates that used higher application rates than current labels allow. In this manner, we are ensuring that exposure estimates are based on current pesticide labels.

Table 71. EPA estimated concentrations of diflubenzuron in surface water associated with registered uses of diflubenzuron products

Use Scenario	Application Method	Rate (lbs a.i./A)	Peak (µg/L)	21-d average (µg/L)	60-d average (µg/L)	Model
Manure[A]	Ground	1.31[3]	7.29	5.63	3.65	GENEEC
Manure[A]*	Ground	1.31[3]	2.27	1.34	0.78	PRZM-EXAMS
Manure[A]**	Ground	1.31[3]	0.62	0.30	0.16	PRZM-EXAMS
Beech nut[B]	Ground	0.0408[1]	0.11	0.06	0.03	PRZM-EXAMS
Beech nut[B]	Aerial	0.0408[1]	0.21	0.13	0.08	PRZM-EXAMS
Brassica[B]	Ground	0.0313[1]	0.07	0.04	0.02	PRZM-EXAMS
Citrus[B]	Ground	0.3125[3]	0.24	0.10	0.05	PRZM-EXAMS
Citrus[B]	Aerial	0.125[1]	0.50	0.30	0.18	PRZM-EXAMS
Cole crop[B]	Ground	0.25[2]	0.36	0.20	0.14	PRZM-EXAMS
Cotton[B]	Ground	0.3125[2]	0.15	0.11	0.06	PRZM-EXAMS
Cotton[B]	Aerial	0.3125[2]	0.82	0.47	0.30	PRZM-EXAMS
Forests[ABCD]	Ground	0.22048[1]	0.58	0.31	0.19	PRZM-EXAMS
Forests[ABCD]	Aerial	0.25[3]	0.80	0.45	0.26	PRZM-EXAMS
Stone fruits[B]	Ground	0.3125[1]	0.26	0.14	0.08	PRZM-EXAMS
Stone fruits[B]	Air blast	0.3125[1]	0.66	0.43	0.26	PRZM-EXAMS
Stone fruits[B]	Aerial	0.3125[1]	1.36	0.86	0.54	PRZM-EXAMS
Nuts[B]	Ground	0.3125[2]	0.90	0.46	0.25	PRZM-EXAMS
Nuts[B]	Air blast	0.3125[2]	1.26	0.70	0.42	PRZM-EXAMS
Nuts[B]	Aerial	0.3125[2]	0.85	0.49	0.28	PRZM-EXAMS
Grains[B]	Ground	0.25[2]	1.34	0.78	0.56	PRZM-EXAMS

Use Scenario	Application Method	Rate (lbs a.i./A)	Peak (µg/L)	21-d average (µg/L)	60-d average (µg/L)	Model
Grains[B]	Aerial	0.125[2]	0.93	0.52	0.41	PRZM-EXAMS
Nursery[D]	Ground	0.25[3]	0.20	0.11	0.08	PRZM-EXAMS
Nursery[D]	Aerial	0.25[3]	0.65	0.38	0.30	PRZM-EXAMS
Pasture[A]	Ground	0.25[2]	0.12	0.08	0.04	PRZM-EXAMS
Pasture[A]	Air blast	0.25[2]	0.29	0.17	0.09	PRZM-EXAMS
Pasture[A]	Aerial	0.25[2]	0.55	0.25	0.13	PRZM-EXAMS
Pistachio[B]	Ground	0.75[2]	0.15	0.08	0.06	PRZM-EXAMS
Residential[D]	Ground	0.25[3]	0.08	0.05	0.03	PRZM-EXAMS
Residential[D]	Aerial	0.25[3]	0.40	0.30	0.25	PRZM-EXAMS
Rights-of-way[ABCD]	Ground	0.25[3]	5.55	3.04	1.61	PRZM-EXAMS
Rights-of-way[ABCD]	Air blast	0.25[3]	8.13	4.39	2.32	PRZM-EXAMS
Rights-of-way[ABCD]	Aerial	0.25[3]	5.59	3.07	1.63	PRZM-EXAMS
Row crops[B]	Ground	0.3125[2]	0.02	0.01	0.01	PRZM-EXAMS
Row crops[B]	Aerial	0.3125[2]	1.15	0.77	0.58	PRZM-EXAMS
Urban[D]	Aerial	0.25[3]	34.13	21.98	15.28	PRZM-EXAMS
Squash[B]	Ground	0.25[4]	1.15	0.65	0.38	PRZM-EXAMS
Squash[B]	Aerial	0.25[4]	1.87	1.18	0.79	PRZM-EXAMS
Turf[D]	Ground	0.25[2]	0.01	0.01	0.00	PRZM-EXAMS
Turf[D]	Air blast	0.25[2]	0.21	0.13	0.08	PRZM-EXAMS
Turf[D]	Aerial	0.25[2]	0.61	0.37	0.20	PRZM-EXAMS
Turf[D]	Ground	0.0313[3]	0.02	0.01	0.01	PRZM-EXAMS
Turf[D]	Air blast	0.0313[3]	0.07	0.04	0.02	PRZM-EXAMS
Turf[D]	Aerial	0.0313[3]	0.13	0.08	0.05	PRZM-EXAMS
Turnip[B]	Ground	0.25[2]	0.72	0.35	0.19	PRZM-EXAMS

* EPA estimate 4-19-2013

** Chemtura estimate 4-10-2013

*** GENEEC is a meta-model of the PRZM-EXAMS used by EPA as a tier 1 screen. It incorporates assumptions consistent with a site vulnerable to runoff. The size of the treated area and aquatic habitat (farm pond) are the same as described for PRZM-EXAMS.

A. Land Use Category: Agriculture-noncrop. Maximum single application use rate: 8.2 lbs a.i./A indoor animal holding areas. The application rate of 1.3 lbs/A is an estimate based on the concentration of diflubenzuron in manure that is applied to agricultural fields (Appendix F, (EPA 2009c))
B. Land Use Category: Agriculture-crop. Maximum single application rate of 0.75 lbs a.i./A pears.
C. Land Use Category: Undeveloped. Maximum single application rate of 0.25 lbs a.i./A public and private forests, aquatic habitats in California for midge control, other uses
D. Land Use Category: Developed. Maximum single application rate of 0.25 lbs a.i./A landscaping, parks, others

Differences from current label:

1- Less than maximum single application rate currently allowed for site and Land Use Category
2- More than maximum single application rate currently allowed for site, ≤ rate allowed for Land Use Category
3- Equivalent to maximum single application rate currently allowed for site
4- Application to this use site is not authorized but use rate is consistent with other sites with Land Use Category

In 2007 EPA conducted an assessment for the section 3 registration of fenbutatin oxide on pistachios (EPA 2007a). This assessment provides model estimates for aquatic concentrations associated with all fenbutatin oxide uses registered at that time (Table 72).

Table 72. EPA estimated concentrations of fenbutatin oxide in surface water associated with registered uses of fenbutatin oxide products

Use scenario	Application method	Rate (lbs a.i./a)	Peak (µg/L)	21-d average (µg/L)	60-d average (µg/L)	90-d average (µg/L)	Model
Almond[A]	Aerial	1[1]	6.38	4.08	3.87	3.83	PRZM-EXAMS
Almond[A]	Ground	1[1]	3.3	1.73	1.66	1.65	PRZM-EXAMS
Fruit (stone) [A]	NS	1[3]	5.42	3.11	2.9	2.87	PRZM-EXAMS
Fruit (stone) [A]	NS	0.75[1]	4. 06	2.33	2.18	2.15	PRZM-EXAMS
Grape[A]	NS	1[1]	34.11	25.7	25.46	25.4	PRZM-EXAMS
Citrus (CA) [A]	NS	1.5[1]	8.13	4.46	4.35	4.3	PRZM-EXAMS
Citrus (FL) [A]	NS	1[1]	12.56	9.72	9.38	9.4	PRZM-EXAMS
Citrus (FL) [A]	NS	1[1]	32.21	24.5	23.86	23.77	PRZM-EXAMS
Peach[A]	NS	0.75[1]	8.45	6.32	6.19	6.18	PRZM-EXAMS
Cherry[A]	NS	1.125[1]	28.34	19.19	18.95	18.89	PRZM-EXAMS
Pecan[A]	NS	1[1]	20.02	16.70	16.51	16.49	PRZM-EXAMS
Strawberry-CA[A]	NS	1.5[3]	55.2	45.68	45.04	44.98	PRZM-EXAMS
Strawberry[A]	NS	1[3]	24.52	20.09	20.01	19.99	PRZM-EXAMS
Tomato[A]	NS	1.5[4]	69.26	55.73	55.08	54.92	PRZM-EXAMS
Berry[A]	NS	1[3]	9.15	6.75	6.59	6.55	PRZM-EXAMS
Turf[A]	NS	2[4]	19.08	14.75	14.23	14.1	PRZM-EXAMS
Christmas trees[A]	NS	1[3]	4.91	2.46	2.35	2.33	PRZM-EXAMS

A. Land Use Category: Agriculture-crop. Maximum single application use rates: 2 lbs a.i./A California citrus, 1.5 lbs a.i./A eggplant, cherry, and California strawberry.
B. Land Use Category: Developed. Maximum single application rate of 1 lbs a.i./A. nurseries
NS - Not specified

1- Less than maximum single application rate currently allowed for site and Land Use Category
2- More than maximum single application rate currently allowed for site, ≤ rate allowed for Land Use Category
3- Equivalent to maximum single application rate currently allowed for site
4- Use is not authorized on current labels. Rate assumed is ≤ rate allowed for Land Use Category.
5- Labels for ornamentals do not specify a maximum application rate, although UPI has indicated they will modify labeling to include a maximum rate of 1 lb a.i./A per application, a maximum of 4 lb a.i./year, and minimum application interval of 21 d. Additionally, this will apply to commercial production of ornamentals as established ornamentals will be removed as an approved use.

Table 73 summarizes EPA aquatic exposure estimates using PRZM-EXAMS modeling of propargite in its assessment for threatened California red-legged frog (EPA 2008).

Table 73. EPA estimated concentrations of propargite in surface water associated with registered uses of propargite products

Use scenario	Application method	Rate (lbs a.i./A)	Peak (µg/L)	21-d average (µg/L)	60-d average (µg/L)	Model
Alfalfa[A]	Ground	2.456[3]	1.75	0.69	0.54	PRZM-EXAMS
Alfalfa[A]	Aerial	2.456[3]	8.68	2.00	1.55	PRZM-EXAMS
Bean[A]	Ground	2.456[3]	9.23	2.25	1.66	PRZM-EXAMS
Bean[A]	Aerial	2.456[3]	9.62	2.74	2.20	PRZM-EXAMS
Berry[A]	Ground	1.92[3]	11.48	2.84	1.84	PRZM-EXAMS
Citrus[A]	Ground	3.36[3]	1.51	0.31	0.26	PRZM-EXAMS
Citrus[A]	Aerial	2.456[3]	8.07	1.35	1.04	PRZM-EXAMS
Clover[A]	Ground	1.6375[1]	1.11	0.44	0.35	PRZM-EXAMS
Clover[A]	Aerial	1.6375[1]	5.81	1.28	1.03	PRZM-EXAMS
Corn[A]	Ground	2.625[3]	8.79	2.45	1.73	PRZM-EXAMS
Corn[A]	Aerial	2.625[3]	9.48	2.56	1.86	PRZM-EXAMS
Cotton[A]	Ground	2.456[4]	3.89	1.30	1.01	PRZM-EXAMS
Cotton[A]	Aerial	2.456[4]	9.11	2.27	1.91	PRZM-EXAMS
Forestry (plantation) [AB]	Ground	2.4[3]	25.07	5.98	4.62	PRZM-EXAMS
Forestry (plantation) [AB]	Aerial	2.4[3]	24.99	6.60	5.22	PRZM-EXAMS
Grapes[A]	Ground	2.88[3]	21.00	5.14	3.36	PRZM-EXAMS
Hops[A]	Ground	1.5[3]	7.67	2.32	1.95	PRZM-EXAMS
Jojoba[A]	Ground	1.6375[3]	0.95	0.33	0.30	PRZM-EXAMS
Jojoba[A]	Aerial	1.6375[3]	5.56	1.05	0.66	PRZM-EXAMS
Mint[A]	Ground	2.25[3]	5.47	1.79	1.30	PRZM-EXAMS
Mint[A]	Aerial	2.25[3]	8.52	2.99	1.98	PRZM-EXAMS
Nectarine[A]	Ground	2.88[3]	2.16	0.53	0.39	PRZM-EXAMS
Nectarine[A]	Aerial	2.88[3]	9.94	2.00	1.58	PRZM-EXAMS
Ornamental shrub[B]	Ground	1.6[3]	32.11	7.23	5.01	PRZM-EXAMS
Ornamental shrub[B]	Aerial	1.6[3]	31.75	7.58	5.26	PRZM-EXAMS
Other Ornamental[B]	Ground	0.48[3]	9.63	2.17	1.50	PRZM-EXAMS
Peanut[A]	Ground	1.6375[3]	6.05	1.48	1.09	PRZM-EXAMS
Peanut[A]	Aerial	1.6375[3]	6.43	2.22	1.48	PRZM-EXAMS
Sorghum[A]	Aerial	1.6375[3]	5.74	1.22	0.82	PRZM-EXAMS
Strawberry[A]	Ground	1.92[3]	7.12	2.37	1.99	PRZM-EXAMS
Tree Fruit[A]	Ground	1.92[1]	1.44	0.35	0.26	PRZM-EXAMS
Tree Nut[A]	Ground	1.92[1]	3.34	0.88	0.63	PRZM-EXAMS

A. Land Use Category: Agriculture-crop. Maximum single application use rates: 3.36 lbs a.i./A California citrus, 3.2 lbs a.i./A walnuts.
B. Land Use Category: Developed. Maximum single application rate of 2.4 lbs a.i./A. conifer nurseries; 1.92 lbs a.i./A other nursery plants
1- Less than maximum single application rate currently allowed for site and Land Use Category
2- More than maximum single application of 3.2 lbs/A proposed by Chemtura Corporation, 12/13/2012
3- Equivalent to maximum single application rate currently allowed for site

4- More than maximum single application rate currently allowed for site, ≤ rate allowed for Land Use Category

9.1.5.2 Utility of EPA-derived exposure estimates 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 consider the range in concentrations to which individuals of the population may be exposed. The highest concentrations in aquatic habitats are typically associated with direct application to water, or off-target deposition of pesticides into shallow habitats in close proximity to the application site. Pacific salmonids utilize a variety of aquatic habitats (Table 64). All listed Pacific salmonid species use shallow, low flow habitats at some point in their life cycle.

EPA Rice Model. EPA used the tier 1 Rice model to estimate aquatic concentrations of diflubenzuron from applications to rice crops and ornamental ponds. The model assumed direct application of 0.625 lbs diflubenzuron /acre to surface water 0.1 m deep. This equates to an average initial concentration of 701 µg/L. However, the model also assumes instantaneous partitioning to the sediment, in this case based on the average K_{oc} of 3961 L/kg. Additionally, the model assumes no dissipation. Therefore both acute and chronic concentrations were estimated to be equivalent at 113 µg diflubenzuron /L (EPA 2009c).

We do not expect listed Pacific salmonids to use flooded rice fields, although they may be exposed to diflubenzuron that is discharged to their habitats or manages to seep from rice fields into drainage canals and ultimately into salmonid habitats. Therefore, water concentration estimates in flooded rice fields, particularly at discharge, can be used as an upper bound for runoff concentrations for salmon and their designated critical habitat. Adjusting for current labeled application rates, the Rice Model simulation paired with field dissipation calculations provide a reasonable estimate of concentrations that may be discharged or runoff into habitats where listed salmonids reside. Rice model results are also useful for estimating likely concentrations associated with diflubenzuron application directly to some habitats that contain listed salmonids (EPA Reg. No. 400-465, CA-970021). Below, we provide additional estimates to account for direct application of diflubenzuron to surface water that account for variability in habitat depth. Current labels restrict the maximum application rate to 0.25 lbs a.i. /A in rice (EPA Reg. No. 400-461). This equates to an average initial concentration of 280 µg/L if applied to 0.1 m of water, and 45 µg/L assuming instantaneous partitioning based on the average K_{oc}. Additionally, the labels require treated flood waters to be held a minimum of 14 d before discharge. Dissipation of diflubenzuron during that period will depend on site specific variables. EPA reports diflubenzuron dissipation half-lives of 2-14 d in aquatic field studies (EPA 2009c). Considering an initial concentration of 280 µg/L, partitioning with sediment, and comparable dissipation suggest diflubenzuron concentrations of approximately 0.35 – 10 µg/L in discharge after a 14-d hold period.

EPA Reg. No. 400-469 allowing application of diflubenzuron to ornamental ponds is in the cancelation process and is not considered part of the federal action. Additionally, there are special local needs registrations that allow direct application of diflubenzuron to water that may contain salmonids. These uses are also being canceled. EPA Reg. No. 400-465 allows for diflubenzuron treatment of ponds used in commercial fish production at rates of up to 55 µg a.i./L. Applying the Rice model partitioning assumptions discussed above results in estimated aquatic concentrations of 9 µg a.i./L. Similar to rice production, these ponds are closed systems not accessible to salmonids. The label specifies a 14 d hold period prior to discharge. Degradation during that interim is expected to further reduce diflubenzuron concentrations prior to discharge. Additionally, EPA Reg. No. 400-543 includes application of effervescent tablets to holding water receptacles or standing water sites around the home. The recommended treatment rate of 1 tablet per 2.5 gallons of standing water equates to more than 1000 µg a.i./L which could be transported to salmonid habitats through runoff. Although the target concentration is high, we expect concentrations in salmonid habitats that might result from any runoff would be reduced by several orders of magnitude through dilution because applications will likely be confined to treatment of relatively small volumes of water (≤100 gallons).

EPA AgDrift model estimates. EPA used the AgDrift model to determine buffer distance to listed species habitats required to prevent adverse impacts to California red legged frog through exposure caused by primary drift of diflubenzuron and propargite. Model simulations assumed EPA default inputs and modeled drift to a 2-m deep farm pond (EPA 2009c, EPA 2008). The physical characteristics of EPA's farm pond provide a reasonable representation of some of the habitats used by salmonids. NMFS used AgDrift to estimate concentrations of the three a.i.s in other important habitats, such as floodplain habitats with lower dilution capacity. In aggregate, the modeled habitats bracket the potential range of exposure among individual salmonids (*NMFS exposure estimates for flood habitat).*

EPA PRZM-EXAMS model estimates. PRZM-EXAMS was the primary tool used to by EPA to assess aquatic concentrations of the three a.i.s in recent assessments (EPA 2009c, EPA 2008, EPA 2007a). PRZM-EXAMS was used to evaluate applications of the three a.i.s to crops. Additionally, the model was used to assess other diflubenzuron uses including applications to manure, pastures, residential areas, and rights-of-way. PRZM-EXAMS assumes application of the pesticide to a 10 hectare field that drains into a one hectare pond that is 2 m deep with no outlet. According to EPA, exposure estimates generated using this model are intended to represent a wide variety of vulnerable water bodies. However, there are a number of factors that may produce overestimates, or underestimates of exposure when actual site specific factors are considered. Examples include depth and flow rate of aquatic habitat, soil type, slope, meteorological conditions, area treated, and area of aquatic habitat (EPA 2009c, NMFS 2011).

As previously indicated, many of the model estimates do not reflect the current action and we omitted those estimates that assumed rates greater than currently allowed. We retained estimates that assessed less than the maximum rate but we note that these scenarios are likely to underestimate exposure when the maximum rate is applied.

9.1.5.3 NMFS exposure estimates for floodplain habitats exposed to pesticide drift

In this section, we provide model estimates intended to bracket the range of exposure among salmonids for all approved uses. Airborne drift of pesticides during application is a transport mechanism that can result in offtarget deposition. Aquatic habitats adjacent to treated fields, including shallow floodplain habitats where juvenile salmonids rear and shelter are particularly vulnerable to drift. We derived exposure estimates for floodplain habitats using the AgDrift model to estimate downwind deposition from pesticide drift (Teske 2001).

AgDrift is a field-scale model that can be used to estimate concentrations in surface water near the pesticide treated site (≤ 1,000 ft). The drift estimates derived represent average initial concentrations projected for the day of application. Once in the aquatic system, we expect the concentrations of the a.i. will dissipate in floodplain habitats and other surface waters at variable rates based on chemical and site specific factors (*e.g.* partitioning with sediment, rates of degradation, dispersal via flow, *etc*.). Although AgDrift reasonably predicts drift, drift is inherently variable. It is influenced by site-specific conditions as well as application equipment/method (Bird et al. 2002).

9.1.5.4 Diflubenzuron concentrations in floodplain habitats

Our model inputs incorporated application requirements specified on current labels of diflubenzuron (Table 74). Outdoor spray application of diflubenzuron typically requires a no-spray buffer to aquatic habitats of 25 ft for ground and 150 ft for aerial applications (*Description of Action* 4.2). Floodplain habitat estimates for diflubenzuron range between 0.02 and 53 µg/L.

Table 74. Estimated average initial diflubenzuron concentrations in a floodplain habitat that is 2 m wide and of variable depths using AgDrift 2.0.05 (specific inputs are foot noted below)

Application method	Example Label; Use	Land Use	No-spray Buffer[1] (feet)	Simulation: Rate in lbs a.i./A	Habitat Depth (m)	Average Initial Concentration in Surface Water (µg/L)
Ground	EPA Reg No. 400-465; Pear[3]	Agriculture	25	0.75	0.1	11.9
					2	0.60
Aerial	EPA Reg No. 400-465; Pear[4]	Agriculture	150	0.75	0.1	52.5
					2	2.63
Ground	EPA Reg No. 400-461; Citrus[3]	Agriculture	25	0.3125	0.1	4.95
					2	0.25
Aerial	EPA Reg No. 400-461; Citrus[4]	Agriculture	150	0.3125	0.1	21.9
					2	1.09
Ground	EPA Reg No. 400-465; crops, landscaping, forests, rights-of-way[3]	All	25	0.25	0.1	3.96
					2	0.20
Aerial	EPA Reg No. 400-465; crops, landscaping, forests, rights-of-way[4]	All	150	0.25	0.1	17.5
					2	0.88
Ground	EPA Reg No. 400-474; crops, landscaping, forests, rights-of-way[3]	All	25	0.125	0.1	1.98
					2	0.10
Aerial	EPA Reg No. 400-474; crops, landscaping, forests, rights-of-way[4]	All	150	0.125	0.1	8.76
					2	0.44
Ground	EPA Reg No. 400-461; crops[3]	Agriculture	25	0.0625	0.1	0.99
					2	0.05
Aerial	EPA Reg No. 400-461; crops[4]	Agriculture	150	0.0625	0.1	4.38
					2	0.22
Ground	EPA Reg No. 400-461; rangeland/pastures[3]	Agriculture, Undeveloped	25	0.03125	0.1	0.50
					2	0.02
Aerial	EPA Reg No. 400-461; rangeland/pasture[4]	Agriculture, Undeveloped	150	0.03125	0.1	2.19
					2	0.11

1 No-spray buffers to aquatic habitat;
2 Tier 1 ground, Low ground boom spray, ASAE fine to medium/coarse distribution, 50th percentile estimate;
3 Tier 1 ground, Low ground boom spray, ASAE very fine to fine distribution, 50th percentile estimate;
4 Tier 1 aerial (Agricultural), ASAE fine to medium distribution, 50th percentile estimate.

9.1.5.5 Fenbutatin oxide concentrations in floodplain habitats

To reduce the risk of aquatic contamination in Florida citrus, ground and aerial applications of fenbutatin oxide are prohibited from occurring within 25 and 150 ft of streams, rivers, lakes, marshes, and estuaries. Current labeling allows fenbutatin oxide products to be applied adjacent to listed Pacific salmonids habitat without a no-spray buffer for ground and aerial applications. However, we understand that the label for fenbutatin oxide technical material will be modified to reduce potential exposure to salmonids and their habitat by requiring changes to all existing end-use product labels (United Phosphorus 2013). These label changes are reflected in the exposure estimates presented below (Table 75). They include prohibition of aerial applications and institution of a 25 foot no-spray buffer for all applications in California, Idaho, Oregon, and Washington. Other pending label modifications are reflected in the *Description of the Proposed Action*. The estimated initial concentrations of fenbutatin oxide predicted in floodplain habitats range from 0.12 – 67 µg/L.

Table 75. Estimated average initial fenbutatin oxide concentrations in a floodplain habitat that is 2 m wide and of variable depths using AgDrift 2.0.05 (specific input are foot noted below)

Application method	Example Label; Use	Land Use	No-Spray Buffer[1] (feet)	Application Rate in lbs a.i./A	Habitat Depth (m)	Average Initial Concentration in Surface Water (µg/L)
Ground (airblast)	EPA Reg No. 7056-211[5]; California citrus [2]	Agriculture	25	2	0.1	66.5
					2	3.32
Ground (airblast)	EPA Reg No. 7056-211[5]; Cherry[3]	Agriculture	25	1.5	0.1	3.51
					2	0.18
Ground (airblast)	EPA Reg No. 7056-211[5]; CA Almond, CA pecan, CA pistachio, walnut [2]	Agriculture	25	1.25	0.1	41.54
					2	2.08
Ground (airblast)	EPA Reg No. 7056-211[5]; Grape [3]	Agriculture	25	1.25	0.1	2.92
					2	0.15
Ground (airblast)	EPA Reg No. 7056-211[5]; Apple, pear,	Agriculture	25	1	0.1	2.34
					2	0.12

Application method	Example Label; Use	Land Use	No-Spray Buffer[1] (feet)	Application Rate in lbs a.i./A	Habitat Depth (m)	Average Initial Concentration in Surface Water (□g/L)
	peach, plum prune, nectarine, WA/ORChristmas tree [3]					
Ground (boom)	EPA Reg No. 7056-211[5]; CA strawberry [4]	Agriculture	25	1.5	0.1	23.8
					2	1.19
Ground (boom)	EPA Reg No. 7056-211[5]; strawberry, CA eggplant, WA/OR raspberry, field grown ornamental[4]	Agriculture	25	1	0.1	15.85
					2	0.79

1- No-spray buffer to aquatic habitat;
2- Tier 1 orchard airblast, dense (citrus, tall trees);
3- Tier 1 orchard airblast, normal (stone and pome fruit, vineyard);
4- Tier 1 ground, Low ground boom spray, ASAE very fine to fine distribution, 50th percentile estimate;
5- Proposed label modification to reflect change in the federal action proposed by UPI on January 16, 2013.

9.1.5.6 Propargite concentrations in floodplain habitats

Estimates for propargite concentrations in floodplain habitats incorporated application requirements specified on current labels, as well as label modification proposed by Chemtura Corporation (Table 76). EPA Reg. No. 400-89 requires no-spray buffers to lakes, reservoirs, rivers, permanent streams, marshes or natural ponds; and estuaries and commercial fish ponds. Irrigation canals and waterways as well as man-made irrigation conveyance structures and impoundments do not require a no-spray buffer, unless they contain water year-round. Chemtura's proposed label changes are factored into the AgDrift modeling estimates presented below include (a) no use of Comite II (EPA Reg No. 400-154) in California, Idaho, Oregon, and Washington; (b) apply propargite-containing products only when wind speed is 2-10 mph at the application site; (c) use of nozzle and pressure combinations that produce a medium or coarse droplet size (>250 microns volume median diameter), and (d) reduction of the maximum application rate in walnuts from 4.5 lbs a.i./A to 3.2 lbs a.i./A. Based on these label changes estimated concentrations of propargite for floodplain habitats range from approximately 0.11 – 269 µg/L. Aerial applications resulted in much higher concentrations of propargite than ground applications. Air blast sprays were simulated for nut and citrus crops with predicted concentrations of approximately 3 - 55 µg/L depending on rate and habitat depth (Table 76).

Table 76. Estimated average initial propargite concentrations in a floodplain habitat that is 2 m wide and of variable depths using AgDrift 2.0.05 (specific inputs footnoted below)

Application method	Example Label; Use	Land Use	Buffer[1] (feet)	Simulation: Rate in lbs a.i./A	Habitat Depth (m)	Average Initial Concentration in Surface Water (µg/L)
Ground (airblast)	EPA Reg No. 400-104; Walnut, Citrus[2]	Agriculture	50	3.2	0.1	52.4
					2	2.62
Aerial	EPA Reg No. 400-104; Walnut[3]	Agriculture	75	3.2	0.1	269
					2	13.4
Aerial	EPA Reg No. 400-89 Almond[3]	Agriculture	75	3.0	0.1	252
					2	12.6
Ground	EPA Reg No. 400-427; Grape[4]	Agriculture	50	2.88	0.1	13.2
					2	0.66
Aerial	EPA Reg No. 400-427; Nectarine[3]	Agriculture	75	2.88	0.1	242
					2	12.1
Ground	EPA Reg No. 400-104; crops[4]	Agriculture	50	2.46	0.1	11.3
					2	0.57
Aerial	EPA Reg No. 400-104; crops[3]	Agriculture	75	2.46	0.1	206
					2	10.3
Ground	EPA Reg No. 400-104; Potato[4]	Agriculture	50	2.05	0.1	9.43
					2	0.47
Aerial	EPA Reg No. 400-104; Potato[3]	Agriculture	75	2.05	0.1	172
					2	8.60
Ground	EPA Reg No. 400-104; crops[4]	Agriculture	50	1.5	0.1	6.90
					2	0.34
Aerial	EPA Reg No. 400-104; crops[3]	Agriculture	75	1.5	0.1	126
					2	6.29
Ground	EPA Reg No. 400-427; Conifers[4]	Developed	50	2.4	0.1	11.0
					2	0.55
Ground	EPA Reg No. 400-427; Rose[4]	Developed	50	1.6	0.1	7.36
					2	0.37
Ground	EPA Reg No. 400-427; ornamentals[4]	Developed	50	0.48	0.1	2.21
					2	0.11

1- No-spray buffer to aquatic habitat
2- Tier 1 orchard airblast, dense, 50th percentile estimate
3- Tier 1 aerial (Agricultural), ASAE medium to course distribution, 50th percentile estimate
4- Tier 1 ground, Low ground boom spray, fine to medium course distribution, 50th percentile estimate

9.1.6 Monitoring Data: measured concentrations of diflubenzuron, fenbutatin oxide, and propargite in surface waters

We reviewed two types of pesticide monitoring data: 1) ambient monitoring that measures concentrations of pesticides in surface waters where sampling is not targeted at the field scale with any specific pesticide application, and 2) targeted monitoring that measures concentrations of pesticides in surface waters adjacent to the site of application and sampling is targeted both spatially and temporally with specific applications of a pesticide.

We evaluated data from USGS' NAWQA database and state databases maintained by California, Oregon, and Washington. Information provided by the four databases includes ambient monitoring data with sampling stations distributed across a range of land uses; some of the stations are in salmonid bearing aquatic areas. They may also include studies that investigated water quality impacts associated with specific pesticide uses. All of these data are considered historical as there is a lag time between collection, detection, and reporting. Idaho does not maintain a state surface water monitoring database for pesticide studies. However, we reviewed the eleven pesticide monitoring reports available on the Idaho State Department of Agriculture web site and found no records of analysis of the three active ingredients (http://www.agri.idaho.gov/Categories/Environment/water/swReports.php).

We also reviewed a targeted monitoring study that investigated surface water concentrations associated with application of fenbutatin oxide on citrus in Florida (Wallace et al. 1993). We found no other targeted monitoring studies evaluating surface water concentrations of the three active ingredients.

9.1.6.1 Monitoring data considerations

Surface water monitoring data provide useful exposure information such as real-world environmental exposure of the active ingredients in aquatic systems. The data also provide information on the occurrence of environmental mixtures. A primary consideration in evaluating monitoring data is whether the objective of the monitoring program/study is addressed by the study design. For determining exposure to salmonids, we evaluate whether the study design a priori targets salmonid presence and application of diflubenzuron, fenbutatin oxide, and propargite. If the study design meets these criteria then we may use the data in quantitative and potentially probabilistic analyses. However if the data do not meet these criteria we use the information qualitatively. Unfortunately, we find that ambient monitoring programs do not track the application of the three insecticides and do not target salmonid habitats based on the presence of threatened and endangered salmonid populations.

The available monitoring studies were conducted under a variety of protocols and for varying purposes. Ambient water quality monitoring conducted in larger streams and rivers frequently does not capture "peak" concentrations because it is not correlated with applications and/or storm events following those applications and not all habitat types are sampled. This is one of the reasons NMFS did not use available monitoring data for probabilistic modeling (*i.e.*, it likely does not contain the complete range of possible concentrations).

As discussed above, the ambient monitoring programs we have were not designed to evaluate the potential range of pesticide exposure to threatened and endangered salmonids. Common aspects that limit the utility of the ambient 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 relationship on which populations of listed salmonids overlap with sampling sites and application events, and (3) lack of representativeness for current and future pesticide uses and conditions. We discuss each of the topics below.

9.1.6.2 Ambient monitoring protocols not designed to capture peak exposure.

The NAWQA program provides a considerable dataset that is useful for evaluating trends in water quality (Hirsch et al. 1988). The NAWQA design does not result in an unbiased representation of surface waters, which limits the ability to make statistical extrapolations to waters not sampled. Sampling by NAWQA and studies contained in the state water quality monitoring databases were generally not conducted in coordination with specific applications of the diflubenzuron, fenbutatin oxide, and propargite at the field scale. Similarly, sampling was not designed with consideration of salmon distributions or to target salmonid habitats most likely to contain the greatest concentrations of pesticides. Given the relatively rapid dissipation of pesticides in flowing water habitats, it is not surprising that pesticide concentrations detected in these datasets are generally much lower than those predicted by modeling efforts and those that monitor targeted pesticide applications at the field scale. Although these data are useful for documenting the occurrence of pesticides in salmonid habitats they likely underestimate the magnitude of potential exposure.

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. Surface water attributes 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 and temporal variability. Given these factors, extrapolating from ambient monitoring data in one geographic area to non-monitored areas is not recommended.

Representativeness of current and future uses. Pesticide use varies annually depending on regulatory changes, market forces, cropping patterns, and pest pressure. Changes in use may result in either increases or decreases in use of pesticide products for specific crops or other uses. While use of diflubenzuron has generally increased in California in recent years (Figure 64), use of fenbutatin oxide and propargite has declined substantially (Figure 65, Figure 66). Idaho, Oregon, and Washington do not require pesticide use reporting and therefore equivalent data do not exist for these states. This is a major data gap that introduces large uncertainties when trying to link monitoring data to pesticide application data. Prediction of future use of pesticides is complicated by changing agricultural patterns and fluctuating pest pressures. Both are further complicated by predicted changes in climate. Additionally, changes in registration due to regulatory decisions may affect which pesticides can be used (e.g. cancelations) and how they are used. Such changes complicate the ability to predict future water quality conditions from historical monitoring information.

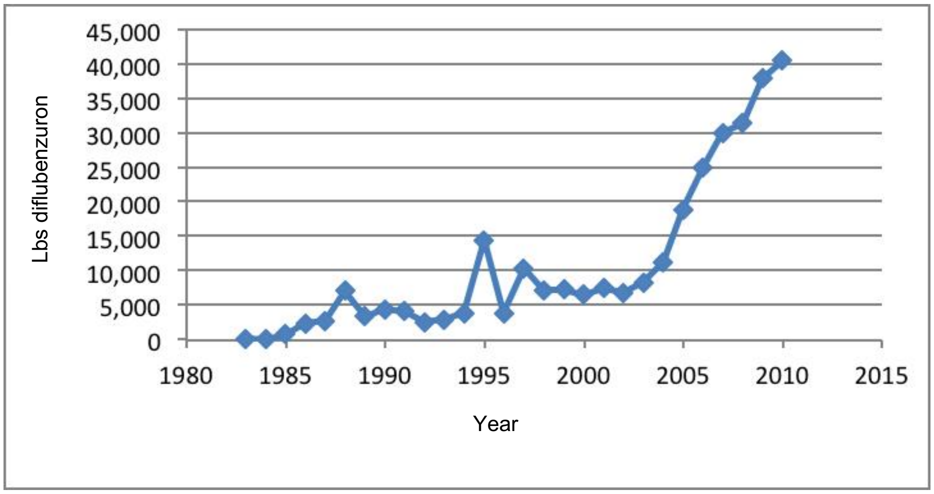

Figure 64. Pounds of diflubenzuron applied in California. Data from the California Pesticide Use Reporting Database 1983 – 2010.

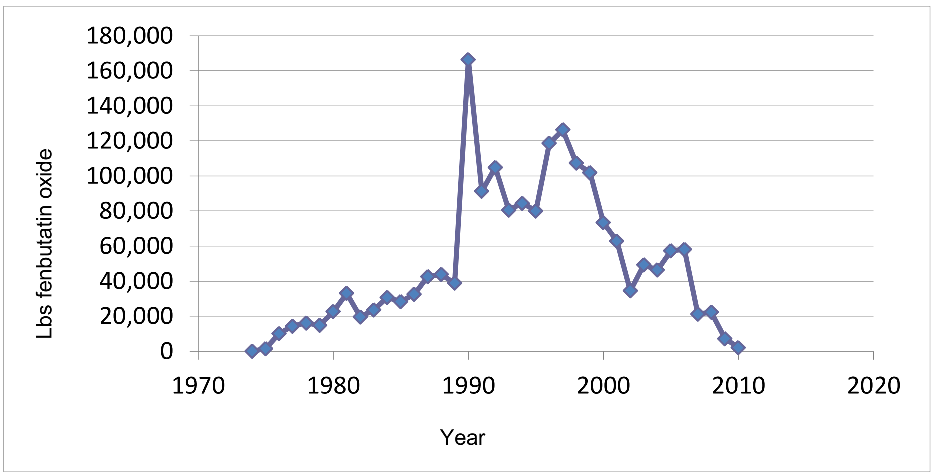

Figure 65. Pounds of fenbutatin oxide applied in California. Data from the California Pesticide Use Reporting Database 1974 – 2010.

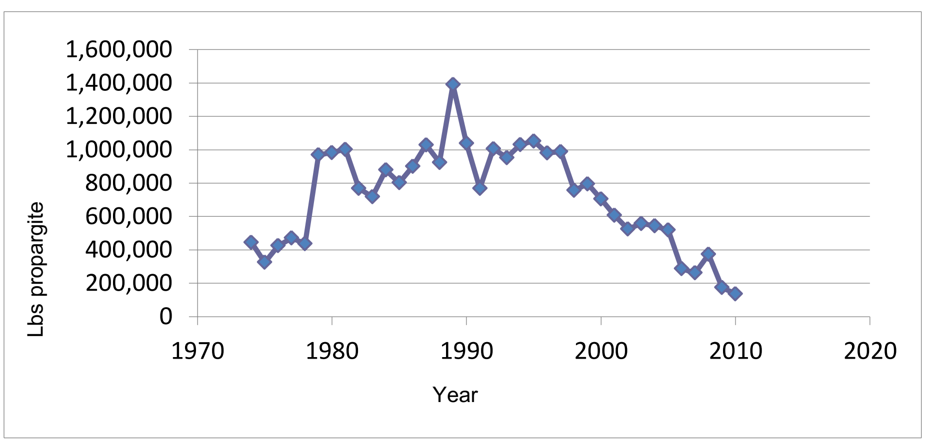

Figure 66. Pounds of propargite applied in California. Data from the California Pesticide Use Reporting Database 1974 – 2010.

9.1.6.3 USGS NAWQA and state monitoring data

We obtained the most current monitoring data from the USGS NAWQA, California DPR's surface water monitoring database (http://www.cdpr.ca.gov/docs/emon/surfwtr/surfdata.htm), Oregon Department of Environmental Quality's Laboratory Analytical Storage and Retrieval Database (http://deq12.deq.state.or.us/lasar2/), and Washington Department of Ecology's Environmental Information Management database (http://www.ecy.wa.gov/eim/) to evaluate the occurrence of the three a.i.s and degradation products in surface waters monitored in California, Idaho, Oregon, and Washington. Of the three a.i.s and their degradates (identified in Section 9.1.3), propargite is the only analyte that was monitored. No monitoring data were available for diflubenzuron or fenbutatin oxide. Propargite records from the four monitoring databases were combined and duplicate information reported for the same samples eliminated. The database queries resulted in more than eight thousand surface water samples obtained from 427 unique locations across the four states during a period of more than 20 years, 1992 – 2012.

Although propargite is approved almost exclusively for use on crop lands, land uses associated with the sampling stations included agriculture, forest, rangeland, urban, industrial, residential, mixed, and other uses (Figure 67). Some water bodies and/or basins sampled do not contain listed salmonids and several of the species have had no sampling for propargite within their freshwater and coastal habitats (Table 77). Sampling effort varied considerably among the sample locations. More than 50% of the data were collected from only 20 monitoring locations (Figure 67); approximately 40% of sites were sampled only once during the 20 year monitoring period. The highest concentrations of propargite and greatest detection frequencies were observed in California and Washington (Table 78). However, caution should be used in extrapolating this information as a number of differences in sampling are likely to lead to erroneous conclusions. For example, the intensity of sampling effort was much greater in California and Washington than that in Idaho and Oregon and all of the sampling conducted in

Idaho was outside the distribution of listed salmonids (Figure 67). Overall, the sampling effort does not correspond well with the distribution of listed salmon or the distribution of propargite use areas. Consequently, we do not expect the data set to be representative of potential exposure for listed salmonids.

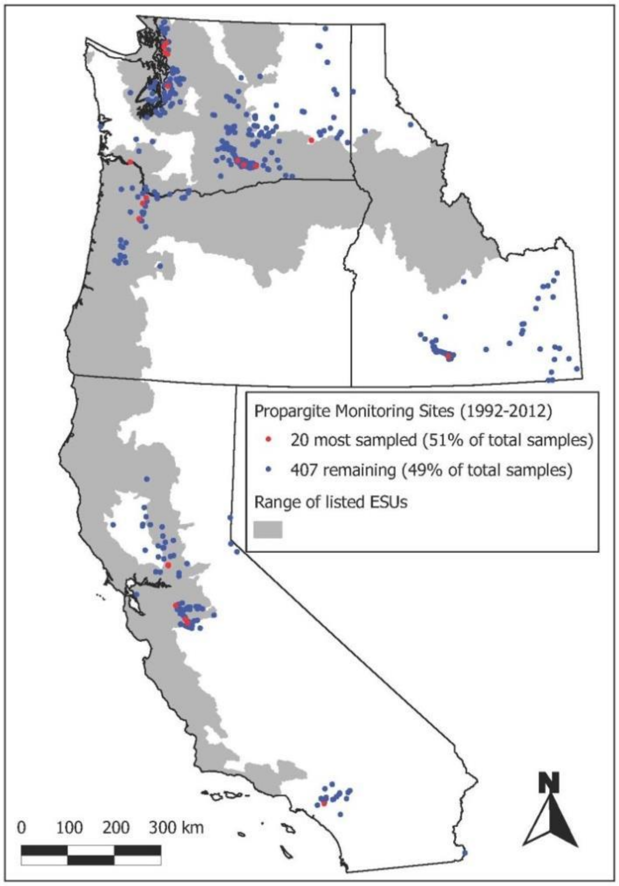

Figure 67. Distribution of monitoring sites that have sampled for the presence of propargite relative to the range of threatened and endangered Pacific salmonids.

Table 77. Number of monitoring sites within the distribution of listed Pacific salmonids as determined through GIS analysis.

Species	ESU	Kilometers of Stream Inhabited	Sites within ESU range[1]
Chinook	Puget Sound	3,639.65	80
	Lower Columbia River	2,443.29	19
	Upper Columbia River Spring - Run	1,646.75	9
	Snake River Fall - Run	1,370.44	1
	Snake River Spring/Summer - Run	5,288.23	0
	Upper Willamette River	3,013.85	26
	California Coastal	2,422.44	0
	Central Valley Spring - Run	2,212.94	16
	Sacramento River Winter - Run	546.84	16
Chum	Hood Canal Summer - Run	141.89	2
	Columbia River	1,162.18	14
Coho	Lower Columbia River	3,307.78	19
	Oregon Coast	10,220.00	0
	Southern Oregon and Northern California Coast	5,619.58	0
	Central California Coast	1,287.78	0
Sockeye	Ozette Lake	70.98	0
	Snake River	1,493.94	0
Steelhead	Puget Sound	3,849.64	80
	Lower Columbia River	4,302.03	18
	Upper Willamette River	3,063.07	17
	Middle Columbia River	10,196.80	84
	Upper Columbia River	2,143.15	17
	Snake River	13,423.40	1
	Northern California	5,324.31	0
	Central California Coast	4,620.72	1
	California Central Valley	4,273.66	55
	South-Central California Coast	5,104.56	0
	Southern California	3,015.86	0

1- Monitoring points within freshwater habitats as defined by range maps presented in *Status of Listed Resources*.

Table 78. Detections and concentrations of propargite monitored in California, Idaho, Oregon, and Washington.

Statistic	California	Idaho	Oregon	Washington	Total
Number of Stations	98	57	41	231	427
Number of Observations	2492	453	1062	4013	8020
Detects	188	1	3	59	251
Percent Detections	7.54	0.22	0.28	1.47	3.1
Median detected (µg propargite /L)	0.07	0.15	0.01	0.04	0.06
Range (µg propargite/L)	0.005-20	0.15	0.007-0.05	0.004-2.14	0.005-20
LRL (µg propargite/L)	0.005-1	0.013-0.07	0.007-0.07	0.004-0.6	0.005-1
Year range	1992 - 2012	1993-2012	1992-2012	1993-2012	1992-2012

Washington Surface Water Monitoring Program: 2009-2011 Triennial Report

The ambient water quality data presented above provide useful information documenting the presence of propargite in some of the species' baseline habitats from previous use of the pesticide. This information has helped Washington, and other states, monitor trends for some pesticides, identify exceedance of water quality standards and ecological thresholds, and apply adaptive management to address risk. A subset of the information includes a series of reports documenting pesticide concentrations in some listed Pacific salmonid habitats conducted by Washington State Departments of Ecology (WDOE) and Agriculture (WSDA) (http://www.ecy.wa.gov/programs/eap/toxics/pesticides.htm). Below we summarize the most recent report and discuss its utility for this consultation (Sargeant et al. 2013a).

Surface Water Monitoring Design

The study design is useful for identifying potential problems (e.g. when surface water concentrations exceed ecological thresholds) and for evaluating localized trends. However, this study does not provide the field scale information on actual use of pesticides (proximity to monitoring site, application rate, etc.) that is required to explicitly evaluate the relationship between pesticide use and surface water concentrations. Actual use of the pesticides at the sample collection sites preceding sampling is unknown. Therefore, this monitoring program does not provide reasonable evidence to conclude that past label restrictions are sufficient to prevent harm to salmonids or their habitat due to the potential for future pesticide use.

Composite surface water samples were collected to determine the concentrations of more than 170 analytes (including pesticides, degradates, adjuvants) and measure conventional water quality parameters (temperature, dissolved oxygen, pH). The monitoring program included analysis for propargite, but did not assess diflubenzuron or fenbutatin oxide, or any of the degradates of the three a.i.s. Sampling to assess whether sediments were contaminated with the three active ingredients also was not within the scope of the study. Sampling was designed to evaluate pesticide concentrations in surface waters associated with both agricultural and urban land uses and has been conducted by Washington State since 2003.

The initial site selection to evaluate agricultural uses of pesticides was based on the percent crop area and the diversity of crops; site selection to evaluate urban pesticide use was based on population density and the amount of impervious surface (Johnson and Cowles 2003). Sampling intensity and location has varied to some extent since 2003 depending upon several factors. Many sites have remained consistent for multiple years as one of the primary goals of the program is to monitor trends of pesticides and other contaminants in surface water. In recent years (2009-2011), weekly samples were collected from 16 sample stations during Washington States' growing season (March – September). Although this period may correspond well with the peak season of pesticide applications, most pesticide product labels do not restrict the timing of applications to specific months and can legally be applied outside of the March – September timeframe, including the three a.i.s. Pesticide use reporting is not required in Washington and therefore there is uncertainty regarding the location, frequency, and timing of pesticide applications in the state. However, WSDA estimates that more than 90% of pesticide applications in agriculture occur during this period. The intent of the sampling design was to provide state specific data to evaluate pesticide concentrations resulting from four pesticide use categories: (1) urban, (2) tree fruit agriculture, (3) irrigated agriculture, and (4) western Washington agriculture. The number of locations sampled within each of these four categories ranged from two – five (
Table **79**).

Table 79. Pesticide sample location information

Pesticide Use Category	Number of sample stations	Basin location of sampling	Sample Stations Identifier
Urban	2	Cedar-Sammamish	TC-3
		Green-Duwamish	LC-1
Tree Fruit Agriculture	5	Wenatchee (0.07 – 13% of area in agricultural production)	BR-1 MI-1 PE-1 WE-1
		Entiat (<1% of area in agricultural production)	EN-1

Pesticide Use Category	Number of sample stations	Basin location of sampling	Sample Stations Identifier
Irrigated Agriculture	4	Lower Yakima (39 – 72% of area in agricultural production)	MA-2 SU-1 SP-3 SP-2
Western Washington Agriculture	5	Skagit-Samish (8 – 91% of area in agricultural production)	BD-1 BD-2 BS-1 IS-1 SR-1

The four categories of pesticide use evaluated do not include all types of pesticide use permitted in the State. Rather, WSDA sampling is meant to capture the uses that are most likely to affect salmon based on the amounts of pesticides used, the diversity of pesticide uses, and the proximity of pesticide use to salmon bearing waters (Tuttle 2013). While the frequency of pesticide application may be relatively high on agricultural lands compared to other uses, pesticides are also applied to undeveloped lands (e.g. forests, range lands, grass lands, etc.) which account for 74% of the area in Washington state. These land use categories have substantial overlap with listed salmonids but are not a focus of the sampling effort. Sampling occurred in watersheds dominated by undeveloped lands (e.g. Entiat and Wenatchee), but the sampling sites within these basins were located immediately adjacent to agricultural use sites (Figure 68, Table **80**). While these sample sites may detect pesticides used on undeveloped lands in the watershed, the sampling is not designed to define likely exposure from these pesticide these uses.

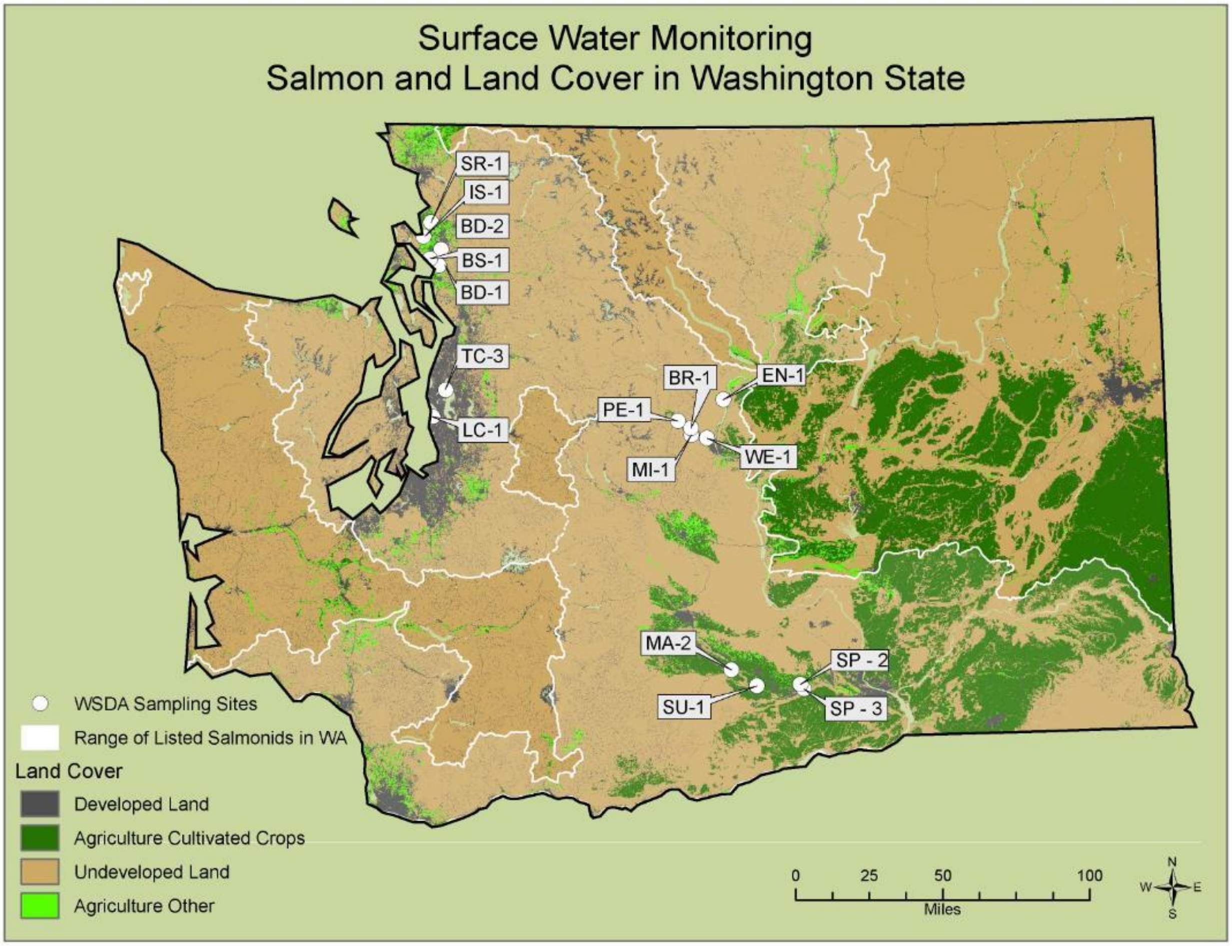

Figure 68. Monitoring station locations in Washington State

Table 80. Area and monitoring effort for land use categories where pesticides may be applied in Washington State

10 Land Use Category	11 Area in Washington 12 Km^2 (%)	13 Number (samples station within category)
14 Developed Land	15 12,598 Km^2 (7.1%)	16 2 (TC-3, LC1)
17 Undeveloped Land	18 130,200 Km^2 (73.7%)	19 0
20 Agriculture-Cultivated Crops	21 26,025 Km^2 (14.7%)	22 9 (SR-1, IS-1, BD-1, PE-1, BR-1, MA-2, SU-1, SP-2, SP-3)
23 Agriculture-Other	24 3595.5 Km^2 (2.0%)	25 5 (BD-2, BS-1, EN-1, WE-1, MI-1)

Information that would allow us to conclude that the sampling effort was sufficient to account for the variability that likely exists within a given Pesticide Use Category (e.g. due to site specific differences in pesticide use and transport) was not within the scope of the report. Only two sample sites (LC-1, monitored since 2008; TC-3, monitored since 2003) are used to evaluate all urban applications while the types and amounts of pesticides authorized for use in urban areas depends on the site-specific classification (e.g. residential, industrial, recreational, etc.). The information provided does not allow us to conclude that the sample design was sufficient to account for these and other sources of site-specific variability (soil type, slope, meteorological conditions, etc.). The monitoring sites selected for this monitoring program do not provide the necessary variability to address the broad range of salmon habitats that include all potential cropping patterns, pesticide use patterns, and aquatic habitat characteristics necessary to evaluate all potential routes of exposure.

The distribution of monitoring sites is not sufficient to fully characterize pesticide exposure that likely occurred in any of the critical habitats supporting listed salmonids. Sampling was very limited with respect to the distribution of listed salmonids. No sampling was conducted in the habitats occupied by 12 of the 17 listed Pacific salmonid species that occur within Washington state (including 3 species whose use of waters in Washington state is confined to migratory and rearing habitats in the Columbia river). Only four to seven sample sites were located within the ranges of each of the five listed salmonids, while these species each occupy thousands of kilometers of streams.

Although previous sampling by WSDA and WDOE suggest weekly sampling was sufficient to capture peak concentrations of pesticides at the sites sampled, many pesticides degrade quickly in aquatic environments with half-lives of less than 7 days. Additionally, dissipation may occur rapidly in habitats with high flow rates suggesting weekly sampling may not be adequate to capture peaks at some sites. Surface waters with smaller volumes have less dilution capacity and therefore will have greater pesticide concentrations from a pesticide drift event; those with higher flow rates will have faster dissipation rates, reducing the duration of exposure and the likelihood of detection with monitoring. The current study provides data that strongly supports the inverse relationship between stream flow and the number and magnitude of detections. Monitoring locations included streams with average flow rates as low as two ft^3/s (cfs) and rivers with average flow rates as high as 5690 cfs. Salmonids use habitats with flow rates that extend both above and below this range. Monitoring sites on the lower end of the flow range represent examples of habitats that may experience high pesticide concentrations due to their physical characteristics. However, the resulting pesticide concentration also depends on the mass of pesticide that is transported to the water body which is a function of a number of application-specific variables (e.g., application rate, droplet size distribution, wind speed). Pesticide transport decreases as distance from the site of pesticide application increases (Bird et al. 2002, Wallace et al. 1993).

While the data provided in this report are valuable for tracking trends in pesticide concentrations and applying adaptive management, they are not useful for assessing potential exposure that may result to salmon based on FIFRA labeling as that was outside the scope of the study. The temporal and spatial relationship between field level pesticide applications and monitoring are unknown. Propargite, for example, may be applied 50 ft from salmonid habitat. However, the possibility that it occurred on a field that was in close proximity to any of the sixteen monitoring stations is low because monitoring was not coordinated at the field scale with pesticide applications. Additionally, based on label restrictions, it is likely that it would not be legal to even apply propargite in the vicinity of several of the monitoring site. The likelihood of detecting the peak concentration of pesticides that occurred within the basin is also low because sampling only occurred once a week and did not target the times when peak concentrations are expected (i.e. during pesticide applications and during likely runoff events).

The Report's results

The report indicates that pesticide detections generally increased from the start of monitoring in March to a peak in May, and then declined through the remainder of the monitoring period which terminated in September. However, temporal patterns in the number and types of pesticides were site specific. This is expected as use and transport of pesticides are influenced by land use classifications and meteorological conditions; both vary depending on site.

Propargite was detected in Marion Drain three times, where 34 samples were collected each year of monitoring (2009-2011). Concentrations detected were 0.110, 0.089, and 0.870 μg/L. Propargite was not detected in any sample at the 15 other monitoring stations. The low detection rates may reflect limited use of propargite during the three year monitoring period. Propargite is primarily registered for use on agricultural crops and therefore detection at the two urban monitoring sites was not expected. Nine stations were located in close proximity to crops (

Table **80**). In California, propargite use has declined by approximately 90% in recent years compared to its peak use in the late 1980s and 1990s when over one million acres were treated with propargite annually in California. Use of pesticides in Washington state is expected to differ from California which has different climates, cropping patterns, and pest pressures. The specific extent of use of propargite in Washington during the monitoring period is unknown because Washington State does not have a mandatory pesticide use reporting system. According to recent estimates from Washington Department of Agriculture, only 1% of the potential market share of the mint crop and the russet potato crop are typically treated with propargite in Washington (Appendix 6) which equates to < 2,000 total acres statewide. Estimates were not available for other commodities that are labeled for propargite use (see *Description of Action*). Future usage of propargite during the 15-year duration of the action is uncertain

The report detailed that seventy-four analytes were detected including 34 herbicides, 21 insecticides, 13 degradates, 4 fungicides, one wood preservative, and one pesticide synergist. Pesticides that were detected at concentrations above assessment criterion for potential impacts to aquatic invertebrates included the organophosphate and carbamate insecticides: chlorpyrifos, diazinon, malathion, methomyl, ethoprop, dichlorvos, and methiocarb; the pyrethroid insecticide bifenthrin; legacy organochlorine pesticides; and the herbicide metolachlor. NMFS previously concluded that use of many of these same pesticides are likely to jeopardize listed salmonids and adversely modify or destroy their designated critical habitat (NMFS 2008e, NMFS 2009d, NMFS 2010a). The report also compared concentrations to levels observed in previous monitoring. Increasing concentration trends were observed for 10 pesticides, while decreasing concentration trends were observed for 16 pesticides. The reported trends for insecticides were site specific, i.e., only azinphos-methyl had significant trends at more than one monitoring station which limits the utility of the data to represent other areas. Increasing trends were observed for 5 herbicides, and decreasing trends for two herbicides at two or more sites.

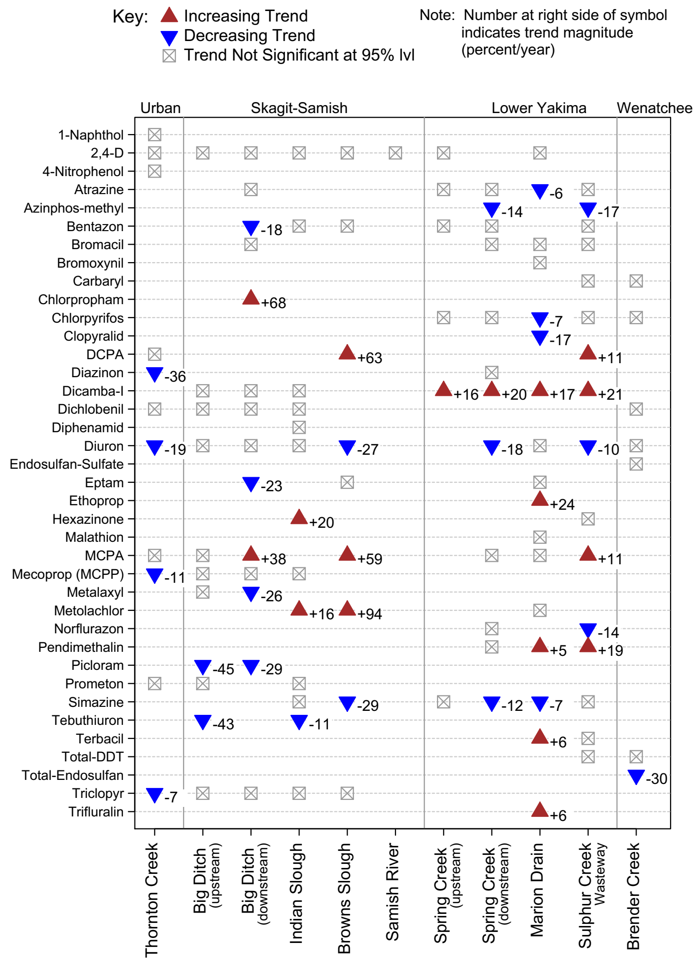

Figure 69. Summary of increasing and decreasing trends in pesticide concentrations by location.

Most pesticides were detected in samples that contained mixtures with other pesticides. Samples contained up to 14 different analytes. Five sites had relatively low detection rates for pesticides, yet 4-11% of the samples from these locations contained mixtures of pesticide ingredients or degradates. The occurrence of mixtures was much greater at the other 11 monitoring sites where 53-99% of the water samples contained mixtures of pesticide contaminants. The report evaluated risk of chemical mixtures by assuming additive toxicity and comparing it to various acute and chronic thresholds for fish and aquatic invertebrates. This is common approach for assessing risk of mixtures, especially for compounds that share the same mechanism of action. The report concluded that although cumulative concentrations of pesticides were of concern, this was generally due to the high concentration of a single pesticide. This conclusion was based on the observation that 68% of the time, these mixtures included a pesticide whose individual concentration alone would indicate a concern. However, the statistics also indicated that 32% of the time a problem may be overlooked if the potential additive effects of the mixtures were not considered. Only one of the chemical mixtures with cumulative concentrations of pesticides of concern included propargite. This particular sample also included two cholinesterase inhibitors (malathion and methomyl) and chlorothalonil. The cumulative concentration of this mixture was 6.4 times the threshold identified for potential chronic effects to invertebrates.

Although, exceedance of criteria for toxicological effects to fish was relatively infrequent, the majority of sites evaluated documented episodes where pesticide concentrations exceeded toxic thresholds for aquatic invertebrates. Insecticides were the class of pesticides that most frequently did not meet water quality criteria or exceeded toxic unit thresholds. Of the conventional pesticides evaluated, the organophosphate and carbamate insecticides were disproportionately represented as compounds responsible for exceedance of thresholds. The organophosphate insecticides that exceeded thresholds included chlorpyrifos, diazinon, malathion, ethoprop, and dichlorvos. These compounds have been shown to cause additive or synergistic effects in aquatic species. Samples from 6 of 14 agricultural monitoring stations exceeded thresholds due to the chlorpyrifos concentrations, and at some sites were detected at concentrations elevated above threshold during consecutive weeks. Carbamates exceeding thresholds included methomyl and methiocarb. Bifenthrin, a pyrethroid insecticide, also exceeded the chronic criteria for invertebrates at several locations. The data suggest that these pesticides, as well as organochlorine legacy pesticides, have recently been documented at concentrations in salmonid habitats that may be harmful to salmonid prey. Additionally, temperature and dissolved oxygen concentrations frequently did not meet water quality standards and monitored levels were often in a range considered harmful to salmonids.

We use these data to show that propargite does find its way to surface waters in Washington State. These data also show that other pesticides are found in salmonid a habitat which informs the baseline condition of the action area.

25.1.1.1 Targeted monitoring

No studies were found that evaluated actual applications of diflubenzuron or propargite to determine aquatic concentrations associated with edge-of-field drift, runoff, or discharge from treated areas. We did locate a targeted monitoring study evaluating off target movement and aquatic concentrations of fenbutatin oxide in surface water that we discuss below.

25.1.1.2 Off target movement of fenbutatin oxide in citrus

A field study was conducted to characterize off-site movement of fenbutatin oxide from real world applications of Vendex 4L (Wallace et al. 1993). The applications were made to five citrus groves within three different regions of Florida. Each site was selected based on its proximity to aquatic habitats. Aquatic habitats evaluated included a pond and drainage canals adjacent to treated areas (1-30 m; 3-98 ft).

Application of fenbutatin oxide. Fenbutatin was applied to the five orchards using air blast spray. Average wind speeds during application ranged from 0.7 – 4 mph (Table 13 *in* Wallace et al. 1993). Each grove received 2 applications at 60 d intervals. Fenbutatin oxide was applied at the rate of 2 lbs a.i. /A in each application. Current labeling of fenbutatin oxide products in California citrus also allows for a maximum single application rate of 2 lbs a.i./A and two applications. The current label also allows for a shorter (30 d) application interval while restricting the maximum seasonal application rate to 3 lbs a.i./A. The label does not specify wind speed restrictions but states that off-site drift potential increases at wind speeds of less than 3 mph and at more than 10 mph (EPA Reg. No. 70506-211). This label also allows for aerial applications in citrus which are predicted to cause more drift than observed in this study. United Phosphorus, Inc. (UPI) indicated they will disallow aerial applications of fenbutatin oxide by modifying the labels (January 16, 2013).

Aquatic habitats monitored. The aquatic habitats monitored had physical characteristics similar to some floodplain habitats used by listed salmonid species during freshwater residence for spawning, rearing, and migration. The pond was relatively large (0.44 ha) and deep (average depth of 4.1 m). The nearest citrus trees were 5-15 m (16 – 49 ft) from the pond. UPI has indicated they will the modify product labeling to require a 7.6 m (25 ft) buffer to salmon bearing waters. Vegetation around the pond included grasses, shrubs, and small trees 3-4 m in height. The canals were variable is size, volume, and bank vegetation. Canals were located 1-

30 m (3-98 ft) from citrus trees. Depths ranged from 0.5 – 2 m, and widths ranged from 1-20 m. Mean flow estimate ranged from zero to several hundred thousand m^3/d. Vegetation along canals was also variable, in some cases it only included grasses <0.75 m in height, in other instances it included trees up to 15 m in height.

Drift to aquatic habitat. Drift to and across aquatic habitats on the day of pesticide application was measured by samples collected with spray deposition cards and laboratory analysis using gas chromatography. Drift observed across the pond was generally less than that observed to canals. This is likely a function of the greater distances to, and across the waterways. The maximum deposition observed in samples obtained from transects across the pond was 2.78% of the target application rate, while median drift was 0.15-0.48% of target (2 lbs a.i. /A). The maximum deposition observed at canal sites ranged between 2 and 36% of target while median values extended from approximately 0.5 to 6 % of target site rates (Table 81).

Table 81. Drift of fenbutatin oxide over aquatic habitats adjacent to treated citrus trees (groves 2-5).

	Drift (% Target Application Rate)				
Grove	1[a]	2[b]	3[b]	4[b]	5[b]
Median	0.15-0.48	1.86-2.66	4.63-6.31	0.90-1.00	0.49-0.56
Minimum	0.04-0.18	0.03-0.11	0.07-0.45	0.01-0.03	0.05-0.11
Maximum	0.25-2.78	7.36-14.02	12.62-13.32	2.38-36.46	2.10-2.52

a denotes 0.44 ha (1.08 Acre) pond located 5 -15 m from treated citrus trees
b denotes perimeter canals 1-30 m from treated citrus trees

Measured concentrations in surface water. Sampling for fenbutatin oxide began before the first application of fenbutatin oxide and extended to 60 d after the second application. Overall, including samples collected from lateral canals within the perimeter of the field, perimeter canals, and a pond:

- 19% of samples had fenbutatin-oxide residues ≥ 0.20 ppb
- 8% of samples had fenbutatin-oxide residues ≥ 1 ppb
- 1% of samples had fenbutatin-oxide residues ≥ 10 ppb

Pond. Concentrations in the pond (grove #1) were substantially lower than those observed in lateral and perimeter canals near the target application site. This is not surprising since the pond had an average depth of more than 4 m while canals were 0.3-2.1 m in depth. On the day of the first application, fenbutatin oxide was detected in 100% of the samples stations along the pond transect that spanned the width of the 1 acre pond (median 0.29 μg/L, range 0.25-0.38 μg/L). Seven percent of the application day samples were >1 μg/L. The maximum concentration observed in the pond was 1.7 μg/L collected on the day of the second application. The median concentration observed during that sampling period was 0.24 μg/L, with a range 0.13-1.7 μg/L. Fourteen percent of samples exceeded the level of quantification (LOQ = 0.2 μg/L) two days after application. Fenbutatin oxide was not detected in the pond above the LOQ at four days post treatment.

Canals. Fenbutatin oxide was detected at relatively high concentrations in lateral and perimeter canals of all groves with peak concentrations of 3.90 – 68.0 μg/L observed on application days (

Table 82). The highest concentration of fenbutatin oxide detected in surface water was 111 µg/L collected in a perimeter canal one day after the second application. The fenbutatin oxide concentration declined to 13 µg/L the following day. Detailed information regarding the dissipation of fenbutatin oxide at this site is not available because the samples collected on day 4 and 7 were not analyzed (Appendix 15.9 *in* Wallace et al. 1993). Dissipation at sampling sites appeared to be highly variable, and presumably associated with the variable flow rates. Fenbutatin oxide concentrations frequently declined to less than 0.20 µg/L within a day or two of application. At other times relatively high concentrations persisted. For example, concentrations at one station remained ≥0.48 µg/L for at least 30 d. Additionally, concentrations remained ≥1.2 µg/L for at least 20 d (grove 5, station #8). All groves had one or more samples with fenbutatin oxide concentrations ≥ 19 µg/L during the monitoring period.

Table 82. Maximum concentrations of fenbutatin oxide detected in orchard lateral and perimeter canals

Days after application	Maximum concentration of fenbutatin oxide detected (µg/L) [1]			
	Grove			
	2	3	4	5
0	19	68	39	20
1	11	5	16	111
2	0.82	25	6.8	13
4	<0.20	3.4	2.6	0.68
7	<0.20	0.65	3.2	0.45
14	<0.20	4.2	2.3	0.95
21	<0.20	0.21	1.2	<0.20
30	<0.20	0.54	0.58	1.8
60	<0.20	<0.20	<0.20	<0.20

1- Maximum concentration observed after 1st or 2nd application

Median concentrations for canal samples stations ranged from 0.29 – 4.85 µg/L on the day of application (Table 83). On the day of application, 81% of all samples collected (N=75) had fenbutatin oxide concentrations > 0.20 µg/L, 40% had concentrations > 1.0 µg/L, and 9% had concentrations > 10 µg/L (Appendix 15.6-15.9 *in* Wallace et al 1993).

Table 83. Median concentrations of fenbutatin oxide detected in orchard lateral and perimeter canals

Days after application	Median concentration of fenbutatin oxide detected (µg/L) [1]			
	Grove			
	2	3	4	5
0	2.25-0.65	2.05-4.85	0.61-1.5	0.29-1.5
1	<0.20-0.35	<0.20-1.8	0.20-0.69	0.35-0.91
2	<0.20-0.28	<0.20	0.26-0.26	<0.20-0.53
4	<0.20	<0.20	<0.20	0.21-NR[2]

1- Range includes medians measured after first and second application

2- Median not reported for day 4 following the second application on grove 5

Sediment. Concentrations of fenbutatin oxide in sediment of aquatic habitats near the treated groves were also evaluated at depths of up to 5 cm. The highest fenbutatin-oxide concentration observed was 4.9 ppm collected in the top cm of sediment from a pond on grove 1, two days before the first application. However, the concentrations in pond sediment tended to be lower than that observed in lateral and perimeter canals where the median post-application concentrations over all sample segment depths, times, and groves ranged from 0.018 to 0.320 ppm. The maximum concentration observed in the canal sediments was 1.6 ppm, observed at the last sampling time point (60 days after the second application).

Adverse biological effects. No fish kill or other apparent adverse biological effects attributable to fenbutatin oxide were observed. However, the authors characterize these observations as qualitative. Our review suggests any conclusions based on these observations should not be given much weight as they appear to be primarily incidental notes rather than a scientific evaluation of response. The report suggests "observations of potential nontarget effects were made by walking along grove canals, or around grove pond 1, and recording dead or moribund organisms observed." However, the extent, location, or timing of any biological surveys that may have been conducted during the course of this study were not reported.

Conclusions. The study authors concluded that off-target aerial deposition was the major route of transport of fenbutatin oxide to aquatic habitats, and the level of exposure was dependent upon the proximity to the target area. Surface water near the target sites had peak concentrations as high as 100 μg/L and frequently exceeded 1 μg/L. In some cases fenbutatin oxide persisted in surface water (>0.20 μg/L) for several days or weeks. The study authors noted that fenbutatin-oxide concentrations in sediment were variable and tended to reflect the levels of exposure observed in the spray depositions and water sample matrices. They also suggest a possible increasing trend in fenbutatin oxide in canal sediments over the duration of the study which would be expected given its persistence. We anticipate salmonids will be exposed to fenbutatin concentrations similar to those observed in the study. Floodplain habitats have similar hydrological characteristics as the study's canals and ponds. Application conditions (wind speeds <10 mph), methods (air blast), and rates (2 lbs a.i./A) are comparable to current labeling for fenbutatin oxide indicating that similar concentrations are likely in floodplain habitats proximate to application areas.

Current labels specify an interval of 30 days before the next spray event with a maximum season application of 3 lbs a.i. /A, whereas the study applied 4 lbs a.i. /A during the study with an interval of 60 days between applications. The aerial drift of fenbutatin oxide into aquatic areas observed in this study was comparable to that predicted using AgDrift modeling. For example, simulations for dense orchards evaluating drift to a 1-acre pond with no-spray buffers of 5-15 m (as reported for this study) predict drift of 0.5 – 0.9% of the target application rate, versus the observed maximum drift of 0.25 – 2.78% for the pond (Grove 1). Additionally, drift to a 2 m wide canal with reported no-spray buffers of 1-30 m predict maximum drift of 0.6 – 8% of target, versus observed maximums of 2 – 36% of target observed in canals (Groves 2-5). Although the AgDrift model slightly underestimates the actual peak concentrations observed during this study, particularly for the canals, it provided reasonable estimates for off-site transport of the pesticide.

In summary, the study results are applicable to anticipated application events throughout the action area and in particular where floodplain habitats are proximate to use areas. We anticipate that concentrations may achieve levels as high as 100 μg/L fenbutatin oxide 24 hours after application which is two orders of magnitude greater that the lowest reported LC_{50} for salmonids.

25.1.2 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.

25.1.2.1 Metabolites and degradates of three a.i.s

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

25.1.2.2 Other ingredients in formulated products

NMFS reviewed all of the active labels for the three a.i.s and found one label that contained multiple a.i.s (EPA Reg. # 61483-91). This is a diflubenzuron product formulated with permethrin that is wiped, spray-applied, or poured on to cattle and horses. Transport to surface water may occur from washoff when animals are submerged in water or via runoff from rainfall. We are uncertain to what degree such application methods may contribute to exposure of aquatic organisms to diflubenzuron and permethrin, but we presume it is significantly less than that predicted for broadcast spray applications modeled above. None of the fenbutatin or propargite products contained more than one active ingredient. End-use products that contain one of the three a.i.s are also composed of other chemicals that frequently account for a significant portion of the formulation (*e.g.* 10 - 99 % by weight).

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

Table 84. 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 three 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, chloropicrin, chlorothalonil). Inerts can be more than 50% of the mass of pesticide products, and millions of pounds of these products are applied to the landscape each year (Koplin et al. 2002). This equates to large contaminant loads of inerts that may adversely affect salmon or their habitat. Uncertainty regarding exposure to these ingredients will be qualitatively incorporated into our analysis.

25.1.2.3 Tank Mixtures

Tank mixtures are authorized unless specifically prohibited on the product label. Tank mixture ingredients are considered part of the action because they are authorized by EPA's approval of the FIFRA label. Exposure to, and risk associated with, ingredients in tank mixtures were not addressed in EPA's BEs and remain a significant source of uncertainty. We evaluated the existing California pesticide use information and found that diflubenzuron, fenbutatin oxide, and propargite were co-applied[17] with other pesticides 29%, 58%, and 70% of the time since 1999. Additionally, we found that the three a.i.s were frequently applied with other pesticides that NMFS has concluded are likely to jeopardize listed salmonids and adversely modify their designated critical habitat (Table 96). For example, propargite has been co-applied with chlorpyrifos more than 9000 times in California since 1999 (~8% of propargite applications). The data also indicate that each of the three a.i.s have been applied in hundreds of unique pesticide mixtures.

[17] Co-applications include applications of more than one pesticide as either a tank mixture or as separate applications of more than one pesticides to the same field, on the same day.

25.1.2.4 Environmental Mixtures

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

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

25.1.3 Use of Best Available Scientific and Commercial Data to Define Exposure to Listed Pacific Salmonids and Their Designated Critical Habitat

We consider several lines of exposure evidence that constitutes the best available scientific and commercial data. As discussed earlier, each type of information has inherent limitations and uncertainties. While each source of information contributes to the exposure characterization, the available information is not equivalent in terms of its scientific quality and relevance for assessing the action, EPA's authorization of pesticide use through FIFRA product labeling. Table 85 provides information describing how we weighed the available information based on its quality and relevance. Estimates that were determined to be most useful for characterizing exposure to salmonids and their designated critical habitat were given the most weight (primary). These included estimates derived from AgDrift, the RICE model, and measured values associated with targeted surface water monitoring. PRZM-EXAMS estimates were given a moderate amount of weight (secondary). While pertinent, these values are less useful for assessing the potential range of exposure because salmonids occupy habitats that can achieve higher concentrations and maximum application rates allowed were not assessed for several authorized uses. Ambient monitoring was given minimal weight (tertiary) in our exposure characterization. Although the ambient data are useful for documenting baseline conditions and the occurrence of pesticides associated with past product use, past product use is not a reliable indicator of future product use and the values derived with ambient monitoring are not considered useful for predicting either the magnitude or extent of exposure to listed salmonids or their habitat.

Table 85. Qualitative consideration of exposure information

Estimate Type	Exposure Underpredict	Exposure Overpredict	Weight[1, 2, 3]	Supporting
Fate and Transport Models				
AgDrift	With buffer <1000 ft	With buffer >1000 ft	Primary	Labels Field Trials: (Bird et al. 2002) Targeted monitoring
PRZM-EXAMS	Site specific e.g. floodplain habitats and small streams	Site specific e.g. higher volume habitat and high flow habitats	Secondary	Labels EPA SAP Field trials
RICE model discharge concentration	No[1]	No[1]	Primary	Labels Dissipation rate
Monitoring				
Targeted	No[1]	No[1]	Primary (fenbutatin oxide only)	Labels Field study: (Wallace et al. 1993) Site-specific use
Ambient	Consistently	Rarely	Tertiary (propargite only)	Usage estimates
Co-occurrence				
Spatial and Temporal overlap	New uses added to label	Not used where permitted	Primary	Labels GIS Data: Species Range Critical Habitat Range USDA NLCD

1- We do not expect over, or under prediction of exposure other than normal site-specific and application-specific variability

25.1.4 Exposure Conclusions

Diflubenzuron, fenbutatin oxide, and propargite have registered uses that occur within the distribution of all 28 listed Pacific salmonids. 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. Product labels for the three a.i.s allow application of pesticides in close proximity to these habitats. Therefore we expect some individuals of each of these species, and their designated critical habitats, will be exposed to the three a.i.s and other stressors during the 15 year duration of the action. The level of exposure will vary among the three a.i.s and species. We expect salmonid exposure to fenbutatin oxide and propargite will be primarily associated with applications to crops on agricultural lands. Whereas, exposure to diflubenzuron will occur from applications to developed, undeveloped, and agricultural lands for a variety of different uses. All three a.i.s may be spray applied in close proximity to salmonids and their designated critical habitat suggesting drift is a likely pathway for exposure. We also consider runoff a likely pathway of exposure although environmental fate characteristics suggest it may be less important than drift as none of the three a.i.s are particularly mobile in soil. Additionally, discharge of surface water is a relevant exposure pathways for diflubenzuron in California where it is approved for use in rice. All other aquatic uses, including applications to control mosquitos and midges will be canceled.

Table 86. Active ingredient ranges in exposure data inform monitoring data and modeling estimates

Active Ingredient	Land Use	Exposure Value (µg/L)	Source
Diflubenzuron	All	Rights-of-way Acute exposure: 6 – 8 Chronic exposure: 2 – 4	PRZM-EXAMS (9.1.5.1)
		Rights-of-way Acute exposure: 0.02 – 18	AgDrift (9.1.5.3)
	Agriculture	Rice Discharge day 14: 0.35 - 10	Estimation of rice discharge (9.1.5.2)
		Noncrop Acute exposure: 0.12 - 7 Chronic exposure: 0.04 - 4	PRZM-EXAMS (9.1.5.1)
		Noncrop Acute exposure: 0.02-18	AgDrift (9.1.5.3)
		Crops Acute exposure: 0.02 – 2 Chronic exposure: 0.01 – 1	PRZM-EXAMS (9.1.5.1)
		Crops Acute exposure: 0.25 - 53	AgDrift (9.1.5.3)
	Undeveloped	Forests Acute exposure: 0.58 – 0.80 Chronic exposure: 0.19 – 0.45	PRZM-EXAMS (9.1.5.1)
		Rangeland, Forests Acute exposure: 0.02 – 18	AgDrift (9.1.5.3)
	Developed	Residential, Urban Acute exposure: 0.08 - 34 Chronic exposure: 0.03 - 22	PRZM-EXAMS (9.1.5.1)
		Landscaping, Parks Acute exposure: 0.02 – 18	AgDrift (9.1.5.3)
Fenbutatin oxide	Agriculture	Crops Acute exposure: 3 - 69 Chronic exposure: 1 - 55	PRZM-EXAMS (9.1.5.1)
		Crops Acute exposure: 0.12 - 67	AgDrift (9.1.5.3)
		Citrus Maximum/site: 19 - 111 Day 0, Median: 0.15 – 4.85	Targeted Monitoring (25.1.1.1)
	Developed	Nursery Acute: 0.12 - 2	AgDrift (9.1.5.3)
Propargite	All	Range: 0.003 - 20	Ambient Monitoring (9.1.6.3)
	Agriculture	Crops Acute: 0.95 - 25 Chronic: 0.26 - 7	PRZM-EXAMS (9.1.5.1)
		Crops Acute: 0.34 - 269	AgDrift (9.1.5.3)
	Developed	Ornamentals Acute: 10 - 32 Chronic: 1.5 - 5	PRZM-EXAMS (9.1.5.1)
		Ornamental/Conifer Nurseries Acute: 0.11 - 11	AgDrift (9.1.5.3)

Monitoring data may reflect pesticide applications proximate to the water body (i.e., targeted monitoring), or resulting from more distant uses in the watershed (ambient monitoring). The targeted monitoring of fenbutatin oxide show a similar range in concentrations to those predicted with PRZM-EXAMS and AgDrift modeling. The environmental fate characteristics of this a.i. suggest that it will persist and accumulate in the environment. For this reason, we assess accumulation by salmonids and their prey as a risk hypothesis.

While a significant amount of ambient surface water monitoring has been conducted for propargite, similar data were not available for diflubenzuron, fenbutatin oxide, degradates of the three a.i.s, or other stressors of the action. The propargite data were not designed to assess the proposed product label restrictions. The spatial and temporal relationships between propargite use and monitoring are either unknown or not defined at a useful scale. Additionally, sampling is not consistent with the spatial and temporal distribution of salmonids. Therefore, we use this information cautiously as part of the best available information recognizing it is not sufficient to quantify either the likely range of exposure or the probability of exposure. We conclude that previous use of propargite products have resulted in detections in surface waters within the four states where listed Pacific salmonids are distributed, suggesting exposure to listed salmonids and their designated critical habitat has occurred.

We assume that the exposure estimates provided by EPA in the BEs, and additional modeling and targeted monitoring information provided above, represent realistic exposure levels for some individuals of each of the listed Pacific salmonids (Table 86).

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 salmonids occupy habitats that could contain high concentrations of these pesticides at one or more life stages. However, the extent of potential pesticide use and the time spent in these areas varies among the species. We are unable to accurately define exposure distributions for the three a.i.s and the other stressors of the action given limitations of the available information. We assume the highest probability of exposure occurs in freshwater habitats near locations where the pesticides are applied.

Defining exposure of the stressors of the action to the listed species is complicated by uncertainty associated with the following factors:

- Information to accurately characterize exposure to pesticides, degradates and other ingredients is largely unavailable;
- Exposure estimates are not available for some application methods (e.g. green house applications);
- Available PRZM-EXAMS estimates frequently did not assume application rates consistent with current or proposed product labels (Estimates based on application rates that exceed the maximum allowable single application rate for a particular Land Use category were censored from the analysis. Estimates that are less than the maximum allowable rate were still considered);
- Product labels authorize the application of chemical mixtures that are not specified or clearly defined (*e.g.*, the ingredients of pesticide formulations are not fully disclosed, labels recommend tank mixture applications with other products, and tank mixtures with other pesticides are permitted unless specifically stated otherwise);
- Information to accurately characterize past and current use of pesticide in Idaho, Oregon, and Washington is unavailable. Pesticide use patterns are likely to change during the 15 year duration of the action. Historic information on the frequency of use, locations of use, and the amounts of pesticides applied may not reflect future use of the three a.i.s.

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

25.2 Response Analysis

In this section we evaluate toxicity information from the stressors of the action and organize them into assessment endpoints which target potential effects to individual salmonids and their supporting habitats. The assessment endpoints represent biological and habitat attributes that, when adversely affected, lead to reduced fitness of individual salmonids or degrade attributes of the Primary Constituent Elements (PCEs) such as prey abundance and water quality (evaluated in R*isk Characterization for Designated Critical Habitat* section). Uncertainties in the available toxicity information are discussed as they are encountered and identified at the end of this section. Following the response analysis, we compare anticipated environmental concentrations described in the exposure analysis with assessment endpoints to evaluate whether individual fitness or habitat endpoints might be compromised. Salmonid and designated critical habitat risk hypotheses are evaluated separately in the *Effects of the Proposed Action on Designated Critical Habitat* section.

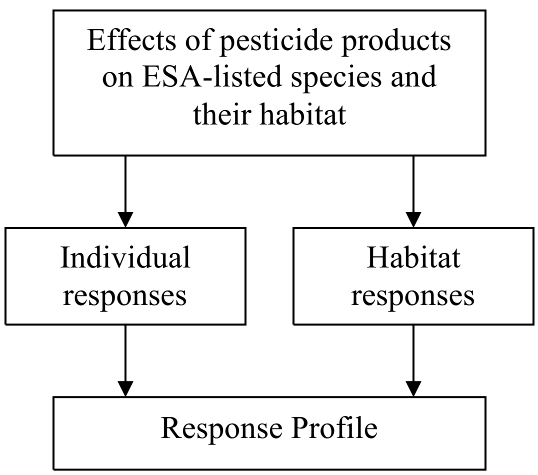

Figure 70. Response Analysis Conceptual Model

We begin the response analysis by describing the toxic mode and mechanism of action of diflubenzuron, fenbutatin oxide, and propargite which sets the stage for which biological endpoints we assess. Next, we summarize the toxicity data presented by EPA including salmonid BEs, REDs, IREDs, the California Red Legged Frog BE, EFED science chapters, industry-supplied data, and open literature (Table 88, Table 90, and Table 91). The information is organized by assessment endpoints (*e.g.*, survival, growth, migration, reproduction, etc.). The information provided by EPA primarily addressed aspects of survival, growth, and reproduction of aquatic species measured by laboratory methods following exposure to the three insecticides. Other information from field experiments on ecological endpoints was sometimes discussed within documents. Little information regarding formulation, other ingredients, and mixture toxicity was reported. We supplemented the information found in EPA documents with additional response data obtained through literature reviews.

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

25.2.1 Modes and Mechanisms of Action

The three insecticides have different modes of toxic action (Table 87). Each is acutely toxic to aquatic invertebrates including insects and crustaceans. Propargite and fenbutatin oxide are highly toxic to fish as measured by acute lethality experiments. Little is known about the specific mechanism of toxicity within invertebrates and fish for both fenbutatin oxide and propargite.

Diflubenzuron is a benzylphenylurea insecticide, and is a member of a larger group of chemicals known as insect growth regulators. Diflubenzuron inhibits the biosynthesis of chitin, which is a principal component of the tough outer covering (or cuticle) of arthropods (including aquatic and terrestrial insects and crustaceans) as well as the internal structures of some invertebrates. The exact mechanism of action is still not fully elucidated (Merzendorfer and Zimoch 2003), and research supports several hypotheses (Marx 1977, Ivie 1978, Crookshank et al. 1978, Cunningham 1986). However, it is clear that the disruption of the normal chitin-producing pathway in chitin-producing organisms results in the improper formation and deposition of new cuticle. Death occurs when the organism can neither shed its old cuticle nor properly deposit new cuticle. This mode of action is critical to immature stages of arthropods that are actively molting (i.e., shedding their old cuticle) as they grow and develop into mature stages. Additionally, at lower concentrations, diflubenzuron can cause malformed mouthparts, which results in insects dying of starvation due to an inability to feed. Vertebrates (including mammals and fish) do not synthesize chitin, and therefore generally show low toxicity to diflubenzuron. In mammals, diflubenzuron causes formation of abnormal hemoglobins called methemoglobin and sulfhemoglobin. These compounds have a reduced ability to transport oxygen and can result in anemia. This could be a relevant toxic pathway for fish, but there are few studies to support this mode of action.

Fenbutatin oxide is an organotin whose precise mechanism of toxic action to vertebrates and invertebrates is unknown. However, its mode of action is assumed to be the inhibition of adenosine triphosphate (ATP) synthase resulting in disruption of cellular respiration (i.e., oxidative phosphorylation). This affects metabolic processes and energy production at the cellular level (i.e., the electron-transport chain ceases to operate). Mites and spiders are extremely susceptible to fenbutatin oxide and are the primary target organisms (i.e. non-systemic acaricide). In addition, fenbutatin oxide appears to inhibit photophosphorylation in chloroplasts, the chlorophyll-bearing subcellular units, and could therefore affect primary production (Ma 2005). While toxicity data exist for other organotins (e.g. tributyltin), organotin compounds are very diverse due to large variation in their organic moiety and do not necessarily share the same mechanisms or modes of toxic action. For this reason, we do not report the toxicity of other organotins.

Propargite is an organo sulfite compound with ATPase-inhibiting properties (Sanchez-Bayo 2011). It is the only organo sulfite chemical that is subject to reregistration under FIFRA and we have located no other pesticides that have a similar chemical profile (EPA RED Propargite). Thus we do not anticipate propargite sharing a similar mode or mechanism of action with current-use pesticides.

Table 87. Information on toxicological properties of diflubenzuron, fenbutatin oxide and propargite

Insecticide	Pesticide Class	Mode; mechanism of action	Description of toxicological effect to target organisms	References
Diflubenzuron	Benzylphenyl urea	Insect growth regulator; inhibits chitin biosynthesis	Inhibition of chitin biosynthesis, resulting in improper formation and deposition of new cuticle. Organisms die when old cuticle is not shed or new cuticle is improperly deposited. Targets arthropods, crustaceans, and aquatic and terrestrial insects.	(Farlow 1976, Marx 1977, Ivie 1978)
Fenbutatin oxide	Organotin	Assumed to disrupt cellular respiration; unknown	Not provided; non-systemic pesticide that targets mites and spiders	(Ma 2005)
Propargite	Organosulfite	Mitochondrial ATPase inhibitor; unknown	Not provided	(Sanchez-Bayo 2011)

25.2.2 Temperature and toxicity

Changes in temperature are known to affect the magnitude of toxic effects in fish. Differences in toxicity due to temperature have been attributed to differences in respiration rate, chemical absorption, metabolism, and binding affinity. Toxicity increases with temperature for pesticides that are transformed/metabolized in fish to more toxic metabolites such as organophosphates (Mayer and Ellersieck 1986). Alternatively, by decreasing the binding affinity to the target (e.g. pyrethroid binding to sodium channels; (Harwood et al. 2009)) increased temperature can decrease toxicity. We located no information showing specific effects of temperature on diflubenzuron or propargite toxicity. The only available data on temperature for fenbutatin oxide shows a slight decrease in toxicity in channel catfish with elevated temperature (LC_{50}s of 1.5 µg/L and 4.0 µg/L at temperatures of 17 °C and 22 °C, respectively (Mayer and Ellersieck 1986).

As discussed in the *Environmental Baseline*, temperature, in and of itself, is a recognized stressor to salmonids in the Central Valley and other salmonid-supporting waters of the West Coast (Myrick and Cech 2005). Water temperatures in the lower Sacramento River regularly exceed 20 °C by late spring, and statistical studies of coded wire-tagged juvenile Chinook show increased mortality as a function of temperature (Baker et al. 1995). Water temperatures higher than optimum levels can kill salmonids, increase physiological stress making them more susceptible to other stressors, increase predation, and affect salmonids' prey base. Thus, temperature directly affects survival, growth rates, distribution, and developmental rates. We therefore discuss salmonid fitness implications in the context of temperatures potentially exacerbate the effects caused by the three insecticides.

25.2.3 pH and toxicity

Propargite degrades (hydrolyzes) more rapidly under alkaline hydrolytic conditions (half-life = 2.2 d) compared to both neutral (half-life = 75 d) and acidic (pH 5 half-life = 120 d) conditions. Although we found no information that toxicity is affected by pH, the longer it persists in the aquatic environment the greater the probability that salmonids and their habitats may be exposed and negatively affected. Aquatic habitats throughout salmonids' distribution experience acidic, neutral, and alkaline pHs, typically pH may range from 6 to 9. Fenbutatin oxide appears stable to changes in pH. Diflubenzuron's persistence is affected by pH, but much less so than propargite. Diflubenzuron is fairly stable to abiotic degradation. Hydrolysis half-lives range from 30-433 days for pHs 5-9.

25.2.4 Insecticide effects to salmonids and their habitats

Fish can consume a very high proportion of the invertebrate secondary production in aquatic habitats (Huryn 1996, Huryn 1998). Juvenile salmonids consume a wide range of invertebrates, including those from all functional feeding groups. Changes in abundance of any of these groups could change prey availability for these fish. Insecticides may kill or injure aquatic insects and other macroinvertebrates that serve as food for rearing juvenile salmonids of all five species and adult steelhead. Lack of food may affect a salmonid's growth and development, ultimately affecting their ability to complete their life cycle.

Juvenile salmonids are generally opportunistic drift-feeders, and are therefore sensitive to factors that influence the general quantity and quality of invertebrate prey items. If, for instance, there were reductions in the production of invertebrate grazers or the inputs of invertebrate prey from riparian vegetation, salmonids may be forced to alter their foraging behavior (*e.g.,* take more risks, select less energy-rich prey). Alternatively, changes in abundance and composition may have minimal impacts to salmonids if they do not alter the overall quality or quantity of prey, or impact foraging behaviors. Whether or not production of prey decreases or shifts (or increases) after exposure to insecticides will depend in part on the composition of the community (structure and function) and the relative sensitivities of those taxa to diflubenzuron, fenbutatin oxide, and propargite. Multiple experiments conducted in mesocosms have demonstrated that the particular composition of the community at the time of exposure influences the magnitude of the impact as well as the trajectory of the recovery (Colville et al. 2008, Downing et al. 2008, Heckmann and Friberg 2005, Hessan et al. 1994, Lytle and Lytle 2002, Maund et al. 2009, Rohr and Crumrine 2005, Schulz et al. 2003b, Schulz et al. 2003a, Van den Brink et al. 1996b, Van den Brink et al. 2007) and this would likely be the case in salmonid habitats.

Mixtures of pesticides present a particular challenge in assessing impacts on salmon habitat. Most of the experiments described above were conducted in mesocosms with a single exposure of a single insecticide. In field surveys in the United States as well as throughout Europe, herbicides are often among the most concentrated pesticides detected, but they are almost always found in mixtures with insecticides and fungicides (Gilliom 2007, Schafer et al. 2007).

A final consideration and uncertainty in how insecticides may impact salmonids and their habitats is the question of resiliency of these aquatic ecosystems. The recovery of secondary production – to rates observed prior to exposure – depends on the communities themselves and the exposure. For example, univoltine species of macroinvertebrates (i.e. that produce one generation/year) will require a long time to recover. Additionally, if insecticides persist in the landscape, exposures may occur repeatedly (or continuously) depending on application rate, precipitation, and conditions in the watershed. In habitats that receive insecticide inputs repeatedly throughout the year, salmonid prey may be chronically suppressed.

25.2.5 Direct Effects to Salmonids

We evaluate effects to salmonids based on toxicity information presented in many sources including Pesticide Registration Eligibility Decisions (EPA 1994a, EPA 1997b, EPA 2001b), salmonid Biological Evaluations (EPA 2002a, EPA 2002c, EPA 2004a), the California red-legged frog Biological Evaluations (EPA 2008, EPA 2009c), other EPA reports (EPA 2007b, EPA 2009a), government agency reports (SERA (Syracuse Environmental Research Associates 2004, USFWS 1992), international agency reports (Authority 2010), ECOTOX database, industry-submitted studies, and open literature.

25.2.5.1 Survival

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

In the case of ESA-listed Pacific salmonids, there are several surrogates including hatchery reared coho salmon, Chinook salmon, steelhead, and chum salmon, as well as rainbow trout[18]. We consider the range in response observed in evaluations with these surrogates to characterize the likely response of listed salmonids. We evaluate the likelihood of concentrations that are expected to kill fish and apply qualitative and quantitative methods to infer population-level responses of ESA-listed salmonids within the *Risk Characterization* section. The three insecticides (in technical products and formulations) have been evaluated in acute lethality toxicity tests with numerous fish species (Table 2, Table 3, Table 4).

25.2.5.2 Growth and Reproduction

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

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

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

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

25.2.5.3 Swimming

Swimming is a critical function for anadromous salmonids to complete their life cycle. Impairment of swimming may affect feeding, migrating, predator avoidance, and spawning. It has been used to assess behavioral responses of fish to various toxicants, including pesticides (Little and Finger 1990). Swimming capacity is a measure of orientation to flow as well as the physical capacity to swim against it (Howard 1975, Dodson and Mayfield 1979). Swimming activity includes measurements of frequency and duration of movements, speed and distance traveled, frequency and angle of turns, position in the water column, and form and pattern of swimming. Little and Finger (1990) concluded that swimming-mediated behaviors are frequently adversely affected at 0.3 – 5.0 % of reported fish LC_{50}s, and that 75% of reported adverse effects to swimming occurred at concentrations lower than reported LC_{50}s. We located no studies that measured impacts to salmonid swimming behaviors from exposure to diflubenzuron, fenbutatin oxide and propargite.

25.2.5.4 Olfaction

Olfaction conveys critical environmental information that fishes use to mate, locate food, discriminate kin, avoid predators, and home (i.e., navigate). Any or all of these essential olfactory-mediated behaviors may be affected by exposure to contaminants such as pesticides (reviewed by(Tierney et al. 2010). For example, copper impairs and destroys salmonid olfactory sensory neurons in a matter of minutes at low μg/L levels and effects persist for hours to weeks depending on exposure concentration and duration (Baldwin et al. 2003). Measured behavioral effects in salmonids from impaired olfaction include compromised alarm response, loss of ability to avoid copper, interrupted spawning migrations, loss of homing ability, and delayed and reduced downstream migration of juveniles (Baldwin et al. 2003, Baldwin et al. 2011, Hansen et al. 1999, McIntyre et al. 2008, Mebane and Arthaud 2010, Sandahl et al. 2004). Disruption of these essential behaviors reduces the likelihood of an individual salmonid completing its life cycle. We located no study results that evaluated olfactory responses in fish following exposure to the three insecticides.

25.2.5.5 Insecticide effects on riparian vegetation and aquatic primary producers

We evaluate the available information to assess whether riparian vegetation and aquatic primary producers may be affected by the three insecticides. Insecticides do not typically have herbicidal activity, however some may. Riparian vegetation is important for providing shade to the stream, stabilizing the stream banks, reducing sedimentation, and providing organic material inputs, both in terms of plant material and terrestrial insects. Riparian vegetation is a major focus of restoration efforts within California, and when present can reduce pesticide loading into aquatic resources. Riparian vegetation is an important assessment endpoint for herbicidal impacts on salmon habitats. Generally there are sparse data regarding the effects of herbicides (and much less with insecticides) on wild plants within riparian systems, other than weed species. EPA requires submission of crop effects data as part of the registration process for herbicides (EPA 1996). This information currently provides the only basis for evaluating effects on herbaceous plants unless data are available from other sources. The overall assumption is that the sensitivity of plant species tested (typically plants used in agriculture) in the registrant-provided guideline studies will be representative of riparian species. There is no way to know this is the case, therefore a high degree of uncertainty regarding the toxicity of the three insecticides to riparian vegetation exists. We note that an EPA Science Advisory Panel in 2001 (approximately 12 years ago) was convened to review non-target plant toxicity tests under the North American Free Trade Agreement. A host of recommendations were made as well as several research recommendations to fill identified data gaps. We could not confirm if any of the recommendations or research initiatives were undertaken, although if implemented, several would have informed the current consultations with effects on aquatic and riparian plants (EPA 2002b).

We also evaluate if and to what extent aquatic primary producers are affected by the stressors of the action. Primary producers including periphyton, diatoms, marcrophytes, and plankton are integral components of aquatic food chains, serving as food for salmonid prey. Reductions in primary productivity may lead to impacts to salmonid prey. Although insecticides are typically not tested for effects to freshwater and marine primary producers, we search for and evaluate any information on effects to primary producers.

25.2.6 Toxicity of diflubenzuron (Assessment Endpoints)

We located numerous studies that measured standard assessment endpoints in several species of fish, freshwater and estuarine invertebrates, and aquatic plants (Table 2). Studies described acute (short-term) and chronic (longer-term) exposures to diflubenzuron, the end-use product Dimilin, and degradates including 2,6-difluorobenzoic acid, 4-chlorophenyl urea, and 4-chloroaniline. Most data were from registrant submitted guideline studies, EPA reports, and open literature studies. Very little information was located on the toxicity of diflubenzuron's degradates. We located a few open literature studies investigating the toxicity of diflubenzuron when mixed with other pesticides. Many mesocosm and field studies were located that measured toxicity to aquatic communities including aquatic invertebrates, insects, and fish. In particular, we located five studies that directly measured fish growth following diflubenzuron application. Despite the large number of relevant studies located, significant data gaps still exist on biological and ecological assessment endpoints such as swimming, olfactory-mediated behaviors, migration, spawning, time-to-first feeding, smoltification, and impacts to riparian vegetation.

25.2.6.1 Direct Effects to Salmonids

25.2.6.1.1 Survival

We located numerous studies that measured fish survival following short-term (96 h or 24 h) diflubenzuron exposure. Twenty-two studies tested the lethality of diflubenzuron to salmonids including cutthroat trout, rainbow trout, Atlantic salmon, brook trout, steelhead trout, and coho salmon. All of the tested fish species appeared insensitive to diflubenzuron. Six of the studies used technical diflubenzuron, while the other sixteen studies used various formulations. Salmonid LC_{50}s ranged from 57,000 to greater than 1,000,000 μg/L, and had a mean and median value of about 190,000 μg/L. There was no clear trend of increased or decreased toxicity between the technical product and formulations. Survival data was located for many non-salmonid species including yellow perch, fathead minnow, bluegill sunfish, channel catfish, common carp, guppy, sheepshead minnow, silver catfish, and mummichog from twenty-four studies. Twelve of those used technical diflubenzuron, while the other twelve used various formulations. Reported 96 h LC_{50}s ranged from 32,990 to greater than 1,000,000 μg/L with a mean value of 292,000 μg/L and a median value of 255,000 μg/L. As with salmonids, there was no obvious trend of increased or decreased toxicity between technical and formulated products.

25.2.6.1.2 Reproduction or larval survival

Two registrant-submitted studies were located that measured reproduction in fathead minnow and mummichog. These studies reported a NOEC of 100 and 50 μg/L, respectively. However, the study using mummichog was noted in the EPA RED (1997) as “did not provide adequate test of the effects of diflubenzuron on reproductive success". Therefore, this study was not carried forward in our analysis of diflubenzuron.

25.2.6.1.3 Growth

We located one peer-reviewed study that measured fish growth after long-term exposure. A NOEC of 45 µg/L, the highest concentration used in this study, was measured in steelhead trout following a 9 week exposure to technical diflubenzuron (Hansen and Garton 1982a). Two additional peer-reviewed studies were located that measured fish growth in laboratory conditions. One study found no effect on adult mosquitofish growth at exposure concentrations up to 78 ng/L (Zaidi and Soltani 2011), while the second study (Draredja-Beldi and Soltani 2003) measured reduced growth in juvenile mosquitofish exposed to 78 ng/L diflubenzuron. These studies are discussed further in Table 3.

Additionally, four peer-reviewed mesocosm, enclosure and pond studies measuring growth in bluegill sunfish (Hanratty and Liber 1996, Tanner and Moffett 1995, Boyle et al. 1996) and striped bass (Ludwig 1993), as well as one agency-submitted report measuring growth in bluegill sunfish (MRID 44386201) were located. Although these studies did not measure standard toxicity endpoints (e.g. LC_{50}), they reported effects on fish growth at concentrations ranging from 2.5 to 30 µg/L. The duration of these studies ranged from about 8 to 16 weeks, and all used a 25% diflubenzuron formulation. Decreased growth was attributed to changes in prey abundance. These studies are discussed further in the following *Risk Characterization* section.

25.2.6.1.4 Development

One peer-reviewed study was located that measured development of fathead minnow eggs following 30 days of exposure to technical diflubenzuron. The authors reported no statistically significant effects and concluded that the NOEC was 45 µg/L. We also note that diflubenzuron did not affect survival, morphology, or length in a zebrafish early life stage experiment at concentrations up to 2 mg/L (Appendix 3).

25.2.6.1.5 Salmonid prey

Numerous studies were located that measured the survival and abundance of aquatic insects following exposure to diflubenzuron. Many of these studies indicate that early developmental stages of crustaceans and aquatic insects are more sensitive to diflubenzuron than adult stages because immature stages molt frequently as they develop. Therefore, tests using immature invertebrates typically had much lower toxicity values than those using adult stages. Approximately fifteen registrant-submitted and peer-reviewed studies were found on species including crustaceans, amphipods, chironomids, cladocerans, copepods, and other insects. Diflubenzuron appears much more toxic to aquatic invertebrates than to fish. No consistent differences were observed between the toxicity of technical diflubenzuron and formulated products. Acute LC_{50} values ranged from 0.15 to 1937 µg/L for 48 h tests; 0.75 – 57,000 µg/L for 96 h tests; and 0.0028 – 2123 µg/L for 24 h tests. Three longer studies (5 and 7 d) were located that report LC_{50} values of 1.02 and 1.79 µg/L in two species of midge and 50 µg/L in dragonfly nymphs. An additional 21-d life-cycle study using *Daphnia magna* calculated an LC_{50} of 0.062 µg/L. Six agency-submitted reports measuring NOELs and LOELs in aquatic invertebrate life-cycle tests reported values ranging from 0.04 to >10 µg/L. Species tested include waterfleas, brine shrimp, and mysids. The test durations were not stated, but life-cycle tests are typically 21 d.

25.2.6.1.6 Estuarine prey survival

We located several studies measuring the survival of estuarine invertebrates including *Palaemonetes, Mysidopsis* and *Eurytemora*. These invertebrates may be important salmonid prey for juveniles during outmigration and estuarine rearing periods. LC_{50} values (48, 72 or 96 h) ranged from 0.78 to 2.95 µg/L, and no differences were seen between the toxicity of technical product and formulations. In chronic studies using mysids, one study on *Mysidopsis* reported a 21 d LC_{50} of 1.24 µg/L, while a second chronic study on *Americamysis* (the new genus name for *Mysidopsis*) reported a LOEL of 0.086 µg/L and a NOEL of 0.045 µg/L.

25.2.6.1.7 Herbicidal effects

Three studies measuring effects on aquatic plants were found. Plants tested included algae, freshwater diatoms, duckweed, phytoplankton, periphyton, and macrophytes. Aquatic plants were generally not affected by diflubenzuron, as evidenced by EC_{50} values of 5000 µg/L for survival and LOEC values greater than 30 µg/L. The actual threshold for effects to plants are unknown as NOECs were not found. One study measuring the toxicity of diflubenzuron to a marine diatom reported a NOEC of 270 µg/L following a 5 d exposure. No studies were located using diflubenzuron formulations.

Table 88. Diflubenzuron toxicity values (µg/L) for aquatic organisms and plants reported in EPA salmonid BE, CRLF BE, RED, agency reports, ECOTOX, and open literature studies. Abbreviations as follows: a.i. = active ingredient; NR = Not Reported; T= Technical grade; F = Formulated product; WP = wettable powder, sw = estuarine/marine species; [ ] = 95% Confidence interval.

Assessment Endpoint		Concentration (µg/L aquatic tests; lbs a.i./acre terrestrial tests)		
		Diflubenzuron		Degradates of diflubenzuron (PCA, DFBA, PCPU)
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
Survival	Salmonid LC_{50} (24 h)		Cutthroat trout (*Salmo clarki)* (25% WP) = 77,000 [62,700-94,600]; (Mayer and Ellersieck 1986) Cutthroat trout (*Salmo clarki)* (25% WP) = 75,000 [61,000-92,100]; (Mayer and Ellersieck 1986) Rainbow trout (*Salmo gairdneri)* (25% WP) = 560,000 [408,000-769,000]; (Mayer and Ellersieck 1986)	
Survival	Salmonid LC_{50} (96 h)	Rainbow trout (*O. mykiss*) (T) = 140,000; MRID 00056150 Rainbow trout (*O. mykiss*) (95%; T) = >100,000; MRID 40094602 (Johnson and Finley 1980) Rainbow trout (*Salmo gairdneri)* (95%; T) = >100,000; (Mayer	Cutthroat trout (*O. clarki*) (25% WP) = 57,000 [48,000-67,000]; MRID 40098001 (Johnson and Finley 1980) Cutthroat trout (*Salmo clarki)* (25% WP) = 57,000 [48,200-67,400]; (Mayer and Ellersieck 1986) Cutthroat trout (*Salmo clarki)* (25%	4-Chlorophenyl urea: Rainbow trout (*O. mykiss*) (25% WP) = 72,000 [57,000-90,000]; (Julin and Sanders 1978) 2-6 Difluorobenzoic acid: Rainbow trout (*O. mykiss*) (25% WP) = >100,000;

Assessment Endpoint		Concentration (µg/L aquatic tests; lbs a.i./acre terrestrial tests)		
		Diflubenzuron		Degradates of diflubenzuron (PCA, DFBA, PCPU)
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
		and Ellersieck 1986) Atlantic salmon (*Salmo salar*) (95%; T) = >50,000; MRID 40098001 Brook trout (*Salvelinus fontinalis*) (95%; T) = >50,000; MRID 40094602 (Johnson and Finley 1980) Steelhead trout (*Salmo gairdneri*) (99.5 %T) = > 45; (Hansen and Garton 1982a)	WP) = 75,000 [61,000-92,100]; (Mayer and Ellersieck 1986) Rainbow trout (*O. mykiss*) (25% WP) = 240,000 [201,000-286,000]; MRID 00041709 (Johnson and Finley 1980) Rainbow trout (*Salmo gairdneri)* (25% WP) = 240,000 [201,000-286,000]; (Mayer and Ellersieck 1986) Rainbow trout (*O. mykiss*) (25%; F) = 342,000; 25WP; MRID 00060384 Rainbow trout (*O. mykiss*) (25%; F) = 195,000; 25WP; MRID 00056150 Rainbow trout (*O. mykiss*) (1% granular) = >1,000,000; MRID 00060380 Rainbow trout (*O. mykiss*) (79.4%; F) = >129,000; 25WP; MRID 45252203 Rainbow trout (*O. mykiss*) (25%; WP) = 190,000; MRID 2018264	(Julin and Sanders 1978) 4-Cholroaniline: Rainbow trout (*O. mykiss*) (25% WP) = 14,000 [11,000-16,000]; (Julin and Sanders 1978)

Assessment Endpoint		Concentration (µg/L aquatic tests; lbs a.i./acre terrestrial tests)		
		Diflubenzuron		Degradates of diflubenzuron (PCA, DFBA, PCPU)
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
			Rainbow trout (*O. mykiss*) (25%; WP) = 240,000 [200,000-290,000]; (Julin and Sanders 1978) Rainbow trout (*Salmo gairdneri)* (Dimilin W-25) = >150,000; (McKague and Pridmore 1978) Coho salmon *(O. kisutch)* (Dimilin W-25) = >150,000; (McKague and Pridmore 1978)	
Survival	Non-salmonid freshwater, estuarine, and marine fish LC_{50} (96 h)	Yellow perch (*Perca flavescens)* = > 25,000; (Johnson and Finley 1980) Fathead minnow (*Pimephales promelas)* = > 500,000; MRID 00060376 Bluegill sunfish (*Lepomis macrochirus*) = 129,000 [116,000 to 142,000]; Slope: 4.7 [3.5 to 5.9]; MRID 00056150	Fathead minnow (*Pimephales promelas)* (25% WP) = >100,000; MRID 40094602 (Johnson and Finley 1980) Channel catfish (*Ictalurus punctatus)* (25% WP) = >100,000; MRID 40094602 (Johnson and Finley 1980) Bluegill sunfish (*Lepomis macrochirus*) (25% WP) = >100,000; MRID 40094602 (Johnson and Finley 1980)	**4-Chlorophenyl urea:** Fathead minnow (*Pimephales promelas)* (25% WP) = >100,000; (Julin and Sanders 1978) Channel catfish (*Ictalurus punctatus)* (25% WP) = >100,000; (Julin and Sanders 1978) Bluegill sunfish (*Lepomis macrochirus*) (25% WP) =

Assessment Endpoint		Concentration (µg/L aquatic tests; lbs a.i./acre terrestrial tests)		
		Diflubenzuron		Degradates of diflubenzuron (PCA, DFBA, PCPU)
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
		Bluegill sunfish (*Lepomis macrochirus*) (T) = 135,000; MRID 00056150 Channel catfish (*Ictalurus punctatus)* (T) = >100,000; MRID 40094602 (Johnson and Finley 1980) Bluegill sunfish (*Lepomis macrochirus)* (T) = >100,000; MRID 40094602 (Johnson and Finley 1980) Bluegill sunfish (*Lepomis macrochirus)* (T) = >100,000; MRID 00056035 (Johnson and Finley 1980) Yellow perch (*Perca flavescens)* (T) = >50,000; (Mayer and Ellersieck 1986) Fathead minnow (*Pimephales promelas)* (T) = >500,000; MRID 00060376	Bluegill sunfish (*Lepomis macrochirus*) (25% WP) = 230,000; MRID 00056150 Common carp (*Cyprinus carpio)* (25% WP) = 390,000; MRID 00060384 Bluegill sunfish (*Lepomis macrochirus*) (1% granular) = >1,000,000; MRID 00060380 Channel catfish (*Ictalurus punctatus)* (25% WP) = 370,000 [280,000-490,000]; (Julin and Sanders 1978) Fathead minnow (*Pimephales promelas)* (25% WP) = 430,000 [360,000-510,000]; (Julin and Sanders 1978) Bluegill sunfish (*Lepomis macrochirus*) (25% WP) = 660,000 [540,000-810,000]; (Julin and Sanders 1978) Silver catfish (*Rhamdia quelen*) = >1,000,000; (Kreutz et al. 2008) Mummichog *(Fundulus heteroclitus)* (25% WP) = 255,000; MRID 56150	>100,000; (Julin and Sanders 1978) **2-6 Difluorobenzoic acid:** Fathead minnow (*Pimephales promelas)* (25% WP) = 69,000 [55,000-87,000]; (Julin and Sanders 1978) Channel catfish (*Ictalurus punctatus)* (25% WP) = >100,000; (Julin and Sanders 1978) Bluegill sunfish (*Lepomis macrochirus*) (25% WP) = >100,000; (Julin and Sanders 1978) **4-Chloroaniline:** Fathead minnow (*Pimephales promelas)* (25% WP) = 12,000 [7000-18,000]; (Julin and

Assessment Endpoint		Concentration (µg/L aquatic tests; lbs a.i./acre terrestrial tests)		
		Diflubenzuron		Degradates of diflubenzuron (PCA, DFBA, PCPU)
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
		Sheepshead minnow *(Cyprinodon variegatus)* (97.6% T) = >13; MRID 42940101 Sheepshead minnow *(Cyprinodon variegatus)* (96.5% T) = >130; MRID 40262701 Fathead minnow *(Pimephales promelas)* (99.5 %T) = > 45; (Hansen and Garton 1982) Guppy *(Lebistes reticulatus)* (99.5 %T) = > 45; (Hansen and Garton 1982)	Mummichog *(Fundulus heteroclitus)* (25% WP) = 32,990 [29,010-37,250]; (Lee and Scott 1989)	Sanders 1978) Channel catfish (*Ictalurus punctatus)* (25% WP) = 23,000 [18,000-29,000]; (Julin and Sanders 1978) Bluegill sunfish (*Lepomis macrochirus*) (25% WP) = 2400 [1800-3200]; (Julin and Sanders 1978)
Repro-duction or larval survival	NOEC/ LOEC	Fathead minnow *(Pimephales promelas*) (T, 99.4%) = 100 (NOEL); MRID 00099755	Mummichog (*Fundulus heteroclitus*) (TH6040 formulation) = 50 (NOEL); MRID 00099722.	
Fish growth	NOEC	Steelhead trout *(Salmo gairdneri)* juveniles (99.5% T) = > 45 (NOEC; 30 d exposure); (Hansen and Garton 1982)		

Assessment Endpoint		Concentration (µg/L aquatic tests; lbs a.i./acre terrestrial tests)		
		Diflubenzuron		Degradates of diflubenzuron (PCA, DFBA, PCPU)
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
Fish development	NOEC	Fathead minnow *(Pimephales promelas)* eggs (99.5% T) NOEC = > 45 (30 d exposure); (Hansen and Garton 1982)		
Habitat: Salmonid prey	Invertebrate survival (48 h EC/LC_{50})	Waterflea (*Daphnia magna*) (T) = 3.7; MRID 43665801 Waterflea (*Daphnia magna*) (97.6% T) = 7.1[5-10], NOEC = 0.45; MRID 40840502 (Kuijpers 1988) Waterflea (*Daphnia magna*) (% NR; T) = 4.42 [2.79-8.63] at 200 mg/L hardness, 6.89 [3.98 – 18.5] at 100 mg/L hardness, 4.55 [2.58-12.71] at 50 mg/L hardness; (Hansen and Garton 1982) Waterflea (*Daphnia magna*) (99% T) = 2.6; MRID 45252204 Waterflea (*Daphnia magna*) (79.4%T) = 3.2; MRID 45252204	Fiddler crab, juvenile (*Uca pugilator*) = 2 (NOEC); (Cunningham and Myers 1987) Waterflea (*Daphnia magna*) 1st instar (25% WP) = 15 [10-24]; MRID 40098001 (Julin and Sanders 1978) Waterflea (*Daphnia magna*) 1st instar (25% WP) = 15.5 [12-20]; (Mayer and Ellerseick 1986) Waterflea (*Daphnia magna*) 1st instar (25% WP) = 15 [10-22]; (Mayer and Ellerseick 1986) Waterflea (*Daphnia*) mixed stages (25% WP) = 1.5; (Miura and Takahash.Rm 1974) Clam shrimp (*Eulimnadia* spp.) (25% WP) = 0.15; (Miura and Takahashi	**4-Chlorophenyl urea:** Midge (*Chironomus plumosus*) (25% WP) = >100,000; (Julin and Sanders 1978) **2-6 Difluorobenzoic acid:** Midge (*Chironomus plumosus*) (25% WP) = >100,000; (Julin and Sanders 1978) **4-Cholroaniline:** Midge (*Chironomus plumosus*) (25% WP) = 43,000 [36,000-51,000]; (Julin and Sanders 1978)

Assessment Endpoint		Concentration (µg/L aquatic tests; lbs a.i./acre terrestrial tests)		
		Diflubenzuron		Degradates of diflubenzuron (PCA, DFBA, PCPU)
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
			1974) Midge (*Chironomus plumosus*), 4th instar larvae (25 % WP) = 560 [460-680]; (Julin and Sanders 1978) Waterflea (*Ceriodaphnia dubia*) (25% WP) = 1.7 [1.36-2.02]; MRID 40130601 (Hall 1986) Fairy shrimp (*Streptocephalus sudanicus*) (Dimilin 4L) = 0.74 [0.6-0.88]; (Lahr et al. 2001) Backswimmer (*Anisops sardeus*) (Dimilin) = 1937 [1800-2020]; (Lahr et al. 2001) Waterflea (*Daphnia magna*) (25% WP) neonate = 0.75 [0.33-1.17]; (Majori et al. 1984) Waterflea (*Daphnia magna*) (25% WP) adult = 23.45 [10.75-36.15]; (Majori et al. 1984)	

Assessment Endpoint		Concentration (µg/L aquatic tests; lbs a.i./acre terrestrial tests)		
		Diflubenzuron		Degradates of diflubenzuron (PCA, DFBA, PCPU)
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
	Invertebrate survival (96 h EC/LC_{50})	Amphipod (*Gammarus pseudolimnaeus*) (95% T) = 45 [34-59]; MRID 40098001 (Mayer and Ellerseick 1986) Amphipod (*Gammarus pseudolimnaeus*) (95% T) = 30; MRID 40094602 Amphipod (*Gammarus pseudolimnaeus*) (95% T) = 30 [21-43] mg/L; (Johnson and Finley 1980) Stonefly (*Skwala* sp.) (95% T) = 57,500 [46,300-71,300]; MRID 40098001 (Mayer and Ellerseick 1986) *Hyalella azteca* (99.5% T) = 1.84 [0.05-3.71] (flow-through exposure); (Hansen and Garton 1982)	Amphipod (*Gammarus pseudolimnaeus*) (25% WP) = 25 [16-40]; (Mayer and Ellerseick 1986) Waterflea (*Daphnia magna*) (25% WP) = 16 [12 – 20]; (Johnson and Finley 1980) Amphipod (*Gammarus pseudolimnaeus*) (25% T) = 25 [16-40]; (Johnson and Finley 1980) Stonefly (*Skwala* sp.) (25% WP) = 57,000 [48,200-67,400]; MRID 40098001 (Mayer and Ellerseick 1986) Midge (*Chironomus*) (25% WP) = 560 [470-670]; (Johnson and Finley 1980) Amphipod (*Gammarus pseudolimnaeus*) (25% WP) mature = 30 [19-45] ; (Julin and Sanders 1978)	
	Invertebrate survival (5d	Midge *(Cricotopus sp.)* (99.5% T) 7d LC_{50} = 1.79 [1.48 – 2.13];	Dragonfly nymphs (*Orthemis* and *Pantala* spp.) (25% WP) 168 h LC_{50} =	

Assessment Endpoint		Concentration (µg/L aquatic tests; lbs a.i./acre terrestrial tests)		
		Diflubenzuron		Degradates of diflubenzuron (PCA, DFBA, PCPU)
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
	or 7d LC_{50})	(Hansen and Garton 1982) Midge *(Tanytarsus dissimilis)* (99.5% T) 5d LC_{50} = 1.02 [0.56 – 1.47]; (Hansen and Garton 1982)	50; (Miura and Takahashi 1974)	
	Invertebrate survival (24 h EC/LC_{50})	Amphipod (*Gammarus pseudolimnaeus*) (95% T) = 87 [65-117]; (Mayer and Ellerseick 1986) Waterflea (*Daphnia magna*) (97.6% T) = 68 [38-180]; MRID 408405-02, (Kuijpers 1988) Mosquito larvae (*Aedes caspius*) (90.1% T) = 1.0 [1.01-1.45]; (Porretta et al. 2008)	Amphipod (*Gammarus pseudolimnaeus*) (25% WP) = 88 [66-117]; (Mayer and Ellerseick 1986) Tadpole shrimp (*Triops longicaudatus*) (25% WP) EC_{40} = 0.75; (Miura and Takahashi 1974) Mosquito larvae (*Aedes albopictus*), 2nd instar (25% WP) = 0.0028 {0.0012-0.0055]; (Ho et al. 1987a) Mosquito larvae (*Aedes albopictus*), 3rd instar (25% WP) = 0.21 [0.014-1.005]; (Ho et al. 1987) Backswimmer (*Anisops sardeus*) (Dimilin) = 2123 [1960-2210]; (Lahr et al. 2001)	

Assessment Endpoint		Concentration (µg/L aquatic tests; lbs a.i./acre terrestrial tests)		
		Diflubenzuron		Degradates of diflubenzuron (PCA, DFBA, PCPU)
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
			Fairy shrimp (*Streptocephalus sudanicus*) (Dimilin 4L) = 13.3 [12.8-14]; (Lahr et al. 2001) Blackfly larvae (*Simulium vittatum*) (Formulation?) = 1.3 (EFED database 2000) Mosquito larvae (*Culex pipiens*) (25% WP) = 2.2; (Kasai et al. 2007)	
	Invertebrate repro-duction (LC_{50}, 21 d life-cycle test)	Waterflea (*Daphnia magna*) (% NR; T) = 0.062 [0.051-0.071] (LC_{50}); (Hansen and Garton 1982)		
	Invertebrate repro-duction (NOEL/LOEL life-cycle test)	Waterflea (*Daphnia magna*) (% NR; T) NOEL = <0.09, LOEL = 0.09; MRID 00010865 Waterflea (*Daphnia magna*) (99% T) NOEL = <0.06, LOEL = 0.06; Beltsville Lab Test 2424 (EPA) Brine shrimp (*Artemia salina*)		

Assessment Endpoint		Concentration (µg/L aquatic tests; lbs a.i./acre terrestrial tests)		
		Diflubenzuron		Degradates of diflubenzuron (PCA, DFBA, PCPU)
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
		(100% T) NOEL = >10, LOEL = >10; MRID 00073933 Mysids (*Mysidopsis bahia*) (99% T) LOEL = 0.075; MRID 43662001 Mysids (*Mysidopsis bahia*) (97.6% T) NOEL = 0.093, LOEL = >0.093, MATC = 0.067; MRID 40237501 Waterflea (*Daphnia magna*) (97.6% T) NOEL = 0.04, LOEL = 0.093; MRID 40840501		
Habitat: Estuarine prey survival	Invertebrate survival (LC_{50}, LOEL)	Mysids (*Mysidopsis bahia*) (95% T) 96 h LC_{50} = 2.1 [1.6-2.7]; (Mayer 1987) Grass shrimp (*Palaemonetes pugio*) (98.4% T) 72 h LC_{50} = 2.95 [3.3-2.66]; (Wilson and Costlow 1986) Grass shrimp (*Palaemonetes pugio*) (98.4% T) 96 h LC_{50} =	Mysids (*Mysidopsis bahia*) (25% WP) 96 h LC_{50} = 2.1[1.6-2.7]; MRID 43662001 (Nimmo et al. 1979) Copepod (*Eurytemora affinis*) (25% WP) 48 h LC_{50} = 2.2; (Savitz et al. 1994) Copepod (*Eurytemora affinis*) (25% WP) 48 h LOEL = 0.78; (Savitz et al. 1994)	

Assessment Endpoint		Concentration (µg/L aquatic tests; lbs a.i./acre terrestrial tests)		
		Diflubenzuron		Degradates of diflubenzuron (PCA, DFBA, PCPU)
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
		1.84 [2.08-1.64]; (Wilson and Costlow 1986) Grass shrimp (*Palaemonetes pugio*) (99% T) 96 h LC_{50} = 1.11 [0.88-1.34]; (Touart and Rao 1987) Grass shrimp (*Palaemonetes pugio*) (99% T) 24 h LC_{50} = 3.4 [1.72-5.04]; (Touart and Rao) 1987 (after exposure, transferred to clean seawater until ecdysis complete)	Grass shrimp (*Palaemonetes pugio*) (25% WP) 72 h LC_{50} = 2.83 [3.27-2.49]; (Wilson and Costlow 1986) Grass shrimp (*Palaemonetes pugio*) (25% WP) 96 h LC_{50} = 1.39 [1.54-1.27]; (Wilson and Costlow 1986) Grass shrimp (*Palaemonetes pugio*) larvae (25% WP) 96 h LC_{50} = 1.44; (Wilson and Costlow 1987) Grass shrimp (*Palaemonetes pugio*) post- larvae (25% WP) 96 h LC_{50} = 1.62; (Wilson and Costlow 1987)	
Habitat: Estuarine prey survival	Invertebrate survival (21 d LC_{50})	Mysids (*Americamysis bahia*) (97.6% T) LOEL = 0.086, NOEL = 0.045; MRID 40197001	Mysids, adult (*Mysidopsis bahia*) (25% WP) = 1.24 [0.84-1.8]; (Nimmo et al. 1979)	
Habitat: In-stream Primary Productivity	Aquatic plant growth NOEC, NOEL	Freshwater algae (*Selenastrum capricornutum*) (% NR; T) NOEC = 45; no effect on growth after 120 h exposure; (Hansen and Garton 1982)		

Assessment Endpoint		Concentration (µg/L aquatic tests; lbs a.i./acre terrestrial tests)		
		Diflubenzuron		Degradates of diflubenzuron (PCA, DFBA, PCPU)
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
		Freshwater algae (*Selenastrum capricornutum*) (% NR; T) NOEC = 300; 5 d exposure; MRID 42940106 (Thompson & Swigert 1993) Freshwater diatom (*Navicula pelliculosa*) (% NR; T) NOEC = 380; 5 d exposure; MRID 42940106 (Thompson & Swigert 1993) Duckweed (*Lemna gibba*) (% NR; T) NOEL = 190; 14 d exposure; MRID 42940106 (Thompson& Swigert 1993) Green Algae (*Selenastrum* sp.) (% NR; T) NOAEL = 200; MRID 45252205 Phytoplankton (Green, filamentous green, diatom, flagellate, bluegreen, filamentous bluegreen, cryptophyte) (% NR;		

Assessment Endpoint		Concentration (µg/L aquatic tests; lbs a.i./acre terrestrial tests)		
		Diflubenzuron		Degradates of diflubenzuron (PCA, DFBA, PCPU)
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
		T) EC_{50} = 5000; 6 d exposure; (Wurtsbaugh and Apperson 1978) Periphyton (Green, filamentous green, diatom, bluegreen) (% NR; T) EC_{50} = 5000; 6 d exposure; (Wurtsbaugh and Apperson 1978) Macrophytes (% NR; T) EC_{50} = 5000; 6 d exposure; (Wurtsbaugh and Apperson 1978)		
Habitat: Marine Primary Productivity	Aquatic plant growth (NOEC)	Marine diatom (*Skeletonema costatum*) (% NR; T) NOEC = 270; 5 d exposure; MRID 42940106 (Thompson& Swigert 1993)		

25.2.6.1.8 Laboratory studies measuring non-standard assessment endpoints (Diflubenzuron)

We located fifteen studies in the open literature and registrant-submitted reports measuring organism health following diflubenzuron exposure in laboratory experiments that did not report standard assessment endpoints (*i.e.*, LC_{50}s). Measured endpoints included survival, molting success, avoidance, reproductive success, growth, and emergence. Experiments were conducted on fish, crustaceans, arthropods, primary producers, and aquatic insects. Significant effects on adult insect emergence were noted between 0.14 and 4.9 µg/L; on invertebrate survival between 0.01 and 1.5 µg/L; on crustacean reproduction at 0.075 and 0.093 µg/L; and on juvenile fish growth at 0.078 µg/L.

Table 89. Laboratory studies with Diflubenzuron and non-standard assessment endpoints.

Chemical	Taxa/Species	Assessment measures	Concentrations tested	Exposure duration	Effects	Data source
Diflubenzuron (Dimilin)	Chironomid midge (*Cricotopus* spp.)	Adult emergence, Survival	36, 18.9, 8.3, 4.9, 2.0, 1.6, 0.4, 0.1 µg/L; actual	96 h continuous; 7 d static	No molting from 4th instar to pupae at and above 4.9 µg/L in continuous-flow tests; no adult emergence at 1.6 µg/L. No observed adult emergence at 18.9 µg/L and above when 4th instar larvae exposed (static), and significant reduction at 4.9 µg/L. Static tests also showed significant mortality at 4.9 µg/L.	(Nebeker et al. 1983)
	Caddisfly (*Clistoronia magnifica*)	Adult emergence, Survival	0.14 µg/L; actual	4 weeks	Larvae died during the molt period between the 5th larval instar and the pupal stage at concentrations of 0.14 µL and greater. No adult emergence was observed.	
	Chironomid midge (*Tanytarsus dissimilis*)	Larval molting; Survival	36, 18.9, 8.3, 4.9, 2.0, 1.6, 0.7 µg/L	5 d	Significant decrease in larval molting success at and above 4.9 µg/L.	
	Amphipod, juvenile (*Hyalella azteca*)	Survival	36, 18.9, 8.3, 4.9, 2.0 µg/L	96 h	Significantly reduced survival at all concentrations tested.	
	Waterflea, juvenile	Survival	2.0 µg/L	2 d and 6 d	Significant mortality at	

Chemical	Taxa/Species	Assessment measures	Concentrations tested	Exposure duration	Effects	Data source
	(*Daphnia magna*)		lowest tested	(continuous flow)	all concentrations tested.	
Diflubenzuron (Dimilin)	Atlantic salmon, parr (*Salmo salar*)	Avoidance behavior	10 µg/L Dimilin	10 min	Fish spent significantly less time in water treated with 10 µg/L Dimilin. However, same effect noted with carrier (Florex clay) control.	(Granett et al. 1978)
Diflubenzuron (Dimilin WP-25)	Rainbow Trout (*Salmo gairdneri*) (average weight = 9.7 ± 3.3 g)	Blood serum parameters	Up to 10 mg/L	96 h	Significantly lower levels of glutamate oxaloacetate transaminase (aspartate aminotransferase; µg/mL) in exposed fish. Other blood parameters (lipid, hematocrit, glucose, sodium) not significantly different.	(Madder and Lockhart 1978)
Diflubenzuron (TH 6040 formulation)	Waterflea (*Daphnia* spp.); Clam shrimp (*Eulimnadia* spp.) Tadpole shrimp (*Triops longicaudatus*); Mayfly nymphs (*Callibaetis* spp.); Midge larvae (*Chironomus* spp.); Dragonfly nymphs (*Orthemis* and *Pantala*); Mosquitofish (*Gambusia affinis*)	Survival Abundance	Laboratory tests with various concentrations	24 – 240 h	Waterflea 50% mortality at 1.5 µg/L. Clam shrimp 50% mortality at 0.15 µg/L. Tadpole shrimp 40% mortality at 0.75 µg/L . Mayfly nymphs 90% mortality at 10 µg/L. Midge larvae 90% mortality at 10 µg/L. Dragonfly nymph mortality 50% at 50 µg/L. No effect on Mosquitofish survival at high dose.	(Miura and Takahash.Rm 1974)

Chemical	Taxa/Species	Assessment measures	Concentrations tested	Exposure duration	Effects	Data source
Diflubenzuron (Dimilin, 25% WP)	Mayfly nymphs (*Callibaetis* spp. and *Siphlonurus*)	Survival	2 µg/L	168 h	Mayfly nymphs showed 50% mortality at 2 µg/L, 90% mortality at 3 µg/L.	(Miura and Takahashi 1975)
Dimilin 25% WP	*Mysidopsis bahia* (estuarine crustacean)	Reproductive success	0.075, 0.25, 0.5, 0.75 µg/L nominal (<0.4, 0.55, 0.91 µg/L measured)	21 d life cycle test, continuous flow, 22-28 ppt salinity	Significantly fewer young/female were produced at 0.075 (<0.4 measured) µg/L.	(Nimmo et al. 1979)
^{14}C-Diflubenzuron (radiolabeled)	Waterflea (*Daphnia magna*)	Growth Reproduction Survival	Up to 0.093 µg/L (reported as actual)	21 d chronic, flow-through exposure	Survival significantly reduced to 50% at 0.093 µg/L. No offspring produced per female at 0.093 µg/L. Mean body length significantly shorter (3.8 mm) at 0.093 µg/L than controls (4.6 – 4.8 mm).	Surprenant, 1988; MRID 40840501
Dimilin 25% WP	Tropical freshwater fish (*Prochilodus lineatus*)	AChE activity Hematological parameters Liver histopathology	25 mg/L active ingredient	6, 24, 96 h	Reduction in the number of erythrocytes and hemoglobin content after 96 h exposure. Decrease in muscle AChE activity at all time-points compared to controls.	(Maduenho and Martinez 2008)
Dimilin 25% WP	Mosquitofish (*Gambusia affinis*)	Survival Growth	16 and 78 ng/L	Exposed for 28 d followed	No effects on mortality, length, weight, or	(Zaidi and Soltani 2011)

Chemical	Taxa/Species	Assessment measures	Concentrations tested	Exposure duration	Effects	Data source
		GSH and GST activity Metric indexes		by 8 d recovery. Sampled at 0, 7, 14, 21, 28 d exposure and 1, 2, 4, and 8d recovery.	condition factor. Significant reductions in glutathione (GSH) at 16 ng/L (starting at 14 d) and 78 ng/L (starting at 7 d). No difference from controls following recovery. Significant induction of GST (glutathione S transferase) starting at day 7 at 78 ng/L. No difference from controls following recovery.	
Diflubenzuron	Mysid (*Mysidopsis bahia*)	Survival Reproduction	28, 48, 93 ng/L	21 d continuous exposure	Significant reduction in reproductive success (number of offspring per female per day) at 93 ng/L.	Breteler, 1987; MRID 40237501
Dimilin 25% WP	Larval crab (*Rhithropanopeus harrisii*) Larval shrimp (*Palaemonetes pugio*)	Survival	10 µg/L initial, decreasing to less than 1 µg/L over 71 days.	3 week exposure to spiked water that had aged for 0, 7, 14, 19, and 32 d.	Crab larvae in treatment without sediment did not survive to the post-larval stage even when the seawater solution had aged 32 days prior to exposures. Survival of crab larvae in treatment with sediment was 0% in 0 and 7 d aged water, 13% in 14 d aged water, and	(Cunningham and Myers 1987)

Chemical	Taxa/Species	Assessment measures	Concentrations tested	Exposure duration	Effects	Data source
					same as controls in 19 d aged water. Shrimp larvae did not survive in any treatment exposure without sediment, even after seawater solution had aged 71 days. Larvae survival was 0% in water with sediment aged 10, 12, and 16 days. Survival was 34 % in sediment water aged 19 days, and reached control levels at 22 and 63 d aged water.	
Dimilin	Waterflea (*Daphnia magna*)	Survival	Exposed to concentrations ranging from 0.001 - 100 µg/L, nominal	Six d acute toxicity test. Static exposures with 100% test solution renewal on day 3.	Survival of adults significantly decreased at 0.01 µg/L. The LC_{50} fell between 0.1 and 0.01 µg/L. Lower concentrations elicited no adverse effects on growth, molting, or reproduction.	(Kashian and Dodson 2002)
Dimilin	Copepod (*Acartia tonsa*)	Survival Fecundity Egg viability	0.5, 1, 5, 10, 100, 1000 µg/L	Static 5 d exposure (survival); Static 4 d exposure (fecundity); Adults exposed for	No effect on survival. No effect on fecundity (number of eggs per adult females). Eggs from females held in 10 µg/L for 60 h	(Tester and Costlow 1981)

Chemical	Taxa/Species	Assessment measures	Concentrations tested	Exposure duration	Effects	Data source
				60 h, then laid eggs moved to clean water for 48 h.	showed a decrease in % hatch from 93.4 to 0% by 24 h. Eggs from females held in 1 µg/L for 60 h showed a decrease in % hatch from control levels to near 0% by 36 h. Reduction in hatching viability continued for at least 30 h after adults moved to clean seawater.	
Dimilin 25% WP	Mosquitofish, juveniles and adult females (*Gambusia affinis*)	Growth Condition index	78 ng/L	Juveniles exposed for 24 h, moved to clean water for 45 d. Adult females exposed for 30 d.	Juvenile length and weight reduced relative to controls on 30 and 45 d post-exposure. No effect on adult female condition index, reductions in ovarian protein and glutathione levels after 15 and 30 d.	(Draredja-Beldi and Soltani 2003)
Dimilin	Marine diatoms (*Thalassiosira weissflogii,Thalassiosira nordenskioeldii, Cyclotella cryptica, Skeletonema costatum*) Harpacticoid copepod (*Tigriopus californicus*)	Survival Photosynthesis	0.1 – 5000 µg/L	Diatoms exposed for 11 – 14 d Copepods exposed for up to 71 d	14C-photosynthesis measurements showed little effect on all diatoms at concentrations up to 1000 µg/L. At concentrations as low at 1 µg/L, adult copepods showed steadily diminishing	(Antia et al. 1985)

Chemical	Taxa/Species	Assessment measures	Concentrations tested	Exposure duration	Effects	Data source
					abundance with no evidence of nauplii production.	

25.2.7 Toxicity of fenbutatin oxide (Assessment Endpoints)

We located study results that measured standard assessment endpoints in several species of fish, freshwater and estuarine/marine invertebrates, terrestrial and aquatic plants, and algae and diatoms following short and longer-term exposures to fenbutatin oxide. Most of the toxicity results were from registrant-supplied studies used to support fenbutatin oxide registration. We located no studies in the open literature or in the gray literature that addressed assessment endpoints for salmonids. We also located no studies that tested fenbutatin oxide mixed with other pesticides. We found no toxicity information for degradates of fenbutatin oxide (9.1.3.2). Thus, significant data gaps exist on biological and ecological assessment endpoints such as swimming, olfactory-mediated behaviors, migration, spawning, time-to-first feeding, smoltification, etc. We therefore are faced with extrapolating from standard toxicity test results to potential effects to salmonid habitats and recognize the complexities and inherent uncertainties introduced.

EFFECTS TO SALMONIDS

25.2.7.1 Direct effects to salmonids: Survival

We located several studies that measured survival to fish following short-term exposures. Seven studies tested the lethality of fenbutatin oxide to rainbow trout of which five studies used technical fenbutatin oxide and two used end-use products. One of the formulations tested was a 50% wettable powder (WP). Currently there are only two end-use products registered and both are 50% WP formulations. The other formulation tested was a liquid. The tests employed standard experimental designs to measure 96 h acute mortality. Several of the studies did report measured concentrations of fenbutatin oxide. Salmonid LC_{50}s ranged from 1.1 – 6.6 µg/L for the technical product and were 33 and 52 µg/L for the formulated products. The lowest LC_{50} reported, 1.1 µg/L, was from a recent registrant-submitted study that measured fenbutatin oxide concentrations (2012; MRID 48815502). The mean and median 96 h LC_{50}s (n=7) for rainbow trout was 13.9 µg/L and 1.7 µg/L, respectively. Results indicate that the two end-use products tested may be slightly less toxic than fenbutatin oxide alone with respect to salmonid survival, however it is difficult to say with high confidence as no measures of variability were reported.

Three studies reported 24 h LC_{50}s of 4.4 - 14.2 µg/L. We located ten 96 h LC_{50}s from freshwater and estuarine non-salmonids (bluegill sunfish, sheepshead minnow, fathead minnow, and channel catfish) that ranged from 1.5 – 30 µg/L.

25.2.7.2 Direct effects to salmonids: Reproduction and growth

Three registrant-submitted studies reported LOECs for rainbow trout and sheepshead minnow larval growth and survival following multi-week exposures to technical fenbutatin oxide. All three studies measured fenbutatin oxide concentrations with analytical chemistry. Following a 60 d flow-through exposure, rainbow trout LOECs for larval growth, larval survival, and embryo hatching were 0.61, 0.61, and 1.3 µg/L (respectively). At these exposure concentrations, mean larval weights were reduced from 309 mg (solvent controls) to 62 mg, larval survival was reduced from 83 % (solvent controls) to 10 %, and embryo hatching success was reduced from 78 % (solvent controls) to 44 %, respectively. Following a 32 d flow-through exposure, the LOECs for sheepshead minnow larval growth and survival were both 2.1 µg/L. Following a 36 d flow-through exposure, the LOECs for sheepshead minnow larval growth, larval survival, and embryo hatching were all 5.7 µg/L. We also note that fenbutatin oxide did not affect survival, morphology, or length in a zebrafish early life stage (larval) experiment at concentrations up to 10 µg/L (Appendix 3). We located no LOEC or EC_{50} data for the effects of fenbutatin oxide on growth and reproduction of other life stages.

25.2.7.3 Effects to salmonid prey

Eight studies were found that evaluated short-term (48 h) toxicity of fenbutatin oxide to the waterflea (*D. magna*). Five used technical fenbutatin oxide, while three used formulations. LC_{50}s ranged from 6.4 – 2184 µg/L. One study of technical product and two studies of formulations used analytical chemistry to measure the exposure concentrations. These three studies reported LC_{50}s of 31 µg/L (technical product) and 6.4 and 13 µg/L (formulated products). We found two study results on waterfleas using a 21 d life-cycle test to determine effects on reproduction. Neither determined a survival EC/LC_{50}. Instead, NOECs of 4 (LOEC of 25 µg/L) and 16 µg/L were reported (EFSA 2010; MRID 40525901).

Five studies were found examining the effects of fenbutatin oxide on estuarine/marine invertebrates. One study used a 24 h exposure of formulated product to brine shrimp and found an LC_{50} of 50 µg/L (nominal concentration). A second study using a 96 h exposure to a mysid shrimp reported a LC_{50} of 2.8 µg/L and measured exposure concentrations. A third study using a 48 h exposure to Eastern oyster measured an EC_{50} for developmental effects of 0.37 µg/L. Two studies on mysid shrimp reported LOECs for effects on reproduction of 0.88 µg/L (following a 32 d flow-through exposure) and 0.55 and 0.32 µg/L (for young/female and F1 survival, respectively, following a 28 d flow-through exposure).

We found four studies describing the sediment toxicity of fenbutatin oxide. One study using an estuarine amphipod (*L. plumulosus*) and a 28 d exposure to contaminated sediment found an EC_{50} for effects on reproduction of 9.1 mg a.i./kg. Three studies using chironomid midges (*C. riparius* and *dilutus*) exposed to contaminated sediments for 10 – 63 d (depending on study) reported NOECs of 90 mg a.i./kg or greater for effects on survival, reproduction, and growth. The studies used contaminated sediment with initially clean water. Fenbutatin oxide concentrations in the overlying waters were measured in two studies and found to be initially 23 and 41 µg/L from sediment exposures of 110 and 90 mg a.i./kg, respectively.

25.2.7.4 Herbicidal effects

Two registrant-provided studies were found on the effect of fenbutatin oxide on terrestrial plants. The studies used standardized tests for vegetative vigor and seedling emergence. A spray application of formulated product at an exposure rate simulating 4 lbs a.i./acre produced no effect on measures of vegetative vigor or measures of seedling emergence. A variety of crop-related species (monocots and dicots) were evaluated such as bean, onion, lettuce, corn, etc. An additional study on the aquatic plant, duckweed, showed LOECs following a 7 d exposure to formulated product of 290, 190, and 190 µg/L for biomass, growth rate, and frond density (respectively). One study found the 96 h EC_{50} for cell density of a marine diatom to be 100 µg/L. A study on a freshwater diatom found 96 h EC_{50}s of 14 and 27 µg/L for biomass and growth rate (respectively). Three studies on algal species found 96 h EC_{50}s ranging from 166.1 - 7434 µg/L for changes in biomass.

Table 90 Fenbutatin oxide toxicity values (µg/L) for aquatic organisms and plants reported in EPA salmonid BE, RED, EFED Problem Formulation, ECOTOX, EFSA Review, and open literature. Abbreviations as follows: a.i. = active ingredient; NR = Not Reported; T= Technical grade; F = Formulated product (wettable powder); F* = Discontinued formulated product; [ ] = 95% Confidence interval; mea = measured concentration; nom = nominal concentration.

Assessment Endpoint		Concentration (µg/L aquatic tests or lbs a.i./acre terrestrial tests)		
		Fenbutatin oxide [ ] = 95% Confidence Interval		Degradates of fenbutatin oxide:
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
Survival	salmonid LC_{50} (96 h)	Rainbow trout (*O. mykiss*) (98%; T, mea) = 1.1 [0.59, 2.6] MRID 48815502 Rainbow trout (*O. mykiss*) (% NR; T, mea) = 1.14 [NR] (EFSA 2010) Rainbow trout (*O. mykiss*) (100%; T, nom) = 1.7 [1.3, 2.4] MRID 40098001 Rainbow trout (*O. mykiss*) (95%; T, nom) = 1.7 [1.4, 2.2] MRID 113075 Rainbow trout (*O. mykiss*) (98.6%; T, nom) = 6.6 [5.8, 7.8] MRID 40473506	Rainbow trout (*O. mykiss*) (%NR; F, mea) = 52 [NR] ((EFSA) 2010) Rainbow trout (*O. mykiss*) (42%; F*, mea) = 33 [28, 37] MRID 40473507	NA
	salmonid LC_{50} (24 h)	Rainbow trout (*O. mykiss*) (100%; T, nom) = 4.4 [3.4, 5.8] (24 h) MRID 4009801 Rainbow trout (*O. mykiss*) (98.6%; T,	Rainbow trout (*O. mykiss*) (50%; F, nom) = 14 [9.7, 20] (24 h) MRID 4009801	

Assessment Endpoint		Concentration (µg/L aquatic tests or lbs a.i./acre terrestrial tests)		
		Fenbutatin oxide [ ] = 95% Confidence Interval		Degradates of fenbutatin oxide:
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
	salmonid LC_{50} (48 h) salmonid LC_{50} (72 h)	nom) = 14.2 [NR] (24 h) MRID 40473506 Rainbow trout (*O. mykiss*) (98%; T, mea) = >2.6 [NR] (24 h) MRID 48815502 Rainbow trout (*O. mykiss*) (98.6%; T, nom) = 7.7 [5.9, 9.1] (48 h) MRID 40473506 Rainbow trout (*O. mykiss*) (98%; T, mea) = 1.6 [1.1, 2.6] (48 h) MRID 48815502 Rainbow trout (*O. mykiss*) (98.6%; T, nom) = 6.9 [5.4, 8.1] (72 h) MRID 40473506 Rainbow trout (*O. mykiss*) (98%; T, mea) = 1.4 [1.1, 2.6] (72 h) MRID 48815502		
Survival	Non-salmonid freshwater and estuarine	Bluegill sunfish (*Lepomis macrochirus*) (95%; T, nom) = 6.9 [5, 9.5] MRID 113076	Fathead minnow (*Pimephales promelas*) (50%; F, nom, 17 °C) = 1.9 [1.0, 3.5] MRID 4009801	NA

Assessment Endpoint		Concentration (µg/L aquatic tests or lbs a.i./acre terrestrial tests)		
		Fenbutatin oxide [ ] = 95% Confidence Interval		Degradates of fenbutatin oxide:
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
	fish LC_{50} (96 h)	Bluegill sunfish (*Lepomis macrochirus*) (100%; T, nom) = 4.8 [2.5, 9.3] MRID 4009801 Sheepshead minnow (*Cyprinodon variegatus*) (99%; T, mea) = 21 [20, 21.8] MRID 41483301 Sheepshead minnow (*Cyprinodon variegatus*) (98%; T, nom) = 20.5 [15, 25] MRID 40590506 Sheepshead minnow (*Cyprinodon variegatus*) (98%; T, mea) = 24 [20, 21.8] MRID 48815503 Fathead minnow (*Pimephales promelas*) (% NR; T, nom) = 1.8 [NR] (EFSA 2010)	Channel catfish (*Ictalurus punctatus*) (50%; F, nom, 17 °C) = 1.5 [0.9, 2.7] MRID 4009801 Channel catfish (*Ictalurus punctatus*) (50%; F, nom, 22 °C) = 4.0 [2.7, 5.6] MRID 4009801 Bluegill sunfish (*Lepomis macrochirus*) (42%; F*, mea) = 30 [28, 42] MRID 40473508	
Larval growth	LOEC	Rainbow trout (*O. mykiss*) (98.6%; T, mea) = 0.61 (60 d flow-through) MRID 40473512 Sheepshead minnow (*Cyprinodon		NA

Assessment Endpoint		Concentration (µg/L aquatic tests or lbs a.i./acre terrestrial tests)		
		Fenbutatin oxide [ ] = 95% Confidence Interval		Degradates of fenbutatin oxide:
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
	NOEC	*variegatus*) (98%; T, mea) = 2.1 (32 d flow-through) MRID 48861401 Sheepshead minnow (*Cyprinodon variegatus*) (99%; T, mea) = 5.7 (36 d flow-through) MRID 41551401 Rainbow trout (*O. mykiss*) (98.6%; T, mea) = 0.31 (60 d flow-through) MRID 40473512 Sheepshead minnow (*Cyprinodon variegatus*) (98%; T, mea) = 0.98 (32 d flow-through) MRID 48861401 Sheepshead minnow (*Cyprinodon variegatus*) (99%; T, mea) = 1.6 (36 d flow-through) MRID 41551401		
Embryo hatching	LOEC	Rainbow trout (*O. mykiss*) (99%; T, mea) = 1.3 (60 d flow-through) MRID 40473512 Sheepshead minnow (*Cyprinodon		

Assessment Endpoint		Concentration (µg/L aquatic tests or lbs a.i./acre terrestrial tests)		
		Fenbutatin oxide [ ] = 95% Confidence Interval		Degradates of fenbutatin oxide:
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
	NOEC	*variegatus*) (99%; T, mea) = 5.7 (36 d flow-through) MRID 41551401 Rainbow trout (*O. mykiss*) (99%; T, mea) = 0.61 (60 d flow-through) MRID 40473512 Sheepshead minnow (*Cyprinodon variegatus*) (99%; T, mea) = 1.6 (36 d flow-through) MRID 41551401		
Larval survival	LOEC NOEC	Rainbow trout (*O. mykiss*) (99%; T, mea) = 0.61 (60 d flow-through) MRID 40473512 Sheepshead minnow (*Cyprinodon variegatus*) (98%; T, mea) = 2.1 (32 d flow-through) MRID 48861401 Sheepshead minnow (*Cyprinodon variegatus*) (99%; T, mea) = 5.7 (36 d flow-through) MRID 41551401 Rainbow trout (*O. mykiss*) (99%; T, mea) = 0.31 (60 d flow-through) MRID		

Assessment Endpoint		Concentration (µg/L aquatic tests or lbs a.i./acre terrestrial tests)		
		Fenbutatin oxide [ ] = 95% Confidence Interval		Degradates of fenbutatin oxide:
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
		40473512 Sheepshead minnow (*Cyprinodon variegatus*) (98%; T, mea) = 0.98 (32 d flow-through) MRID 48861401 Sheepshead minnow (*Cyprinodon variegatus*) (99%; T, mea) = 1.6 (36 d flow-through) MRID 41551401		
Fish growth (life stage NR, European studies not available to NMFS)	NOEC	Rainbow trout (*O. mykiss*) (% NR; T, nom) = 1.27 (28 d flow-through) (EFSA 2010) Rainbow trout (*O. mykiss*) (% NR; T, nom) = 0.2 (62 d flow-through) (EFSA 2010) Fathead minnow (*Pimephales promelas*) (% NR; T, nom) = 0.3 (35 d flow-through) (EFSA 2010)		NA
Bioconcentration	BCF	Bluegill sunfish (*Lepomis macrochirus*) (99%; T, mea) MRID 48973501 693x (edible tissue) 1875x (non-edible tissue)		NA

Assessment Endpoint		Concentration (µg/L aquatic tests or lbs a.i./acre terrestrial tests)		
		Fenbutatin oxide [ ] = 95% Confidence Interval		Degradates of fenbutatin oxide:
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
	Kinetic BCF	1350x (whole body) Bluegill sunfish (*Lepomis macrochirus*) (99%; T, mea) MRID 48973501 757x (edible tissue) 2186x (non-edible tissue) 1506x (whole body)		
Habitat: salmonid prey	Invertebrate survival (48 h EC/LC_{50})	Waterflea (*Daphnia magna*) (% NR; T, nom) = 47.6 [NR] (EFSA 2010) Waterflea (*Daphnia magna*) (98%; T, mea) = 31 [24, 53] MRID 48815501 Waterflea (*Daphnia magna*) (%NR; T, nom) = 83 [51, 185] MRID 113077 Waterflea (*Daphnia magna*) (99%; T, nom) = 26 [23, 30] MRID 40473509 Waterflea (*Daphnia magna*) (99%; T, nom, fed) = 220 [190, 260] MRID 40473510	Waterflea (*Daphnia magna*) (50%; F, mea) = 13 [NR] (EFSA 2010) Waterflea (*Daphnia magna*) (50%; F, nom) = 2184 [1268, 6675] MRID 113077 Waterflea (*Daphnia magna*) (42%; F*, mea) = 6.4 [4.8, 8.6] MRID 40473511	NA

Assessment Endpoint		Concentration (µg/L aquatic tests or lbs a.i./acre terrestrial tests)		
		Fenbutatin oxide [ ] = 95% Confidence Interval		Degradates of fenbutatin oxide:
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
	Invertebrate reproduction (21 d life-cycle test) NOEC LOEC	Waterflea (*Daphnia magna*) (99% ; T, mea) = 16 MRID 40525901 Waterflea (*Daphnia magna*) (% NR; T, mea) = 4 (EFSA 2010) Waterflea (*Daphnia magna*) (99%; T, mea) = 25 MRID 40525901		NA
Mesocosm (details unavailable)	NOAEC		(50%, F, nom) 100 (118 d static, corresponding to 1.23 mg a.i./kg sediment) (EFSA 2010)	NA
Microcosm (details unavailable)	NOEC		(50%, F, nom) 10 (28 d static) (EFSA 2010)	NA
Habitat: Estuarine/marine invertebrates	Survival (LC_{50}) Development (EC_{50}) Reproduction (LOECs)	Mysid shrimp (*Americamysis bahia*) (98%; T, mea) = 2.8 [2.3, 4.6] (96 h) MRID 40590508 Eastern oyster (*Crassostrea virginica* larvae) (98%; T, mea) = 0.37 [0.35, 0.42] (48 h) MRID 40590507 Mysid shrimp (*Americamysis bahia*) (99%; T, mea) = 0.88 (32 d flow-	Brine shrimp (*Artemia sp.*) (55%; F*, nom) = 50 [40, 60] (24 h) (Machera et al. 1996)	NA

Assessment Endpoint		Concentration (µg/L aquatic tests or lbs a.i./acre terrestrial tests)		
		Fenbutatin oxide [ ] = 95% Confidence Interval		Degradates of fenbutatin oxide:
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
		through) MRID 48933901 Mysid shrimp (*Americamysis bahia*) young/female (99%; T, mea) = 0.55 (28 d flow-through) MRID 41551402 Mysid shrimp (*Americamysis bahia*) F1 survival (99%; T, mea) = 0.32 (28 d flow-through) MRID 41551402		
Habitat: Freshwater sediment invertebrates	Survival NOEC Growth NOEC	Midge (*Chironomus riparius*) (%NR, T, nom) >1000 mg a.i./kg sediment (28 d) (EFSA 2010) Midge (*Chironomus dilutus*) (97%, T, mea) = 110 mg a.i./kg sediment (20 d) (overlying water initially 23 µg/L) MRID 47910408 Midge (*Chironomus dilutus*) (97%, T, mea) = 90 mg a.i./kg sediment (10 d) (overlying water initially 41 µg/L) MRID 47910407		NA

Assessment Endpoint		Concentration (µg/L aquatic tests or lbs a.i./acre terrestrial tests)		
		Fenbutatin oxide [ ] = 95% Confidence Interval		Degradates of fenbutatin oxide:
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
	Reproduction NOEC	Midge (*Chironomus dilutus*) (97%, T, mea) = 110 mg a.i./kg sediment (20 d) MRID 47910408 Midge (*Chironomus dilutus*) (97%, T, mea) = 90 mg a.i./kg sediment (10 d) MRID 47910407 Midge (*Chironomus dilutus*) (97%, T, mea) = 110 mg a.i./kg sediment (63 d) MRID 47910408		
Habitat: Estuarine sediment invertebrates	Survival NOEC Growth LOEC Reproduction LOEC	Amphipod (*Leptocheirus plumulosus*) (98%, T, mea) = 99 mg a.i./kg sediment (28 d) MRID 48861402 Amphipod (*Leptocheirus plumulosus*) (98%, T, mea) = 20 mg a.i./kg sediment (28 d) MRID 48861402 Amphipod (*Leptocheirus plumulosus*) (98%, T, mea) = 10 mg a.i./kg sediment (28 d) MRID 48861402		

Assessment Endpoint		Concentration (µg/L aquatic tests or lbs a.i./acre terrestrial tests)		
		Fenbutatin oxide [ ] = 95% Confidence Interval		Degradates of fenbutatin oxide:
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
	Reproduction EC_{50}	Amphipod (*Leptocheirus plumulosus*) (98%, T, mea) = 9.1 mg a.i./kg sediment (28 d) MRID 48861402		
Habitat: Riparian Vegetation	Vegetative vigor (lbs a.i./acre): shoot length, shoot dry weight NOEC	NA	(50%, F, mea) = 4 (lbs a.i./A) for: Bean (*Phaseolus vulgaris*) Corn (*Zea mays*) Cucumber (*Cucumis sativus*) Lettuce (*Lactuca sativa*) Oats (*Avena sativa*) Oilseed rape (*Brassica napus*) Onion (*Allium cepa*) Perennial ryegrass (*Lolium perenne*) Soybean (*Glycine max*) Tomato (*Lycopersicon esculentum*) MRID 47910406	NA
Habitat: Riparian Vegetation	Seedling emergence (lbs a.i./acre): percent emergence, shoot length, shoot dry weight	NA	(50%, F, mea) = 4 (lbs a.i./A) for: Bean (*Phaseolus vulgaris*) Corn (*Zea mays*) Cucumber (*Cucumis sativus*) Lettuce (*Lactuca sativa*) Oats (*Avena sativa*) Oilseed rape (*Brassica napus*) Onion (*Allium cepa*)	NA

Assessment Endpoint		Concentration (μg/L aquatic tests or lbs a.i./acre terrestrial tests)		
		Fenbutatin oxide [ ] = 95% Confidence Interval		Degradates of fenbutatin oxide:
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
	NOEC		Perennial ryegrass (*Lolium perenne*) Soybean (*Glycine max*) Tomato (*Lycopersicon esculentum*) MRID 47910405	
Habitat: Primary Productivity	Aquatic algal biomass (96 h): EC_{50}	Freshwater blue-green alga *(Anabaena flosaquae*) (96%, T, nom) = 188.8 [NR] (Ma 2005) Freshwater blue-green alga *(Microcystis aeruginosa*) (96%, T, nom) = 168.4 [NR] (Ma 2005) Freshwater blue-green alga *(Microcystis flosaquae*) (96%, T, nom) = 166.1 [NR] (Ma 2005) Green algae (*Pseudokirchneriella subcapitata*) (96%, T, nom) = 7434 [NR] (Ma 2005) Green algae (*Chlorella pyrenoidosa*) (96%, T, nom) = 1738 [NR] (Ma 2005) Green algae (*Chlorella vulgaris*) (96%,	Green algae (*Pseudokirchneriella subcapitata*) (50%, F, mea) > 1,200 MRID 47910402 Freshwater diatom (*Navicula pelliculosa*) (50%, F, mea) = 14 [10, 18] MRID 47910403 Freshwater blue-green alga *(Anabaena flosaquae*) (50%, F, mea) = 600 [180, 800] MRID 47910404	NA

Assessment Endpoint		Concentration (µg/L aquatic tests or lbs a.i./acre terrestrial tests)		
		Fenbutatin oxide [ ] = 95% Confidence Interval		Degradates of fenbutatin oxide:
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
		T, nom) = 3094 [NR] (Ma 2005)		
	NOEC	Green algae (*Scenedesmus obliquus*) (96%, T, nom) = 1509 [NR] (Ma 2005) Green algae (*Scenedesmus quadricauda*) (96%, T, nom) = 554.5 [NR] (Ma 2005)		
	Aquatic algal growth rate (96 h): EC_{50}		Green algae (*Pseudokirchnerialla subcapitata*) (50%, F, mea) = 1,200 MRID 47910402 Freshwater diatom (*Navicula pelliculosa*) (50%, F, mea) = 4.8 MRID 47910403 Freshwater blue-green alga *(Anabaena flosaquae*) (50%, F, mea) 430 MRID 47910404	
	NOEC		Green algae (*Pseudokirchnerialla subcapitata*) (50%, F, mea) > 1,200 MRID 47910402 Freshwater diatom (*Navicula pelliculosa*) (50%, F, mea) = 27 [26, 29] MRID	

Assessment Endpoint		Concentration (µg/L aquatic tests or lbs a.i./acre terrestrial tests)		
		Fenbutatin oxide [ ] = 95% Confidence Interval		Degradates of fenbutatin oxide:
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
	Aquatic plant biomass (7 d): EC_{50} NOEC LOEC Aquatic plant growth rate (7 d): EC_{50} NOEC		47910403 Freshwater blue-green alga *(Anabaena flosaquae*) (50%, F, mea) > 930 MRID 47910404 Green algae (*Pseudokirchnerialla subcapitata*) (50%, F, mea) = 700 MRID 47910402 Freshwater diatom (*Navicula pelliculosa*) (50%, F, mea) = 11 MRID 47910403 Freshwater blue-green alga *(Anabaena flosaquae*) (50%, F, mea) > 430 MRID 47910404 Duckweed (*Lemna gibba*) (50%, F, mea) = 700 [560, 800] MRID 47910401 Duckweed (*Lemna gibba*) (50%, F, mea) = 190 MRID 47910401	

Assessment Endpoint		Concentration (µg/L aquatic tests or lbs a.i./acre terrestrial tests)		
		Fenbutatin oxide [ ] = 95% Confidence Interval		Degradates of fenbutatin oxide:
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
	LOEC		Duckweed (*Lemna gibba*) (50%, F, mea) = 290 MRID 47910401	
	Aquatic plant frond density (7 d): EC_{50}		Duckweed (*Lemna gibba*) (50%, F, mea) = > 910 MRID 47910401	
	NOEC		Duckweed (*Lemna gibba*) (50%, F, mea) = 44 MRID 47910401	
	LOEC		Duckweed (*Lemna gibba*) (50%, F, mea) = 190 MRID 47910401	
	Marine diatom cell density (96 h): EC_{50}		Duckweed (*Lemna gibba*) (50%, F, mea) = 400 [280, 520] MRID 47910401	
			Duckweed (*Lemna gibba*) (50%, F, mea) = 44 MRID 47910401	
			Duckweed (*Lemna gibba*) (50%, F, mea) = 190 MRID 47910401	
			Marine diatom (*Skeletonema costatum*)	

Assessment Endpoint		Concentration (µg/L aquatic tests or lbs a.i./acre terrestrial tests)		
		Fenbutatin oxide [ ] = 95% Confidence Interval		Degradates of fenbutatin oxide:
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
			(50%; F, mea) = 100 [57, 180] MRID 48815504	

25.2.8 Toxicity of propargite (Assessment Endpoints)

We located study results that measured standard assessment endpoints in several species of fish, freshwater invertebrates, and aquatic plants following short and longer-term exposures to propargite, an end-use product (Omite), and a degradate, 2-[4-(1,1-dimethylethyl) phenoxy] cyclohexane-1-ol, also referred to as 2-(p-tertiarybutyl) phenoxycyclohexanol (TBPC). All toxicity results were from registrant-supplied studies used to support propargite registration in the U.S. and European Union. We located no studies in the open literature or in the gray literature that addressed assessment endpoints for salmonids or their habitat. We also located no studies that tested propargite mixed with other pesticides. Several degradation/metabolism studies identified degradates, however no toxicity information was found. Thus, significant data gaps exist on biological and ecological assessment endpoints such as swimming, olfactory-mediated behaviors, migration, spawning, time-to-first feeding, smoltification, riparian function, foodweb condition, etc. No mesocosm or field studies were located. We therefore are faced with extrapolating from standard toxicity test results to potential effects to salmonid habitats and recognize the complexities and inherent uncertainties introduced.

25.2.8.1 Direct Effects to Salmonids: Survival

We located several studies that measured survival to fish following short-term exposures. Seven studies tested the lethality of propargite to rainbow trout of which two studies used technical propargite and five used end-use products. The tests employed several experimental designs including 96 h continuous flow through exposure to a single overhead spray to chambers containing water and sediment. Salmonid LC_{50}s ranged from 24 – 445 µg/L. The lowest LC_{50} reported, 24 µg/L, resulted from a 21 d exposure wherein at 96 h in the same experiment an LC_{50} of 43 µg/L was reported (MRID 41458301). The mean and median 96 h LC_{50}s (n=7) for rainbow trout was 159 µg/L and 143 µg/L, respectively. Survival from experiments with end-use products reported LC_{50}s of 47- 445 µg/L indicating no clear trend of whether end-use products are more or less toxic with respect to survival. Several studies reported 24 h LC_{50}s ranging from 100 – 216 µg/L. We located one 96 h LC_{50} (55 µg/L) to an estuarine/marine species, a sheepshead minnow. We also note that propargite did not affect survival or morphology of a zebrafish early life stage experiment at concentrations up to 150 µg/L (Appendix 3).

25.2.8.2 Direct effects to salmonids: Reproduction and growth

Two registrant-submitted studies measured responses in fathead minnows following multi-week exposure to propargite. Within 35 d of hatching 100% of fathead minnows had died following exposure to 11 µg/L propargite. No effects to fry survival or reproduction were observed at 16 µg/L treatment. In the second study, fry length and weight were reduced at 28 µg/L, 66% of eggs successfully hatched at 27 µg/L, and 100% of juveniles died within 30 d following exposure to 27 µg/L. No adverse effects were recorded at 5.7 µg/L. Both studies measured treatment concentrations with analytical chemistry methods. Propargite reduced the length of zebrafish larvae following a 5 d exposure to 150 µg/L (Appendix 3).

25.2.8.3 Effects to salmonid prey

Two species of aquatic invertebrates were exposed to propargite, the waterflea (*D. magna*) and a chironomid (*C. riparius*). Four studies evaluated short-term exposures (48 h) to waterfleas to determine LC_{50}s. Daphnia LC_{50}s ranged from 14 – 91 µg/L. Two of the results tested technical propargite (LC_{50}s = 14 and 91 µg/L) and two tested formulations (LC_{50}s = 74 and 74 µg/L). In a separate experiment, 21 d continuous exposure of propargite to *D. magna* reduced reproduction at 14 µg/L (LOEC) and no such effects were reported at 9 µg/L (NOEC). A sediment toxicity test where *C. riparius* were exposed for 35 d following introduction of propargite to the overlying water reported an EC_{50} of 1770 µg/L for survival and emergence, a NOEC of 320 µg/L for larval survival, and a NOEC of 1000 µg/L for emergence. Although radiolabeled propargite was used in the experiment, values were reported as nominal.

25.2.8.4 Herbicidal effects

No experimental results were located on the effect of propargite on terrestrial plants. We did locate three results from experiments with green algae. Propargite reduced green algae growth with reported EC_{50}s of 66.2, 106, and 105,500 µg/L. A large variation in sensitivity exists as the EC_{50}s span more than four orders of magnitude. One aquatic plant was tested, duck weed, and showed very little sensitivity to propargite, i.e., LOEC = 75,000 µg/L.

Table 91 Propargite toxicity values (µg/L) for aquatic organisms and plants reported in EPA salmonid BE, CRLF BE, RED, EFED science chapter, and ECOTOX. Abbreviations as follows: a.i. = active ingredient; NA= Not Available; NR = Not Reported; T= Technical grade; F = Formulated product; sw = estuarine/marine species; [ ] = 95% Confidence interval.

<table>
<tr><th rowspan="3">Assessment Endpoint</th><th></th><th colspan="3">Concentration (µg/L aquatic tests or lbs a.i./acre terrestrial tests)</th></tr>
<tr><th></th><th colspan="2">Propargite
[ ] = 95% Confidence Interval</th><th>Degradates of Propargite: TBPC</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>salmonid LC_{50} (96 h)</td><td>Rainbow trout (O. mykiss) (% NR; T) = 118; probit slope = 4.5 [2-9;] MRID 0066498 (Kuc 1995)</td><td>Rainbow trout (O. mykiss) (76.2%; F) = 143; MRID 43759001 (Davis 1995)

Rainbow trout (O. mykiss) (30%; F) = 445; MRID 00043552

Rainbow trout (O. mykiss) (88.4%; T):
96 h = 43 [38-49];
72 h = 53 [46-62];
48 h = 84 [70-110];
24 h = >100;
21 d = 24, [21-32]
21 d NOEC = <14 µg/L (partial or complete loss of equilibrium, darkened pigmentation). 100% mortality at 21 ds in 100, 52, and 32 µg/l treatments. Note: measured concentration reported, flow through exposure.
MRID 41458301

Rainbow trout (O. mykiss) (52.8%; F) = 160 a.i., Omite 570EW, over spray to static water body with sediment, Hargreaves 2003. Unpublished report</td><td>Rainbow trout (O. mykiss) (100%) = 1.49 mg/L; NOEC = 306 µg/L (qualitative observations); LC_{50} = 3.6 mg/L (2 h exposure); NOEC = 0.96 mg/L (96 h survival). Values reported as measured concentrations. MRID 48557302</td></tr>
</table>

Assessment Endpoint		Concentration (µg/L aquatic tests or lbs a.i./acre terrestrial tests)		
		Propargite [ ] = 95% Confidence Interval		Degradates of Propargite: TBPC
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
			No. 22233, 08.05.2003 Rainbow trout (*O. mykiss*) (53.3%; F) Omite 570EW: 96 h LC50 = 47 [NA], 72 h LC50 =53 [NA], 48 h LC50 = 91 [48-165], 24 h LC50 = 216 [120-441]. Note: Measured concentration reported. Knight 2002. OECD guideline 203/EC CI. Unpublished report No. 20482, 10.05.2002 Rainbow trout (*O. mykiss*) (53.3%; F) = 160, NOEC = 110, Omite 570EW. Hargreaves 2003US EPA OCSPP Guideline Unpublished report No. 850.1075/1925. [static test systems contained 1.5 cm sediment and 800 mL water]	
Survival	Non-salmonid freshwater, estuarine, and marine fish LC_{50} (96 h)		Channel catfish (*Ictalurus punctatus*) (90.9%; T) = 40.4, NOEC = 18 (survival). Author not stated 1979. Unpublished report No. 11506-97, 15.03.1979. No chemical analyses of propargite.	

Assessment Endpoint		Concentration (µg/L aquatic tests or lbs a.i./acre terrestrial tests)		
		Propargite [ ] = 95% Confidence Interval		Degradates of Propargite: TBPC
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
			Sheepshead minnow (*Cyprinodon variegatus*) (87.4%; T): 96 h LC50 = 55 [32 – 60], survival NOEC 32; 72 h LC50 = 48 h LC50 = 70 [55-86]; 24 h LC50 = 100 [79 – 130], 10% mortality @ 24 h in 60 µg/L treatment. Note: measured concentrations reported. Omite technical, Static test. MRID 40514001 Bluegill sunfish (*Lepomis macrochirus*) (57 %; F) = 31; MRID 00112368 Bluegill sunfish (*Lepomis macrochirus*) (90.2%; T) = 81 (96 h), survival NOEC = 60 (survival), observed effects NOEC = 40. 72 h survival = 81, survival NOEC = 60, observed effects NOEC = 40. 48 h survival = 136 [85-200], survival NOEC = 60, observed effects NOEC = 40. 24 h survival = 361, survival NOEC = 241, observed effects NOEC = 60. Note: measured concentrations	

Assessment Endpoint		Concentration (µg/L aquatic tests or lbs a.i./acre terrestrial tests)		
		Propargite [ ] = 95% Confidence Interval		Degradates of Propargite: TBPC
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
			reported. MRID 46073301 Carp (*Cyprinus carpio*) (35%; F) = 330, MRID 00090718.	
Reproduction or larval survival	NOEC/ LOEC		Fathead minnows (*Pimephales promelas*) (86.3%; T) LOEC = 28 (100% mortality of hatched fry within 35 d); NOEC = 16 (no effect fry survival, length, weight). Note: Measured concentrations were used. MRID 00126739	
Fish growth and development	NOEC/ LOEC		Fathead minnows (*Pimephales promelas*) (89.9%; T): LOEC = 11 (length and weight); 27 µg/L reduced egg hatching (66% hatched), survival (0% survival of larvae post 30 days exposure), egg/spawn, spawn/female, growth; NOEC = 5.7 µg/L (growth of F_1 larvae). Note: measured concentrations reported. MRID 440866801	
Habitat: salmonid prey	Invert-ebrate	Waterflea (*D. magna*) (100%; T) = 91, MRID 00068752	Waterflea (*D. magna)* (76.2%; F) = 74, MRID 43759002 (Davis 1995)	Waterflea (*D. magna*) (100%; NA) = 3.35 mg/L;

Assessment Endpoint		Concentration (μg/L aquatic tests or lbs a.i./acre terrestrial tests)		
		Propargite [ ] = 95% Confidence Interval		Degradates of Propargite: TBPC
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
	survival (48 h EC/LC_{50})	Chironomid (*Chironomus riparius*) (99.4%, radio-labeled [^{14}C]) =1770 μg/L (28 d EC_{50} survival and emergence); NOEC = 320 μg/L (larvae survival); NOEC = 1000 μg/L (emergence); 28 d sediment test. Note: no confidence intervals reported due to lack of fractional responses at tested concentration; Concentration is reported as nominal concentrations added to test chambers on day 1. MRID 48557301	Waterflea (*D. magna*) (90.2%; T) = 14 [11-19], survival NOEC = 4; 24 h EC_{50}= 38 [30-46], survival NOEC = 15. Note: measured concentrations reported. MRID 46015901 Waterflea (*D. magna*) (53.3%; F) = 74; survival LOEC=38. Knight 2002. Omite 570EW. OECD 202 Part 1/EC C2 (European Commission Directive 92/69/EEC). Unpublished report no. 20930, 10.05.2002.	survival NOEC = 1.55 mg/L; MRID 48557303
	Invert-ebrate repro-duction NOEC/ LOEC (21 d life-cycle test)		Waterflea (*D. magna*) (88%; F) : NOEC = 9, LOEC =14 (reproduction), MRID 00126738 (Forbis 1983)	
Habitat: Riparian Vegetation	Vegetative vigor (lbs	NA	NA	NA

Assessment Endpoint		Concentration (µg/L aquatic tests or lbs a.i./acre terrestrial tests)		
		Propargite [ ] = 95% Confidence Interval		Degradates of Propargite: TBPC
	Assessment measure	> 95% a.i. (% a.i.)	< 95% a.i. (% a.i.)	
	a.i./acre):			
Habitat: Riparian	Seedling emergence (lbs a.i./acre): shoot length mortality EC_{25}	NA	NA	NA
Habitat: aquatic primary production	Aquatic plant growth: cell density EC_{50} & NOEC		Duckweed (*Lemna gibba*) (76.2%; F) = 75,000; NOEC = 28000, MRID 43885805 (Davis 1995). Green algae (*Kirchneria subcapitata*) (88.2%; NR) = >105500; NOEC = 4300, MRID 43414542 (Brock 1992) Freshwater diatom (*Navicula pelliculosa*) (76.2%; F) = 106, NOEC = 99, MRID 43885807 (Davis 1995) Freshwater green alga *(Selenastrum capricornutum*) (76.2%; F) = 66.2, NOEC =5, MRID 43885807 (Davis 1995)	

25.2.9 Bioconcentration and Bioaccumulation

Salmonids and their prey may accumulate the insecticides from the water (bioconcentrate) or from their food and water (bioaccumulate). We describe several studies below that tested accumulation of the insecticides.

25.2.9.1 Diflubenzuron:

We located five open-literature studies that measured tissue concentrations in fish following water-borne exposure to diflubenzuron. The first study (Apperson et al. 1978) measured diflubenzuron concentration in white crappie (*Pomoxis annularis*) tissues. A California lake was dosed with 5 µg/L diflubenzuron (25% formulation) and fish were sampled over the following month. Tissue concentrations increased to a maximum of 355 ppb at 4 d post-treatment, and declined to 62 ppb at day 21 and 0.4 ppb on day 35. Relative to average water concentrations, residues in fish represented a 49- to 123-fold increase. A second study (Schaefer et al. 1979) measured tissue accumulation in a laboratory setting to white crappie (*Pomoxis annularis*) and bluegill sunfish (*Lepomis macrochirus*). They found that diflubenzuron is accumulated from water into fish tissues at levels up to 80-fold after exposure for 24 hours to 10 µg/L. Fish showed modest reductions in tissue concentrations by 72 hours post-exposure. A third study (Colwell and Schaefer 1980) dosed several ponds with diflubenzuron to achieve a mean water concentration of 13.2 µg/L. Tissue residues in black crappie (*Pomoxis nigromaculatus*) and brown bullhead (*Ictalurus nebulosus*) were as high as 466.3 ng/g and 387.5 ng/g 1 d post-treatment, respectively. Residues in both species of fish decreased to non-detectable concentrations by 7 d post-treatment. They calculated tissue accumulations of 33- and 23-fold relative to water concentrations in black crappie and brown bullhead, respectively. The fourth study (Schaefer et al. 1980) showed that fish accumulated diflubenzuron up to 160-fold higher than water concentrations, and that tissue concentrations declined steadily over time. The final study (Booth and Ferrell 1976) measured tissue concentrations in channel catfish (*Ictalurus punctatus*) exposed Dimilin-treated soils in laboratory aquaria. The authors used radio-labeled dimilin (25% WP formulation; ^{14}C) at concentrations of 550 and 7 ppb and sampled fish for 28 d. Tissue concentrations rose to about 20 ppb after 4 d in the 550 ppb exposure. Concentrations in catfish viscera (i.e., digestive organs) plateaued and were 48 ppb on 28 d. Catfish meat concentrations were also about 20 ppb on day four, but steadily decreased to about 2 μg/L by 28 d. The data show that diflubenzuron did not bioaccumulate in channel catfish under the conditions of this experiment. This same study (Booth and Ferrell 1976) also measured accumulation of diflubenzuron by the blue-green algae *Plectonema boryanum,* a common constituent of aquatic systems, after exposure to a 100 μg/L aqueous concentration. The algae yielded a concentration of 145 ppm after 1 h, and rapidly eliminated the residue to 8 ppb by 4 d. Therefore, lower trophic levels in aquatic systems such as blue-green algae are not expected to transfer accumulated diflubenzuron to higher trophic levels such as fish. The studies summarized here demonstrate that while diflubenzuron is accumulated by fish from contaminated water, tissue concentrations decline rapidly once exposure ceases and bioaccumulation does not rise to a level of concern.

25.2.9.2 Fenbutatin oxide:

Several studies were found reporting that fish accumulated fenbutatin oxide (bioconcentration factors >1000). This is also consistent with other organotins. The most recent study (MRID 48973501) is considered of high quality since 1) radio-labeled fenbutatin oxide concentrations were measured both in tissues and water, 2) the exposure was maintained until after steady-state accumulation was reached, and 3) fish were transferred to clean water and the elimination of fenbutatin oxide was measured. Juvenile bluegill were continuously exposed to measured concentrations of either 2.8 or 34 ng/L ^{14}C-labeled fenbutatin oxide for 65 d and then transferred to clean water for 61 d of depuration. Fish were fed daily, so in addition to the water exposure some exposure may have been through the diet. No mortalities were observed during the experiment. For fish exposed to 2.8 ng/L, bioconcentration factors (BCFs) of 285, 1241, and 805 were calculated for edible tissue, non-edible tissue, and whole fish (respectively). The accumulation during this exposure was insufficient to calculate kinetic BCFs, time to reach 90% steady state, and time to reach 50% clearance. For fish exposed to 34 ng/L, the observed BCFs were 693, 1875, and 1350 for edible tissue, non-edible tissue, and whole fish (respectively). For this higher exposure, kinetic BCFs were calculated to be 757, 2186, and 1506 for edible tissue, non-edible tissue, and whole fish (respectively). The times to reach 90% steady state were calculated to be 48.7, 61.9, and 58.9 d for edible tissue, non-edible tissue, and whole fish (respectively). The times to reach 50% clearance were calculated to be 14.7, 18.7, and17.7 days for edible tissue, non-edible tissue, and whole fish (respectively). These results demonstrate that fenbutatin oxide is accumulated by bluegill and that it takes just over two weeks in uncontaminated water to eliminate the material. Importantly, this accumulation occurred at very low concentrations of fenbutatin oxide (ng/L rather than µg/L). Although negative effects in bluegill from accumulation of fenbutatin oxide were not measured, whether they are possible is a source of uncertainty.

25.2.9.3 Propargite:

A study measured accumulation of propargite by juvenile bluegill. Fish were continuously exposed to 3.1 µg ^{14}C-Omite [technical formulation, 87.4% propargite] /L (nominal) for five weeks after which the bluegill were transferred to tanks with uncontaminated water for a two week depuration period (MRID 48545818). Fish were fed twice daily, thus bluegill experienced two types of exposure, through the water column across the gills and through their diet via feeding. Mean measured concentrations averaged 4.1 µg/L propargite during the five week exposure. Steady state was reached at 7 and 10 d for edible tissue and non-edible tissue, respectively. No mortality was observed during the experiment. A concentration factor of 1550 was determined based on measuring ^{14}C propargite in exposure water and in non-edible tissue. For edible tissue, a factor of 260 was determined. After two weeks spent in clean water, propargite-contaminated fish had eliminated approximately 87.3% of accumulated propargite. These study results demonstrate that propargite is accumulated by bluegill and that it takes more than two weeks in uncontaminated water to eliminate the material. Although negative effects in bluegill from accumulation of propargite were not measured, they are possible. Generally, bioconcentration factors greater than 1000 are considered a concern for fish while factors greater than 5000 are recommended for banning use.[19]

25.2.10 Degradate Toxicity

25.2.10.1 Diflubenzuron:

[19] Category for Persistent, Bioaccumulative, and Toxic New Chemical Substances, Federal Register: November 4, 1999 (Volume 64, Number 213), pages 60194-60204.

Several degradation products of diflubenzuron are likely to occur in water due to abiotic and microbial processes (Metcalf et al. 1975, Ivie et al. 1980)). We located one study (Julin and Sanders 1978) that measured the toxicity of three degrades of diflubenzuron, 4-chlorophenylurea, 2,6-difluorobenzoic acid, and 4-chloroaniline. The study evaluated the toxicity to fish species including rainbow trout, fathead minnow, channel catfish and bluegill sunfish as well as an aquatic invertebrate (*Chironomus plumosus*). These compounds are not acutely toxic to fish or aquatic invertebrates, as evidenced by 48 h or 96 h LC_{50} values ranging from 2400 µg/L to > 100,000 µg/L. The most sensitive organism was the bluegill sunfish with a 96 h LC_{50} of 2400 µg/L to 4-chloroaniline. However, based on current application rates of diflubenzuron, this degradate concentration is not expected to be found in salmonid habitats.

25.2.10.2 Fenbutatin oxide:

No toxicity information was located for the few known degradates. Fenbutatin oxide is quite stable in water, soil, and tissue.

25.2.10.3 Propargite:

We located a registrant-submitted study that tested the acute toxicity of the propargite degradate TPBC on rainbow trout survival and one that tested survival of *D. magna* following exposure to TPBC. The 96 h LC_{50} was 1.49 mg/L for rainbow trout (MRID 48557302) and the 48 h survival EC_{50} for *D. magna* was 3.35 mg/L (MRID 48557303). Both studies suggest that TPBC is toxic to the two species in the mg/L range. Several other degradates were identified but since we located no toxicity information on them, a data gap exists.

25.2.11 Mixtures

We located no information on the effects of these insecticides in combinations with one another or in combinations with other chemicals (e.g. in formulations, tank mixtures, or environmental mixtures). Therefore, the potential effects are difficult to ascertain.

25.2.12 *Adjuvant Toxicity*

Although no data were provided in the EPA available documents related to adjuvant toxicity, an abundance of toxicity information is available on the effects of the alkylphenol polyethoxylates, a family of non-ionic surfactants used extensively in combination with pesticides as dispersing agents, detergents, emulsifiers, adjuvants, and solubilizers (Xie et al. 2005). Two types of alkylphenol polyethoxylates, nonylphenol (NP) ethoxylates and octylphenol ethoxylates, degrade in aquatic environments to the more persistent, toxic, and bioaccumulative degradates, NP and octylphenol, respectively. We did not receive information on the presence or absence of alkylphenol polyethoxylates in formulations of the three insecticides. 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 reports provided by EPA.

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

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

These results demonstrate NP is of concern to aquatic life, particularly salmonid endocrine systems involved in reproduction and smoltification. This summary is for one of the more than 4,000 inert/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 diflubenzuron, fenbutatin oxide, and propargite formulations. Consequently, the effects that these other ingredients may have on listed salmonids and designated critical habitat remain an uncertainty and are a recognized data gap in EPA's action under this consultation.

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

No incidents were reported in EPA's incident database for the three insecticides.

25.2.14 Data Gaps and uncertainties identified from review of available toxicity information for diflubenzuron, fenbutatin oxide, propargite

1. No information on effects to salmonid behaviors and associated assessment endpoints such as swimming, olfaction, endocrine system, immune-system, migration, spawning, and smoltification;
2. No field study results on the responses of threatened and endangered salmonids or their designated critical habitats;
3. No incident information;

4. No field studies in aquatic environments that tracked effects of real-world applications of propargite;
5. No empirical data on effects to riparian plant species from exposure to diflubenzuron or propargite; and,
6. No toxicity information on other ingredients within pesticides formulations containing the insecticides.

25.2.15 Evaluation of data available for response analysis

We summarize the available toxicity information by assessment endpoint in

Table 92,

Table 93, and

Table 94. Data and information reviewed for each assessment endpoint were assigned a 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 three insecticides was available for acute lethality to salmonids and aquatic invertebrates, and highly variable for the other assessment endpoints. However, much less information was available for other ingredients, due in part to the lack of formulation information provided in EPA’s reports as well as the statutory mandate under FIFRA for toxicity data on diflubenzuron, fenbutatin oxide, and propargite to support registration. We did locate a substantial amount of data on one group of adjuvants/surfactants, the NP ethoxylates. We received minimal information detailing tank mixes and other ingredients within formulations.

Table 92. Summary of Toxicity Data for Diflubenzuron

Assessment Endpoint	Evidence of adverse responses (yes/no)	Concentration range of observed effect or concentrations tested showing absence of effect (µg/L)	Degree of confidence in effects (low, moderate, high)
Salmonid survival (LC_{50})	No	57,000 - >1,000,000 (n=22)	high
Growth (NOEC)	No	>45 (n=1)	high
Reproduction (NOEL)	No	50 - 100 (n=2)	moderate
Respiration	-	-	-
Swimming	-	-	-
Olfactory-mediated behaviors	-	-	-
Prey survival (LC_{50}, EC_{50})	Yes	0.0028 – 57,500 (n=48)	high
Prey reproduction and growth (LC_{50}, LOEL)	Yes	0.062 (n=1) 0.04 - >10 (n=6)	high
Aquatic primary production (LOEC, EC_{50})	No	>30 - 380 (n=5) 5000 (n=3)	high
Riparian vegetation (terrestrial)	-	-	-
Additive toxicity	-	-	-
Degradate Toxicity	No	2,400 - >100,000 (n=12)	high

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

Table 93: Summary of Toxicity Data for Fenbutatin oxide

Assessment Endpoint	Evidence of adverse responses (yes/no)	Concentration range of observed effect or concentrations tested showing absence of effect (µg/L)	Degree of confidence in effects (low, moderate, high)
Salmonid survival (LC_{50})	Yes	1.1 - 52 (n=7)	high
Growth (LOEC)	-	-	-
Reproduction (LOEC) (larval growth, survival, hatching)	Yes	0.61 - 5.7 (n=3)	high
Respiration	-	-	-
Swimming	-	-	-
Olfactory-mediated behaviors	-	-	-
Endocrine disruption	-	-	-
Prey survival (EC_{50})	Yes	2.8 - 2184 (n=9)	high
Prey reproduction and growth (LOEC)	Yes	0.32 - 25 (n=4)	high
Aquatic primary production (EC_{50})	Yes	14 - 7434 (n=5)	high
Riparian vegetation (terrestrial, NOEC)	No	4 lbs a.i./Acre	high
Additive toxicity	-	-	-
Degradate Toxicity	-	-	-

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

Table 94: Summary of Toxicity Data for Propargite

Assessment Endpoint	Evidence of adverse responses (yes/no)	Concentration range of observed effect or concentrations tested showing absence of effect (µg/L)	Degree of confidence in effects (low, moderate, high)
Salmonid survival (LC_{50})	Yes	24 - 445 (n=14)	high
Growth (LOEC)	Yes	11 (n=1)	high
Reproduction (LOEC)	Yes	28 (n=1)	high
Respiration	-	-	-
Swimming	-	-	-
Olfactory-mediated behaviors	-	-	-
Prey survival (EC_{50})	Yes	14 -1770 (n=4)	high
Prey reproduction and growth (LOEC)	Yes	14 (n=1)	high
Aquatic primary production (EC_{50})	Yes	66.2 -75,000 (n=4)	moderate
Riparian vegetation (terrestrial, EC_{25})	-	-	-
Additive toxicity	-	-	-
Degradate Toxicity	Yes	1490 (n=1) rainbow trout 1550 (n=1) waterflea	high

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

25.3 Risk Characterization

In this section we integrate our exposure and response analyses to evaluate the likelihood of adverse effects to individuals and populations (Figure 71). First we evaluate any information that addresses effects to aquatic organisms from outdoor studies such as mesocosms and other field studies. We then combine 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.

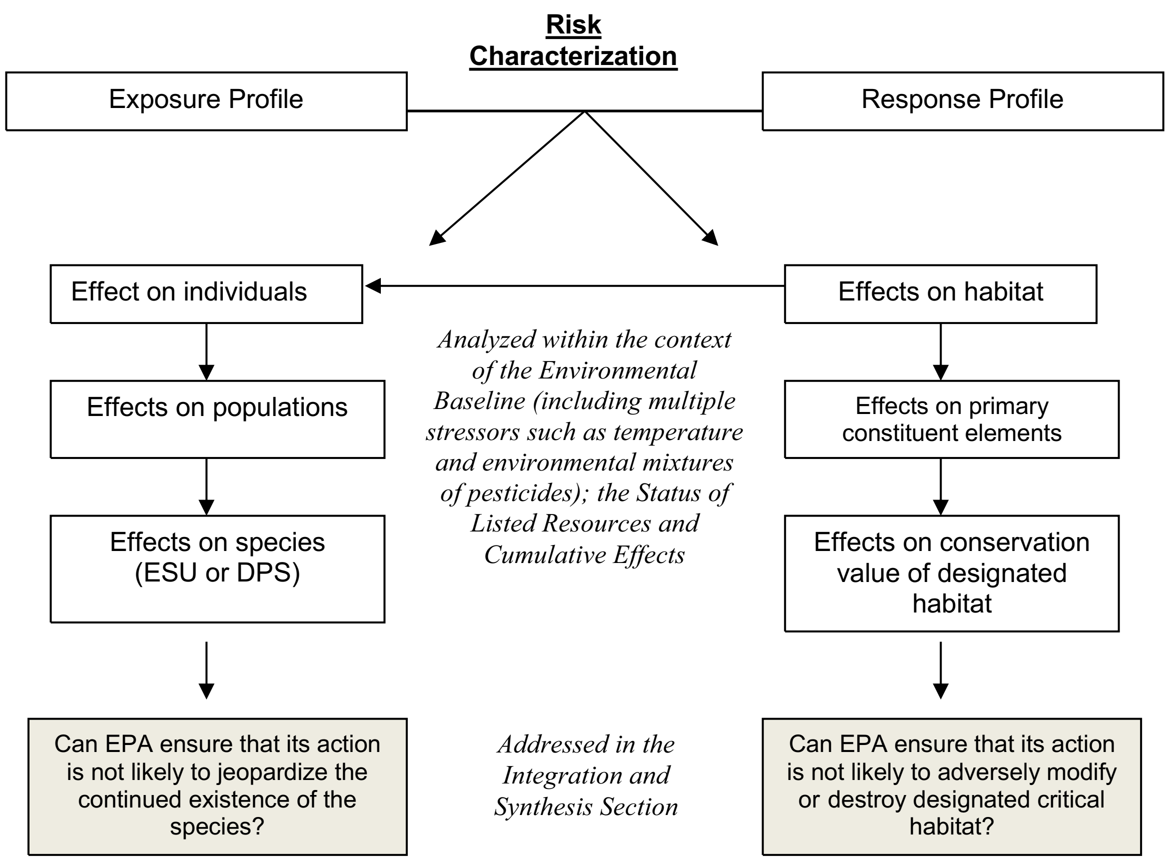

Figure 71. Schematic of risk characterization phase

25.3.1 *Relationship of pesticide use to effects in the field*

We located no field studies with propargite that evaluated exposure and response in outdoor aquatic areas.

We found reference to two field studies in a European review of fenbutatin oxide (EFSA 2010). The review reports two results using formulated product and without measured exposure concentrations. The NOEC for mesocosms following a 118 d exposure was 110 µg/L, while the NOEC for microcosms following a 28 d exposure was 10 µg/L. The original studies cited in the review were not available for NMFS review, so details of the study design and results could not be assessed. We also noted earlier in the Exposure Section that a study in Florida measured concentrations of fenbutatin oxide in aquatic systems, but did not report on biological responses of aquatic organisms. No other mesocosm or microcosm studies were found.

Over twenty studies were located that measured the response of various aquatic organisms in field situations following exposure to diflubenzuron. Exposed environments included outdoor mesocosms, stream channels, ponds, lakes, and holding ponds. Most studies observed aquatic invertebrate communities for several weeks to months following one or more diflubenzuron applications. In general, invertebrate communities were affected at low concentrations (low µg/L range), toxic effects were measurable within days, and population recovery took weeks to months.

Additionally, we located three open literature studies (Boyle et al. 1996, Tanner and Moffett 1995, Ludwig 1993) and one EPA report (Moffett et al. 1995) that measured fish growth following exposure to diflubenzuron at concentrations well below published fish LC_{50} values. Ludwig (1993) studied the growth of striped bass (*Morone saxatilis*) and the abundance of zooplankton in culture ponds treated once with diflubenzuron at 30 µg/L. In treated ponds, juvenile fish survival was significantly reduced to 15.1% (compared to 44% and 59% survival in controls) and total weight of fish was only 8.6 g (compared to 67.5 and 36.4 g in controls) after 1 month. Cladoceran abundance was initially reduced, but reached control levels after 2.5 weeks. Few copepod nauplii were present in the treated ponds throughout the experiment. The author concludes that fry survival rates were related to the concentration and size of zooplankton, particularly rotifers, cladocerans, and copepods, available during the critical juvenile growth period.

In a second study by Boyle et al. (1996), the authors treated replicate outdoor mesocosms with 10 μg/L diflubenzuron (nominal concentration) either monthly (5 total applications) or bi-monthly (9 total applications). Abundance, survival, and growth of invertebrates (insects and zooplankton) and fish (bluegill sunfish and largemouth bass) were measured for 16 weeks. Both monthly and bi-monthly treatments produced similar results. Zooplankton populations were significantly reduced, as cladocerans, copepods, and rotifers were nearly eliminated within 4 weeks of diflubenzuron application and remained depressed for the duration of the experiment. Diflubenzuron also reduced population abundance and species richness of emergent insects (dipterans). Average weight and biomass of juvenile bluegill were reduced 50% in the diflubenzuron treatments. Additionally, average condition factor of juvenile largemouth bass was significantly reduced to 0.9 (compared to1.0 in controls). No significant differences were observed in adult fish between treatments. The authors concluded that reductions in juvenile growth occurred because of apparent decreases in invertebrate food resources (zooplankton). Although dietary analysis was not performed, the authors further conclude that adult fish growth was not reduced due to shifts to alternate prey items, which is possible given their larger mouth gape and swimming ability. Therefore, diflubenzuron could cause significant ecological restructuring of aquatic ecosystems due to direct reductions of chitin-producing invertebrates, and indirect reductions in fish populations due to changes in prey availability.

In a third study by Tanner and Moffett (1995), the authors studied the diets and growth of larval bluegill sunfish (*Lepomis macrochirus*), the reproductive success of adult bluegill sunfish, and zooplankton population abundance in six enclosures in the littoral zone of a pond in Minnesota. Diflubenzuron was applied twice at nominal concentrations of 2.5 and 30 μg/L, and ponds monitored over an approximate 4 month period. Reproductive behavioral effects on adult bluegill were not observed. The number of spawning events declined after diflubenzuron application, but it was unclear if this was due to the chemical or a natural seasonal pattern. Growth rates of larval bluegill decreased by 56% and 86% in the 2.5 μg/L replicate enclosures, and growth decreased by 88% and 97% in the 30 μg/L replicate enclosures relative to growth in a control enclosure. Additionally, three major larval bluegill food items, cladocerans, copepods (nauplii) and chironomids, were reduced by 92%, 90%, and 55% (respectively) on 6, 6, and 21 days (respectively) after the first diflubenzuron application of 2.5 µg/L. Larval bluegill growth was significantly correlated to zooplankton and macroinvertebrate abundances. Bluegill diet analysis revealed dietary differences between treated and control fish five days after application, and for the remainder of the study. The authors concluded that diflubenzuron reduced larval bluegill growth rates indirectly by eliminating or reducing their preferred prey (cladocerans and copepods). In fact, six days following application, just when larvae were large enough to consume cladocerans, the cladoceran abundance was reduced by 92%. This study suggests that bluegill larvae are more sensitive to the indirect effects of diflubenzuron than older fish.

The final fish growth study we located was an EPA document (Moffett et al. 1995) that reported the effects of diflubenzuron on juvenile bluegill (*Lepomis macrochirus*) and indigenous fishes (*Culaea inconstans, Phoxinus eos,* and *Umbra limi*). Replicate pond enclosures were dosed twice at concentrations of 0.7, 2.5, 7, and 30 µg/L. No effect of diflubenzuron on abundance, biomass, and weight was evident on the three indigenous species. However, these species were only sampled at the end of the study, and growth rates of juveniles were not measured, so the conclusion of no effect should be made with caution. The authors did find a significant effect of diflubenzuron on the growth of juvenile bluegill. They calculated a LOEC of 2.5 µg/L and a NOEC of 0.7 µg/L for growth (length and weight), and juvenile bluegill growth was significantly lower at >2.5 µg/L diflubenzuron than controls. Additionally, the authors report a direct correlation of juvenile growth rates with crustacean zooplankton abundance (Cladocera and Copepoda). Invertebrates that are less preferred juvenile prey, such as Chironomidae and Rotifera, showed little or no correlation to bluegill growth rates. Also, diet analysis confirmed that juvenile bluegill consumed prey that were more abundant, and that their diet changed over the course of the experiment. The authors further conclude that this study clearly demonstrates the indirect effects of diflubenzuron on fish populations, and that similar results can be expected in other aquatic systems.

Table 95 Summary of diflubenzuron field and mesocosm studies on fish and various freshwater, estuarine, and marine invertebrates

Chemical	Taxa/Species	Assessment measures	Concentrations tested	Exposure duration	Effects	Data source
Dimilin 25% WP	Bluegill (*Lepomis macrochirus*); Macroinvertebrates (chironomids); Zooplankton (cladocera, copepoda, rotifera)	Reproductive success Abundance Growth	Applied twice to littoral enclosures at 2.5 and 30 µg/L a.i., nominal	Weekly sampling for about 3 months (June to September)	Increasing concentrations of diflubenzuron significantly reduced larval bluegill growth rates (length and weight) indirectly by eliminating or reducing prey (cladocerans and copepods). 2.5 µg/L reduced growth of juvenile bluegill by 56 and 86% (replicate enclosures) and by 88 and 97% at 30 µg/L. Abundance of cladocerans, copepods and chironomids were reduced at both concentrations relative to controls. Young of the year bluegill growth significantly correlated to zooplankton (especially cladoceran) and macroinvertebrate abundance.	(Tanner and Moffett 1995)

Dimilin 25% WP	Bluegill sunfish, young-of-the-year (*Lepomis macrochirus*)	Growth	Applied twice to littoral enclosures at 0.7, 2.5, 7, 30 µg/L, nominal.	70 d	NOEC = 0.7 µg/L LOEC = 2.5 µg/L Bluegill Y-O-Y growth significantly lower than controls at 2.5 µg/L. Bluegill growth rates were directly correlated to the density of invertebrate taxa (Cladocera and Copepoda). Size, abundance, and biomass of indigenous fish species not affected up to 30 µg/L.	(Moffett et al. 1995)

Diflubenzuron (Dimilin 25W)	Outdoor mesocosms: Zooplankton Insects Bluegill sunfish (adults and juveniles) Largemouth bass (juvenile)	Insects: Density Species abundance Fish: Mortality Growth Reproduction Phytoplankton: Biomass Productivity	10 µg/L (9.9 µg/L actual) applied monthly or biweekly	Exposed monthly or biweekly for 5 months, measured weekly for following 16 weeks.	Significant reductions in zooplankton populations; however, significant seasonal trend also present. Cladocera, Copepoda, and Rotifers nearly eliminated in all treatments within 4 weeks. Number of species lower in treated mesocosms. Number of insects and number of insect species lower in treatments vs. control, but not significant due to time (confounding variable). Chlorophyll a significantly higher in treatments vs. control, but increased over time in both treatments and control. In diflubenzuron treatments: Average weight, condition and biomass of juvenile bluegill significantly reduced. Average condition of juvenile largemouth bass significantly reduced. Weight, biomass and condition of adult bluegill declined, but not significantly. Overall, lower total fish biomass.	(Boyle et al. 1996)

Diflubenzuron (25% WP)	Striped bass (*Morone saxatilis*) Zooplankton (rotifers, copepods, copepod nauplii, ostracods, cladocerans)	Abundance	One application at 30 µg/L to holding ponds; subsequent water concentrations not measured	Approximately 1 month	Chemically treated ponds had lower fish survival and lower total weights. Cladoceran populations reduced compared to control. Ponds treated with diflubenzuron had the fewest copepods, and the lowest mean daily concentration of copepods and copepod nauplii.	(Ludwig 1993)
Diflubenzuron (Dimilin 25 WP)	Chironomidae Ephemeroptera	Abundance	30, 7.0, 2.5, 0.7 µg/L	Pond enclosures treated, sampled up to 56 d later	For chironomids at 30 µg/L, there was a 79% reduction in abundance on 15 d and a 60% reduction in abundance on 28 d. At 7.0 µg/L, decrease of 58% on 15 d. NOEC = 2.5, LOEC = 7.0 µg/L. For Ephemeroptera, significant reductions on 57 d at 2.5 µg/L and higher concentrations (NOEC = 0.7, LOEC = 2.5 µg/L).	(O'Halloran et al. 1996)

Diflubenzuron (Dimilin)	Ephemeroptera (mayfly) Plecoptera (stonefly) Diptera (Chironomid)	Survival Density Diversity	0.1, 1.0, 10, 50 µg/L dissolved in dimethylformamide (DMF)	Stream channels, continuous exposure for 5 months	Direct toxic effects at concentrations greater than 1 µg/L. Total insect diversity significantly reduced at 10 and 50 µg/L by 1 month, mildly reduced at 1.0 µg/L. Chironomids (Dipterans) density reduced 10-75% at 10 µg/L by 1 month, but survived at low densities for 5 months before being eliminated. Mayfly populations eliminated at 1.0 µg/L and higher by 1 month. Did not recover by end of 5 month study. Stonefly populations reduced 50-90% at 1.0 µg/L or eliminated by 1 month. Did not recover by end of 5 month study. No effects on Trichoptera, Coleoptera, Oligochaeta and Gastropoda. No increased drift. Evidence of abnormal metamorphosis.	(Hansen and Garton 1982b)

Dimilin WP-25	Zooplankton (Daphnidae and Bosminidae) Amphipods Benthic invertebrates	Abundance, Mortality	Max aerial application rate (Canada); highest concentration was 13.82 µg/L one hour post-spray.	Invertebrates monitored up to 110 d post-aerial spray.	Zooplankton (cladocera and copepoda) populations were reduced 3 d after treatment and remained suppressed for 2-3 months. Mortality of caged amphipods was approximately twice as high in the treated ponds (82-90%) as in the control pond (42-53%). Abundance of insect species including mayfly, dragonflies, and damselfly were significantly reduced after application. Reductions generally not apparent until 21 to 34 d post-application. Numbers returned to pre-treatment or control levels by 68 to 110 d after treatment.	(Sundaram et al. 1991)
Dimilin 4L formulation (40.4 % a.i.)	Stoneflies Mayflies Crane flies	Abundance	Not measured	Monthly sampling for one year post-treatment.	Densities of some taxa in treated watersheds showed either population depressions or no increases.	(Hurd et al. 1996)

Dimilin 25% WP	Insects Chironomids	Emergence	0.7, 2.5, 7.0, 30 µg (a.i.)/L	Applied to pond enclosures on July 9 and August 11. Insects sampled 10 d prior to treatment until 53 d post treatment	Insecta EC_{50} = 1.2 µg/L (probit 1.4 [1.1-1.7]) NOEC = 1.0 µg/L LOEC = 1.9 µg/L Chironomidae EC_{50} = 1.2 µg/L (probit 1.3 [0.7-1.9]) NOEC = 1.0 µg/L LOEC = 1.9 µg/L	(Liber et al. 1996)
Dimilin 25% WP	Mayflies (Heptageniidae) Stonefly (*Peltoperla arcuata*)	Abundance, molting	0.0, 0.6, 5.6, 55.7, 557.2 µg/L (mayflies); 0, 1.0, 10.2, 101.5, 1015 µg/L (stoneflies)	Exposed for 96 h, followed by clean water for 36 d	Mayflies had significantly reduced survival after 4 days to all concentrations tested. Behavioral changes included decreased swimming speed, altered avoidance behavior, and sporadic shaking. Far fewer successful molts, especially at higher concentrations. Stoneflies did not show dose-response relationship. No behavioral changes observed.	(Harrahy et al. 1994)

Dimilin 25% WP	Stoneflies (*Peltoperla arcuata* and *Pteronarcys proteus*)	Survival, molting	Concentrations on leaves ranged from 416 to 80 µg/L	Consumed treated leaves for 24 d, observed for 60 or 90 d.	Stoneflies (*P. arcuata*) fed contaminated leaves showed significantly lower survival at 60 days. No effect on molting. *P. proteus* survival not significantly different from controls. Few animals molted, no difference from controls.	(Harrahy et al. 1994)
Diflubenzuron	Black crappie (*Pomoxis nigromaculatus*) Brown bullhead (*Ictalurus nebulosus*) Zooplankton	Mortality Growth Condition factor	Mean concentration 13 µg/L	Experimental ponds treated, then sampled over the following 3 months	For 1 month following treatment, stomach content analyses indicated major alterations in diet. Growth rates and condition factors of fish 3 months post-treatment similar to controls. Zooplankton (cladocera and copepod) decreased within a few days after treatment, populations returned to pre-treatment levels between 7 d and 4 wk later.	(Colwell and Schaefer 1980)

Diflubenzuron	Littoral enclosures including: Phytoplankton, Periphyton, Macrophytes, Cladocerans, copepods, Rotifers, Ostracods, Bluegill sunfish, Chironomidae, and Ephemeroptera	Biomass Abundance Taxa/Species composition	0.7, 2.5, 7.0, 30.0 µg/L	Two applications 1 month apart, sampled up to 37 d after second application	Cladoceran abundance significantly reduced by 89% at 0.7 µg/L, recovered to near control levels 29 d post-treatment. Periphyton NOEC = 2.5, LOEC = 7 (80% reduction) Cladocerans NOEC = <0.7, LOEC = 0.7 (89% reduction in abundance) Copepods NOEC = <0.7, LOEC = 0.7 (81% reduction in abundance) Ostracods NOEC = 2.5, LOEC = 7 (84% reduction in abundance) Bluegill sunfish NOEC = 0.7, LOEC = 2.5 (35% reduction in growth) Chironomidae (larvae) NOEC = 2.5, LOEC = 7 (58% reduction in abundance) Ephemeroptera NOEC = 0.7, LEOC = 2.5 (100% reduction in abundance) Amphipods NOEC = <0.7, LOEC = 0.7 (88% reduction in abundance) Chironomidae adult emergence NOEC = 0.7, LEOC = 2.5 (89%	(Hanratty and Liber 1996)

Diflubenzuron 25% WP	Cladocerans (*Daphnia, Ceriodaphnia, Bosmina*) Copepods (*Cyclops, Diaptomus*) *Hyalella azteca* Mayfly (*Caenis* sp.)	Survival Abundance	Applied at 156 g a.i./ha (12 µg/L)	Two applications: April 26th and August 24th	Cladoceran populations completely eliminated within one week post-treatment. *Daphnia* and *Ceriodaphnia* did not recover after 6 months, *Bosmina* reappeared after 11 weeks. Copepods eliminated within one week post-treatment. *Cyclops* recovered 6 weeks after treatment, *Diaptomus* recovered after 4 months. *Hyalella* eliminated within 4 weeks, absent for more than 6 months post-treatment. Mayfly nymphs reduced 99%, recovering in 6 weeks post-treatment. Second pesticide treatment had similar results.	(Ali and Mulla 1978a)

Diflubenzuron (granular formulation)	Cladocerans (*Daphnia* sp.) Amphipods (*Hyalella azteca*) Copepods (*Diaptomus*) Ostracods (*Cyprinotus* sp.) Oligochaete worms	Survival Abundance	Applied at 0.11 kg a.i./ha (3.7 µg/L) and 0.22 kg a.i./ha (7.4 µg/L) to a lake surface.	Low conc. applied to a southern CA lake surface on June 22, high conc. applied on August 27, both monitored for 4 weeks post-treatment.	Low concentration: *Daphnia* reduced 62-75% within 7 d, returned to pretreatment level by 2 weeks. *Diaptomus* reduced 30% after 2 d. *Hyalella* reduced 97%, remained low for study duration. High concentration: *Daphnia* completely eliminated for 3 months following treatment. *Diaptomus* eliminated within 7 d, but returned to pretreatment levels within 2 weeks. *Hyalella* reduced 32-100% during the 2.5 months post-treatment.	(Ali and Mulla 1978b)

Diflubenzuron (TH 6040 formulation)	Waterflea (*Daphnia* spp.); Clam shrimp (*Eulimnadia* spp.) Tadpole shrimp (*Triops longicaudatus*); Mayfly nymphs (*Callibaetis* spp.); Midge larvae (*Chironomus* spp.); Backswimmers (*Notonecta* and *Buenoa*); Mosquitofish (*Gambusia affinis*)	Survival Abundance	Mesocosms treated with 5 µg/L Ponds treated with 0.1, 0.025, 0.02, 0.01, and 0.005 lbs a.i./acre	Observations for 2 months following one or two treatments.	In outdoor mesocosms: Waterflea (Cladoceran) abundance markedly reduced on 4th day post-treatment, recovered in 3 weeks. Only mature waterfleas noticed in treated mesocosms until 16 d post-treatment. Similar reductions were seen in copepods, but magnitude of reduction was smaller and recovery time was shorter. In treated mesocosms, backswimmer abundance same as controls but neither eggs nor nymphs were observed for 2 months post-treatment. In treated ponds: All clam and tadpole shrimp gone 3 d post-treatment at 0.01 and 0.025 lb A.I./acre. Waterflea (Cladoceran) populations also temporarily reduced.	(Miura and Takahashi 1974)

Diflubenzuron (TH 6040)	Aquatic community	Survival	0.02 to 0.05 lb a.i./acre	Various pasture ponds treated in 19 aerial applications, monitored for 1 week; Bioassays for 5 days	In treated pasture ponds: Cladocerans and mayfly nymph populations reduced, but recovered a few days post-treatment. Bioassays: Daphnia mortality was 100, 55.5, 40 and 31.2% in water treated with 0.02 lb a.i./acre and held in the lab prior to testing for 0, 24, 48 and 72 h. Daphnia mortality was 100, 100, 84.4 and 74.7% at 0.04 lb a.i./acre at same holding times.	(Miura and Takahashi 1975)

Diflubenzuron (25% WP)	Gnat (*Chaoborus astictopus*) Cladocerans Copepods Rotifers Bluegill sunfish (*Lepomis macrochirus*)	Survival Emergence	2.5, 5, 10 µg/L to surface of farm ponds. 5 µg/L to a lake surface.	One treatment to each waterbody	Recovery interval for *Daphnia* sp. was about 1 month in the 5 and 10 µg/L farm pond treatments, and about 2.5 months in the lake treatment. Copepod (especially nauplii) declined after treatment, returned to pre-treatment levels after 3 to 6 weeks. In 5 and 10 µg/L farm ponds, emergence reduced by 98% by 2 and 7 d post-treatment, substantial emergence did not reoccur until 4.5 – 6 weeks later. Lake treated at 5 µg/L reduced emergence by 95-100%. *C. astictopus* larval counts reduced by 96, 88, 44% in farm ponds, and 99% in treated lake. No effects observed on rotifers. Bluegill sunfish stomach contents in pre-treatment lake were predominantly cladocerans, copepods, and chironomid pupae and adults. Post-treatment stomach contents were predominantly chironomids and terrestrial insects	(Apperson et al. 1978)

Dimilin	Cranefly (*Tipula abdominalis)* Caddisfly (*Platycentropus radiatus*)	Survival Growth	Animals fed Dimilin treated leaves (6.4 mg/L)	330 and 450 d; fed treated leaves continuously	Survival much lower in animals fed treated leaves (20% vs. 80-100% survival). Growth significantly lower in animals (*Platycentropus*) fed treated leaves.	(Swift et al. 1988)
Diflubenzuron (25% WP)	Copepoda (*Cyclops* spp.) Coleoptera (larval) Diptera (*Chaoborus* sp.; larval)	Survival	One treatment at 28 (6 µg/L) and 56 (12 µg/L) g a.i./ha in experimental ponds	One treatment, then sampled weekly for 5 weeks.	Both concentrations reduced 96-100% of *Cyclops* populations for at least 9 d. Larval *Chaoborus* reduced for 1 week at low concentration, reduced and remained very low at high concentration for 5 weeks. Coleoptera larvae reduced at both concentrations for 5 weeks.	(Ali and Lord 1980)

Dimilin (25% WP) and Dimilin (0.5% granular)	Chironomids (*Tanytarsus* and *Pentaneura*) Mayflies (*Baetis* sp.) Copepods (*Cyclops* sp, *Diaptomus* sp.) Cladocerans (*Daphnia*) Ostracods	Survival Emergence	0.25 lb/acre (80 µg/L) 0.05 lb/acre (16 µg/L)	One treatment in a pasture pond, then sampled for 2 weeks.	Chironomid emergence markedly depressed up to 11 – 15 d after WP treatment. Copepod populations reduced starting 2 – 4 d after treatment, increased at 11 d. Cladoceran density almost reached zero at 4 d post-treatment, increased 11 – 15 d post-treatment.	(Mulla et al. 1975)
Dimilin (25% WP)	Hydropsychidae larvae (Trichoptera)	Survival Emergence	2.5, 25, 250 µg/L a.i.	Exposed for 15 days 4 different times in a year	Significantly more larvae survived in control tank than in all treated tanks. Control survival averaged 44.7% in four trials, and 1.9% (at 2.5 µg/L), 2.2 % (at 25 µg/L), and 0.6% (at 250 µg/L) in treated tanks. In control tank 71% of survivors pupated and emerged as adults, while no larvae pupated or emerged in the experimental tanks.	(Bradt and Williams 1990)

Diflubenzuron (Dimilin)	Copepod (*Eurytemora affinis*)	Survival % reaching adulthood % of females with broods Nauplii abundance	Range between 0.21 – 1.7 ppb	**Early**- exposed for 6.5 days, then clean water until day 14. **Late**- clean water until day 7, then exposed until day 14 **Continuous**- continuous exposure for 14 days.	In early exposure: significantly reduced survival at 0.78 and 0.93 µg/L; significantly fewer nauplii per female at 0.93 µg/L. Fewer, but not significantly, % reaching adulthood and % females with broods at 0.93 µg/L. In late exposure: significantly fewer % females with broods at 0.98 and 0.78 µg/L; no nauplii produced at 0.98 and 0.78 µg/L. In continuous exposure: Significantly lower survival at 0.5, 0.78 and 0.93µg/L; significantly fewer % reaching adulthood at 0.78 and 0.93 µg/L; no females with broods at 0.78 and 0.93 µg/L. Individuals at higher concentrations had attached exuviae, associated with exposure during the later copepodite stages.	(Wright et al. 1996)

Diflubenzuron 25% WP	Copepod (*Eurytemora affinis*)	Survival	Range of concentrations from 0.33 to 7.73 µg/L	Exposed 5 to 6 days	Significantly lower survival at 0.78 µg/L in two experiments, and at 0.93 µg/L in one experiment.	(Savitz et al. 1994)
Dimilin 25 WP	Zooplankton Benthic invertebrates Adult insects	Survival Emergence	0.56 kg a.i./ha applied to a pond (canal led to a lake)	Sampled for 56 days post-treatment	Most organisms showed no difference between treatments and controls. No significant suppression of Chironomid emergence.	(Ali et al. 1988)
Diflubenzuron (UBI 6958; Dimilin 80 WG)	Aquatic ecosystems (zooplankton, macro-invertebrates, neuston)	Survival Emergence	Two applications	Sampled up to 84 days post-application	NOEC for Daphnia < 0.16 µg/L. Negatively impacted at all exposure concentrations. Zooplankton NOEC 0.64 to 1.3 µg/L Negative impact on zooplankton diversity at highest two concentrations. Corixidae NOEC < 0.16 µg/L. Full population recovered by day 84 at 1.3 µg/L exposure. Invertebrate community NOEC considered > 2.0 µg/L.	MRID 49038503

25.3.2 Exposure and Response Integration

We show the overlap between exposure estimates for the three insecticides and concentrations that affect assessment endpoints (
Figure 72, Figure 73, and Figure 74). This portion of the analysis does not take into consideration other stressors of the action or environmental conditions that may contribute to toxicity. The figure shows the exposure concentration ranges (minimum to maximum values) gleaned from the three sources of exposure data we analyzed: EPA's estimates presented in the BEs; NMFS' modeling estimates for flood plain habitats; and surface water monitoring data from ambient monitoring programs and from targeted monitoring (if available). In addition to the salmonid BEs submitted to NMFS, we also considered the exposure estimates developed by EPA in the BEs for the California red-legged frog. The effect concentrations are values taken from the toxicity data reviewed in the *Response Analysis section*. For the survival assessment endpoint, effect concentrations are LC_{50}s, thus death of individuals occurring at concentrations below LC_{50}s are not represented. Consequently, when LC_{50} effect concentrations are not exceeded by the exposure estimates, it does not mean that no salmonids died. Thus for those instances where LC_{50}s do not overlap or are not exceeded by exposure estimates, we discuss the difference in magnitude of the two metrics and apply best professional judgment on whether death of individuals is anticipated. This coarse analysis does not present temporal aspects of exposure nor does it show the distribution of toxicity values. It is predicated primarily on standard toxicity endpoints as we located little ecologically relevant sublethal information on the stressors of the action. We did locate a robust body of studies that evaluated the effect of diflubenzuron on aquatic invertebrates (Table 95) however we located few outdoor studies on fenbutatin oxide and none on propargite. The exposure and response integration analysis allows us to systematically address which assessment endpoints are likely to be affected by exposure to diflubenzuron, fenbutatin oxide, and propargite.

Modeled concentrations of diflubenzuron exceed toxicity thresholds that kill salmonid prey and that affect prey reproduction and growth. Given that fish LC_{50}s are in the upper mg/L range, we do not anticipate salmonids dying from short term exposures to authorized uses of diflubenzuron.

Fish reproduction and growth were not directly affected by the highest concentrations tested i.e. 100 and 45 µg/L (NOECs), respectively; thus we lack data on effect thresholds for these endpoints. We located studies where fish growth was affected due to effects on the prey base. These effects occurred as low as 2.5 µg/L. We located no ambient surface water monitoring results for diflubenzuron.

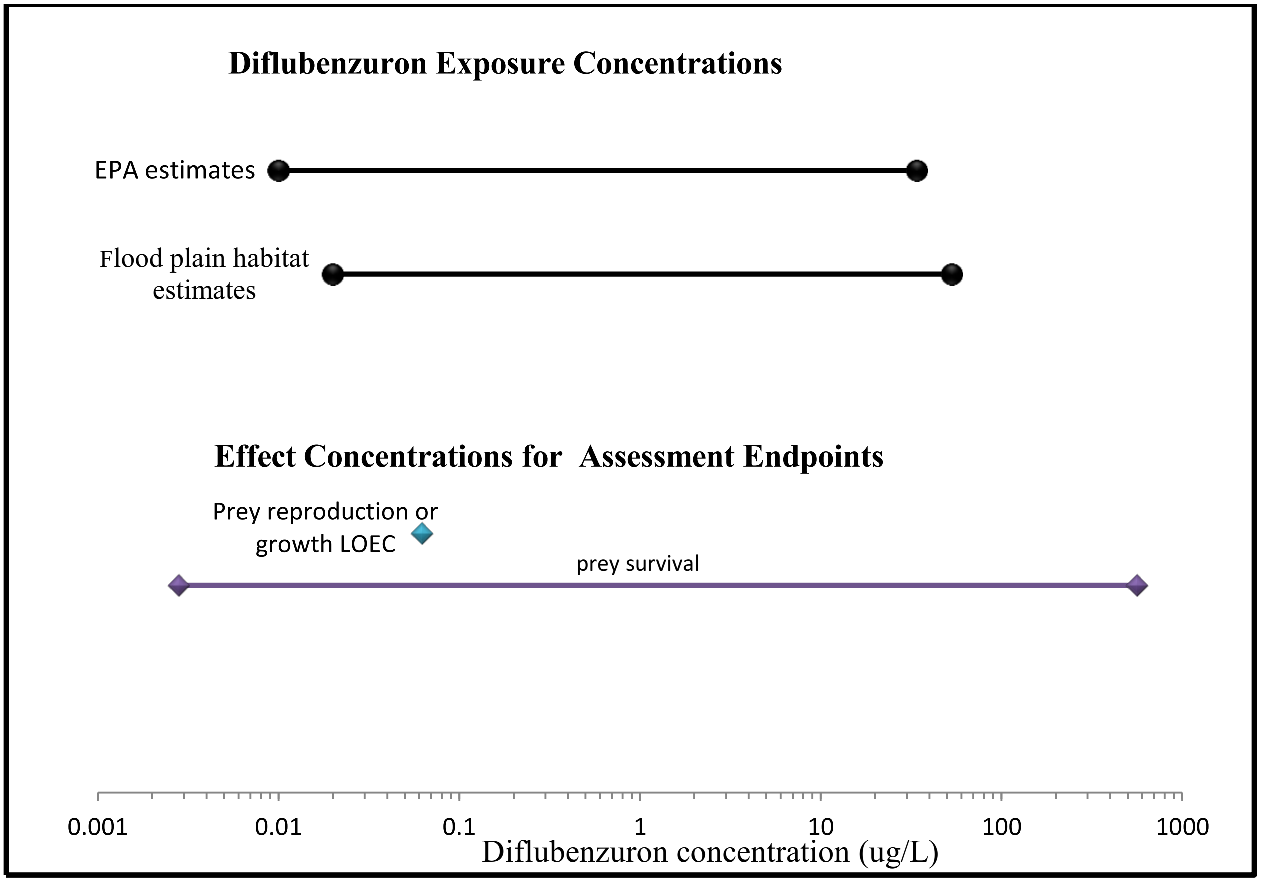

Figure 72 Comparison of exposure concentrations and salmonid assessment endpoint effect thresholds for diflubenzuron

Measured and modeled concentrations of fenbutatin oxide exceeded toxicity thresholds for salmonid and habitat assessment endpoints. The maximum concentrations from each of the types of exposure data (targeted monitoring, EPA model estimates, floodplain habitat estimates) exceeded all of the LC_{50} salmonid and fish reproduction values. The lower range of primary producers is exceeded by exposure concentrations.

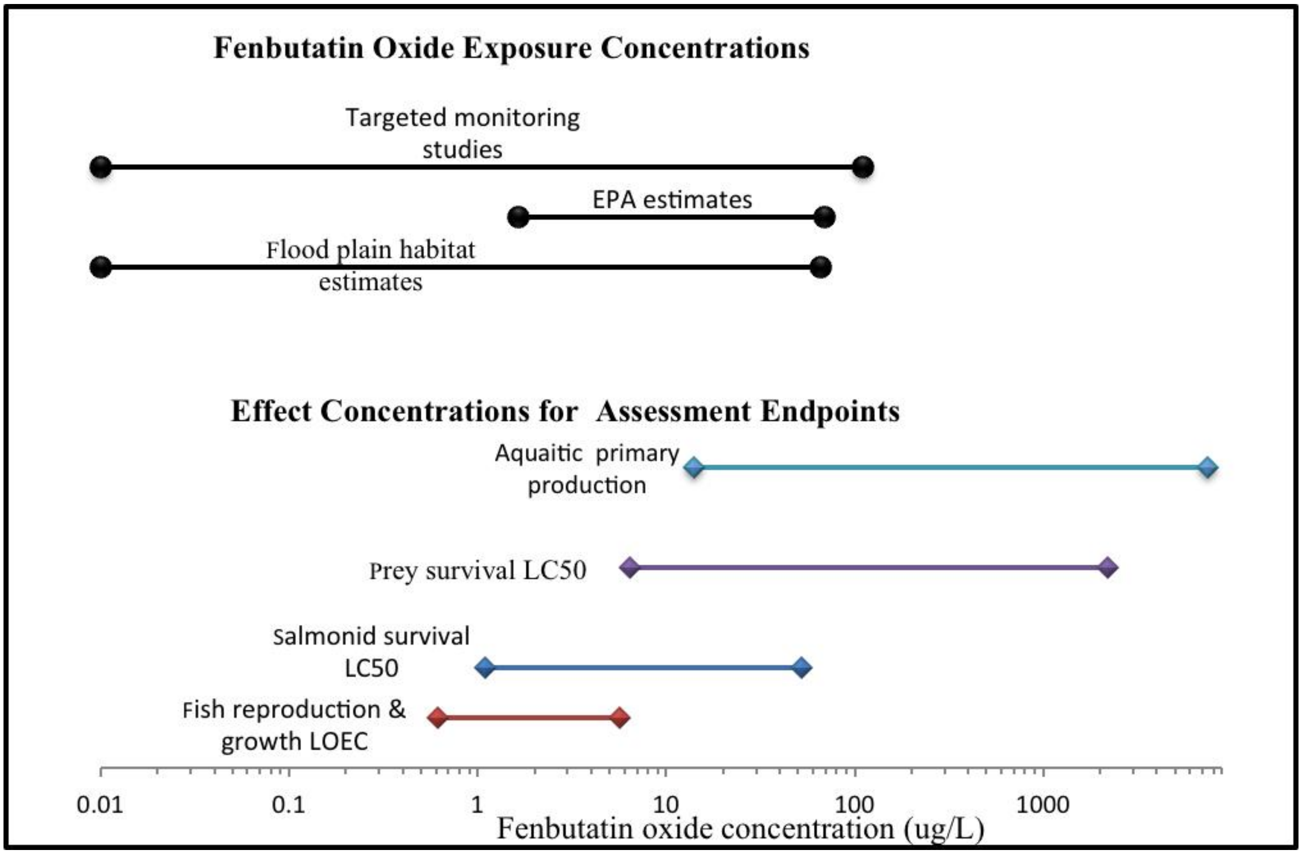

Figure 73 Comparison of exposure concentrations and salmonid assessment endpoint effect thresholds for fenbutatin oxide.

Measured and modeled concentrations of propargite are near or exceed toxicity thresholds that kill salmonids and their prey as well as impact fish growth and reproduction. The exposure estimates for flood plain habitats exceed all of the assessment endpoints indicating that salmonids and their prey are particularly vulnerable in these habitats. Monitoring data exceed some of the assessment endpoints near the lower end of the range. The lowest salmonid LC_{50} (24 µg/L) is very close to the maximum monitoring value reported (20 µg/L) as is the fish reproduction LOEC (28 µg/L). The upper range of EPA's exposure estimates for a 2 m deep, 2 hectare water body exceed all assessment endpoints, particularly those at the lower end of their ranges. Notable data gaps include no available information on sublethal effects to fish or prey. This introduces uncertainty especially if sublethal effects occur well below LC_{50} estimates.

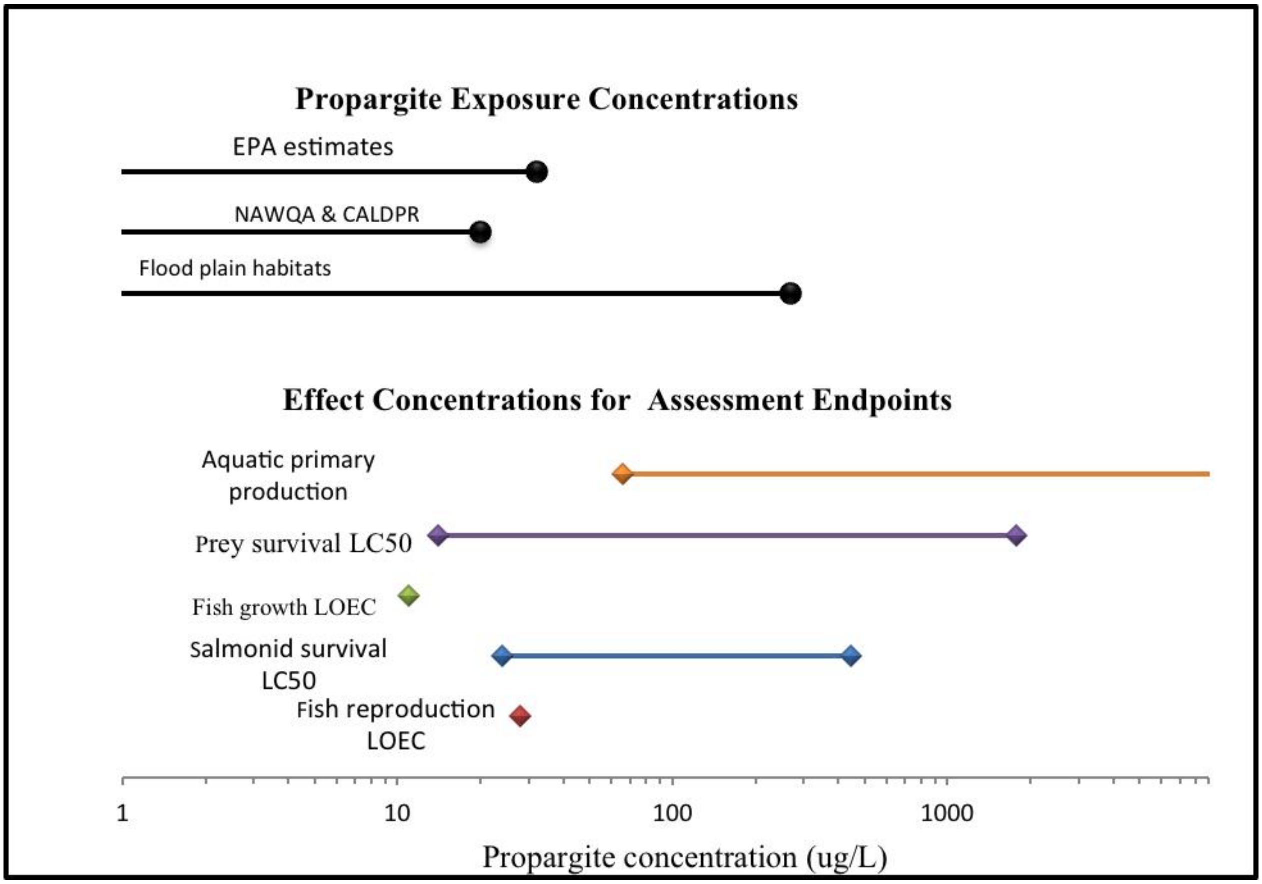

Figure 74 Comparison of exposure concentrations and salmonid assessment endpoint effect thresholds for propargite

25.3.3 Mixtures containing the stressors of the action

California's Department of Pesticide Regulation maintains an extensive dataset on pesticide use (1999-2010) and in particular pesticides that are co-applied, i.e. applied on the same day to the same area. We used this information to evaluate whether the three insecticides are applied as mixtures and if so what types of pesticides are co-applied (**Table 96**). All three insecticides are frequently applied with other pesticides; 29 -70% of reported applications are applied with other pesticides. Many of the pesticides that are applied on the same day as diflubenzuron, fenbutatin oxide, and propargite have undergone ESA Section 7 consultation regarding their effects to listed Pacific Salmonids. All of the organophosphate and carbamate insecticides listed in **Table 96** were found to jeopardize Pacific salmonids and adversely modify their designated critical habitat. No measures have been put in place to limit these pesticides from entering salmonid habitats; therefore we anticipate exposure to salmonids and their habitats is occurring and will continue to occur. The three insecticides being evaluated in this Biological Opinion are likely to be co-applied with these pesticides and therefore salmonids will likely be exposed. The effects of the three insecticides in various combinations of mixtures have not been assessed, although we anticipate that salmonids and their prey will experience additional stress and toxicity from these pesticides.

Table 96 California Pesticide use summary for 1999-2010 for pesticides applied to the same area on the same day (co-application).

	Diflubenzuron	Fenbutatin oxide	Propargite
Total Number of Applications	109,757	35,721	120,819
Total Number of Applications as Mixtures	31,357	20,700	84,568
Percentage of Applications as Mixtures (%)	29	58	70
Total Amount of Active Ingredient (lbs)	223,361	536,446	10,319,345
Total Amount of Other Active Ingredients (lbs)	4,699,101	3,411,899	5,762,342
Total Amount of Specific A.I.s Co-Applied (lbs)			
Diazinon	6,191	8,901	48,083
Malathion	9,186	34,266	33,788
Chlorpyrifos	181,136	254,451	1,041,854
Methomyl	74	13,187	40,972
Carbaryl	2,440	16,672	33,009
Dimethoate	14,952	2,794	103,094
Naled	1,605	2,363	201,538
Methidathion	12,108	288	5,489
Phosmet	17,662	260,982	419,959
Copper Sulfate	233,101	367	135
Esfenvalerate	475,041	331,101	208,997

25.3.4 Risk Hypotheses

We examine the weight of evidence to determine whether a given risk hypothesis is supported or refuted (Table 97). This is not a statistical analysis, but rather a qualitative assessment of the available exposure and response information. We also highlight uncertainties and data gaps associated with the data as they relate to risk hypotheses. In some instances no information for a risk hypothesis may be available on one or more of the three insecticides. In such circumstances we typically broaden the net and evaluate other pesticides that share a mode of toxic action. For example, in previous Biological Opinions with organophosphate (OPs) insecticides, we extrapolated from the available data on other OPs to fill data gaps in the response analysis. Given that diflubenzuron, fenbutatin oxide, and propargite have different, and in some cases, unknown modes of toxic action to aquatic organisms we do not extrapolate from other pesticides' toxicity data.

When we find evidence that a risk hypothesis is supported, i.e., salmonids' individual fitness is compromised, we then evaluate the potential consequences at the population level- the next step in the analysis following evaluation of risk hypotheses. When we find evidence that a risk hypothesis is unsupported, we terminate the analysis of that assessment endpoint and do not evaluate consequences at the population level. For those hypotheses that have no information available (either in support or against), we label this as an uncertainty and describe how the uncertainty affects the analysis.

We characterized exposure to the stressors of the action by evaluating surface water monitoring data and estimates from pesticide transport models (runoff and drift). We combine this information with the distribution and life-history characteristics of ESA-listed Pacific salmonids by ESU. As discussed in the *Exposure Analysis* section above, each source of information has inherent limitations and uncertainties. For example, the pesticide monitoring data were not designed to quantify peak exposure concentrations or distributions of exposure in salmonid habitats within California, Idaho, Washington, and Oregon. Additionally, for diflubenzuron no monitoring data are available and for fenbutatin oxide data from only a single, targeted monitoring study is available. Consequently, we used pesticide exposure models to supplement monitoring data, and together this information was used to describe the potential range of pesticide concentrations that salmonids may experience. We conducted AgDrift model runs to provide estimates for concentrations resulting from drift to floodplain habitats, often shallow and narrow bodies of water. Small streams and many floodplain habitats are more susceptible to higher pesticide concentrations as their physical characteristics provide less dilution compared to larger, high flow systems and many are proximate to high pesticide use areas.

We recognize that pesticide concentrations will vary markedly among salmonid habitats throughout the four states and that exposure duration and concentration will be reduced by higher water volumes and velocities.

Standardized toxicity tests for pesticide registration are typically poor predictors of real world aquatic ecosystems as test organisms are exposed to constant pesticide concentrations for arbitrary durations (*e.g.,* acute, 96-h and chronic, 21- or 28-d) that may poorly reflect field exposures, which tend to be repeated exposures. Additionally, the tests do not account for exposure to stressors, pesticides and others, already in the water (discussed in the Environmental Baseline section). The response of fish and their prey to different durations of exposure, potentially multiple exposures, and exposure representing different dissipation patterns of the stressors of the action is a prominent data gap. Exposure durations sufficient to elicit toxicological responses can occur at durations much shorter than standard toxicity tests. We therefore did not average exposure concentrations over time, *i.e.*, time-weighted averages, because adverse responses to short term 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 life stage and its life history are important considerations in its use of aquatic habitats. This natural variation is overlaid with the inherent variation of environmental factors including climate (*e.g.,* precipitation patterns), habitat stressors, and land use. Given this biological and environmental variability, it is difficult to predict the precise exposure for any one individual, let alone for independent populations or ESUs/ DPSs (species). Consequently, we used the life history information of the species to evaluate potential exposure in their myriad aquatic habitats.

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 focused on habitats used for rearing and migrating including first order streams, floodplain habitats, estuaries and near-shore marine areas, and large rivers. These habitats comprise critical elements to ensure successful adult migration, development and growth of young fish, and provide safe passage to and from the ocean.

This framework allows us to evaluate risk hypotheses based on the spatial and temporal nature of exposure to the stressors of the action and Pacific salmonid ESUs. Below we evaluate the evidence for each risk hypothesis and make a finding as to whether fitness of individual salmonids is compromised (**Table 97** and **Table 98**). If fitness is compromised, we conduct an analysis at the population level; if fitness is not compromised, we do not anticipate population level effects and therefore do not evaluate potential population scale consequences.

DRIFT:

Aerial drift of the stressors of the action occurs during their application period, which may occur anytime throughout the year. Aquatic concentrations depend on proximity to application area, application method, droplet size, release height, wind speed/direction, receiving water volume/flow, and interception by riparian vegetation. Based on drift studies, modeling exercises, and surface water monitoring studies, concentrations may range from non-detectable to 280 μg/L for diflubenzuron, non-detectable to 111 μg/L for fenbutatin oxide, and non-detectable to 269 μg/L for propargite.

RUNOFF:

Runoff following applications may result in the three insecticides entering salmonid habitats. Final concentrations within salmonid habitats receiving runoff will be dependent on factors including application rate, existence of vegetated filter strips, riparian areas, receiving water flow and depth. We located no information on runoff concentrations of the three insecticides into salmonid habitats.

Table 97 Risk hypotheses: Effects to Salmonids

Effects to salmonids
1. Exposure to diflubenzuron, fenbutatin oxide, or propargite via drift or runoff is sufficient to:
a. Kill salmonids from direct exposure
b. Reduce salmonid survival through impacts to growth
c. Reduce salmonid survival through impacts to reproduction
d. Reduce salmonid growth through impacts on the availability and quantity of prey
e. Impair swimming
f. Accumulate in salmonids impairing salmonids' fitness
Effects from other stressors of the action and contributing environmental factors
2. Exposure to degradates of diflubenzuron, fenbutatin oxide, or propargite will cause adverse effects to salmonids and their habitats.
3. Exposure to adjuvants, tank mixtures and other chemicals within pesticide products containing diflubenzuron, fenbutatin oxide, or propargite cause adverse effects to salmonids and their habitats.
4. Exposure to other pesticides present in the action area can act in combination with the three insecticides to increase effects to salmonids and their habitats.
5. Exposure to elevated temperatures enhances the toxicity of the stressors of the action.

25.3.5 Evaluation of Risk Hypotheses

1. Exposure to diflubenzuron, fenbutatin oxide, and propargite is sufficient to:

a. Kill salmonids from direct exposure

We located robust laboratory assays that evaluated acute toxicity of the three insecticides. Salmonids were well studied (n = 22) and appeared very tolerant to acute concentrations of diflubenzuron, as LC_{50} estimates were in the mg/L range, i.e., at concentrations that are not anticipated in salmonid habitats. Based on these data and diflubenzuron's toxic mode of action, we do not expect salmonids to be killed by short-term exposures to currently authorized uses of diflubenzuron. Therefore, we do not evaluate risk to populations from death of individuals.

A substantial body of laboratory toxicity data indicates that 50% of a test population of salmonids die following short term (<96 h) exposures to both fenbutatin oxide and propargite in the low µg/L range. In comparison to other insecticides that have gone through ESA section 7 consultations, fenbutatin oxide is one of, if not, the most acutely toxic. Salmonids are extremely sensitive to fenbutatin oxide and are expected to die at concentrations well below 1 µg/L. Propargite is slightly less acutely toxic to salmonids compared to fenbutatin oxide and it is expected to kill salmonids in the low µg/L range. In real world aquatic systems exposure duration and health condition of individuals are key determinants of lethality, as are the presence of other stressors of the action and stressors present in the environmental baseline. We located no fish lethality experiments conducted in the field and no reported incidents of fish die offs were reported by EPA.

We expect that adults and juveniles will die if exposed to fenbutatin oxide- and propargite containing drift or runoff from applications near or proximate to salmonid-bearing habitats. The most pronounced effects will likely be experienced by juveniles and adults occupying shallow habitats adjacent to application areas. We evaluate the implications from death of juveniles and adults at the population scale.

b. Reduce salmonid survival through impacts to growth

We located several standardized laboratory study results that measured growth in a variety of fish species. Reductions in larval growth occurred at low µg/L concentrations and in the case of fenbutatin oxide at 0.61 µg/L (LOEC) following 60 d exposure to rainbow trout. Diflubenzuron did not affect juvenile steelhead growth over 30 d at the highest concentration tested, 45 µg/L (NOEC). Propargite affected fish growth at 11 µg/L (LOEC). With fenbutatin oxide and propargite, exposure concentrations may attain or exceed toxicity thresholds for growth and therefore we find support for this risk hypothesis. Individual fitness of exposed juveniles will likely be compromised for those juveniles that are exposed for multiple days or to repeated applications of the two insecticides. For these reasons, we evaluate the impacts of reduced growth at the population scale from exposure to fenbutatin oxide and propargite.

c. Reduce salmonid survival through impacts to reproduction

Limited data were available to evaluate the effects on salmonid reproduction. Registrant-submitted studies for a variety of fish species indicated that fish reproduction could be affected by fenbutatin oxide and propargite. That said, uncertainty regarding time to effect and exposure duration makes it difficult to translate the laboratory results to field situations. We located no additional peer-reviewed studies from the open literature that evaluated reproduction at the individual or population scales. Diflubenzuron did not elicit effects on reproduction or development up to 45 µg/L in rainbow trout (NOEC). The limited study results indicate that reproduction may be affected by fenbutatin oxide and propargite and therefore we evaluate the potential consequences at the population scale. For diflubenzuron, we have a limited dataset coupled with minimal information on diflubenzuron's mode of action in fish. We do not anticipate fish reproduction being affected at the scale of the individual and therefore do not evaluate potential effects at the population scale.

d. Reduce salmonid growth through impacts on the availability and quantity of prey

We located many studies that measured the effects of diflubenzuron on aquatic invertebrate abundance and fish growth. Experiments were conducted in outdoor ponds, mesocosms or lake enclosures at concentrations within the range expected to be found in salmonid habitats. Macroinvertebrate taxa were extremely sensitive to diflubenzuron, with significant population decreases observed at low µg/L concentrations. Fish growth was significantly decreased at concentrations ranging from 0.7 – 30 µg/L, and all studies linked reduced fish growth directly to reductions in available invertebrate prey. Larval and juvenile life stages of fish were especially vulnerable to changes in their prey base, since they were not able to shift to alternate prey items due to physiological (e.g. gape and mouth size) and behavioral (e.g. swimming ability) limitations. Therefore, these studies showed that larval and juvenile fish grew less following diflubenzuron application due to a reduction in abundance of their invertebrate prey, thereby providing strong evidence in support of this hypothesis.

No results were found that tested fish growth in aquatic systems where either propargite or fenbutatin oxide were introduced. However, reported invertebrate LC_{50} values do overlap with modeled environmental concentrations. We anticipate that when concentrations of propargite and fenbutatin oxide attain levels that reduce salmonid prey abundance (low µg/L), juvenile salmonid growth will be compromised. Therefore, we evaluate the effect of reduced salmon growth from application of the three pesticides at the population level.

e. Impair swimming

Compromised swimming may lead to reduced growth (via reductions in feeding), delayed and interrupted migration patterns, reduced survival (via inability to escape predators), and reproduction (reduced spawning success) (Little and Finger 1990). No studies were located that evaluated swimming behaviors following exposures to the three insecticides. The absence of any data creates a sizeable data gap and introduces uncertainty. We also have no chemical surrogates for these three insecticides as they each have unique or unknown modes of toxic action. Typically swimming may be impacted at concentrations well below 96 h LC_{50} values for fish. A literature review summarized effects of contaminants on swimming behavior and found that swimming behaviors are frequently affected at 0.7 – 5.0 % of fish LC_{50}s (Little and Finger 1990). If this is the case, swimming would be impaired at concentrations as low as 0.008 µg/L fenbutatin oxide and 0.2 µg/L propargite. We are uncertain at what concentrations below the LC_{50} swimming is actually impacted by these two compounds. Given that adverse effects to swimming frequently occurs below LC_{50} estimates and that LC_{50} estimates are exceeded by anticipated exposure concentrations, we discuss potential population level impacts.

f. Accumulation of the three pesticides from the water and/or prey impairs salmonids' fitness.

We located bioaccumulation data for each of the three insecticides. Diflubenzuron and propargite are not expected to elicit fitness level consequences from accumulation based on each compound's moderate tendency to accumulate. Fenbutatin oxide, however, is expected to accumulate in salmonids at concentrations that may degrade a salmonid's fitness. The organotin pesticide is persistent in aquatic habitats and accumulates in fish.

Although we located no studies that assessed the responses of aquatic organisms following accumulation of fenbutatin oxide, the toxicity and physical/chemical properties suggest that organism health may be compromised. Therefore, we discuss the implications of bioaccumulation by salmonids at the population level.

25.3.6 Risk hypotheses regarding other stressors of the action and contributing environmental factors

2. **Exposure to degradates of diflubenzuron, fenbutatin oxide, and propargite cause adverse effects to salmonids and their habitat**

We located toxicity information on diflubenzuron and propargite degradates from registrant-submitted studies, however no information was available for fenbutatin oxide degradates. No information was provided on potential exposure concentrations for any of the degradates. Based on toxicity data for propargite and diflubenzuron, salmonids and their prey appear tolerant as effect levels were in the mg/L range and above. These concentrations are several orders of magnitude higher than the anticipated levels in salmonid habitats for parent compounds. No information was located for degradates of fenbutatin oxide. When introduced to water, fenbutatin oxide rapidly divides into two identical molecules, which are persistent, and responsible for toxicity. We therefore rely on the toxicity of the parent compound to address risk and do not anticipate that its degradates are responsible for additional toxicity. We do not expect individual fitness to be compromised substantially by degradates of the three insecticides.

3. **Exposure to adjuvants, tank mixtures and other ingredients within pesticide products containing diflubenzuron, fenbutatin oxide, or propargite cause adverse effects to salmonids and their habitats.**

Salmonids and their habitats are likely exposed to other stressors of the action including other chemicals in formulated products and tank mixtures. More than four thousand inert or other ingredients are approved for use in end-use pesticide products by EPA, as well as adjuvants, such as surfactants and other products that are applied within tank mixes. Once a mixture is introduced into the environment, physicochemical properties influence their rates of partitioning and fates within aquatic and terrestrial systems. We anticipate some percentage of these stressors will be present in salmonid habitats from spray drift deposition and from runoff events following application. We also expect that salmonids and their habitats exposed to multiple chemical stressors will show a greater response than laboratory animals exposed to one pesticide active ingredient. The available toxicity information generally underestimates the response from a field-applied application.

One active label for diflubenzuron contained another active ingredient, permethrin. The label authorizes applications to livestock to kill flies and lice. We do not anticipate substantial off-site movement for this application.

In addition to other/inert ingredients that are listed on pesticide labels, thousands of other compounds are approved by EPA for addition to pesticide end-use formulations without any specific requirement for the compound identity or amount to be listed on the labels. One example of these ingredients is the nonylphenol polyethoxylates, which have been linked to endocrine disruption and were addressed at length in previous Biological Opinions on EPA pesticide registrations (NMFS 2008e, NMFS 2009b, NMFS 2010b).

4. Exposure to other pesticides present in the action area act in combination with the three pesticides to increase effects to salmonids and their habitats

We expect that aquatic habitats within the action area receive inputs of numerous toxic chemicals including pesticides that vary in space and time. Which combinations and at what concentrations will lead to interactions with the three insecticides is unknown. Aquatic organism responses to mixtures can be divided into three categories of toxicity; synergism, additivity, or antagonism. In previous BiOps we found support for additivity and synergism when mixtures contained chemicals with the same mode or mechanism of action. None of the current compounds share a mode of action or mechanism of action with other pesticides making it difficult to forecast potential effects with any degree of certainty. Our default assumption is for chemicals to behave in an additive manner. We, therefore, expect fitness consequences that may occur from exposure to the three insecticides will be worse and potentially magnified by the presence of additional insecticides that have already been found to jeopardize salmonid species and their habitats. The results of co-application as mixtures are of particular concern given the risk associated with the insecticides presented in Table 96. We find substantial support for this hypothesis and discuss potential consequences to populations.

5. Exposure to elevated temperatures enhances the toxicity of stressors of the action

We located no information on whether elevated temperatures affect the toxicity of the three insecticides. As these three insecticides have unique and unknown modes of toxic action, chemical surrogates are unavailable and ill-advised. This is a data gap that produces a large uncertainty as many of the aquatic habitats that salmonids utilize are listed as impaired due to elevated temperatures. We do not discuss potential population level effects given the scarcity of data and lack of available evidence.

Table 98. Summary of individual-based risk hypotheses

Risk Hypotheses	Is individual fitness of exposed salmonids compromised?		
1. Exposure to diflubenzuron, fenbutatin oxide, or propargite via drift or runoff is sufficient to:	**diflubenzuron**	**fenbutatin oxide**	**propargite**
a. Kill salmonids from direct exposure	not anticipated	anticipated	anticipated
b. Reduce salmonid survival through direct impacts to growth	not anticipated	anticipated	anticipated
c. Reduce salmonid survival through impacts to reproduction	not anticipated	anticipated	anticipated
d. Reduce salmonid growth through impacts on the availability and quantity of prey	anticipated	anticipated	anticipated
e. Impair swimming	not anticipated	anticipated	anticipated
f. Accumulate in salmonids impairing salmonids' fitness	not anticipated	anticipated	anticipated
Effects from other stressors of the action and contributing environmental factors			
2. Exposure to degradates of diflubenzuron, fenbutatin oxide, or propargite will cause adverse effects to salmonids and their habitats.	not anticipated	not anticipated	not anticipated
3. Exposure to adjuvants, tank mixtures and other chemicals within pesticide products containing diflubenzuron, fenbutatin oxide, or propargite cause adverse effects to salmonids and their habitats.	anticipated	anticipated	anticipated
4. Exposure to other pesticides present in the action area can act in combination with the three insecticides to increase effects to salmonids and their habitats.	anticipated	anticipated	anticipated
5. Exposure to elevated temperatures enhances the toxicity of the stressors of the action.	not anticipated	not anticipated	not anticipated

25.3.7 Salmonid Population Models

We selected four life-history strategies to model (Appendix 7). 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 were unable to construct a steelhead (*O. mykiss*) life-history model due to lack of demographic information. Stream-type Chinook salmon were used as a surrogate for steelhead as they share many of the same life history attributes. 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 7.

While models were constructed for different salmon species and life-history strategies, they were not specific to a particular salmon population. For many salmon populations, the necessary demographic data are unavailable. Additionally, while there may be differences in some demographic rates among individual populations, the similarity of response within a life-history strategy is well established (Winemiller and Rose 1992, Schaaf et al. 1987, Schaaf et al. 1993). Generic models, therefore, are still useful to estimate risk across populations that share a life history strategy.

We use these models to develop informed estimates of how a hypothetical population of salmonids may respond to exposure from each of the three insecticides, i.e., at what concentration a population's intrinsic rate of growth (lambda) is affected. As such, this threshold concentration should not be interpreted as a threshold for all specific independent populations comprising an ESU. Rather we compare the threshold with the available exposure information for each pesticide and make a determination as to whether concentrations described in the exposure section are above or below the threshold value that impacts the hypothetical population.

In this sense, we qualitatively compare and discuss whether or not real-world populations are likely exposed to concentrations that elicit reductions in a population's lambda.

One frequent area of misunderstanding regarding population models in past pesticide BiOps is the concept that the population modeling results were based on (or used) exposure data presented in the *Exposure Analysis Section*, and therefore, that the population models are deterministic and the results apply directly to all listed populations. This is incorrect and a misunderstanding of how population modeling results were and are used in pesticide Opinions. The threshold values derived from a hypothetical population are qualitatively compared to the range of exposure data presented in the *Exposure Analysis section* to provide a weight-of-evidence approach on the likelihood of population level consequences to real-world populations.

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 fenbutatin oxide and propargite. Diflubenzuron was not modeled because acute lethality of juveniles is not expected from short-term exposures (see Response section for toxicity information). The acute toxicity models excluded sublethal and indirect effects of the pesticide exposures and focused on the population-level outcomes resulting from acute, short term exposures to juveniles in a population to a single exposure of the active ingredient. The model does not account for the potential impact of additional acute exposures and will, therefore, underestimate impacts if fish experience more than one acute exposure during their juvenile phase. Death of juveniles was implemented as a change in first-year survival rate for each of the salmon life-history strategies modeled. Exposure to pesticides likely varies from year to year depending on overlap of salmonids and application periods as well as differences in seasonal weather patterns. Therefore, we evaluated population level responses resulting from varying the proportion of a population exposed (17, 25, 50, 75, and 100 per cent) to account for a range of exposure scenarios. This information helps us estimate the likely impacts to a population based on the fraction of the population exposed.

25.3.7.1 Acute exposures to sub-yearling salmonids

We use the results of laboratory LC_{50} experiments to establish dose-response relationships to predict the impact of acute exposure at different concentrations to fenbutatin oxide and propargite. In this context, acute exposures represent short-term exposures lasting several days or less (i.e. less than a week). Ideally, field toxicity data that evaluated the death of juvenile salmonids across a range of exposure durations (hours to days), habitats, populations, pesticide uses, time periods (multiple months, years) would be available to represent the varying conditions in Idaho, California, Washington, and Oregon. Unfortunately, these types of field studies are expensive, difficult to execute, and, if working with threatened and endangered species, are a threat to ESA-listed populations.

One potential source of information that is robust and used frequently to predict/estimate death of individual organisms from short-term exposures is data from laboratory LC_{50} experiments. The toxicological standard in ecotoxicology for aquatic organisms including fresh water, marine, and estuarine fish is the acute, 96 hour LC_{50}, and as such has been, and still is, used widely to predict survival to acute exposures. It is not a perfect fit for predicting effects in the field since exposure durations vary in the field, and those conditions are difficult to mimic in laboratory settings. How much the LC_{50} will change with different exposure conditions will depend on the species and the contaminant and is a source of uncertainty. However, data on the 96 hour LC_{50} is the most commonly available and, as such, provides a large source of relevant toxicological information on many contaminants and species.

In addition, the use of LC_{50} data is standard practice in ecological risk assessment for assessing the risk of acute, short-term exposures to aquatic species. For example in U.S. regulatory arenas acute LC_{50}s are used in derivation of national water quality criteria and state water quality standards. In U.S. pesticide registration and re-registration, EPA's Office of Pesticide Programs uses acute 96 h LC_{50} data to evaluate the level of risk posed by pesticide active ingredients to threatened and endangered species as well as for comparing against pesticide-modeling exposure estimates. Many other regulatory programs and statutes use LC_{50} data to predict effects to populations including CERCLA and NRDA activities. USGS and USEPA recently developed pesticide thresholds that used LC_{50} data to predict effects to aquatic organisms in the field. Thus laboratory data from 96 hour exposures are used to assess the risk from short-term, acute exposures in the field of various durations and therefore is a convention in each of the above programs. Given the available information on acute toxicity of pesticides to aquatic species, and in particular, to salmonids, we find that using LC_{50} data is appropriate and consistent for estimating population level impacts from death of individual fish.

25.3.7.2 Acute effects to salmonids and populations they comprise

The percent changes in the intrinsic population growth rates (lambdas) increased as concentrations of the two insecticides increased (Table 99). 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 due to the lower 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 LC_{50} and related slope values are important drivers for these results. Therefore, an LC_{50} above or below the ones used here will result in a different dose-response. We selected the lowest reported salmonid LC_{50} from the available information to ensure that risk is not underestimated. However, if the actual environmental LC_{50} is lower, then the model will under predict mortality. If the actual environmental LC_{50} is higher, then the model will over-predict mortality. The amount of time it takes to kill a salmonid is also uncertain as most LC_{50} experiments do not report when mortality of individual fish occurs.

Table 99. Acute salmonid mortality as a function of concentration

Pesticide	Fenbutatin oxide		Propargite	
LC_{50}[1]	1.1		43	
Slope (sigmoid)	4.3[2]		3.6[3]	
Concentration relative to LC_{50}	Concentration (µg/L)	Mortality (%)	Concentration (µg/L)	Mortality (%)
0	0	0	0	0
0.1	0.11	0	4.3	0
0.2	0.55	5	21.5	8
0.8	0.88	28	34.4	31
1.0	1.1	50	43	50
1.2	1.32	69	51.6	66
1.5	1.65	85	64.5	81
2.0	2.2	95	86	92

1 denotes lowest salmonid LC_{50}.
2 sigmoid slope is mean of two slopes estimated using non-linear regression from data set (MRID 113075 and 40473507)
3 sigmoid slope is converted from probit slope of 4.5 (MRID 0066498).

The shape of the dose-response curves for propargite and fenbutatin oxide are shown in Figure 75 and Figure 76. Concentrations between 23 and 32 µg/L propargite are anticipated to be the lower thresholds for reducing the intrinsic rate of growth for the four modeled populations. This illustrates that acute toxicity of propargite at estimated concentrations may reduce the viability of exposed populations of juveniles. The degree to which the acute effects will manifest at the population level in any given year is dependent on the overlap of pesticide loading with fish presence. Likewise, concentrations of fenbutatin oxide between 0.7 and 0.9 are anticipated to be the lower thresholds for reducing the intrinsic rate of growth for the four modeled populations. These are extremely low concentrations and are anticipated from currently authorized uses of fenbutatin oxide.

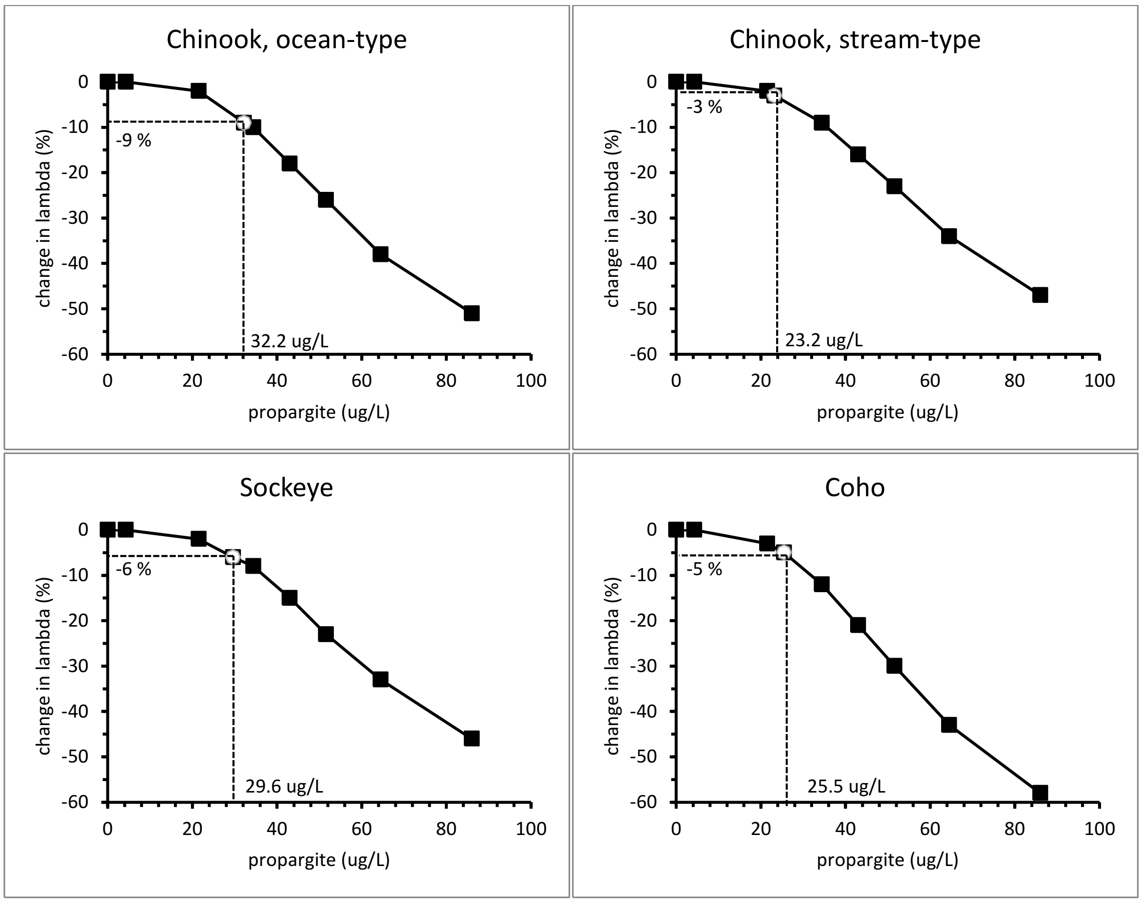

Figure 75. Population dose-response for death of juvenile salmonids following acute exposure to propargite

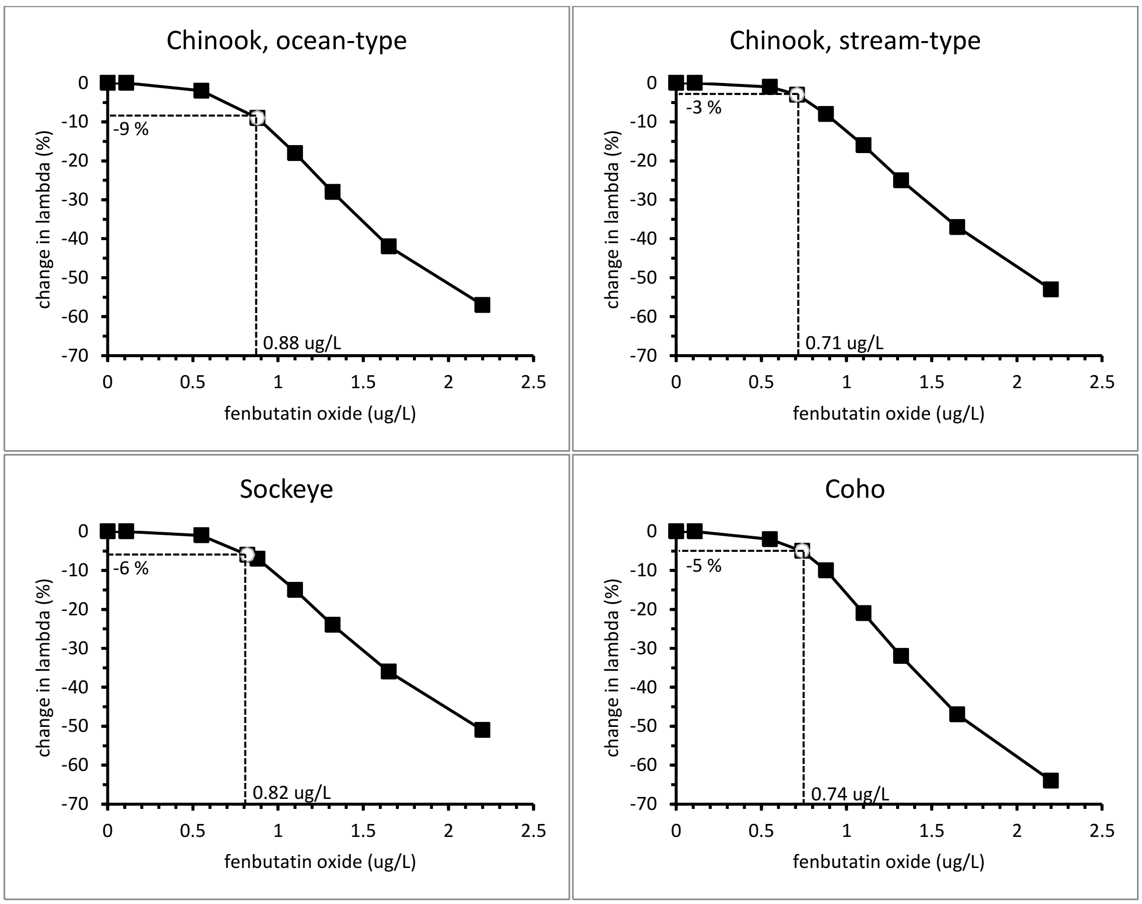

Figure 76. Population dose-response for death of juvenile salmonids following acute exposure to fenbutatin oxide

In the earlier example, the dose-response relationship was modeled based on all juveniles of a given population being exposed once to an acute concentration of either fenbutatin oxide or propargite during their freshwater residency period. Table 100 shows a.i. concentrations that cause significant changes to population growth rate (lambda). Because only a proportion of a population may be exposed, we varied the proportion of the population exposed to determine subsequent thresholds for reduction in lambda (intrinsic rate of growth) from death of juveniles. When 17% of juvenile salmonids from a population are exposed to fenbutatin oxide, population growth rate is impacted at concentrations as low as 1.25 µg/L (stream-type Chinook). When 17% of the population is exposed, propargite appears less toxic than fenbutatin oxide. Population growth was reduced at propargite concentrations at 42 µg/L (stream-type Chinook).

Table 100. Concentrations (µg/L) that cause significant change to population growth rate (lambda)

% of population exposed	Chinook, ocean-type		Chinook, stream-type		Sockeye		Coho	
	Fenbutatin oxide	Propargite	Fenbutatin oxide	Propargite	Fenbutatin oxide	Propargite	Fenbutatin oxide	Propargite
17	5.0	na	1.25	42.6	1.66	68.1	1.38	48.6
25	1.60	60.7	1.08	36.3	1.26	50.2	1.13	41.1
50	1.10	41.2	0.85	28.7	0.99	37.5	0.89	31.7
75	0.96	35.6	0.77	25.3	0.87	32.7	0.80	27.6
100	0.88	32.3	0.71	23.2	0.82	29.6	0.74	25.5

25.3.7.3 Effects to salmonid populations from reduced size of juveniles due to 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 prey availability and somatic growth of individual salmon to the productivity of salmon populations expressed as a percent change in a population's intrinsic rate of growth (lambda). The model scenarios assume a single annual pesticide exposure of the prey community. As with the acute lethality model, we developed the growth model for four life history strategies of salmonids: ocean-type Chinook, stream-type Chinook, sockeye, and coho salmon. We also evaluated population level responses from repeated pulsed exposures and exposures to varying portion of the populations.

The growth model focuses on the potential for reductions in juvenile salmon growth due to reductions in prey abundance, i.e. salmonid food. Salmon are frequently 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 size and growth of juvenile fish following exposure to pesticides (Boyle et al 1996). Based on our review of exposure and response data for the three insecticides, we expect reductions in densities and altered composition of salmonid prey communities.

We modeled reductions in aquatic prey because of the high relative toxicity of insecticides to salmonid prey and the extended duration of effects on aquatic prey communities. Juvenile salmonids are typically opportunistic, feeding on a diverse group 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 most sensitive taxa for which there is toxicity information and within this group a range of sensitivities exists (Table 101). Each of the three active ingredients is toxic to aquatic macroinvertebrates, and concentrations that are not expected to kill salmonids are often lethal for salmon prey. This is particularly the case for diflubenzuron, where concentrations well below 1 μg/L kill the majority of salmonid prey species while salmonids appear tolerant to diflubenzuron's acute toxicity (LC_{50}s are in the hundreds to thousands of mg/L active ingredient).

In particular, prey items that are preferred by small juvenile salmonids (including midge larvae, water fleas, mayflies, and caddisflies) are among the most sensitive aquatic macroinvertebrates. Effects on the prey community can persist for extended periods of time (weeks, months, and years), resulting in effects on fish feeding and growth long after an exposure has ended (Colville et al. 2008, Liess and Schulz 1999, Van den Brink et al. 1996a, Ward et al. 1995)

25.3.7.4 Selection of aquatic invertebrate toxicity values to represent salmonid prey items

The model applies an EC_{50} for each pesticide to represent a 50% reduction in the abundance of salmonid prey and a corresponding slope (Appendix 1). The term "EC_{50}" will be used in this section to describe short-term survival data for aquatic invertebrates (death and/or immobility in laboratory experiments). We looked for EC_{50} concentrations for diflubenzuron, fenbutatin oxide, and propargite from available laboratory studies on aquatic invertebrate species. We found few laboratory studies measuring the toxicity of fenbutatin oxide and propargite to aquatic invertebrates, and therefore did not develop species sensitivity distributions for these pesticides. Instead, we selected the lowest available survival EC_{50} for *Daphnia magna* for each pesticide to represent the salmonid prey community EC_{50}. We followed a similar approach in the 2010 Opinion on 12 OP insecticides. Conversely, for diflubenzuron, we found a robust number of laboratory acute toxicity studies that measured aquatic invertebrate survival for several species at 24, 48, 96, 120 (5 day), and 168 (7 day) hours. These laboratory tests were used to develop a species sensitivity distribution to determine a prey abundance EC_{50} as described below. We did not use the many field and mesocosm toxicity studies for diflubenzuron, since the studies typically did not report effects at comparable time intervals, did not measure a range of concentrations, and/or did not report standard toxicity endpoints (*e.g.* survival EC_{50}).

To determine a single salmonid prey abundance EC_{50} concentration to use for diflubenzuron in the model analysis, a search was completed using the EPA's ECOTOX database (http://cfpub.epa.gov/ecotox/). Additionally, we included data from the many peer-reviewed studies that were located in scientific publications as well as industry-supplied reports (Gagne 2011). Several criteria were used to determine which EC_{50} values were appropriate to use in the final analysis. The data included studies using taxa that are known salmonid prey (or functionally similar to salmonid prey), including a diverse group of aquatic insect larvae, freshwater and estuarine crustaceans, and zooplankton. Studies using exposure durations of at least 24 hours and no longer than 7 days were included. Shorter and longer exposure times are known to affect invertebrates, but these studies were excluded so that estimated EC_{50} values would be comparable and would ensure realistic short-term exposures. Studies reporting mortality or immobilization as the recorded endpoint were included. Data from sublethal endpoints such as growth or reproduction were not included. When more than one EC_{50} value was located for a particular species, we calculated a geometric mean EC_{50} value. Table 101 shows the list of species and their associated EC_{50} concentrations used in our analysis of diflubenzuron. Input parameters for the salmon growth model are shown in Table 102.

Table 101. Species and EC_{50} concentrations used for diflubenzuron species sensitivity distribution. * indicates a geometric mean EC_{50} value.

Species	**Common Name**	**EC_{50}**	**N**	**Source**
Aedes albopictus	mosquito larvae	0.02*	2	Ho et al. 1987
Aedes caspius	mosquito larvae	1	1	Porretta et al. 2008
Anisops sardeus	backswimmer	2027.87*	2	Lahr et al. 2001
Callibaetis sp.	mayfly	2	1	Miura et al 1975
Ceriodaphnia dubia	daphnia	1.7	1	Hall 1986
Chironomus sp.	midge	560*	2	Johnson & Finley 1980; Julin & Sanders 1978
Cricotopus sp.	midge	1.79	1	Hansen & Garton 1982a
Culex pipiens	mosquito larvae	2.2	1	Kasai et al. 2007
Daphnia magna	daphnia	6.79*	14	Miura & Takahashi 1974a; Kuijpers 1988; MRID 43665801; Julin & Sanders 1978; Hansen & Garton 1982a; Mayer & Ellerseick 1986; Johnson & Finley 1980; Majori et al 1984; MRID 45252204
Eulimnadia spp.	clam shrimp	0.15	1	Miura & Takahashi 1974a
Eurytemora affinis (estuarine)	copepod	2.2	1	Savitz & Wright 1994
Gammarus pseudolimnaeus	amphipod	39.41*	8	Mayer & Ellerseick 1986; MRID 40094602; Johnson & Finley 1980; Julin & Sanders 1978
Hyalella azteca	amphipod	1.84	1	Hansen & Garton 1982a
Mysidopsis bahia (estuarine)	mysid shrimp	2.1*	2	Nimmo et al. 1979; Mayer 1987

Orthemis sp. nymphs	dragonfly nymph	50	1	Miura & Takahashi 1974a
Palaemontes pugio (estuarine)	grass shrimp	1.92*	8	Wilson & Costlow 1986; Touart & Rao 1987; Wilson & Soctlow 1987
Simulium vittatum	blackfly larvae	1.3	1	EFED database, 2000
Skwala sp.	stonefly	57500	1	Mayer & Ellerseick 1986
Streptocephalus sudanicus	fairy shrimp	3.14*	2	Lahr et al 2001
Tanytarsus dissimilis	midge	1.02	1	Hansen & Garton 1982a
Triops longicaudatus	tadpole shrimp	0.75	1	Miura & Takahashi 1974a

Species sensitivity distribution of aquatic prey survival toxicity values

We plotted the survival EC_{50} data for diflubenzuron using a cumulative probability distribution. From the distribution of the data, a single prey abundance EC_{50} was derived to best represent the diverse community of prey available in juvenile salmonid freshwater and estuarine habitats. The distribution of EC_{50} values was analyzed to estimate the 10^{th} percentile. Figure 77 shows the distributions of diflubenzuron EC_{50} values. Specifically, a probability plot was used to graph the EC_{50} concentrations normalized to a normal probability distribution. The X axis is scaled in probability (between 0 and 100%) and shows the percentage of entire data whose values is less than the data point. The Y axis displays the range of the toxicity data on a log scale. The results of a linear regression of the log-transformed concentrations are shown and highlight the lognormal distribution of the data. In the regression equation, the norm.s.inv function returns the inverse of the standard normal cumulative distribution. For example, given a percentile value of 50 (i.e., a probability of 0.5), norm.s.inv(50) returns a value of zero. The plots and regressions were generated with KaleidaGraph 4.0 (Synergy Software).

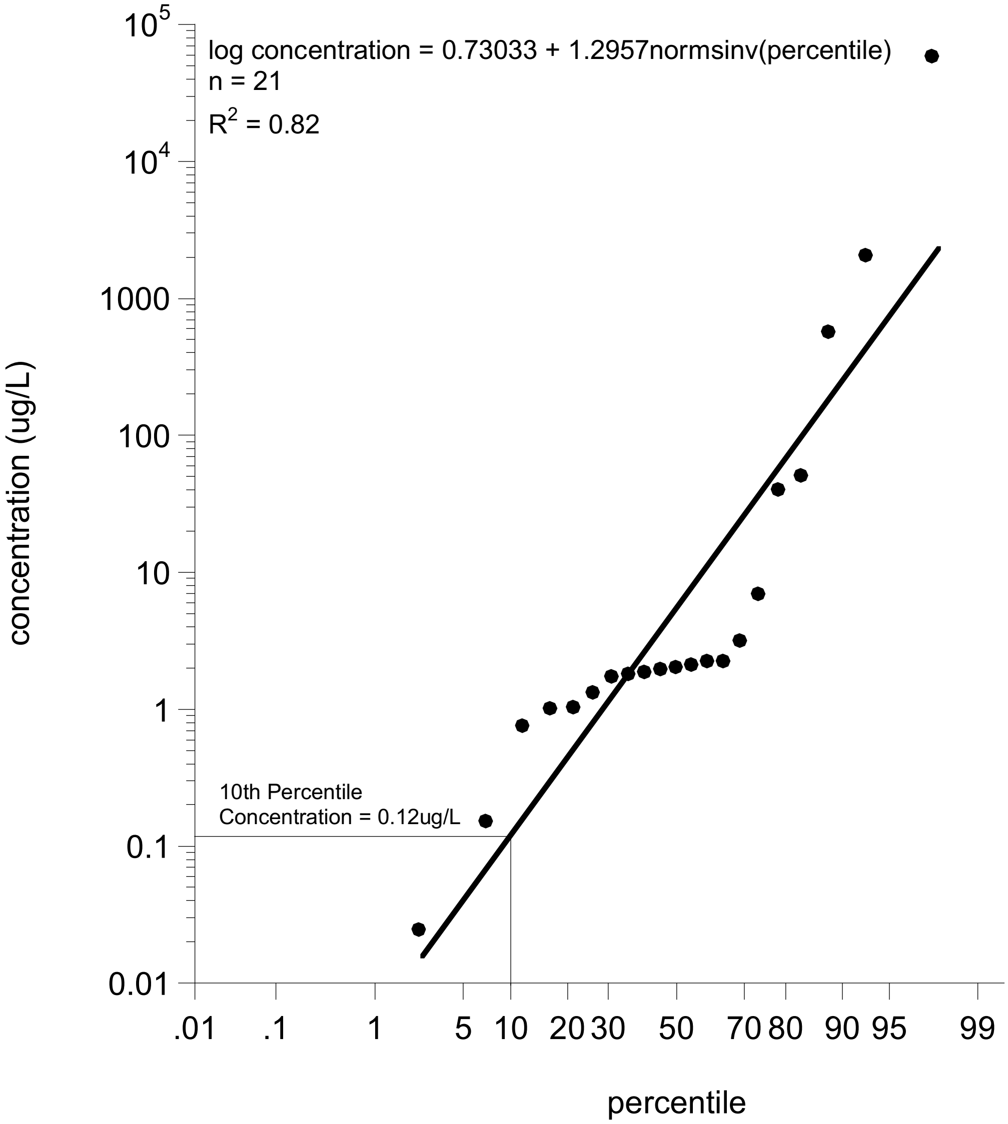

Figure 77. Probability plot for diflubenzuron shows the distribution of EC_{50}s for aquatic invertebrate species. Each point represents the geometric mean of data for individual species. The solid sloping line shows the result of a linear regression. In the regression equation, the norm.s.inv( ) function returns the inverse of the standard normal cumulative distribution. The dashed lines indicate the 10^{th} percentile effect concentration.

We selected the 10^{th} percentile to represent the EC_{50} concentration for salmonid prey abundance. The 10^{th} percentile is a reasonable selection as these data do not account for other effects to prey communities, e.g., sublethal. The data included in the meta-analysis were limited to concentrations that caused mortality/immobilization over a short duration (1-7 days) and did not incorporate other sublethal endpoints such growth or reproduction that may reflect reductions in prey abundance.

A growing number of studies on a variety of insecticides have reported that concentrations well below LC_{50}s 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 *Daphnia magna*, hearty enough to survive laboratory conditions. Studies specifically examining other salmonid prey species, which may be more difficult to rear in the laboratory, have documented relatively low survival EC_{50} values when exposed to current use insecticides (Johnson et al. 2008).

Table 102 Input parameters for salmon growth model

Pesticide	Prey survival EC_{50} (µg/L)	Prey survival sigmoid slope
diflubenzuron	0.12	0.8[1]
fenbutatin oxide	6.4	3.6[2]
propargite	14	3.6[2]

1 denotes geometric mean of five slopes (Ho 1990, Ho et al. 1987b) (Liber et al. 1996) (Majori et al. 1984) (Moffett et al. 1995)
2 denotes standard sigmoid slope from probit slope of 4.5

25.3.7.5 Modeling availability of unaffected prey

Reductions in invertebrate densities following pesticide exposures can lead to long-term reductions in prey availability and reductions in fish growth (Davies and Cook 1993, (Moffett et al. 1995, Tanner and Moffett 1995). That said, pesticide exposures do not usually reduce prey densities 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). The model assumes that fish can and will switch to feeding on these prey items, even if they were not their preferred prey prior to exposure. By making this assumption, we ignore any additional energetic costs or increased risk incurred by the fish in locating or foraging on these alternate prey.

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 available 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).

25.3.7.6 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, or below, 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 7). 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.

25.3.7.7 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).

25.3.8 Growth model results

25.3.8.1 Exposure to single insecticides

Population model outputs for the four salmon populations are summarized as dose-response curves in Figure 78, Figure 79, Figure 80. As expected, greater reductions in population growth resulted from longer durations of exposure.

Similar trends in effects were seen for each pesticide across each of the four life-history strategies modeled. This is apparent by the similar shape of the dose-response curves across species. 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 reached, further reductions in lambda are minimal with increasing concentrations. Diflubenzuron affected salmon populations at concentrations well below one µg/L while fenbutatin oxide affected salmon populations' growth rates at low µg/L concentrations. Propargite affected salmon populations at concentrations as low as 14µg/L.

The modeling results show that population-level consequences are possible, and in some cases, likely (based on the available exposure data) from reductions in abundance of prey during salmonid's sub-yearling freshwater rearing. The models' outputs are informed estimates and should not be construed as definitive. We use the information to help us determine whether or not population level consequences may occur from label authorized uses of the three insecticides. This is but one line of evidence in evaluating effects from these pesticides. We discuss the implications at the species level, ESUs/DPSs, in the *Integration and Synthesis section*.

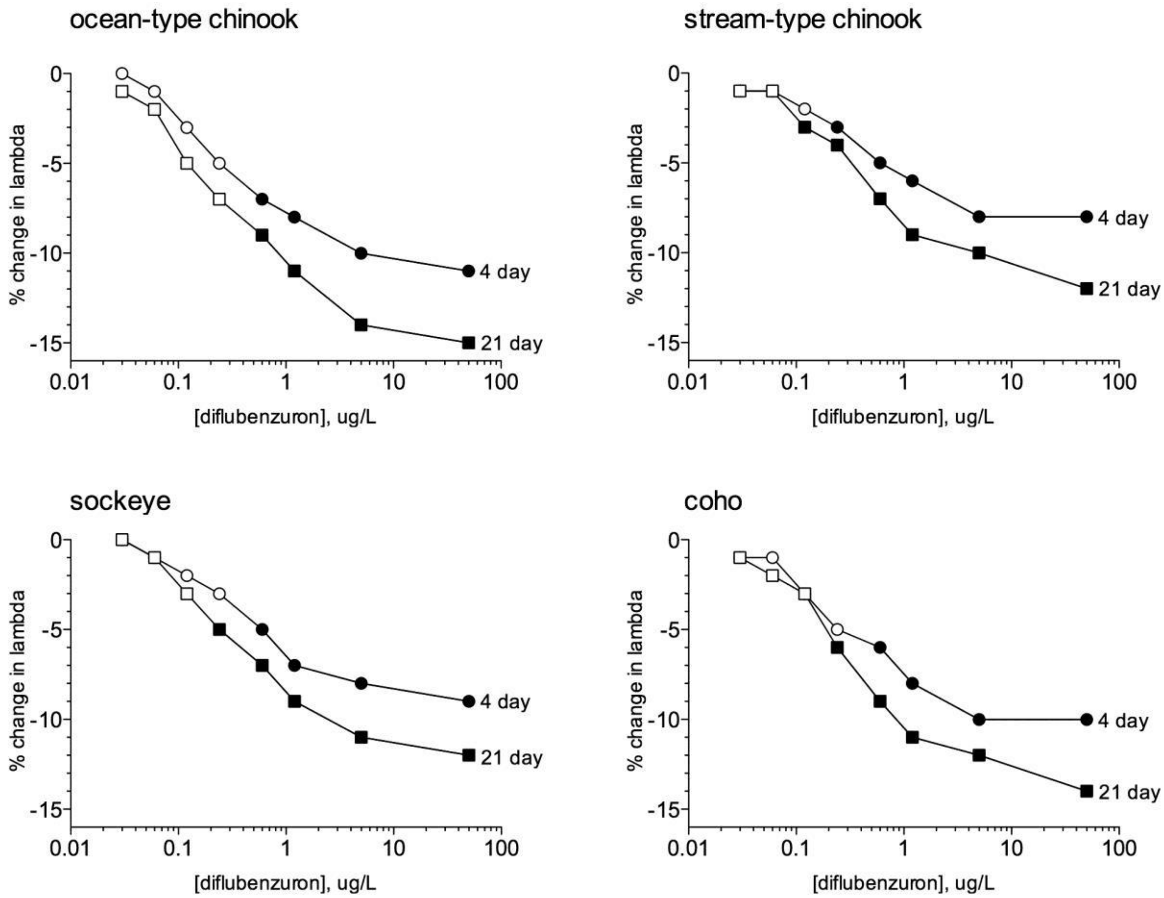

Figure 78. Percent change in lambda for modeled species following acute and chronic exposures to diflubenzuron. Open symbols denote a percent change in lambda of less than one standard deviation from the control population. Closed symbols represent a percent change in lambda of more than one standard deviation from control population.

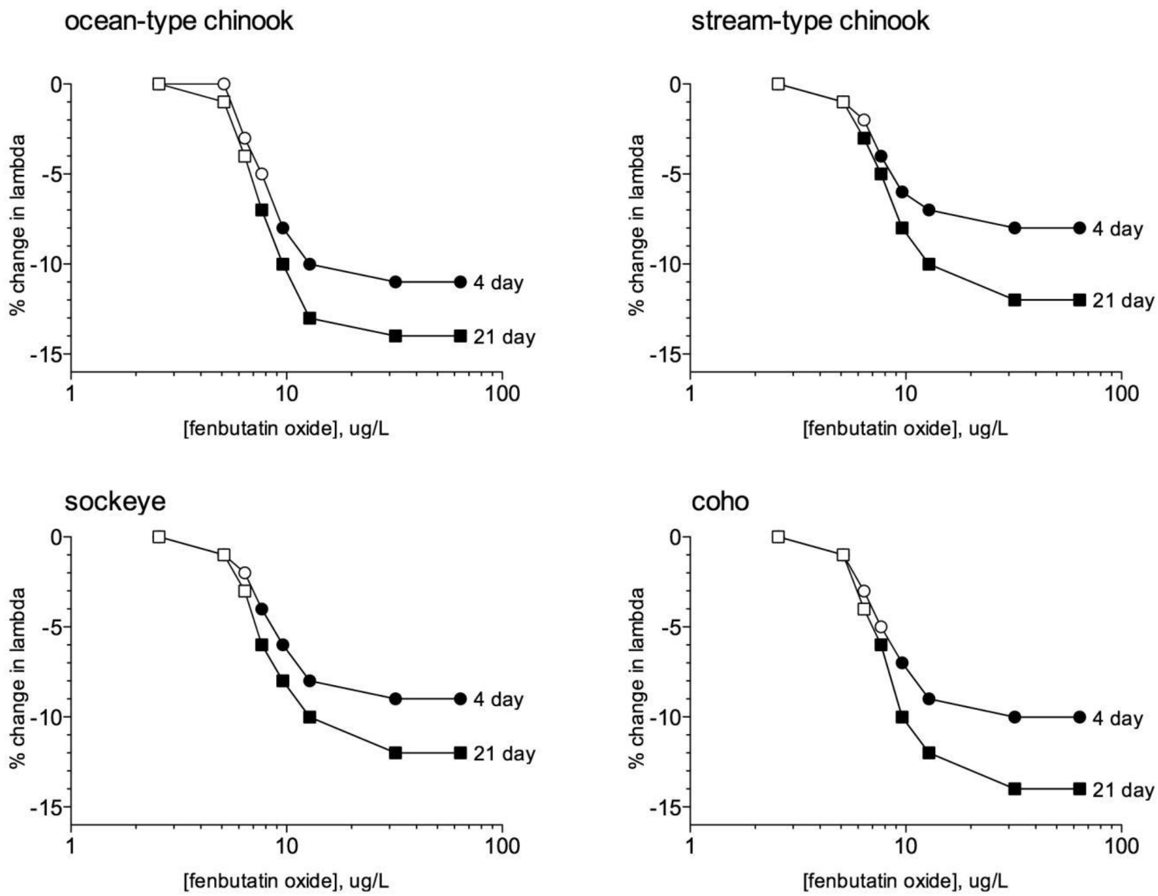

Figure 79. Percent change in lambda for modeled species following acute and chronic exposures to fenbutatin oxide. Open symbols denote a percent change in lambda of less than one standard deviation from the control population. Closed symbols represent a percent change in lambda of more than one standard deviation from control population.

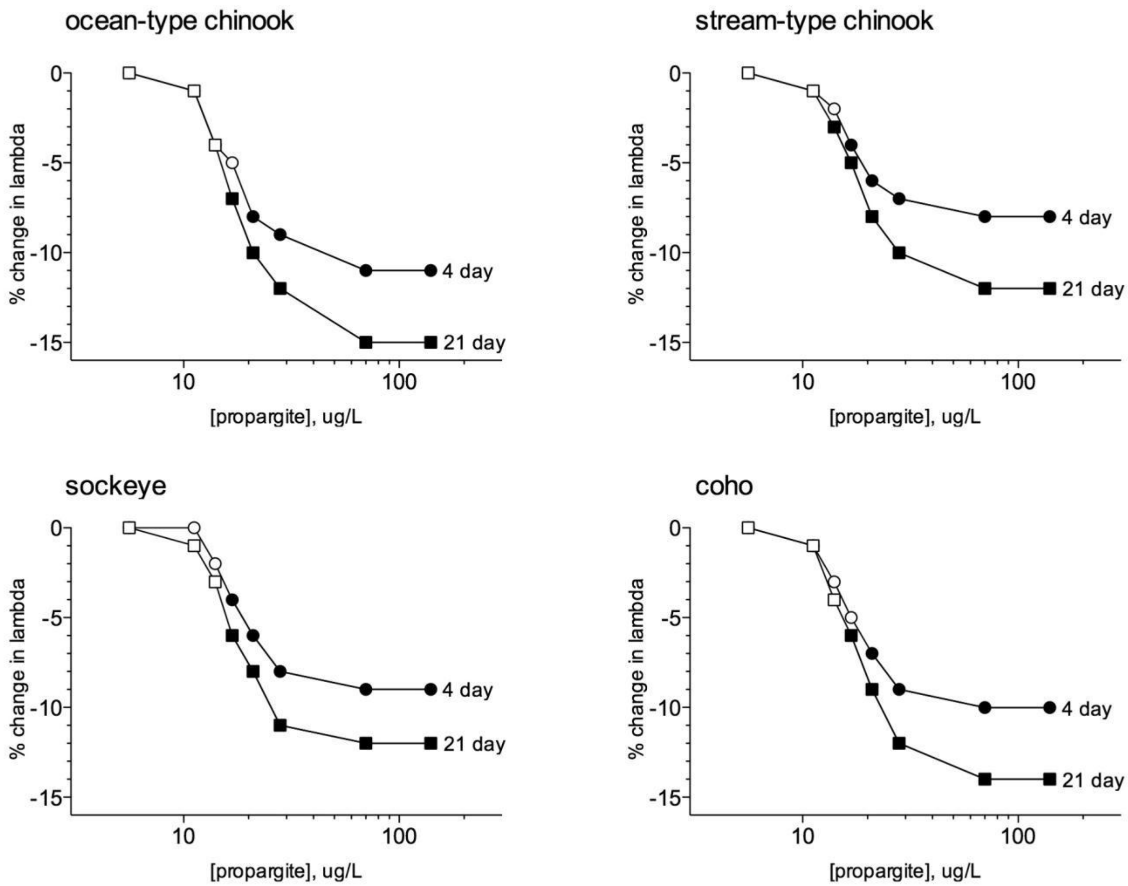

Figure 80. Percent change in lambda for modeled species following acute and chronic exposures to propargite. Open symbols denote a percent change in lambda of less than one standard deviation from the control population. Closed symbols represent a percent change in lambda of more than one standard deviation from control population.

25.3.9 Exposure to multiple applications

All of the currently authorized labels allow at least two or more applications annually and specify a no-application interval (in days) between applications. We anticipate that effects to listed salmonids and their prey will be more pronounced than from a one-time exposure from a single application.

25.3.10 Population level consequences from other affected salmonid assessment endpoints and other stressors of the action

25.3.10.1 Starvation during a critical life stage transition

In the *Population Modeling* section above we assessed impacts from reduced growth of juvenile salmonids associated with reduced prey availability, however the models do not address starvation occurring from lack of a specific size class of prey at a critical life stage transition. Limitations in prey can cause starvation which can further limit a population's 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 (size of their mouths when opened), 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. We found no information on any of the pesticides that tested impacts to swimming or olfaction. This is a notable uncertainty. Prey may be killed outright by the stressors of the action, leading to reduced prey availability or the absence of prey, although this is rare. Floodplain habitats where fry seek shelter and food are highly susceptible to the highest concentrations of the insecticides, 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.

25.3.10.2 Death of returning adults

Earlier, we discussed and analyzed with population models how effects of pesticide exposures to juveniles impact population viability. However, we did not address possible implications of returning adults dying from direct 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 they typically die within hours to days after spawning. We anticipate that returning adults in this condition are likely less tolerant of chemical stressors. However, this is an important data gap, since the available toxicity data are not from returning adults, but typically from juvenile life stages before the transition to seawater. 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 and other toxics may be present in addition to the a.i.s addressed in this Opinion. We are particularly concerned about fenbutatin oxide which is the most acutely toxic of the three a.i.s, with salmonid LC_{50}s at 1 µg/L. EECs from all methods of estimation are in this range, as are monitoring data. Propargite is also expected to kill some returning adults, based on overlaps between the EECs and salmonid LC_{50}s and is less of a risk than fenbutatin oxide as it is less toxic to salmonids. Risk of death to returning adults from applications of diflubenzuron is minimal given EECs and salmonid LC_{50}s. The length of time the adults are exposed may vary widely for fenbutatin oxide and propargite, depending on their persistence 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 and other aquatic areas where salmonids seek a reprieve from high flows during migration to their natal areas.

Another important consideration for returning adults is that 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 system, 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 overlap among cohorts. 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. In cases where a large fish kill occurs, it may also affect distribution via extirpation of sub-populations.

25.3.10.3 Toxicity from other stressors of the action

As described in the individual-level risk hypotheses, we expect identified degradates 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. Additional active ingredients contained in pesticide formulations and tank mixes also 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 a.i.s 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 three insecticide a.i.s. Additionally, there are data gaps regarding the expected concentrations of these chemicals in salmonid habitats and the toxicity of these ingredients. 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.

25.3.11 Conclusions on population level effects

We anticipate many populations of threatened and endangered Pacific salmonids will likely experience reductions in viability, particularly those that have juvenile life histories that include rearing for weeks to years in freshwater habitats found in intensive use areas such as cultivated crop areas (Table 103). Juvenile coho salmon, steelhead, sockeye, 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 endangered populations for each ESU or DPS distributed in high use areas of the pesticides. Effects to abundance and productivity of populations are anticipated from exposure to each of the insecticides, where the geographic ranges of populations overlap with intensive cultivated crop areas. For diflubenzuron, prey communities are at high risk of significant reductions in areas that receive exposures, particularly where high prey abundance is needed to support rearing young-of- the- year salmonids. Risk is compounded as diflubenzuron use is authorized across the landscape for cropped and non-cropped agriculture, forestry, right-of-ways, and developed lands.

Predicted exposure of juvenile salmonids to fenbutatin oxide and, to a lesser degree, propargite can cause population declines through direct acute lethality. Population-level effects from exposure to single a.i.s through acute lethality are not anticipated for diflubenzuron. Additionally, significant population effects due to prey reductions are expected for some populations due to predicted exposure to each of the three pesticides and particularly for diflubenzuron. 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.

Several factors increase the likelihood of population-level effects for the active ingredients: repeated exposures to pesticides due to repeat applications of the a.i.s; exposure to environmental mixtures of pesticides that cause additive or synergistic effects; sublethal effects may include a range of responses such as impaired swimming and olfactory-mediated behaviors that have consequences for survival, migration, and reproduction; exposure to toxic degradates of the active ingredients; and 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.

Table 103. 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	Diflubenzuron	Fenbutatin oxide	Propargite
Death of subyearling juveniles causes reductions in lambda	Not anticipated	Anticipated	Anticipated
Reduced growth of subyearlings results in reduced first year survival causing reductions in lambda	Anticipated	Anticipated	Anticipated
Impaired swimming and olfactory-mediated behavior	Unknown	Unknown	Unknown
Starvation during critical life stage transition	Anticipated	Anticipated	Anticipated
Death of returning adults	Not anticipated	Anticipated	Anticipated
Synergistic or additive toxicity	Unknown	Unknown	Unknown
Toxicity from degradates in combination with the parent compounds	Not anticipated	Not anticipated	Unknown
Toxicity from other stressors of the action: Other actives, inert/other ingredients, and chemicals added to tank mixtures	Anticipated	Anticipated	Anticipated

10 Cumulative Effects

Cumulative effects include the effects of future state, tribal, local, or private actions that are reasonably certain to occur in the action area considered by this Opinion. 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; 2018) are more likely to occur than longer-term projects (10-years; 2023). 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 and withdraw water from 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 existing and additional 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. Continued growth into forested and other natural areas will continue the cycle of altering landscapes to the detriment of salmon habitat. Altered landscapes adversely affect the delivery of sediment and gravel and significantly alter stream hydrology and water quality.

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. Diflubenzuron, fenbutatin-oxide, propargite 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. However, 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). For example, EPA recently released draft *National Rivers and Streams Assessment 2008-2009 – Collaborative Survey* (EPA 2013) revealed only 41.9 percent of rivers and streams in the west were in good shape when looking at overall biological condition. Biological condition is the most comprehensive indicator of water body health. When the biology of a stream is healthy, the chemical and physical components of the stream are also typically in good condition. The EPA assessment indicated that the overall health of the rivers and streams has declined when compared to past surveys. Nationally, the amount of stream length in good quality for macroinvertebrate condition dropped from 27.4 percent in 2004 to 20.5 percent. As growth in population continues, it will take a concerted effort to reverse this trend.

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).

Considering the status of these ESU/DPS, all of which are listed as endangered or threatened and remain at risk, and their degraded designated critical habitat, the effects from the actions in the Environmental Baseline, including EPA's registration of the a.i.s of the past six recent Opinions,[20] 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.

[20] Opinion 1: chlorpyrifos, malathion, and diazinon; Opinion 2: carbaryl, carbofuran, and methomyl; Opinion 3: azinphos-methyl, dimethoate, phorate, methidathion, naled, methyl parathion, disulfoton, fenamiphos, methamidophos, phosmet, ethoprop, and bensulide; Opinion 4: 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil; Opinion 5: oryzalin, pendimthalin, and trifluralin; Opinion 6: thiobencarb.

11 Integration and Synthesis for Threatened and Endangered Pacific Salmonids

This section describes NMFS' assessment of the potential for EPA's registration of diflubenzuron, fenbutatin oxide, and propargite 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 exposure values is the assumption that the pesticide is applied in a location proximate 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 extent of application sites near salmon-bearing waters, NMFS used a GIS overlay containing land use classifications and species' ranges to determine overlap. Because cropping patterns and registered use sites may change over time, land use classifications (agricultural, forestry, urban/developed) are used rather than specific crops. Details of the GIS analysis and the maps are provided in *Appendix 4*. Occurrence of land uses where specific a.i.s could be applied near salmon-bearing waters for each ESU/DPS is shown in Table 50, Table 51, Table 55, Table 56, and Table 57.

The GIS data was used by NMFS in a qualitative approach to assess the likelihood of exposure based on potential co-occurrence of salmon and pesticide use. Important considerations in the approach included the specific locations of likely pesticide use (i.e. land use) with respect to salmon-bearing streams, that salmon will move around within the habitat as they rear and migrate, and the connectedness of salmon-bearing streams within the habitat. Quantitative GIS measures of the area of overlap between salmon habitat and land use were not considered an appropriate estimate of the percent of the population likely to be exposed (Teply et al. 2012). Juvenile salmon will move around within their habitat as they rear and migrate (Healey 1991, Sandercock 1991), and potentially encounter multiple land uses. Agricultural land uses, for example, tend to be in downstream reaches of salmon habitat lower in the watershed. While salmon may spawn and initially rear in upstream habitats that are not used by agriculture, as they continue to rear and migrate they may have to move into reaches adjacent to agriculture as they eventually make their way to the estuary. Consequently, while a small portion of the overall habitat may contain agricultural use it's possible that all of the salmon in that population will have to spend time in reaches adjacent to agricultural land as they move downstream to rear and migrate and, therefore, 100% of the population will potentially be exposed to pesticides. Quantitative GIS measures of overlap between salmon habitat and land use, therefore, are a minimum estimate of any co-occurrence. By minimizing the likelihood of exposure, these estimates maximize the likelihood of a Type II error (in this case assuming there is no co-occurrence when there is co-occurrence). NMFS' qualitative approach is meant to reduce this error by incorporating additional information that reflects the biology of the species and details of the habitat.

Based on the effects at the population scale from each a.i. presented in the *Risk Characterization* section; 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 whether the a.i. as registered, and used, will reduce the reproduction, numbers, or distribution of populations within each ESU/DPS. A qualitative designation for each ESU/DPS of either low, medium, or high is made (Table 104 through Table 131). A summary of the designations for each a.i. for all species is provided in Table 132 at the end of this section.

ESUs and DPSs may be comprised of one to many discrete, independent populations. These independent populations support the survival and recovery of the listed ESU/DPS, but may not all be equally affected by the authorized use of an a.i. We therefore determine the potential for appreciable reduction in the reproduction, numbers, or distribution of the species (ESU/DPS) from the stressors of the action by taking into account the unevenness of pesticide use across the populations within the ESU/DPS, the life history of the populations that co-occur with authorized uses of the a.i.s, and the relative importance of populations to the ESU/DPS for recovery and eventual delisting.

A high ranking is achieved when substantial overlap of a land use- where application of an a.i. is authorized, and salmonid populations comprising a listed ESU/DPS occurs. We used cultivated crops as the primary land use classification to determine overlap for fenbutatin oxide and propargite. For diflubenzuron, we use both cultivated crop and undeveloped (i.e., forestry) classifications as primary land uses. In California, we also note that diflubenzuron was authorized for direct applications to aquatic water bodies including those that contain salmonids. This use in and of itself would be a high risk to salmonid prey items and therefore to salmonid's growth, however the applicants removed the three special local needs label from the action.

For ESUs/DPSs where the geographic ranges of only some of the populations overlap with primary land use classification(s), we conduct a more in-depth analysis. The first step in this process is to identify the independent populations where overlap occurs, and then determine the importance/significance of any exposed populations toward achieving recovery goals for that species (ESU/DPS). We review NOAA NMFS reports on a population's significance and relationship to the species as a whole including population viability information from NOAA's Technical Recovery and Biological Recovery Teams (TRT/BRT) as well as information developed for recovery planning e.g., (NOAA 2007, NOAA 2012). We also contacted several recovery and TRT/BRT team members to discuss population information.

Several criteria were selected from the reports to determine the importance of a given population to the ESU/DPS. The criteria include:

Is the population designated as "core"?

Is the population designated as "genetic legacy"?

Is the population designated as primary, contributing, or stabilizing?

Is the population within a high, medium, or low conservation value for designated critical habitat?

Does the population have a high probability of persistence?

What other limiting factors and stressors found in the environmental baseline are affecting a population (s)?

Where the land use classification of concern overlaps with a population, information on each of these questions is evaluated and if the population is deemed important to the overall health of the ESU/DPS we make a finding of high. For those populations that do not rise to the level of importance to the species, we make a finding of medium risk at the species level. A low ranking is given for those that have very minimal overlap between land uses and population ranges.

In the *Conclusion* section, we present jeopardy and no jeopardy determinations (Table 169). For "threatened" ESUs/DPS we equated "high" designations as jeopardy, that is the potential for reduction in the reproduction, numbers, or distribution at the species level is anticipated. For "endangered" ESUs/DPSs we equated "medium" or "high" designations as jeopardy, that is the potential for reduction in the reproduction, numbers, or distribution at the species level is anticipated. Endangered species are more vulnerable to extinction than threatened species.

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. We focus on abundance and productivity parameters as they may be directly affected by chemical contaminants such as pesticides and other chemicals associated with the application of pesticide end-use products. As exposure to a.i.s during the juvenile life stage is of particular concern for each of the three insecticides, we discuss residence time of juveniles in vulnerable habitats including flood plain habitats and small streams. Young fish need and use these areas to rear and avoid predators, taking advantage of abundant prey resources.

25.3.12 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 extirpated runs 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 Appendix 4 and in Table 55. 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, forested, 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 the stressors of the action that are applied near their habitats.

Abundance and productivity of key populations within this ESU will likely be reduced by exposures to each of the a.i.s subject to this consultation. The Nooksack, Skagit, Stillaguamish, Snohomish/Skokomish, Green/Duwamish, Puyallup/White, and Nisqually have over-lap with cultivated croplands particularly in the lowland reaches where floodplain habitats are more extensive. Recovery of the Chinook populations within each of these river systems is critical for recovery of the ESU. In addition, these and other important river systems have vast forest and urban overlap where diflubenzuron may also be used. NMFS concludes that the potential co-occurrence of use of each of the a.i.s in this consultation with several important populations in the ESU suggests their abundance and productivity may be severely compromised, and as such, there is a high potential for reduction in reproduction, numbers, or distribution of the species as a whole Table 104.

Table 104. Puget Sound Chinook

Pesticide	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Crop	Non-crop	Developed	Undeveloped	Populations	Species
Diflubenzuron	Yes	Yes	Yes	Yes	High	High
Fenbutatin oxide	Yes	No	Yes*	No	High	High
Propargite	Yes	No	Yes*	No	High	High

*In developed lands the only labeled use is for nurseries.

25.3.13 Lower Columbia River (LCR) Chinook Salmon (Threatened Species)

The LCR Chinook salmon ESU includes 21 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 Appendix 4 and in Table 55. 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 several important populations may be exposed to pesticides applied in agriculture, forest, 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. In addition, population status assessments indicate that all LCR tule fall Chinook populations have a baseline persistence probability of low or very low (NOAA 2012). Declines in their persistence probability are related primarily to losses in abundance, productivity, and diversity. While two late fall run populations have high (Sandy River) and very high (Lewis River) persistence probabilities (NOAA 2012), the Sandy has a high over-lap with cultivated crop landcover. The late fall run and spring run on the Sandy are historically more productive and are both core and genetic legacy populations (NOAA 2012). The Clackamas fall run has high over-lap with crop lands and is also a core population. The Hood River fall run Chinook also have high over-lap with crop lands in the lower river and forest lands in the upper river. The Hood River fall run are designated as a primary run for the ESU. In addition, several other populations within this ESU have over-lap with forest and urban lands where diflubezuron may also be applied. NMFS concludes that the potential co-occurrence of use of each of the a.i.s in this consultation with several important populations in the ESU suggests their abundance and productivity may be severely compromised, and as such, there is a high potential for reduction in reproduction, numbers, or distribution of the species as a whole Table 105.

Table 105. Lower Columbia River Chinook

Pesticide	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Crop	Non-crop	Developed	Undeveloped	Populations	Species
Diflubenzuron	Yes	Yes	Yes	Yes	High	High
Fenbutatin oxide	Yes	No	Yes*	No	High	High
Propargite	Yes	No	Yes*	No	High	High

*In developed lands the only labeled use is for nurseries.

25.3.14 Upper Columbia River (UCR) Spring run Chinook Salmon (Endangered Species)

The UCR Spring-run Chinook salmon ESU is comprised of three extant and endangered 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, although in recent years, no trend is apparent. 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. Pesticides have been documented in these waters in past years. For example, 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 Appendix 4 and in Table 55. The percentage of agricultural and developed lands that overlap with UCR Chinook salmon habitat is about 5.4% and 4.7%, respectively and primarily on or near the floodplain. Forested lands make up about 45% of the ESU. Our GIS analysis indicates that all three populations are exposed to pesticides applied in agriculture, forested, 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. The Wenatchee, Entiat, and Methow Rivers each support independent populations of UCR Chinook. Each of these rivers, along with the mainstem Columbia River have over-lap with agricultural crop-lands within the floodplain.

NMFS concludes that the potential co-occurrence of use of each of the a.i.s in this consultation with several key populations in the ESU suggests these populations' abundance and productivity may be compromised, and as such, there is a high potential for reduction in reproduction, numbers, or distribution of the species as a whole (Table 106).

Table 106. Upper Columbia River Chinook

Pesticide	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Crop	Non-crop	Developed	Undeveloped	Populations	Species
Diflubenzuron	Yes	Yes	Yes	Yes	High	High
Fenbutatin oxide	Yes	No	Yes*	No	High	High
Propargite	Yes	No	Yes*	No	High	High

*In developed lands the only labeled use is for nurseries.

25.3.15 Snake River (SR) Fall-run Chinook Salmon (Threatened Species)

The SR Fall-run Chinook salmon ESU consists of one small population that spawns in the lower mainstem Snake River. Two historically large populations: the Marsing Reach, and the Salmon Falls have both been extirpated. The spatial distribution of the Lower Mainstem population 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 the two extirpated 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 Appendix 4 and Table 55. Pesticide use areas for the 3 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. However spawning and early life-history rearing is above Lower Granite Dam (LGD) where there is minimal overlap with croplands. The majority of cropland overlap in this ESU is downstream of LGD and this reach is used by Chinook mostly as a migration corridor. NMFS concludes that the potential co-occurrence of use of each of the a.i.s in this consultation with the one important population in the ESU suggests this population's abundance and productivity may be somewhat compromised, and as such, there is a medium potential for reduction in reproduction, numbers, or distribution of the species as a whole (Table 107).

Table 107. Snake River fall Chinook

Pesticide	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Crop	Non-crop	Developed	Undeveloped	Populations	Species
Diflubenzuron	Yes	Yes	Yes	Yes	Medium	Medium
Fenbutatin oxide	Yes	No	Yes*	No	Medium	Medium
Propargite	Yes	No	Yes*	No	Medium	Medium

*In developed lands the only labeled use is for nurseries.

25.3.16 Snake River (SR) Spring/Summer-run Chinook Salmon (Threatened Species)

This ESU includes 31 historical populations comprising five major population groups (MPGs). 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 Appendix 4 and in Table 55. 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 Grande Ronde, Lostine, Willowa, mainstem Snake and Columbia Rivers. Some agricultural and urban use of the land is also scattered in the Salmon River and its tributaries. The Snake River is a high-volume, high-flow system, and the salmon use it primarily as a migratory corridor.

To achieve a viable Snake River Spring/Summer Chinook ESU, the Technical Recovery Team recommends all extant MPGs meet MPG-level viability criteria (NOAA 2007). The Lower Snake, Grande Rande/Imnaha, South Fork Salmon, Middle Fork Salmon, and Upper Salmon constitute the five MPGs. Within the Lower Snake MPG, the Tucannon River population is the lone extant population. The Tucannon River begins in forest lands and flows through agricultural lands before entering the Snake River near Starbuck, WA. Spring/summer Chinook in Asotin Creek are considered functionally extirpated.

The Grande Rande/Imnaha MPG is comprised of the following extant component populations: Wenaha River, Minam River, Lostine/Wallowa Rivers, Catherine Creek, Upper Grande Ronde, Imnaha River. Also within this MPG are two functionally extirpated populations which are the populations in Big Sheep Creek and Lookingglass Creek. TRT recovery criteria of this MPG would four of these populations meet viability criteria, one of which must meet high viability criteria. The population in the Imnaha River has a unique life history strategy and must be one of the four populations that meet viability criteria. Also two of the three large populations (Lostine/Wallowa Rivers, Catherine Creek, and Upper Gande Ronde) must meet viability criteria (NOAA 2007). The Imnaha has very little agricultural land-use. Therefore, species co-occurrence with the a.i.s in this consultation for agricultural purposes would be unlikely. However, the upper reaches of the Imnaha is forest land where use of diflubenzuron is approved. The Grande Rande, Catherine Creek, and Lostine/Willowa Rivers flow through forest and agricultural lands.

There are four component populations within the South Fork Salmon River MPG. The TRT recommendation is that two of the four be elevated as highly viable and viable and the other two be maintained. The Little Salmon River is the only one that expresses the spring/summer life history trait and this one must be one of the two that achieves this TRT goal. The South Fork Salmon, Secesh River and the East Fork South Fork Salmon River all express the summer run life history trait. All flow through forest lands and the Little Salmon River also flows through developed and agricultural lands. The Little Salmon enters the Salmon River at the town of Riggins, Idaho.

The Middle Fork Salmon MPG and Upper Salmon MPG comprise the final components of this ESU. Very little agricultural lands co-occur within these MPGs. However forest lands are a major component of these MPGs.

Based on the above considerations, NMFS concludes that the potential co-occurrence of use of each of the a.i.s in this consultation with key populations within each MPG in this ESU suggests their abundance and productivity may be severely compromised, and as such, there is a high potential for reduction in reproduction, numbers, or distribution of the species as a whole (Table 108).

Table 108. Snake River Spring/Summer-run Chinook

Pesticide	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Crop	Non-crop	Developed	Undeveloped	Populations	Species
Diflubenzuron	Yes	Yes	Yes	Yes	High	High
Fenbutatin oxide	Yes	No	Yes*	No	High	High
Propargite	Yes	No	Yes*	No	High	High

*In developed lands the only labeled use is for nurseries.

25.3.17 Upper Willamette River (UWR) Chinook Salmon (Threatened Species)

The UWR Chinook salmon ESU is composed of seven populations. Significant natural production occurs only in the McKenzie and Clackamas populations. Abundance is low for all populations, and they are all considered non-viable (NMFS 2011). 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 agricultural, urban and forested areas. Ten of these pesticides exceeded

EPA criteria for the protection of freshwater aquatic life from chronic toxicity.

The percentages of cultivated, developed, and forested lands that overlap with UWR Chinook salmon habitat are 10.4%, 9.1%, and 46.7% respectively. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Appendix 4 and in Table 55. Our GIS analysis indicates all populations in this ESU may be exposed to pesticides applied in agricultural, urban, and forested areas. Notably, juveniles from all populations will rear in off-channel areas along the mainstem Willamette River and in floodplain wetlands during the inundation period before and during their downstream migration from spawning areas. 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.

As shown in Table 109, NMFS concludes that the potential co-occurrence of use of each of the a.i.s in this consultation with populations in the ESU suggests their abundance and productivity may be severely compromised, and as such, there is a high potential for reduction in reproduction, numbers, or distribution of the species as a whole.

Table 109. Upper Willamette River Chinook salmon

Pesticide	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Crop	Non-crop	Developed	Undeveloped	Populations	Species
Diflubenzuron	Yes	Yes	Yes	Yes	High	High
Fenbutatin oxide	Yes	NA	Yes*	NA	High	High
Propargite	Yes	NA	Yes*	NA	High	High

*In developed lands the only labeled use is for nurseries.

25.3.18 *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 Appendix 4 and in Table 50. One percent of cultivated croplands overlap with the entire CC Chinook salmon ESU. Our GIS analysis indicates the Russian River population in this ESU is the population primarily exposed to pesticides applied in agricultural lands where the three a.i.s are approved for use. 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. The Russian River is of particular importance for preventing the extinction and contributing to the recovery of CC Chinook salmon. Because of the lack of population data, viability of the Russian River population is uncertain. However, even if the Russian River population is eventually deemed viable, the lack of other viable populations within the Central Coastal stratum places this stratum at greater risk due to catastrophic events to the mainstem Russian River where most spawning is believed to occur (Spence et al. 2008a). The long-term viability of the Russian River Chinook population is critical to the ESU as a whole for recovery (Ambrose 2013). Therefore, the effects ratings were based primarily on the overlap in this watershed. Juveniles rear in freshwater streams for 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. As shown in Table 110, NMFS concludes that the potential co-occurrence of use of each of the a.i.s in this consultation with a key population within this ESU suggests its abundance and productivity may be severely compromised, and as such, there is a high potential for reduction in reproduction, numbers, or distribution of the species as a whole.

Table 110. California Coastal Chinook Salmon

Pesticide	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Crop	Non-crop	Developed	Undeveloped	Populations	Species
Diflubenzuron	Yes	Yes	Yes	Yes	High	High
Fenbutatin oxide	Yes	No	Yes*	No	High	High
Propargite	Yes	No	Yes*	No	High	High

*In developed lands the only labeled use is for nurseries.

25.3.19 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, altered hydrology because of water management (dams, levees, reservoirs), and impaired water quality. Pesticides are detected in the Sacramento River. The percentage of cultivated croplands and developed lands that overlap with CV Chinook salmon habitat are 21.3% and 10.8%, 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 Appendix 4 and in Table 50. Our GIS analysis indicates all four populations in this ESU are exposed to pesticides applied in agriculture and urban areas (i.e., diflubenzuron). 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. NMFS concludes that the potential co-occurrence of use of each of the a.i.s in this consultation with all four populations in the ESU suggests their abundance and productivity may be severely compromised, and as such, there is a high potential for reduction in reproduction, numbers, or distribution of the species as a whole (Table 111).

Table 111. Central Valley Spring-run Chinook Salmon

Pesticide	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Crop	Non-crop	Developed	Undeveloped	Populations	Species
Diflubenzuron	Yes	Yes	Yes	Yes	High	High
Fenbutatin oxide	Yes	No	Yes*	No	High	High
Propargite	Yes	No	Yes*	No	High	High

*In developed lands the only labeled use is for nurseries.

25.3.20 *Sacramento River Winter-run Chinook Salmon (Endangered Species)*

The Sacramento River Winter-run Chinook salmon ESU is now comprised of a single population. Put another way, the one population is the ESU. 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, altered hydrology because of 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 are frequently detected in the Sacramento River. 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 Appendix 4 and in Table 50. 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. NMFS concludes that the potential co-occurrence of use of each of the a.i.s in this consultation with the Sacramento River Winter-run population suggests this population's abundance and productivity may be severely compromised, and as such, there is a high potential for reduction in reproduction, numbers, or distribution of the species as a whole (Table 112).

Table 112. Sacramento Winter-run Chinook Salmon

Pesticide	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Crop	Non-crop	Developed	Undeveloped	Populations	Species
Diflubenzuron	Yes	Yes	Yes	Yes	High	High
Fenbutatin oxide	Yes	No	Yes*	No	High	High
Propargite	Yes	No	Yes*	No	High	High

*In developed lands the only labeled use is for nurseries.

25.3.21 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 Appendix 4 and in Table 56 Our GIS analysis indicates that both populations of HC Summer-run chum salmon may be exposed to pesticides applied in agriculture and urban areas. As shown in Table 113 below, NMFS concludes that the co-occurrence of use of diflubenzuron is mainly from its application in forest lands within the range of the two populations in this ESU, this suggests their abundance and productivity may be somewhat compromised and that there is a medium potential for reduction in reproduction, numbers, or distribution of the ESU. NMFS further concludes the potential co-occurrence of use of fenbutatin-oxide and propargite with the two populations within this ESU, suggests the abundance and productivity may be only slightly affected, and as such there is a low potential for the reduction in reproduction, numbers, or distribution of the species as a whole (Table 113).

Table 113. Hood Canal Summer-run chum salmon

Pesticide	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Crop	Non-crop	Developed	Undeveloped	Populations	Species
Diflubenzuron	Yes	Yes	Yes	Yes	Medium	Medium
Fenbutatin oxide	Yes	No	Yes*	No	Low	Low
Propargite	Yes	No	Yes*	No	Low	Low

*In developed lands the only labeled use is for nurseries.

25.3.22 *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. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Appendix 4 and in Table 56.

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. NMFS concludes that the co-occurrence of use of diflubenzuron is mainly from its application in forest lands within the range of the populations in this ESU. This suggests their abundance and productivity may be somewhat compromised and that there is a medium potential for reduction in reproduction, numbers, or distribution of the species. NMFS further concludes the potential co-occurrence of use of fenbutatin-oxide and propargite with the two populations within this ESU suggests the abundance and productivity may be only slightly affected, and as such there is a low potential for the reduction in reproduction, numbers, or distribution of the species as a whole (Table 114).

Table 114. Columbia River chum salmon

Pesticide	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Crop	Non-crop	Developed	Undeveloped	Populations	Species
Diflubenzuron	Yes	Yes	Yes	Yes	Medium	Medium
Fenbutatin oxide	Yes	No	Yes*	No	Low	Low
Propargite	Yes	No	Yes*	No	Low	Low

*In developed lands the only labeled use is for nurseries.

25.3.23 Lower Columbia River Coho Salmon (Threatened Species)

Historically, coho salmon spawned in almost every accessible stream system in the lower Columbia River. Out of the 24 populations that make up this ESU, 21 are now considered to have a very low probability of persisting for the next 100 years. This is due to low abundance and productivity, loss of spatial structure, and reduced diversity (NOAA 2012). NMFS has not yet designated core or genetic legacy populations for this ESU. However, the Clackamas and Sandy subbasins contain the only populations in the ESU that have clear records of continuous natural spawning (McElhany et al. 2007). Both populations are well below long-term minimum abundance thresholds.

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.

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. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Appendix 4 and in Table 56. Our GIS analysis indicates both the Sandy and Clackamas populations of LCR coho salmon co-occur in agricultural crop, and forested lands and therefore may be exposed to pesticides applied in these 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). NMFS concludes that the potential co-occurrence of use of the three a.i.s in this consultation with key LCR coho salmon populations suggests these populations' abundance and productivity may be severely compromised, and therefore there is a high potential for reduction in reproduction, numbers, or distribution of the species as a whole (Table 115).

Table 115. Lower Columbia River coho salmon

Pesticide	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Crop	Non-crop	Developed	Undeveloped	Populations	Species
Diflubenzuron	Yes	Yes	Yes	Yes	High	High
Fenbutatin oxide	Yes	No	Yes*	No	High	High
Propargite	Yes	No	Yes*	No	High	High

*In developed lands the only labeled use is for nurseries.

25.3.24 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. Juvenile coho salmon are often found in small streams less than five feet wide and rear in fresh water for up to 18 months. 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 independent populations in other watersheds with less overlap. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Appendix 4 and in Table 56. Our GIS analysis indicates all 13 populations of OC coho salmon may be exposed to pesticides applied in agriculture and forested areas. However, agricultural activities involving crops in this ESU are sparse, while roughly 80% of the Oregon coastal range is forested. NMFS concludes that the potential co-occurrence of use of diflubenzuron is mainly from its application in the vast forest lands within the range of the populations in this ESU. This suggests several populations' abundance and productivity may be severely compromised, and that there is a high potential for reduction in reproduction, numbers, or distribution of the species. NMFS concludes the potential co-occurrence of use of fenbutatin-oxide, and propargite with the coho populations within this ESU, suggests the abundance and productivity may be only slightly affected, and therefore a low potential for the reduction in reproduction, numbers, or distribution of the species as a whole (Table 116).

Table 116. Oregon Coast coho salmon

Pesticide	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Crop	Non-crop	Developed	Undeveloped	Populations	Species
Diflubenzuron	Yes	Yes	Yes	Yes	High**	High**
Fenbutatin oxide	Yes	No	Yes*	No	Low	Low
Propargite	Yes	No	Yes*	No	Low	Low

*In developed lands the only labeled use is for nurseries.

**Due to registered use in forest lands

25.3.25 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 Appendix 4 and Table 56. Areas with more cropland include the Salmon, Scott, and Shasta watersheds in the Klamath basin, and the Upper and Middle Rough River[21] watersheds (NMFS 2012b).

The Salmon River population is a non-core independent population, its recovery target is to recover to as least a moderate risk of extinction. Sufficient spawner densities are needed to maintain connectivity and diversity and continue to represent critical components of the evolutionary legacy of the ESU. The Salmon River has the potential to act as a refugia population within the Interior Klamath because its ecosystem function and habitat values remain relatively intact and is not significantly influenced by hatchery fish (NMFS 2012b).

[21] The Rough River is also be referred to as the Rouge or Rouge River in other publications, maps, or websites

The Scott River population is considered to be an independent population and is a core population due to its location in the most eastern part of the ESU, its delayed interior basin run timing, its large run size compared to other SONCC coho salmon populations, and its unique life history traits. As a core population, the recovery target for the Scott River population is for it to be viable, and to have a low risk of extinction (NMFS 2012b).

The Shasta River population is considered an independent population, meaning that it has been sufficiently large to be historically viable-in-isolation, and its demographics and extinction risk have been minimally influenced by immigrants from adjacent populations (Williams et al. 2006). The Shasta River population is a core population and therefore its recovery target is the low risk of extinction threshold. Besides its role in achieving demographic goals and objectives for recovery, the Shasta River population fulfills other needs within the Interior Klamath basin. The Shasta River population may serve as a source population for the Middle and Lower Klamath River populations, and provides connectivity and diversity with other populations in the larger Klamath Basin (NMFS 2012b).

The Upper Rogue River coho salmon population is considered functionally independent because of the large amount of habitat it contains. As a functionally independent population, it is expected the Upper Rogue River population would contribute recruits to nearby populations, such as those in the Rogue River basin. At present, the capacity of the Upper Rogue River coho salmon population to provide recruits to adjacent independent populations is limited due to its low spawner abundance. Conversely, recruits straying from the nearby Lower Rogue, Middle Rogue/Applegate, and Illinois rivers may enhance recovery of the Upper Rogue River population. Although the extent of agriculture in the Upper Rogue River subbasin is not large, these lands substantially overlap coho salmon habitat. Much of the water withdrawals causing insufficient flow are used for agriculture. Other agricultural impacts include wetland filling, channelization and diking, riparian removal, channel simplification, and chemical application. Herbicide use has resulted in fish kills in the Rogue River basin, including juvenile coho salmon in Bear Creek in 1996 (Ewing 1999). Risk to coho salmon resulting from agriculture chemical use has been identified as a concern throughout 5 the Pacific Northwest

(Laetz et al. 2009), and it is likely that pesticides known to harm salmonids (NMFS 2008e, NMFS 2009b, NMFS 2010a, NMFS 2011, NMFS 2012a) are used in the region.

Considering the above, NMFS concludes that the potential co-occurrence of use of the a.i.s in this consultation with key SONCC coho salmon populations suggests that these populations' abundance and productivity may be severely compromised, and therefore there is a high potential for reduction in reproduction, numbers, or distribution of the species as a whole (Table 117).

Table 117. Southern Oregon, Northern California Coast coho salmon

Pesticide	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Crop	Non-crop	Developed	Undeveloped	Populations	Species
Diflubenzuron	Yes	Yes	Yes	Yes	High	High
Fenbutatin oxide	Yes	No	Yes*	No	High	High
Propargite	Yes	No	Yes*	No	High	High

*In developed lands the only labeled use is for nurseries.

25.3.26 Central California Coast (CCC) Coho Salmon (Endangered Species)

The CCC coho salmon ESU includes 12 independent populations (Spence et al. 2008b). The spatial structure for CCC coho salmon has been substantially modified due to lack of viable source populations and loss of dependent populations. Wild populations of coho salmon are extinct or nearly so in a number of watersheds within the CCC ESU (Good et al. 2005) 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 Russian River is of particular importance for preventing the extinction and contributing to the recovery of CCC coho salmon

The percentage of cultivated croplands and developed lands that overlap with CCC coho salmon habitat are 2.3% and 9.4%, respectively. Whereas evergreen forested lands comprise 46% of the ESU (NLCD 2006). Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Appendix 4 and in Table 50. Much of the development is centered around San Francisco Bay, and there are also developed areas and agriculture in the Russian River. The dominant land uses in the Russian River watershed are urban, agriculture, ranching, and gravel mining. Forested lands (coniferous) make up 12 %, and agricultural lands make up 13%. Coho in 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 12 populations may be exposed to pesticides applied in agricultural, forested, and urban areas. However, the Russian has the highest agricultural use. Considering the above, NMFS concludes that the potential co-occurrence of use of the a.i.s in this consultation with a key coho salmon population suggests that the populations' abundance and productivity may be severely compromised, and therefore there is a high potential for reduction in reproduction, numbers, or distribution of the species as a whole (Table 118).

Table 118. Central California Coast coho salmon

Pesticide	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Crop	Non-crop	Developed	Undeveloped	Populations	Species
Diflubenzuron	Yes	Yes	Yes	Yes	High	High
Fenbutatin oxide	Yes	No	Yes*	No	High	High
Propargite	Yes	No	Yes*	No	High	High

*In developed lands the only labeled use is for nurseries.

25.3.27 Ozette Lake Sockeye Salmon (Threatened Species)

The Ozette Lake sockeye salmon ESU consists of a single population made up of five spawning aggregations. The population is divided between beach spawners and tributary spawners (NMFS

2009c). 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. 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 Appendix 4 and in Table 56. 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 three a.i.s. Upon leaving the ocean, adult spawners enter the Ozette River, which flows directly from Ozette Lake. Their upriver migration is typically from April to early August. Adults may remain in Ozette Lake for extended periods before spawning (October- February). The Ozette River flows through the coastal rain forest, and is bordered by the Olympic National Park and the Ozette Indian Reservation. 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 leaving the lake via the Ozette River to enter 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 identified 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.

Direct effects to fish are a possibility within tributaries. Although no cropland occurred within the 2.5 km area analyzed, some applications of the a.i.s could occur in developed lands along tributaries. We assumed it is unlikely that restricted use pesticides would be applied in these situations. Not likely to adversely affect determinations were made for the a.i.s (fenbutatin oxide and propargite) that are not registered for forestry use and, therefore, have a very low probability of exposure and effects that are expected to be insignificant or discountable (Table 119).

Application of diflubenzuron could occur to forested lands that comprise much of the ESU. While direct effects on fish are considered unlikely for this a.i., applications could potentially lead to reductions in prey. Based on juvenile sockeye's lake rearing, we do not anticipate any reductions in prey within the tributaries to affect juvenile growth. Prey abundance within Ozette Lake itself is expected to be only slightly affected due to reduced exposure concentrations. NMFS concludes that the potential co-occurrence of diflubenzuron with the population suggests its abundance and productivity may be slightly affected, and as such, there is a low potential for reduction in reproduction, numbers, or distribution of the species as a whole (Table 119).

Table 119. Lake Ozette sockeye salmon

Pesticide	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Crop	Non-crop	Developed	Undeveloped	Populations	Species
Diflubenzuron	No	Yes	Yes	Yes	Low	Low
Fenbutatin oxide	No	No	Yes*	No	NLAA**	NLAA**
Propargite	No	No	Yes*	No	NLAA**	NLAA**

*In developed lands the only labeled use is for nurseries.

**At the scale of the individual, effects are anticipated to be discountable.

25.3.28 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 more recent 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 Appendix 4 and in Table 56. Following emergence, fry immediately migrate to the lake and juveniles rear in the lake for one to three years before migrating through the Snake and Columbia Rivers for several hundred miles to the ocean. Our GIS analysis indicates that within the ESU, Snake River sockeye salmon have minimal risk of exposure to pesticides applied for agricultural or urban uses and may be at risk of exposure from forestry related uses. Given the distance traveled between Redfish Lake and the ocean, juveniles and returning adults are at risk of exposure to pesticides applied for all uses near salmonid habitats during migration.

Although more than 50% of the ESU is in evergreen forests, only diflubenzuron is allowed for use on forests. We anticipate effects to macro invertebrates from diflubenzuron, however we expect only slight reductions in juvenile growth. Juvenile sockeye will either be feeding in the lake, where exposure concentrations will be reduced, or be feeding during the migration, where they will be primarily consuming smaller fish as they will be more than a year old during ocean migration. Direct effects to salmonids from diflubenzuron are not anticipated as salmonids are very insensitive. Some applications of the other two a.i.s could occur in developed lands. However, we assume it is unlikely that restricted use pesticides would be applied in these situations and that any use that did occur would be limited.

Outmigrating juveniles and returning adults are at risk of acute exposures to fenbutatin oxide and propargite as they migrate. However, both life stages do not utilize shallows and off-channel habitats as they travel, instead remaining more in the mainstem of the river. This will reduce the exposure concentrations and, therefore, any effects on the population are expected to be slight.

NMFS concludes that the potential co-occurrence of the three a.i.s with the population suggests its abundance and productivity may be slightly affected, and as such, there is a low potential for reduction in reproduction, numbers, or distribution of the species as a whole (Table 120). For diflubenzuron, this is mostly due to the slight effect on juvenile growth expected from forestry use within the ESU. For fenbutatin oxide and propargite, this is mostly due to slight effects on survival expected from uses along the migratory corridor.

Table 120. Snake River sockeye salmon

Pesticide	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Crop	Non-crop	Developed	Undeveloped	Populations	Species
Diflubenzuron	Yes	Yes	Yes	Yes	Low	Low
Fenbutatin oxide	Yes	No	Yes*	No	Low	Low
Propargite	Yes	No	Yes*	No	Low	Low

*In developed lands the only labeled use is for nurseries.

25.3.29 *Puget Sound Steelhead (Threatened Species)*

The Puget Sound (PS) 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 2005d). 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 (Table 57). 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 Appendix 4. Our GIS analysis indicates all populations in this DPS are exposed to pesticides applied in agriculture and urban areas. Steelhead 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.

The Northwest Fisheries Science Center reported the populations in the PS steelhead DPS are showing continued downward trends in estimated abundance, a few sharply so. This DPS remains distributed over a large geographic area but current trends in abundance are concerning. Available new information confirms that this DPS remains at moderate risk of extinction. The forthcoming recovery plan for this DPS will identify specific viability criteria that will need to be met in order for this DPS to be considered recovered.

Due to their long freshwater residency time and their use of freshwater, estuarine, and nearshore habitats, juveniles and migrating adults have a high probability of exposure to pesticides that are applied near their habitats. Considering the above, NMFS concludes that the potential co-occurrence of use of the a.i.s in this consultation with key Puget Sound steelhead populations suggests that their abundance and productivity may be severely compromised, and therefore there is a high potential for reduction in reproduction, numbers, or distribution of the species as a whole (Table 121).

Table 121. Puget Sound steelhead

Pesticide	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Crop	Non-crop	Developed	Undeveloped	Populations	Species
Diflubenzuron	Yes	Yes	Yes	Yes	High	High
Fenbutatin oxide	Yes	No	Yes*	No	High	High
Propargite	Yes	No	Yes*	No	High	High

*In developed lands the only labeled use is for nurseries.

25.3.30 Lower Columbia River Steelhead (Threatened Species)

The Lower Columbia River (LCR) steelhead DPS includes 23 extant populations, 16 of which are considered to be at high or very high risk of extinction. 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 that overlap with the LCR Steelhead DPS is about 7%. Of that, 4.5 % is hay/pasture land and 2.5% is cultivated crop land (Table 57). More than 61% of the DPS is composed of evergreen, deciduous, and mixed forests. Urban/residential developed lands cover 12% of this DPS. Co-occurrence of agriculture, undeveloped, and urban areas with salmonid habitat is shown in Appendix 4 and in Table 57. Our GIS analysis indicates all populations are exposed to pesticides applied in agricultural, undeveloped, 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. Due to their relatively long freshwater residency time and their extensive use of freshwater habitats, juveniles and migrating adults have a high probability of exposure to pesticides that are applied near their habitats. NMFS concludes that the potential co-occurrence of use of the a.i.s in this consultation with key LCR steelhead populations suggests that their abundance and productivity may be severely compromised, and there is a high potential for reduction in reproduction, numbers, or distribution of the species as a whole (Table 122).

Table 122. Lower Columbia River steelhead

Pesticide	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Crop	Non-crop	Developed	Undeveloped	Populations	Species
Diflubenzuron	Yes	Yes	Yes	Yes	High	High
Fenbutatin oxide	Yes	No	Yes*	No	High	High
Propargite	Yes	No	Yes*	No	High	High

*In developed lands the only labeled use is for nurseries.

25.3.31 *Upper Willamette River Steelhead (Threatened Species)*

The Upper Willamette River (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, which currently is at levels observed in the mid-1990s. 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, human population growth, and degraded water quality within the Willamette mainstem and the lower reaches of its tributaries. Past USGS sampling indicated 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 exceeded EPA criteria for the protection of freshwater aquatic life (USGS 2008).

The percentage of cultivated crop lands and developed lands overlapping with this DPS are 14.5% and 10%, respectively (Table 57). Co-occurrence of agriculture, undeveloped, and urban areas with salmonid habitat is shown in Appendix 4 and in Table 57. Our GIS analysis indicates all four populations in this DPS are likely exposed to pesticides applied in agricultural and urban areas. After emergence, steelhead fry typically rear in floodplain habitats associated with their natal rivers and streams for two years. Given their relatively long freshwater residency period and habitat preference, juveniles and migrating adults have a high probability of exposure to pesticides that are applied near their habitat. NMFS concludes that the potential co-occurrence of use of each of the a.i.s with all of the populations in the ESU suggests that their abundance and productivity may be severely compromised, and there is a high potential for reduction in reproduction, numbers, or distribution of the species as a whole (Table 123).

Table 123. Upper Willamette Steelhead

Pesticide	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Crop	Non-crop	Developed	Undeveloped	Populations	Species
Diflubenzuron	Yes	Yes	Yes	Yes	High	High
Fenbutatin oxide	Yes	No	Yes*	No	High	High
Propargite	Yes	No	Yes*	No	High	High

*In developed lands the only labeled use is for nurseries.

25.3.32 Middle Columbia River Steelhead (Threatened Status)

The Middle Columbia River (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. Past NAWQA and Washington State monitoring indicated seventy-six pesticide compounds were detected within the Yakima River Basin (USGS 2008, Sargeant et al. 2013b)

The percentage of cultivated crop lands and developed lands within the range of this DPS are 17% and 3%, respectively (Table 57). Co-occurrence of agriculture, undeveloped, and urban areas with salmonid habitat is shown in Appendix 4. Our GIS analysis indicates all 16 populations are likely exposed to pesticides applied in agricultural and urban areas. Steelhead fry usually inhabit shallow water along stream banks and stream margins, where they rear for approximately two years. Adult steelhead return to spawn at all times of the year, thus adults and rearing juveniles are in freshwater habitats throughout the year. Due to their relatively long freshwater residency time and their extensive use of freshwater habitats, juveniles and migrating adults have a high probability of exposure to pesticides that are applied near their habitats. NMFS concludes that the potential co-occurrence of use of each of the a.i.s with all of the populations in the ESU suggests their abundance and productivity may be severely compromised. Therefore, there is a high potential for reduction in reproduction, numbers, or distribution of all populations and the species as a whole (Table 124).

Table 124. Middle Columbia River steelhead

Pesticide	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Crop	Non-crop	Developed	Undeveloped	Populations	Species
Diflubenzuron	Yes	Yes	Yes	Yes	High	High
Fenbutatin oxide	Yes	No	Yes*	No	High	High
Propargite	Yes	No	Yes*	No	High	High

*In developed lands the only labeled use is for nurseries.

25.3.33 *Upper Columbia River Steelhead (Endangered Species)*

The Upper Columbia River (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 eliminated by the Grand Coulee Dam.

The major obstacles to the survival and recovery of UCR steelhead include hatcheries, dams that block fish migration, altered floodplain and channel morphology, water diversions, loss of in-stream woody debris, 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 (Table 57). 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 in search of 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, undeveloped and urban areas with salmonid habitat is shown in Appendix 4. There are some agricultural lands in the spawning and rearing areas in the Wenatchee, Methow, and Okenogan watersheds. In the Entiat watershed, there is intense agriculture outside the buffer in the Upper Columbia Irrigation District. River 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, but we found no data to suggest how elevated temperatures may affect exposure to the three a.i.s.

Due to their long freshwater residency time and extensive use of freshwater habitats, juveniles and migrating adults have a high probability of exposure to pesticides that are applied near their habitats. As shown in Table 125, NMFS concludes that the potential co-occurrence of use of each of the a.i.s with all of the populations in the ESU suggests their abundance and productivity will be severely compromised and that there is a high potential for reduction in reproduction, numbers, or distribution of the species as a whole.

Table 125. Upper Columbia River steelhead

Pesticide	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Crop	Non-crop	Developed	Undeveloped	Populations	Species
Diflubenzuron	Yes	Yes	Yes	Yes	High	High
Fenbutatin oxide	Yes	No	Yes*	No	High	High
Propargite	Yes	No	Yes*	No	High	High

*In developed lands the only labeled use is for nurseries.

25.3.34 Snake River Basin Steelhead (Threatened Species)

The Snake River (SR) basin steelhead DPS includes 24 populations that are spatially distributed in each of the six major geographic areas (Lower Snake, Clearwater, Grande Ronde, Salmon River, Hells Canyon and Imnaha) 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, and the DPS remains at moderate risk of extinction. 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 time.

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, remain in the river through the winter, and 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, and smolt after two or three years. During their freshwater residence they may be exposed to the three a.i.s from a variety of uses on agricultural, urban, and undeveloped lands. Potential exposure from pesticide use within the SR basin 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%) (Table 57). Our GIS analysis indicates substantial overlap of crop lands with steelhead habitats in the Clearwater, Grande Ronde, and Lower Snake Major Population Groups (MPGs). Additionally, the Clearwater and Grande Ronde MPGs have been identified in the SR recovery plan (NOAA 2007) as needing to meet viability criteria in order to be considered for de-listing as a threatened species. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Appendix 4. Due to their long freshwater residency time, their use of a variety of freshwater habitats, and the substantial co-occurrence with crop lands, juveniles and migrating adults have a high probability of exposure to pesticides that are applied near their habitats. NMFS concludes that the potential co-occurrence of use of each of the a.i.s with key populations in the ESU suggests their abundance and productivity will be severely compromised, and that there is a high potential for reduction in reproduction, numbers, or distribution of the species as a whole (Table 126).

Table 126. Snake River steelhead

Pesticide	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Crop	Non-crop	Developed	Undeveloped	Populations	Species
Diflubenzuron	Yes	Yes	Yes	Yes	High	High
Fenbutatin oxide	Yes	No	Yes*	No	High	High
Propargite	Yes	No	Yes*	No	High	High

*In developed lands the only labeled use is for nurseries.

25.3.35 Northern California Steelhead (Threatened Species)

The Northern California (NC) steelhead DPS includes 15 historically independent populations of winter-run steelhead and 4 extant populations of summer-run 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 less than 1% and 19%, respectively (Table 51). Co-occurrence of agriculture, undeveloped, and urban areas with salmonid habitat is shown in Appendix 4. We expect NC steelhead populations to have limited exposure to fenbutatin oxide and propargite. The majority of registered uses of these chemicals are on crops, and there are few areas of concentrated agriculture in this ESU. 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. Since diflubenzuron is authorized for use on nearly all the land uses and juvenile steelhead have a protracted rearing time in freshwater habitats, we found a strong likelihood for exposure to diflubenzuron when applied near their habitats. Conversely, it is unlikely that steelhead will be exposed to the other two a.i.s based on marginal overlap of habitat with crop lands.

NMFS concludes that the potential co-occurrence of use of each of the a.i.s with all of the populations in the ESU suggests their abundance and productivity will be severely compromised by diflubenzuron but only slightly affected by fenbutatin oxide and propargite. Therefore, diflubenzuron results in a high potential for reduction in reproduction, numbers, or distribution of the species as a whole, and fenbutatin oxide and propargite result in a low potential for reduction in reproduction, numbers, or distribution of the species as a whole (Table 127).

Table 127. Northern California steelhead

Pesticide	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Crop	Non-crop	Developed	Undeveloped	Populations	Species
Diflubenzuron	Yes	Yes	Yes	Yes	High	High
Fenbutatin oxide	Yes	No	Yes*	No	Low	Low
Propargite	Yes	No	Yes*	No	Low	Low

*In developed lands the only labeled use is for nurseries.

**Due to aquatic applications and forestry use.

25.3.36 Central California Coast (CCC) Steelhead (Threatened Species)

The Central California Coast (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 smoltification 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 iteroparity.

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.

The percentages of cultivated crop and developed lands overlapping with CCC steelhead habitat are 4% and 25%, respectively (Table 51). A large proportion of crop lands are along the Russian River, which also has one of the largest steelhead populations. Southern portions of the DPS include the heavily developed areas around San Francisco Bay. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Appendix 4. 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. Due to steelhead's long freshwater residency time, their use of extensive freshwater habitats, and the large overlap of crop and developed lands with steelhead habitat, juveniles and migrating adults have a high probability of exposure to pesticides that are applied near their habitats. NMFS concludes that the potential co-occurrence of use of each of the a.i.s with key populations in the ESU suggests their abundance and productivity will be severely compromised, and that there is a high potential for reduction in reproduction, numbers, or distribution of the species as a whole (Table 128).

Table 128. Central California Coast steelhead

Pesticide	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Crop	Non-crop	Developed	Undeveloped	Populations	Species
Diflubenzuron	Yes	Yes	Yes	Yes	High	High
Fenbutatin oxide	Yes	No	Yes*	No	High	High
Propargite	Yes	No	Yes*	No	High	High

*In developed lands the only labeled use is for nurseries.

25.3.37 California Central Valley (CCV) Steelhead (Threatened Species)

The California Central Valley (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.

The percentage of agriculture, developed, and forested lands that overlap with CCV steelhead habitat are 32%, 10%, and 58%, respectively (Table 51). Heavy use of agricultural pesticides increases the likelihood of negative effects for this species. Co-occurrence of agriculture, undeveloped, and urban areas with salmonid habitat is shown in Appendix 4. Our GIS analysis indicates that CCV steelhead may be exposed to pesticides applied in aquatic, urban, 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 habitats. Adult steelhead return to this system year-round, 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 aquatic invertebrate prey.

Due to steelhead's long freshwater residency time and the large overlap of crop and developed lands with steelhead habitat, juveniles and migrating adults have a high probability of exposure to pesticides that are applied near their habitats. NMFS concludes that the potential co-occurrence of use of each of the a.i.s with key populations in the ESU suggests their abundance and productivity will be severely compromised, and that there is a high potential for reduction in reproduction, numbers, or distribution of the species as a whole (Table 129).

Table 129. California Central Valley steelhead

Pesticide	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Crop	Non-crop	Developed	Undeveloped	Populations	Species
Diflubenzuron	Yes	Yes	Yes	Yes	High	High
Fenbutatin oxide	Yes	No	Yes*	No	High	High
Propargite	Yes	No	Yes*	No	High	High

*In developed lands the only labeled use is for nurseries.

25.3.38 South-Central California Coast (S-CCC) Steelhead (Threatened Species)

The South-Central California Coast (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, and 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 iteroparity. 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 give the benefit of doubt to the species and assume that the populations are predominantly in the mainstem of the Salinas and Pajaro Rivers. Both of these river basins have areas of intensive agriculture and development.

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 percentages of cultivated crop lands and developed lands that overlap with this DPS are 7% and 10%, respectively (Table 51). Co-occurrence of agriculture, undeveloped, and urban areas with salmonid habitat is shown in Appendix 4. Agriculture is the dominant land use in the Salinas River valley, and there are areas of intense agriculture in the Russian River and the Pajaro watersheds as well. Habitats higher in the Salinas and Pajaro watersheds and along some of the coastal areas overlap to a far lesser degree with crop lands. The Russian River is of particular importance for preventing the extinction and contributing to the recovery of CCC steelhead.

Due to the overlap of steelhead habitat with crop lands in the Salinas and Pajaro River valleys, and the protracted amount of time juveniles and migrating adults spend in freshwater habitats, there is a high likelihood that S-CCC steelhead will be exposed to pesticides that are used near their habitats. Therefore, NMFS concludes that the potential co-occurrence of use of each of the a.i.s with key populations in the ESU suggests their abundance and productivity will be severely compromised, and that there is a high potential for reduction in reproduction, numbers, or distribution of the species as a whole (Table 130).

Table 130. South-Central California Coast steelhead

Pesticide	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Crop	Non-crop	Developed	Undeveloped	Populations	Species
Diflubenzuron	Yes	Yes	Yes	Yes	High	High
Fenbutatin oxide	Yes	No	Yes*	No	High	High
Propargite	Yes	No	Yes*	No	High	High

*In developed lands the only labeled use is for nurseries.

25.3.39 Southern California (SC) Steelhead (Endangered Species)

The Southern California (SC) DPS is at the species extreme southern limit of the steelhead range, and 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 than other species. SC steelhead juveniles rear in fresh water streams and rivers 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.

The percentage of cultivated crop lands and developed lands within SC steelhead habitat are about 5% and 34%, respectively as shown in Table 51. Cultivated crop lands are concentrated in the 4 major river valleys (Santa Maria, Santa Ynez, Ventura, and Santa Clara rivers). Co-occurrence of agricultural, developed, and undeveloped areas with salmonid habitat is shown in Appendix 4. All of the rivers are affected by anthropogenic inputs, and certain direct aquatic uses of diflubenzuron are permitted. Given the overlap of SC habitats with crop, urban and aquatic areas where labeled uses of the three a.i.s are permitted, it is likely that steelhead will be exposed to the three pesticides when used near their habitats. Therefore, NMFS concludes that the potential co-occurrence of use of each of the a.i.s with key populations in the ESU suggests their abundance and productivity will be severely compromised, and that there is a high potential for reduction in reproduction, numbers, or distribution of the species as a whole (Table 131).

Table 131. Southern California steelhead

Pesticide	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Crop	Non-crop	Developed	Undeveloped	Populations	Species
Diflubenzuron	Yes	Yes	Yes	Yes	High	High
Fenbutatin oxide	Yes	No	Yes*	No	High	High
Propargite	Yes	No	Yes*	No	High	High

*In developed lands the only labeled use is for nurseries.

Table 132. Summary table of species calls for each active ingredient

Species	ESU	Diflubenzuron	Fenbutatin Oxide	Propargite
Chinook	Puget Sound	High	High	High
	Lower Columbia River	High	High	High
	Upper Columbia River Spring - Run	High	High	High
	Snake River Fall - Run	Medium	Medium	Medium
	Snake River Spring/Summer - Run	High	High	High
	Upper Willamette River	High	High	High
	California Coastal	High	High	High
	Central Valley Spring - Run	High	High	High
	Sacramento River Winter - Run	High	High	High
Chum	Hood Canal Summer - Run	Medium	Low	Low
	Columbia River	Medium	Low	Low
Coho	Lower Columbia River	High	High	High
	Oregon Coast	High	Low	Low
	Southern Oregon and Northern California Coast	High	High	High
	Central California Coast	High	High	High
Sockeye	Ozette Lake	Low	NLAA	NLAA
	Snake River	Low	Low	Low
Steelhead	Puget Sound	High	High	High
	Lower Columbia River	High	High	High
	Upper Willamette River	High	High	High
	Middle Columbia River	High	High	High
	Upper Columbia River	High	High	High
	Snake River	High	High	High
	Northern California	High	Low	Low
	Central California Coast	High	High	High
	California Central Valley	High	High	High
	South-Central California Coast	High	High	High
	Southern California	High	High	High

12 Effects of the Action on Proposed and Designated Critical Habitat

NMFS' critical habitat analysis determines whether the proposed action is likely to 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. A diagram of our analysis framework is shown in Figure 81. 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 whether critical habitat is likely to be exposed to the stressors of the proposed action. If we find that critical habitat is likely to be exposed, we assess the consequences of that exposure on the quality, quantity, or availability of one or more of those primary constituent elements (PCEs) that comprise critical habitat (Table 5). Typically each species' PCEs include freshwater spawning, rearing, migrating, estuarine, and open ocean sites. Salmonids that utilize the Puget Sound in Washington State may also have near shore marine sites as a PCE.

Water quality, forage (prey availability), and natural cover (riparian vegetation) 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 contaminants. Here, we use the presence of chemical contaminants as an indicator of degraded water quality as exposure to sufficient chemical concentrations can affect salmonid prey.

The proposed action would degrade water quality by introducing diflubenzuron, fenbutatin oxide, propargite, and other associated chemicals into salmonid habitats. Therefore, we use the pesticide concentrations likely to adversely affect fish, prey, and terrestrial and aquatic plants as measures of degraded water quality. We also note that PCE's depend on availability and quality of prey. The three insecticides all affect prey at environmentally realistic concentrations. This analysis is conducted by comparing toxicity information (e.g., aquatic invertebrate LC_{50} values) reviewed earlier in the *Response* section with expected pesticide concentrations in salmonid habitats presented in the *Exposure* section.

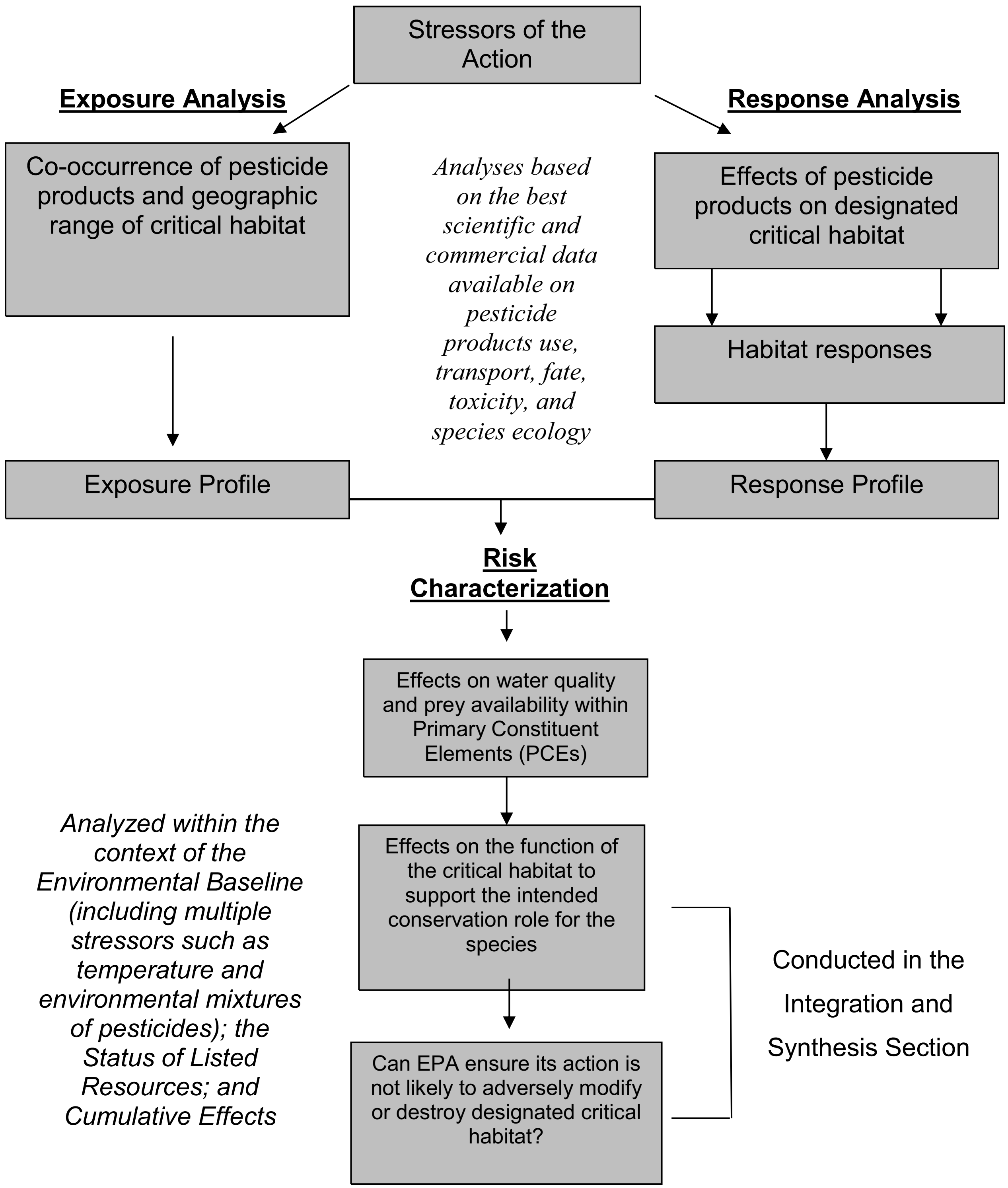

Figure 81. Assessment Framework for Designated Critical Habitat.

We translated each of the PCEs into a risk hypothesis to assess potential impacts on designated critical habitat. The analysis of risk hypotheses is based on: 1) the likely concentrations of the three pesticides that would be observed in critical habitat; and 2) the response of PCEs to those anticipated concentrations.

NMFS used conservation values (high, medium, and low) of watersheds within each ESU/DPS for the PCEs of critical habitat identified for each life stage common to listed salmonids (described in the *Status of Listed Resources* section). Because watersheds with high conservation value are essential to the conservation of the species; reductions in the quantity, quality, or distribution of the PCEs supporting that watershed would be expected to adversely affect the function of critical habitat to support its intended conservation role. We assess these watersheds within the *Integration and Synthesis for Designated Critical Habitat* section.

NMFS has designated, or proposed for designation, critical habitat for each of the species in this Opinion. In January, 2013, NMFS proposed designated critical habitat for lower Columbia River coho, and Puget Sound steelhead. Critical habitat has been proposed for these two species and is expected to be formally designated in late 2013 or early 2014. The action area for this Opinion encompasses all designated and proposed critical habitat for listed Pacific salmonids in Washington, Oregon, California and Idaho. The PCEs for each listed species are described in the *Status of Listed Resources* section of this Opinion. As the species of salmonids addressed in this Opinion have similar life history characteristics, they share many of the same PCEs. These PCEs include sites that support one or more life stages and contain physical or biological features essential to the conservation of the ESU/DPS. PCEs include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, estuarine areas, nearshore marine areas, and offshore marine areas.

Water quality, prey availability and riparian vegetation in freshwater and estuarine areas may be susceptible to pesticide effects where they overlap. Effects to water quality and prey availability will be evaluated to determine the likelihood of reducing the quality of freshwater, estuarine, and nearshore marine areas. Given the use and environmental fate profile of the pesticide formulations containing diflubenzuron, fenbutatin oxide and propargite, we do not expect offshore marine areas to be affected. Therefore, a risk hypothesis was not developed for this area and further evaluation of this PCE is not warranted.

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 adversely affect aquatic organisms and 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 impacted.

We also evaluate effects on salmonid prey because forage is an essential attribute of many PCEs. Freshwater juvenile rearing and migratory habitats as well as estuarine and nearshore marine areas must provide sufficient forage to 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 (EC_{50}/LC_{50}), prey growth, salmonid survival, fish growth and reproduction, and aquatic primary production to determine whether expected concentrations of the stressors of the action are sufficient to affect PCEs for salmonid critical habitats.

12.1 Exposure of proposed and designated critical habitats to the stressors of the action:

Designated critical habitat for the three species is located within the action area. Many freshwater areas overlap with the allowable uses (Appendix 4). The stressors of the action contaminate these habitats via drift and runoff (including from irrigation returns), and to a lesser extent from atmospheric deposition. Once in salmonid habitats, the three active ingredients persist for varying periods of time, depending on the chemical, biological, and physical environment of the contaminated aquatic habitats. The most persistent of the three, fenbutatin oxide (3.5 year half-life in aquatic systems), may accumulate in aquatic food chains affecting organisms beyond those exposed initially from application events. Expected concentrations of other/inert ingredients and adjuvants added to formulations prior to application remain unknown, and are an identified data gap.

Table 133 shows expected concentrations of the three a.i.s that were derived from EPA modeling estimates, 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 available exposure information applies more readily to freshwater habitats compared to estuarine or marine habitats, where much less information is available.

Table 133. Expected concentrations (µg/l) of the three active ingredients in aquatic ecosystems.

	Diflubenzuron	**Fenbutatin Oxide**	**Propargite**
EPA peak PRZM/EXAMS estimates for farm pond	0.1 - 34	1 – 69	0.26 - 32
NMFS AgDrift estimates for floodplain habitat	0.02 - 53	0.12 - 67	0.11 - 269
Monitoring data	n/a	0.15 - 111	0.003 - 20

Responses of salmonid habitats to the stressors of the action

If PCEs are exposed to the stressors, we evaluate the level at which the three a.i.s adversely affect water quality, prey availability, and natural cover (terrestrial and aquatic primary production). For many of the other ingredients contained in formulations of the a.i.s, 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 diflubenzuron, fenbutatin oxide, and propargite can result in concentrations that may reduce salmonid survival, prey survival, prey growth, and fish growth and reproduction. These types of individual fitness consequences demonstrate a degradation of water quality in affected habitats.

We summarized the available toxicity information in the *Response Analysis* (Table 92). A summary of assessment endpoint toxicity values relevant to critical habitat analysis is shown here in Table 134. It is important to note that the toxicity of diflubenzuron, fenbutatin oxide, and propargite is variable depending on the biological endpoint (*e.g.*, acute lethality to fish and invertebrates), the concentrations expected in salmonid habitats, the presence of other pesticides, and the occurrence of other environmental stressor such as elevated water temperatures.

Table 134. Effect concentrations (µg/L) of the three active ingredients in aquatic ecosystems.

Assessment Endpoint	Diflubenzuron	Fenbutatin oxide	Propargite
Salmonid survival (LC_{50})	57,000 - 342,000	1.1 - 52	43 - 445
Fish growth (LOEC)	>45 (NOEC)	n/a	11
Fish reproduction (LOEC)	50 – 100 (NOEC)	0.61 – 5.7	28
Prey survival (EC_{50})	0.0028 – 57,000	6.4 - 2184	14 - 1770
Prey reproduction/growth (EC_{50}, LOEL)	0.04 - >10	25	14
Primary production (EC_{50}, LOEC)	>30 - 5000	14 - 7434	66.2 – 75,000

12.2 Risk Characterization for Proposed and Designated Critical Habitat

NMFS reviews the status of designated critical habitat affected by the proposed action separate from species effects by examining the condition and trends of primary constituent elements (PCEs) throughout the action area. PCEs are the physical and biological features identified as essential to the conservation of listed species. The PCEs for salmonid designated and proposed critical habitat are identified and discussed in the *Status of the Species* section and are summarized in Table 5.

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 designated and proposed critical habitats. 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 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, prey availability, and natural cover are affected (Table 135). We ultimately determine whether the degradation of water quality and reduction in prey availability within freshwater spawning and rearing sites, migration corridors, estuarine areas, and nearshore marine areas will rise to the level expected to reduce the intended conservation role of designated critical habitats, which is evaluated within the *Integration and Synthesis for Designated and Proposed Critical Habitat* section. The final conclusion of whether EPA's proposed action with end-use products containing diflubenzuron, fenbutatin oxide, or propargite are likely to adversely modify or destroy a species' designated critical habitat is provided in the *Conclusion* section.

Table 135. Risk Hypotheses for designated critical habitat.

Risk hypothesis for designated critical habitat
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 resources in freshwater rearing sites.
3. Exposure to the stressors of the action is sufficient to degrade water quality, natural cover, and/or reduce prey resources in freshwater migratory corridors.
4. Exposure to the stressors of the action is sufficient to degrade water quality and/or reduce prey resources in estuarine areas.
5. Exposure to the stressors of the action is sufficient to degrade water quality and/or reduce prey resources in nearshore marine areas.

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 three 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 1) experience multiple applications of the a.i.s, 2) are shallow, low flow systems, and 3) are located in high pesticide use areas such as intensive agricultural watersheds. The best estimators of water quality degradation for spawning habitats are salmonid survival, growth, and reproduction endpoints (Table 136).

Table 136. Analysis of freshwater spawning sites

Analysis Parameters		Diflubenzuron	Fenbutatin Oxide	Propargite
Measured and modeled concentrations in salmonid habitats (µg/L)		0.01-69	0.12-111	0.003-269
Assessment Endpoints used as a measure of degraded water quality	salmonid survival (µg/L)	57,000-342,000	1.1-52	43-445
	fish growth (µg/L)	n/a	n/a	11
	fish reproduction (µg/L)	n/a	0.61-5.7	28
Evidence for degraded water quality?		**no**	**yes**	**yes**

Based on a comparison of relevant water quality endpoints (*i.e.* fish survival, growth, and reproduction) with measured and estimated environmental concentrations (Table 136), we find it possible that water quality will be degraded by fenbutatin oxide and propargite. We find it unlikely that water quality will be degraded by diflubenzuron in freshwater spawning habitats. 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 habitat PCE is likely by fenbutatin oxide and propargite. We evaluate whether the degradation of this PCE, 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 salmon 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 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 areas are diverse, extensive, complex sites that can range from small, shallow, intermittent floodplain habitats to channel edges and shallow bars of large river systems. As such, expected concentrations range from some of the highest estimates (via spray drift into floodplain habitats) to some of the lowest estimates (monitoring results from large rivers). 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. Reductions and removal of prey biomass in rearing habitats may substantially reduce this PCE's role in recovering salmonid populations. The best estimators of water quality degradation and reduced prey abundance in these habitats are prey survival, prey growth and reproduction, fish (juvenile) survival, and fish growth and reproduction endpoints (Table 137).

Table 137. Analysis of freshwater rearing sites

Analysis Parameters		**Diflubenzuron**	**Fenbutatin Oxide**	**Propargite**
Measured and modeled concentrations in salmonid habitats (µg/L)		0.01-53	0.12-111	0.003-269
Assessment Endpoints used as a measure of degraded water quality and reduced prey availability	salmonid survival (µg/L)	57,000-342,000	1.1-52	43-445
	fish growth (µg/L)	n/a	n/a	11
	fish reproduction (µg/L)	n/a	0.61-5.7	28
	prey survival (µg/L)	0.0028-57,000	6.4-2184	14-1770
	prey reproduction and growth (µg/L)	0.062->10	25	14
Evidence for degraded water quality?		**yes**	**yes**	**yes**
Evidence for reduced prey availability?		**yes**	**yes**	**yes**

Based on a comparison of relevant assessment endpoints (*i.e.* prey survival, prey reproduction and growth, fish survival, fish growth, and fish reproduction) with measured and estimated environmental concentrations (Table 4), we find it likely that water quality will be degraded by diflubenzuron, fenbutatin oxide and propargite. Furthermore, we find it highly likely that prey abundance will be reduced by all three insecticides in this habitat. Additionally, there are abundant data from field and mesocosm studies on diflubenzuron showing dramatic and prolonged reductions in prey abundance at low µg/L concentrations (Table 30 in *Risk Characterization* section). The resulting overlap of rearing habitats and application areas combined with expected concentrations and toxicity effect thresholds to aquatic invertebrates indicates that degradation of attributes (*i.e.* water quality and prey availability) of the rearing habitat PCE is likely by all three insecticides. We evaluate whether the degradation of this PCE, in combination with other affected PCEs, reduce the conservation value of the designated critical habitats within the *Integration and Synthesis for Designated Critical Habitat* section.

Risk hypothesis 3. Exposure to the stressors of the action is sufficient to degrade water quality, natural cover, and/or reduce prey availability in freshwater migration corridors.

Freshwater migration corridors require good water quality, natural cover, 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. Rest areas such as undercut banks, side channels, submerged and overhanging large wood, log jams, and beaver dams are often rare 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 without 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, many channel-edge habitats are proximate to application sites of the stressors of the action, thereby 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. The assessment endpoints considered in this risk hypothesis are shown below in Table 138.

Table 138. Analysis of migratory sites

Analysis Parameters		**Diflubenzuron**	**Fenbutatin Oxide**	**Propargite**
Measured and modeled concentrations in salmonid habitats (µg/L)		0.01-69	0.12-111	0.003-269
Assessment Endpoints used as a measure of degraded water quality and reduced prey availability	salmonid survival (µg/L)	57,000-342,000	1.1-52	43-445
	fish growth (µg/L)	n/a	n/a	11
	fish reproduction (µg/L)	n/a	0.61-5.7	28
	prey survival (µg/L)	0.0028-57,000	6.4-2184	14-1770
	prey reproduction and growth (µg/L)	0.062->10	25	14
	aquatic primary production (µg/L)	>30-5000	14-7434	66.2-75,000
Evidence for degraded water quality?		**yes**	**yes**	**yes**
Evidence for reduced prey availability?		**yes**	**yes**	**yes**
Evidence for reduced primary production?		**no**	**no**	**no**

Primary producers are the base of aquatic food webs, and support invertebrate communities that salmon rely upon for food. Reduced abundance of primary producers can have cascading ecological effects throughout the food web, but given the complexities of those relationships, effects can be difficult to predict. Many groups of primary producers (e.g. diatoms, green algae, periphyton) are common in aquatic habitats. While pesticide exposure may reduce the abundance of one or more species, other primary producers may not be affected, resulting in limited reduction in community abundance or biomass and minimal effect, if any, on higher trophic levels. However, if salmonid prey items rely on sensitive primary producers as food, reductions in biomass may limit their growth and survival.

We located laboratory, toxicity data that described short term effects of the three pesticides on aquatic primary producers. Toxicity values show that diflubenzuron is not acutely toxic to aquatic primary producers at anticipated exposure concentrations. EC_{50} values representing 50% reductions in abundance compared to controls are in the mg/L range and do not overlap with expected environmental concentrations. The lower end of reported fenbutatin oxide toxicity values overlap with both monitoring data and concentration estimates for salmonid habitats. Only two EC_{50} values (14 and 27 µg/L), both in the same species of freshwater diatom, are at the low end of the toxicity value range. All other toxicity values are at least an order of magnitude greater. This suggests that there may be a wide range of sensitivities to fenbutatin oxide among aquatic primary producers. It is unclear how this translates to a community-level effect, since no such data were available. The lower range of propargite toxicity values overlap with concentrations expected to be found in flood plain habitats. However, the overlap is only with two data points: a freshwater green algae EC_{50} of 66 µg/L and a diatom EC_{50} of 106 µg/L. The other two data points are at least 1000 times higher, indicating a wide range of sensitivities to propargite among primary producers. Given the wide range of acute toxicity values and the lack of community-level data, we find no compelling evidence suggesting propargite effects on aquatic primary producers will lead to effects at higher trophic levels.

Given the lack of compelling evidence of direct effects on aquatic primary producers, it is unlikely that there will be cascading ecological effects on higher trophic levels. Therefore we find it unlikely that populations of aquatic invertebrates will be reduced due to impacts on primary producers. Additionally, there were no data found for the three pesticides indicating effects on instream cover or riparian vegetation. Given the mode of action of each of the three insecticides, it is unlikely that they would pose a direct toxic threat to riparian vegetation. However, the available aquatic invertebrate toxicity data indicate that expected concentration of the three 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, ranging 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 pesticides from use 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 necessarily representative of estuarine habitats. Pacific estuaries are incredibly variable to one another in terms of size, tidal volume, exchange rate, freshwater input, salinity, watershed land uses, trophic structures, and bathymetry (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.

The available toxicity information for estuarine and marine organisms for the three a.i.s is presented below in Table 139. Diflubenzuron and fenbutatin oxide are toxic to estuarine invertebrates, as evidenced by LC_{50} values in the low µg/L range. No toxicity data for estuarine prey was available for propargite. It is unclear how representative mysid shrimp are of other salmonid prey item.

Table 139. Assessment endpoint values (µg/L) of estuarine invertebrates and fish

Assessment Endpoints	Diflubenzuron	Fenbutatin Oxide	Propargite
Prey survival (Mysid shrimp, chronic LC_{50})	0.086 – 1.24	0.32 – 0.88	n/a
Prey survival (Mysid shrimp, acute LC_{50})	2.1	2.8	n/a
Prey survival (Grass shrimp, acute LC_{50})	1.141 – 3.4	n/a	n/a
Salmonid survival (µg/L)	57,000-342,000	1.1-52	43-445
Fish growth (µg/L)	n/a	n/a	11
Fish reproduction (µg/L)	n/a	0.61-5.7	28

The overlap of allowed use sites with estuaries designated as critical habitat suggests that contamination of estuarine habitats is likely. This expected contamination, when combined with available estuarine toxicity information, supports that degradation of water quality is expected by the three insecticides. Prey resources for juvenile salmonids may be reduced from pulses of diflubenzuron and fenbutatin oxide in high risk areas such as tidal mudflats and channels draining agricultural areas where the pesticide products are applied. It is difficult to determine at what levels forage is affected by propargite given the paucity of toxicity information for that chemical. Adult salmonid forage (small fishes) may be reduced by exposure to fenbutatin oxide and propargite, as evidenced from fish survival LC_{50} values. The highest risk to forage fishes (adult salmonid prey) are in areas to which pesticides are frequently delivered or 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 salmonid growth and maturation. Similar to estuarine sites, we located no monitoring data from marine areas 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 nearshore marine habitats. Available toxicity information from estuarine species can be used to assess toxicity to prey in marine areas, since both groups inhabit saltwater environments. The available toxicity data (Table 139) shows that the stressors of the action are likely to kill and/or reduce the growth of estuarine prey. However, the representativeness of these standard test species (invertebrates and fish) for salmonid prey in nearshore habitats is unknown.

The available toxicity data suggest that these insecticides are toxic to estuarine and marine organisms at low μg/L concentrations. Whether and how frequently the stressors of the action attain toxic levels for sufficient durations within nearshore marine environments remains unknown. Fundamental environmental fate data regarding persistence and degradation rates are lacking, as are data from environmentally realistic exposure scenarios for key salmonid prey taxa including small, forage fish. However, collectively assessing the available toxicity data along with use sites and nearshore habitats, we anticipate some level of degradation of water quality and reduction in available prey resources in these habitats. This may be particularly relevant for habitats within Puget Sound, where chemical deposition and loading might be greater than in other nearshore environments due to a longer water residence time. 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 prey abundance to be reduced in spawning, rearing, migratory, estuarine, and nearshore marine habitats. Next, within the *Integration and Synthesis for Designated Critical Habitat* section, we evaluate whether these adverse changes to PCEs affect the conservation value of designated critical habitat.

13 Integration and Synthesis Analysis for Designated and Proposed Critical Habitat

This section describes NMFS' assessment of the likelihood that EPA's registration of diflubenzuron, fenbutatin-oxide, and propargite will destroy or adversely modify designated critical habitat for the 26 ESUs/DPSs that have designated critical habitat and for the two ESU/DPS that have proposed designations (LCR coho salmon, and Puget Sound steelhead) covered in this Opinion.

All species addressed in this Opinion have similar PCEs. These PCEs are sites supporting one or more life stages and include

1. freshwater rearing sites,
2. freshwater migration corridors,
3. estuarine areas,
4. nearshore marine areas, and
5. offshore marine areas.

These designated areas contain physical or biological features essential to the conservation of the ESU/DPS.

Essential physical and biological features include water quality, substrate, prey availability, and natural cover. Within this section we evaluate whether these adverse changes to PCEs affect the conservation value of designated critical habitat. 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 reductions or community-level modifications in the in- and near-stream plant communities, or reductions in water quality that may cause fish to have impaired health or greater susceptibility to other stressors.

As noted in the salmonid recovery plans and critical habitat designations, 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 cover and forage. Sufficient forage is necessary for juveniles to maintain growth, which subsequently reduces freshwater predation mortality, increases overwintering success, initiates smoltification, and improves their survival at sea. Natural cover, such as over-hanging vegetation and aquatic plants, provides juveniles protective shelters from predation and substrates for prey.

We start with the analyses presented in the *Effects* chapter. Modeling EECs and monitoring data are not ESU/DPS specific. Inherent in the modeling used to determine both PRZM-EXAMS and AgDrift EECs is the assumption that the pesticide is applied in a location next to or draining directly 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. In the *Exposure* NMFS used a GIS overlay containing landuse categories and salmon distributions to determine overlap of the landuse categories and designated critical habitat for each ESU/DPS. During the fifteen year period covered by this Opinion, market or environmental changes, including climate change, could result in shifting or rotation of crops. Therefore landuse categories (agricultural, forestry, urban/developed) are used to determine potential overlap rather than specific crops. Details of the GIS analysis and the maps are provided in *Appendix 4*. In the *Response* section we described the anticipated effects on water quality, primary productivity, riparian vegetation, prey availability and other habitat constituents. Summaries of effects expected based on our analysis of the a.i.s and other stressors of the action are presented below.

In the *Risk Characterization* for Designated Critical Habitat section we discuss how designated critical habitat is affected by the proposed action by examining the condition and trends of primary constituent elements (PCEs) throughout the action area. We use available toxicity information and expected environmental concentrations to evaluate the scientific lines of evidence that support or refute risk hypotheses developed for designated critical habitats. Risk hypotheses relate directly to each PCE, and frame our qualitative analysis of whether the use of the three a.i.s is sufficient to degrade water quality and/or reduce prey availability in each PCE. We ultimately determine whether the degradation of water quality and reduction in prey availability within freshwater spawning and rearing sites, migration corridors, estuarine areas, and nearshore marine areas will rise to the level expected to reduce the intended conservation role of designated critical habitats.

In this section we analyze the likelihood that effects of the a.i.s and other stressors of the action will cause appreciable reduction in the designated critical habitat PCEs for listed Pacific salmon within the context of ESU/DPS- specific considerations discussed in the *Environmental Baseline* and *Status of Listed Resources* sections.

Evaluating the Likelihood of Adverse Effects on PCEs

The likelihood of adverse effects on PCEs was considered low in cases where we did not anticipate reductions or community-level modifications in the in- and near-stream invertebrate communities or reductions in water quality that might impair foraging, fish health, decrease reproduction, or cause greater susceptibility to other stressors. Where the likelihood of adversely affecting PCEs is low, NMFS determines the likely affects to the conservation values of the designated critical habitat is also low.

The likelihood of adverse effects on PCEs was considered medium in cases where we anticipate reductions or community-level modifications in the in- and near-stream invertebrate communities or reductions in water quality which might impair foraging, decrease reproduction, fish health, or cause greater susceptibility to other stressors. Conditions warranting a medium classification included reductions or community-level modifications to in-stream invertebrate communities where the affected areas invertebrate communities are less diverse and abundant, but still likely to provide sufficient forage and energy base for the system. A medium classification is also applied when changes in riparian prey communities affect the amount or type of allochthonous input contributing to the salmonids prey-base. Degradation of water quality is considered medium when chemical concentrations are high enough to affect fish health and susceptibility to other stressors, but not to cause death or visually obvious behavioral modifications.

The likelihood of adverse effects on PCEs was considered high in cases where we anticipate reductions or community-level modifications in the in- and near-stream prey communities or reductions in water quality that might severely impair foraging, fish health or cause greater susceptibility to other stressors. Conditions warranting a high classification included reductions or community-level modifications to in-stream prey communities where affected areas invertebrate communities are less diverse and abundant, and no longer provide sufficient forage and energy base for the system. A high classification is also applied when anticipated reductions in the riparian areas prey communities severely affect the abundance or type of allochthonous input that would otherwise contribute to the salmonids prey-base. Degradation of water quality is considered high when chemical concentrations are high enough to affect fish health and susceptibility to other stressors, and/or causes death or visually obvious behavioral modifications.

Determining Destruction or Adverse Modification of Critical Habitat

In the *Conclusion* section, we present our conclusions regarding whether the proposed action is likely to destroy or adversely modify critical habitat (Table 170). Taking into account the potential unevenness in use of the a.i.s, and the conservation value of the various watersheds, we determined if the proposed action would appreciably reduce conservation value of the critical habitat. We considered the conservation value appreciably reduced if effects were sufficient to cause long-term or permanent shifts in the prey communities, or were anticipated to be temporally persistent due to chemical properties of the a.i. or frequent inputs and occurred in a significant number of watersheds in the ESU/DPS. We considered the conservation value appreciably reduced if degradation of water quality affects fish health or prey availability. Our conclusions regarding the potential for appreciable reductions in the conservation values of salmonid proposed or designated critical habitat are presented in Table 168. A “High” designation indicates we consider the proposed action likely to result in substantial reduction in the conservation value of the proposed or designated critical habitat and therefore the destruction or adverse modification of designated critical habitat. A “medium” designation indicates we consider the proposed action likely to have moderate reductions in the conservation values of proposed or designated critical habitats and may result in the destruction or adverse modification of proposed or designated critical habitat. A “low” designation indicates we consider the proposed action to have minimal effects to the conservation values of designated critical habitat, and in these cases, we do not expect destruction or adverse modifications of proposed or designated critical habitat to result.

13.1 Designated or Proposed Critical Habitat: Specific evaluations for each ESU/DPS 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 the stressors of the action in salmonid spawning, rearing, and migration habitat is of concern, we highlight exposure from the stressors in shallow, more vulnerable habitats. The number of exposed watersheds that co-occur with agricultural, forested, and urban areas is also given. Using both chemical and species habitat information, we determine whether the stressors associated with 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.

Puget Sound Chinook salmon designated critical habitat

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. Most extend into upper watersheds and are exposed to pesticides applied to forests. Among the medium value watersheds, six and nine are exposed to pesticides from agriculture and urban land uses, respectively. All low value areas are exposed to both agricultural and urban land uses. 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, and 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 (e.g., diking, dewatering, fill), reduced substrate conditions for spawning and incubation (e.g., sedimentation), and degraded water quality (e.g., non-point runoff, stormwater, sewage treatment discharge, chemical inputs). 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. 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.

Cultivated crops, including hay and pastures (5%) are primarily distributed on the floodplain and other lowland habitats typical of areas where juvenile Chinook seek refuge in shallow edge habitats along river margins and in side channel floodplain 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.

Overall, the likelihood each of the a.i.s will cause adverse effects on designated critical habitat PCEs is high (Table 140). While not all watersheds in the ESU have agriculture, where all three a.i.s have approved uses, those that do are in areas that overlap with high and medium conservation value Chinook habitat. In addition, most all of the upper watersheds in the ESU support forestry where the use of diflubenzuron is also approved. We also anticipate some input in urban/developed areas and from rights-of-way, adjacent to salmon-bearing waters. We believe significant portions of designated critical habitat will be affected and that the proposed action will appreciably reduce the conservation value of the designated critical habitat by adversely affecting water quality and adversely affecting salmonid prey as described in the *Risk Characterization* for Designated Critical Habitat section.

Table 140. Puget Sound Chinook salmon designated critical habitat

Pesticide	Co-occurrence				Designated critical habitat
	Crop	Non-crop	Developed	Undeveloped	Likelihood to Appreciably reduce conservation values
Diflubenzuron	Yes	Yes	Yes	Yes	High
Fenbutatin oxide	Yes	No	Yes*	No	High
Propargite	Yes	No	Yes*	No	High

*In developed lands the only labeled use is for nurseries.

Lower Columbia River (LCR) Chinook salmon designated critical habitat

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. Most extend into upper watersheds where exposure from the use of diflubenzuron on forest lands is likely. All 13 medium and 4 low conservation watersheds are also exposed to pesticide applications from these 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.

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.

Overall, the likelihood the three a.i.s will cause adverse effects on designated critical habitat PCEs is high (Table 141). Use sites are distributed throughout the ESU/DPS, and we anticipate several high and medium value watersheds within the designated critical habitat will be affected. NMFS anticipates the effects to appreciably reduce the conservation value of the designated critical habitat by adversely affecting water quality and adversely affecting salmonid prey as discussed in the *Risk Characterization* for Designated Critical Habitat section .

Table 141. Lower Columbia River Chinook salmon designated critical habitat

Pesticide	Co-occurrence				Designated critical habitat
	Crop	Non-crop	Developed	Undeveloped	Likelihood to Appreciably reduce conservation values
Diflubenzuron	Yes	Yes	Yes	Yes	High
Fenbutatin oxide	Yes	No	Yes*	No	High
Propargite	Yes	No	Yes*	No	High

*In developed lands the only labeled use is for nurseries.

Upper Columbia River (UCR) Spring-run Chinook salmon designated critical habitat

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. Most extend into upper watersheds where exposure from the use of diflubenzuron on forest lands is likely.

Fish spawn and rear in the major tributaries leading to the Columbia River between Rock Island and Chief Joseph dams. 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.

Overall, the likelihood each of the three a.i.s will cause adverse effects on designated critical habitat PCEs is high (Table 142). Significant watersheds within the designated critical habitat have agriculture, much of which is in orchards. We also anticipate some input in urban/developed areas and from rights-of-way, located adjacent to salmon-bearing waters. We believe significant portions of designated critical habitat will be affected and that the proposed action will appreciably reduce the conservation value of the designated critical habitat by adversely affecting water quality and adversely affecting salmonid prey as discussed in the *Risk Characterization* for Designated Critical Habitat section.

Table 142. Upper Columbia River spring-run Chinook salmon designated critical habitat

Pesticide	Co-occurrence				Designated critical habitat
	Crop	Non-crop	Developed	Undeveloped	Likelihood to Appreciably reduce conservation values
Diflubenzuron	Yes	Yes	Yes	Yes	High
Fenbutatin oxide	Yes	No	Yes*	No	High
Propargite	Yes	No	Yes*	No	High

*In developed lands the only labeled use is for nurseries.

Snake River (SR) Fall-run Chinook salmon designated critical habitat

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. 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.

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.

Overall, the likelihood the three a.i.s will cause adverse effects on designated critical habitat PCEs is medium (Table 143). While there is some overlap with watersheds within the designated critical habitat that have agriculture, spawning and most rearing (spawning and rearing PCEs) takes place above Lower Granite Dam, where with the exception of the town of Lewiston and some nearby agriculture, most of the land is undeveloped. While we anticipate some input in urban/developed areas and from rights-of-way adjacent to salmon-bearing waters, and some inputs from forest applications of diflubenzuron. We believe designated critical habitat will be minimally affected. We do not believe the proposed action will appreciably reduce the conservation value of the designated critical habitat.

Table 143. Snake River fall-run Chinook salmon designated critical habitat

Pesticide	Co-occurrence				Designated critical habitat
	Crop	Non-crop	Developed	Undeveloped	Likelihood to Appreciably reduce conservation values
Diflubenzuron	Yes	Yes	Yes	Yes	Medium
Fenbutatin oxide	Yes	No	Yes*	No	Medium
Propargite	Yes	No	Yes*	No	Medium

*In developed lands the only labeled use is for nurseries.

Snake River (SR) Spring/Summer-run Chinook salmon designated critical habitat

The designated critical habitat for the SR spring/summer-run Chinook salmon is much larger in than the designated critical habitat for the SR fall-run Chinook salmon. As with the fall-run, 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 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 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.

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 designated critical habitat; however those that are present are clustered mostly around the mainstem Snake and Columbia Rivers and in watersheds that support core and genetic legacy populations necessary for the recovery of the species (NOAA 2007). The Snake and Columbia Rivers are high-volume, high-flow systems, and salmon use it primarily as a migratory corridor.

Overall, the likelihood that three a.i.s will cause adverse effects on designated critical habitat PCEs is High (Table 144). Key watersheds supporting core populations of the run have agriculture and forest uses. We also anticipate some input in urban/developed areas and from rights-of-way adjacent to salmon-bearing waters. We believe important portions of designated critical habitat will be affected and that the proposed action will appreciably reduce the conservation value of the designated critical habitat by adversely affecting water quality and adversely affecting salmonid prey as discussed in the *Risk Characterization* for Designated Critical Habitat section.

Table 144. Snake River spring/summer-run Chinook salmon designated critical habitat

Pesticide	Co-occurrence				Designated critical habitat
	Crop	Non-crop	Developed	Undeveloped	Likelihood to Appreciably reduce conservation values
Diflubenzuron	Yes	Yes	Yes	Yes	High
Fenbutatin oxide	Yes	No	Yes*	No	High
Propargite	Yes	No	Yes*	No	High

*In developed lands the only labeled use is for nurseries.

Upper Willamette River (UWR) Chinook salmon designated critical habitat

Of 59 assessed watersheds, 22 are of high, 18 are medium and 19 are low conservation value. 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. In addition, most overlap with forestry uses in the middle and upper portions of the watershed. Of the medium conservation watersheds, 13 and 12 are also exposed to pesticide applications from the above respective land uses. All 19 low value habitats are exposed to urban and developed uses. The percentage of cultivated and develop lands that overlap with UWR Chinook salmon habitat are 27% 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.

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.

Overall, the likelihood the three a.i.s will cause adverse effects on designated critical habitat PCEs is high (Table 145). Use sites are distributed throughout the ESU/DPS, and we anticipate most or all designated critical habitat will be affected. We anticipate pesticide input to designated habitat will occur multiple times due to repeated applications, and/or from multiple sources. Given landuse within the designated critical habitat area within the ESU, we also anticipate exposure to other stressors which exacerbate the effects of the a.i. NMFS anticipates the effects to appreciably reduce the conservation value of the designated critical habitat by adversely affecting water quality and adversely affecting salmonid prey as discussed in the *Risk Characterization* for Designated Critical Habitat section.

Table 145. Upper Willamette River Chinook salmon designated critical habitat

Pesticide	Co-occurrence				Designated critical habitat
	Crop	Non-crop	Developed	Undeveloped	Likelihood to Appreciably reduce conservation values
Diflubenzuron	Yes	Yes	Yes	Yes	High
Fenbutatin oxide	Yes	No	Yes*	No	High
Propargite	Yes	No	Yes*	No	High

*In developed lands the only labeled use is for nurseries.

California Coastal (CC) Chinook salmon designated critical habitat

Of 45 occupied watersheds, 27 and 10 are of high and medium conservation value, respectively. The remaining 8 are of low conservation value. Our GIS analysis indicates 8 and 27 high conservation watersheds are exposed to pesticides from agriculture and urban land uses, respectively. In addition, most overlap with forestry uses in the middle and upper portions of the watershed. Of the medium conservation watersheds, 4 and 10 are exposed to pesticide applications from the above respective land uses. All 8 low are exposed to urban land uses, while 2 are exposed to agriculture 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 low population abundance across the ESU.

As indicated in Table 146, the likelihood each of the a.i.s will cause adverse effects on designated critical habitat PCEs is high. Use sites are distributed throughout the designated critical habitat area within the ESU, and we anticipate important watersheds within the designated critical habitat will be affected. We anticipate pesticide input to habitat will occur multiple times due to repeated applications, and/or input from multiple sources. Given landuse within the designated critical habitat area, NMFS anticipates the effects from the proposed action to appreciably reduce the conservation value of the designated critical habitat by adversely affecting water quality and adversely affecting salmonid prey as discussed in the *Risk Characterization* for Designated Critical Habitat section .

Table 146. California Coastal Chinook salmon designated critical habitat

Pesticide	Co-occurrence				Designated critical habitat
	Crop	Non-crop	Developed	Undeveloped	Likelihood to Appreciably reduce conservation values
Diflubenzuron	Yes	Yes	Yes	Yes	High
Fenbutatin oxide	Yes	No	Yes*	No	High
Propargite	Yes	No	Yes*	No	High

*In developed lands the only labeled use is for nurseries.

Central Valley (CV) Spring-run Chinook salmon designated critical habitat

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. Many of the watersheds have forested origins, an additional approved use site for diflubenzuron. Of the medium conservation watersheds, two and three watersheds are exposed to from the above land uses as well. All low value watersheds are exposed to pesticide applications from urban land uses (diflubenzuron), while only 2 are exposed to agricultural applications.

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 (with the exception of the Yolo bypass) 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 degrade 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. 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.

Overall, the likelihood the three a.i.s will cause adverse effects on designated critical habitat PCEs is high. Use sites are distributed throughout the designated critical habitat within the ESU. We anticipate most or all designated critical habitat will be affected. We anticipate input of these chemicals to habitat will occur multiple times due to repeated applications, and/or input from multiple sources. Given landuse within the designated critical habitat area, we also anticipate exposure to other stressors which exacerbate the effects of the a.i.s. Therefore, NMFS anticipates the effects from the proposed action to appreciably reduce the conservation value of the designated critical habitat by adversely affecting water quality and adversely affecting salmonid prey as discussed in the *Risk Characterization* for Designated Critical Habitat section (Table 147).

Table 147. Central Valley spring-run Chinook salmon designated critical habitat

Pesticide	Co-occurrence				Designated critical habitat
	Crop	Non-crop	Developed	Undeveloped	Likelihood to Appreciably reduce conservation values
Diflubenzuron	Yes	Yes	Yes	Yes	High
Fenbutatin oxide	Yes	No	Yes*	No	High
Propargite	Yes	No	Yes*	No	High

*In developed lands the only labeled use is for nurseries.

Sacramento River Winter-run Chinook salmon designated critical habitat

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. 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.

Overall, the likelihood each of the a.i.s will cause adverse effects on critical habitat PCEs is high. Use sites are distributed throughout the designated critical habitat of Sacramento River winter-run Chinook salmon ESU. We anticipate most or all designated critical habitat will be affected. We anticipate the a.i.s input to habitat will occur multiple times due to repeated applications, and/or input from multiple sources. Given landuses within the designated critical habitat, we also anticipate exposure to other stressors which exacerbate the effects of the a.i. We anticipate the effects from the proposed action to appreciably reduce the conservation value of the designated critical habitat by adversely affecting water quality and adversely affecting salmonid prey as discussed in the *Risk Characterization* for Designated Critical Habitat section (Table 148).

Table 148. Sacramento River winter-run Chinook salmon designated critical habitat

Pesticide	Co-occurrence				Designated critical habitat
	Crop	Non-crop	Developed	Undeveloped	Likelihood to Appreciably reduce conservation values
Diflubenzuron	Yes	Yes	Yes	Yes	High
Fenbutatin oxide	Yes	No	Yes*	No	High
Propargite	Yes	No	Yes*	No	High

*In developed lands the only labeled use is for nurseries.

Hood Canal Summer-run Chum Salmon designated critical habitat

Of 12 assessed watersheds, nine and three are of high and medium conservation value, respectively. Most all of the watersheds have extensive forest lands. 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, and most all are exposed from forest uses. All three medium conservation watersheds are exposed to these 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.

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.

As indicated in Table 149, the likelihood diflubenzuron will cause adverse effects on critical habitat PCEs is medium, and the likelihood fenbutatin-oxide and propargite will cause adverse effects on critical habitat PCEs is low. Only a few watersheds in the designated critical habitat areas within the ESU have much agriculture. Diflubenzuron is approved for use on forests and developed lands. We anticipate some input in forested, urban/developed areas and from rights-of-way located adjacent to salmon-bearing waters. We believe the conservation values of designated critical habitat of Hood Canal summer-run chum salmon will be moderately affected by approved uses of diflubenzuron, and minimally affected by approved uses of fenbutatin-oxide and propargite.

While water quality and prey may be impacted by the use of the a.i.s and diflubenzuron in particular, we anticipate the effects from the proposed action will not appreciably reduce the conservation value of the designated critical habitat.

Table 149. Hood Canal summer-run chum salmon designated critical habitat

Pesticide	Co-occurrence				Designated critical habitat
	Crop	Non-crop	Developed	Undeveloped	Likelihood to Appreciably reduce conservation values
Diflubenzuron	Yes	Yes	Yes	Yes	Medium
Fenbutatin oxide	Yes	No	Yes*	No	Low
Propargite	Yes	No	Yes*	No	Low

*In developed lands the only labeled use is for nurseries.

Columbia River (CR) Chum Salmon designated critical habitat

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.

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.

Overall, the likelihood diflubenzuron will cause adverse effects on critical habitat PCEs is medium, and the likelihood fenbutatin-oxide and propargite will cause adverse effects on critical habitat PCEs is low (Table 150). Only a few watersheds in the designated critical habitat areas within the ESU have agriculture. Diflubenzuron is approved for use on forests and developed lands. We anticipate some input in forested, urban/developed areas and from rights-of-way located adjacent to salmon-bearing waters. We believe the conservation values of designated critical habitat of Columbia River chum salmon will be moderately affected by approved uses of diflubenzuron, and minimally affected by approved uses of fenbutatin-oxide and propargite. While water quality and prey may be impacted by the use of the a.i.s and diflubenzuron in particular, we anticipate the effects from the proposed action will not appreciably reduce the conservation value of the designated critical habitat.

Table 150. Columbia River chum salmon designated critical habitat

Pesticide	Co-occurrence				Designated critical habitat
	Crop	Non-crop	Developed	Undeveloped	Likelihood to Appreciably reduce conservation values
Diflubenzuron	Yes	Yes	Yes	Yes	Medium
Fenbutatin oxide	Yes	No	Yes*	No	Low
Propargite	Yes	No	Yes*	No	Low

*In developed lands the only labeled use is for nurseries.

Lower Columbia River Coho Salmon proposed designated critical habitat

Designated critical habitat was proposed on January 14, 2013 (50 CFR Part 226). Final designations are expected later in 2013 or early 2014. Of 54 assessed watersheds, 33 and 18 are of high and medium conservation value, respectively. Three were deemed low. Our GIS analysis indicates 13 high and 12 medium conservation value watersheds are exposed to pesticide applications from agriculture and developed areas. All high value and medium value watersheds are exposed to pesticide applications from forested lands adjacent to Lower Columbia River coho salmon habitat.

The LCRBRT identified several activities that affect the PCEs within this designated area. They include agriculture, channel modifications/diking, forestry, irrigation impoundments and withdrawals, road building/maintenance, and urbanization. Much of the agricultural practices are in high conservation value watersheds supporting core populations of coho that are critical for the recovery of the species. All core areas designated as proposed critical habitat had significant forest uses of the land.

Overall, the likelihood each of the a.i.s will cause adverse effects on critical habitat PCEs is high. Use sites are distributed throughout the designated proposed critical habitat of the LCR coho salmon ESU. We anticipate significant areas of core population proposed designated critical habitat will be affected. Moreover, we anticipate the a.i.s input to habitat will occur multiple times due to repeated applications, and/or input from multiple sources. Given landuses within much of the designated proposed critical habitat, we also anticipate exposure to other stressors which exacerbate the effects of the a.i. We anticipate the effects from the proposed action to appreciably reduce the conservation value of the designated critical habitat by adversely affecting water quality and adversely affecting salmonid prey as discussed in the *Risk Characterization* for Designated Critical Habitat section (Table 151).

Table 151. Lower Columbia River coho salmon proposed designated critical habitat

Pesticide	Co-occurrence				Designated critical habitat
	Crop	Non-crop	Developed	Undeveloped	Likelihood to Appreciably reduce conservation values
Diflubenzuron	Yes	Yes	Yes	Yes	High
Fenbutatin oxide	Yes	No	Yes*	No	High
Propargite	Yes	No	Yes*	No	High

*In developed lands the only labeled use is for nurseries.

Oregon Coast (OC) Coho Salmon designated critical habitat

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. All 45 of the watersheds are exposed to pesticides from forested areas. Of the medium conservation watersheds, 18 and 23 are exposed to pesticide applications from the above respective land uses. All 27 watersheds are exposed to pesticides from forested areas. Of the 8 low conservation value watersheds, 2 are exposed to pesticide applications from agricultural and 4 are exposed to pesticide applications from urban land uses. All eight are exposed to pesticide applications from forests.

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. 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.

Overall, the likelihood fenbutatin-oxide and propargite will cause adverse effects on critical habitat PCEs is low. There is minimal agriculture in most of the watersheds in the ESU/DPS and/or it is not located adjacent to salmon-bearing waters. We expect the likelihood diflubenzuron will cause adverse effects on critical habitat PCEs to be High (Table 152). This stems from the tremendous amount of forested lands upon which this a.i. is proposed to be approved for. We also anticipate some input in urban/developed areas and from rights-of-way located adjacent to salmon-bearing waters.

We believe designated critical habitat for Oregon Coast coho salmon will be minimally affected by fenbutatin-oxide and propargite. We do not believe the proposed action for these two a.i.s will appreciably reduce the conservation value of this species' designated critical habitat (Table 152).

This is not the case for diflubenzuron. We anticipate the effects from approving the use of diflubenzuron on forest lands will appreciably reduce the conservation value of the designated critical habitat by adversely affecting water quality and adversely affecting salmonid prey as discussed in the *Risk Characterization* for Designated Critical Habitat section.

Table 152. Oregon Coast coho salmon designated critical habitat

Pesticide	Co-occurrence				Designated critical habitat
	Crop	Non-crop	Developed	Undeveloped	Likelihood to Appreciably reduce conservation values
Diflubenzuron	Yes	Yes	Yes	Yes	High
Fenbutatin oxide	Yes	No	Yes*	No	Low
Propargite	Yes	No	Yes*	No	Low

*In developed lands the only labeled use is for nurseries.

Southern Oregon/Northern California Coast (SONCC) Coho Salmon designated critical habitat

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.

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.

Overall, the likelihood the three a.i.s addressed in this Opinion will cause adverse effects on designated critical habitat PCEs is high. Use sites are distributed throughout the designated critical habitat of the Southern Oregon, Northern California Coast coho salmon ESU. We anticipate most or all designated critical habitat will be affected. We anticipate the a.i.s input to habitat will occur multiple times due to repeated applications, and/or input from multiple sources. Given landuses within the designated critical habitat, we also anticipate exposure to other stressors which exacerbate the effects of the a.i. We anticipate the effects from the proposed action to appreciably reduce the conservation value of the designated critical habitat by adversely affecting water quality and adversely affecting salmonid prey as discussed in the *Risk Characterization* for Designated Critical Habitat section (Table 153).

Table 153. Southern Oregon / Northern California Coast coho salmon designated critical habitat

Pesticide	Co-occurrence				Designated critical habitat
	Crop	Non-crop	Developed	Undeveloped	Likelihood to Appreciably reduce conservation values
Diflubenzuron	Yes	Yes	Yes	Yes	High
Fenbutatin oxide	Yes	No	Yes*	No	High
Propargite	Yes	No	Yes*	No	High

*In developed lands the only labeled use is for nurseries.

Central California Coast (CCC) Coho Salmon designated critical habitat

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.

Much of the development is centered around San Francisco Bay, and developed and agricultural areas also occur in the Russian River watershed. The Russian River is of particular importance for preventing the extinction and contributing to the recovery of CCC coho salmon. 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.

Overall, the likelihood each of the a.i.s will cause adverse effects on critical habitat PCEs is high. Use sites are distributed throughout the designated critical habitat of the Central California Coast Chinook salmon ESU. We anticipate most or all designated critical habitat will be affected. We anticipate the a.i.s input to habitat will occur multiple times due to repeated applications, and/or input from multiple sources. Given landuses within the designated critical habitat, we also anticipate exposure to other stressors which exacerbate the effects of the a.i. We anticipate the effects from the proposed action to appreciably reduce the conservation value of the designated critical habitat by adversely affecting water quality and adversely affecting salmonid prey as discussed in the *Risk Characterization* for Designated Critical Habitat section (Table 154).

Table 154. Central California Coast coho salmon designated critical habitat

Pesticide	Co-occurrence				Designated critical habitat
	Crop	Non-crop	Developed	Undeveloped	Likelihood to Appreciably reduce conservation values
Diflubenzuron	Yes	Yes	Yes	Yes	High
Fenbutatin oxide	Yes	No	Yes*	No	High
Propargite	Yes	No	Yes*	No	High

*In developed lands the only labeled use is for nurseries.

Ozette Lake Sockeye Salmon designated critical habitat

The Ozette Lake watershed is of high conservation value. The entire circumference of the lake is within Olympic National Park. Areas along the Ozette Lake shore and portions of three tributaries support spawning 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, and competition with non-native species. Migration habitat is affected by high water temperatures and low water flows in summer.

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. 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.

As indicated in Table 155, the likelihood each of the a.i.s will cause adverse effects on critical habitat PCEs is Low. While logging/forest uses are in tributary habitats, most of the land in the Ozette Lake Basin is National Park. We anticipate designated critical habitat will be minimally affected. We do not anticipate the effects from the proposed action to appreciably reduce the conservation value of the designated critical habitat of Ozette Lake sockeye salmon.

Table 155. Ozette Lake sockeye salmon designated critical habitat

Pesticide	Co-occurrence				Designated critical habitat
	Crop	Non-crop	Developed	Undeveloped	Likelihood to Appreciably reduce conservation values
Diflubenzuron	Yes	Yes	Yes	Yes	Low
Fenbutatin oxide	Yes	No	Yes*	No	Low
Propargite	Yes	No	Yes*	No	Low

*In developed lands the only labeled use is for nurseries.

Snake River Sockeye Salmon designated critical habitat

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.

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.

Overall, the likelihood each of the a.i.s will cause adverse effects on critical habitat PCEs is low (Table 156). We anticipate most or all designated critical habitat will be minimally affected. We do not anticipate the effects from the proposed action to appreciably reduce the conservation value of the designated critical habitat.

Table 156. Snake River sockeye salmon designated critical habitat

Pesticide	Co-occurrence				Designated critical habitat
	Crop	Non-crop	Developed	Undeveloped	Likelihood to Appreciably reduce conservation values
Diflubenzuron	Yes	Yes	Yes	Yes	Low
Fenbutatin oxide	Yes	No	Yes*	No	Low
Propargite	Yes	No	Yes*	No	Low

*In developed lands the only labeled use is for nurseries.

Puget Sound (PS) Steelhead proposed critical habitat

Critical habitat for Puget Sound (PS) steelhead has been proposed, but not yet designated. Of 70 assessed watersheds (HUC 5), 41 were assigned a high and 18 were assigned a medium conservation value. Our GIS analysis revealed considerable overlap of agricultural, undeveloped, and developed lands with proposed critical habitat that has been assigned a medium or high conservation value (Appendix 4).

Migration, spawning, and rearing PCEs in upper watersheds of most river systems, and in the lower alluvial valleys of mid- to southern Puget Sound and the Strait of Juan de Fuca, have been heavily altered by forestry and agricultural land uses and urban development. These activities have resulted in the loss of floodplain habitat, reduced substrate conditions for spawning and incubation, and degraded water quality. Estuary PCEs throughout 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 channels. Elevated water temperatures in this ESU further degrade about 374 km of streams, which make suitable PCE conditions in remaining proposed critical habitat even more important for ensuring long-term species conservation. Cultivated crops, including hay and pastures (5%) are primarily distributed on the floodplain and other lowland habitats typical of areas where juvenile steelhead seek refuge and forage. Heavily developed urban and residential areas are concentrated within river and stream valleys such as the Green-Duwamish, Snohomish, and Puyallup, as well as along the Puget Sound shoreline.

Due to the overlap of proposed critical habitat with labeled uses of the a.i.s, substantial decreases in prey resources and degradation of water quality is anticipated in this ESU. Overall, the likelihood each of the a.i.s will cause adverse effects on the prey base and water quality in proposed critical habitat PCEs is high (Table 157). We conclude that significant portions of proposed critical habitat will be affected and that the proposed action will appreciably reduce the conservation value of the designated critical habitat by adversely degrading water quality and reducing salmonid prey.

Table 157. Puget Sound steelhead proposed critical habitat

Pesticide	Co-occurrence				Proposed critical habitat
	Crop	Non-crop	Developed	Undeveloped	Likelihood to appreciably reduce conservation value
Diflubenzuron	Yes	Yes	Yes	Yes	High
Fenbutatin oxide	Yes	No	Yes*	No	High
Propargite	Yes	No	Yes*	No	High

*In developed lands the only labeled use is for nurseries.

Lower Columbia River (LCR) Steelhead designated critical habitat

Of the 41 watersheds that are designated critical habitat for LCR steelhead, 28 and 11 are of high and medium conservation value, respectively. Our GIS analysis revealed substantial overlap of agricultural, undeveloped, and developed lands with designated critical habitat that has been assigned a medium or high conservation value.

Freshwater migration, spawning, and rearing PCEs in the Lower Columbia River ESU have been heavily altered by agricultural land uses and urban development. These activities have resulted in the loss of floodplain habitat, reduced substrate conditions for spawning and incubation, reduced prey availability, and degraded water quality. Elevated water temperatures in this ESU further degrade PCEs and functionally exclude steelhead from about 342 km of streams, which make suitable PCE conditions in remaining proposed critical habitat even more important for ensuring long-term species conservation. Cultivated crops, including hay and pastures (7%) are primarily distributed on the floodplain and other lowland habitats typical of areas where juvenile steelhead seek refuge and forage. Heavily developed urban and residential areas are concentrated within river and stream valleys. 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 prey production in these aquatic systems also is expected to be diminished.

Overall, the likelihood each of the a.i.s will cause adverse effects on the prey base and water quality in designated critical habitat PCEs is high (Table 158). Because use sites are distributed throughout the ESU, we expect most of the LCR designated critical habitat to be affected. Exposure to the stressors of the action is anticipated to produce substantial decreases in prey resources and degradation of water quality. Therefore, NMFS concludes that the proposed action will appreciably reduce the conservation value of the designated critical habitat by adversely degrading water quality and reducing salmonid prey.

Table 158. Lower Columbia River steelhead designated critical habitat

Pesticide	Co-occurrence				Designated critical habitat
	Crop	Non-crop	Developed	Undeveloped	Likelihood to Appreciably reduce conservation values
Diflubenzuron	Yes	Yes	Yes	Yes	High
Fenbutatin oxide	Yes	No	Yes*	No	High
Propargite	Yes	No	Yes*	No	High

*In developed lands the only labeled use is for nurseries.

Upper Willamette River (UWR) Steelhead designated critical habitat

In UWR designated critical habitat, 14 watersheds have a high conservation value and 6 have a medium conservation value. Our GIS analysis revealed substantial overlap of agricultural, undeveloped, and developed lands with proposed critical habitat that has been assigned a medium or high conservation value. All watersheds showed overlap with agricultural land uses, where all three a.i.s have authorized uses.

Freshwater spawning, rearing, and migration PCEs in the Upper Willamette River ESU have been heavily altered by agricultural land uses and urban development. These activities have resulted in the loss of floodplain habitat, reduced substrate conditions for spawning and incubation, and degraded water quality. Migration PCEs are adversely affected by several dams that obstruct migrating juveniles and adults along their migratory corridor. Existing water quality necessary for PCEs within many watersheds has been impaired by pollutants in agricultural runoff. Consequently, invertebrate prey production and abundance, as well as water quality, in watersheds and the mainstem Columbia River may be effected. Elevated water temperature further prevents UWR steelhead from inhabiting 1,668 km of waters within its range.

Overall, the likelihood each of the a.i.s will cause adverse effects on the prey base and water quality in designated critical habitat PCEs is high (Table 159). Because use sites are distributed throughout the ESU, and all watersheds overlap with agricultural land uses, we expect most of the UWR designated critical habitat to be affected. Exposure to the stressors of the action is anticipated to produce substantial decreases in prey resources and degradation of water quality. Therefore, NMFS concludes that the proposed action will appreciably reduce the conservation value of UWR designated critical habitat by adversely degrading water quality and reducing salmonid prey.

Table 159. Upper Willamette River steelhead designated critical habitat

Pesticide	Co-occurrence				Designated critical habitat
	Crop	Non-crop	Developed	Undeveloped	Likelihood to Appreciably reduce conservation values
Diflubenzuron	Yes	Yes	Yes	Yes	High
Fenbutatin oxide	Yes	No	Yes*	No	High
Propargite	Yes	No	Yes*	No	High

*In developed lands the only labeled use is for nurseries.

Middle Columbia River (MCR) Steelhead designated critical habitat

Of the 106 watersheds in Middle Columbia River designated critical habitat, 73 watersheds have a high conservation value and 24 have a medium conservation value. Our GIS analysis revealed considerable overlap of agricultural, undeveloped, and developed lands with proposed critical habitat that has been assigned a medium or high conservation value. Several watersheds showed overlap with agricultural land uses, where all three a.i.s have authorized uses. A large proportion of this ESU is comprised of undeveloped lands, included forested areas, on which diflubenzuron has permitted uses. The lower Columbia River rearing/migration corridor downstream of the spawning range is also of high conservation value. Several dams in this ESU have further altered freshwater habitats.

Freshwater spawning, rearing, and migration PCEs in the Middle Columbia River ESU have been heavily altered by agricultural and other land uses. This has led to loss of floodplain habitat, reduced substrate conditions for spawning and incubation, and degraded water quality. Migration PCEs are adversely affected by several dams that obstruct migrating juveniles and adults along their migratory corridor. Existing water quality necessary for PCEs within many watersheds has been impaired by pollutants in agricultural runoff. Elevated water temperature prevents MCR steelhead from inhabiting 3,727.9 km of waters within its range. In the Yakima River alone, 72 streams and river segments are also listed as impaired waters and 83% exceed temperature standards.

Overall, the likelihood each of the a.i.s will cause adverse effects on the prey base and water quality in designated critical habitat PCEs is high (Table 160). Because use sites are distributed throughout the ESU, we expect a substantial proportion of the MCR designated critical habitat to be affected. Exposure to the stressors of the action is anticipated to produce substantial decreases in prey resources and degradation of water quality. Therefore, NMFS concludes that the proposed action will appreciably reduce the conservation value of MCR designated critical habitat by adversely degrading water quality and reducing salmonid prey.

Table 160. Middle Columbia River steelhead designated critical habitat

Pesticide	Co-occurrence				Designated critical habitat
	Crop	Non-crop	Developed	Undeveloped	Likelihood to Appreciably reduce conservation values
Diflubenzuron	Yes	Yes	Yes	Yes	High
Fenbutatin oxide	Yes	No	Yes*	No	High
Propargite	Yes	No	Yes*	No	High

*In developed lands the only labeled use is for nurseries.

Upper Columbia River (UCR) Steelhead designated critical habitat

The Upper Columbia River designated critical habitat is comprised of 41 watersheds, of which 31 are of high conservation value and 7 are of medium conservation value. The UCR rearing/migration corridor downstream of the species' spawning range is also of high conservation value. Our GIS analysis showed considerable overlap of these land uses where use of the three a.i.s is permitted with designated critical habitat that has been assigned a high or medium conservation value. Several watersheds showed overlap with agricultural land uses, where all three a.i.s have authorized uses. A large proportion of this ESU is comprised of undeveloped lands, included forested areas, on which diflubenzuron has permitted 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. Agriculture land uses overlap with spawning and rearing areas in the Wenatchee, Methow, and Okenogan watersheds. Intense agriculture occurs in the Upper Columbia Irrigation District within the Entiat watershed, where river water is heavily used and re-used for irrigation.

Overall, the likelihood each of the a.i.s will cause adverse effects on the prey base and water quality in designated critical habitat PCEs is high (Table 161). Because use sites are distributed throughout the ESU, we expect a substantial proportion of the UCR designated critical habitat to be affected. Exposure to the stressors of the action is anticipated to produce substantial decreases in prey resources and degradation of water quality. Therefore, NMFS concludes that the proposed action will appreciably reduce the conservation value of UCR designated critical habitat by adversely degrading water quality and reducing salmonid prey.

Table 161. Upper Columbia River steelhead designated critical habitat

Pesticide	Co-occurrence				Designated critical habitat
	Crop	Non-crop	Developed	Undeveloped	Likelihood to Appreciably reduce conservation values
Diflubenzuron	Yes	Yes	Yes	Yes	High
Fenbutatin oxide	Yes	No	Yes*	No	High
Propargite	Yes	No	Yes*	No	High

*In developed lands the only labeled use is for nurseries.

Snake River (SR) Basin Steelhead designated critical habitat

The Snake River Basin designated critical habitat is comprised of nearly 300 watersheds. Of these, 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 many high and medium conservation value watersheds overlap with agriculture and urban lands where the three a.i.s have permitted 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. 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. A large proportion of this ESU is comprised of undeveloped lands, including forested areas, on which diflubenzuron has permitted uses.

Overall, the likelihood each of the a.i.s will cause adverse effects on the prey base and water quality in SR designated critical habitat PCEs is high (Table 162). Because use sites are distributed throughout the ESU, we expect a substantial proportion of the SR designated critical habitat to be affected. Exposure to the stressors of the action is anticipated to produce substantial decreases in prey resources and degradation of water quality. Therefore, NMFS concludes that the proposed action will appreciably reduce the conservation value of SR designated critical habitat by adversely degrading water quality and reducing salmonid prey.

Table 162. Snake River Basin steelhead designated critical habitat

Pesticide	Co-occurrence				Designated critical habitat
	Crop	Non-crop	Developed	Undeveloped	Likelihood to Appreciably reduce conservation values
Diflubenzuron	Yes	Yes	Yes	Yes	High
Fenbutatin oxide	Yes	No	Yes*	No	High
Propargite	Yes	No	Yes*	No	High

*In developed lands the only labeled use is for nurseries.

Northern California (NC) Steelhead designated critical habitat

Of the 50 assessed watersheds in the Northern California (NC) designated critical habitat, 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 minimal overlap of agricultural and developed land uses, but substantial overlap of undeveloped land uses, with watersheds classified as high and medium conservation value.

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 freshwater rearing PCEs . 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. The few areas of agriculture 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. Diflubenzuron is permitted for use on both of those types of undeveloped lands. Additionally, diflubenzuron can be directly applied to certain aquatic areas in California.

Overall, the likelihood that diflubenzuron will cause adverse effects on the prey base and water quality in NC designated critical habitat PCEs is high (Table 163). Because use sites are distributed throughout the ESU, we expect a substantial proportion of the NC designated critical habitat to be affected. Exposure to the stressors of the action is anticipated to produce substantial decreases in prey resources and degradation of water quality. The likelihood that fenbutatin oxide and propargite will cause adverse effects on NC designated critical habitat is low (Table 163) due to the limited overlap of use sites with steelhead habitats. Therefore, NMFS concludes that the use of diflubenzuron will appreciably reduce the conservation value of Northern California designated critical habitat by adversely degrading water quality and reducing salmonid prey. NMFS further concludes that the use of fenbutatin oxide and propargite will not appreciably reduce the conservation value of NC designated critical habitat.

Table 163. Northern California steelhead designated critical habitat

Pesticide	Co-occurrence				Designated critical habitat
	Crop	Non-crop	Developed	Undeveloped	Likelihood to Appreciably reduce conservation values
Diflubenzuron	Yes	Yes	Yes	Yes	High
Fenbutatin oxide	Yes	No	Yes*	No	Low
Propargite	Yes	No	Yes*	No	Low

*In developed lands the only labeled use is for nurseries.

Central California Coast (CCC) Steelhead designated critical habitat

Central California Coast (CCC) designated critical habitat is comprised of 47 watersheds, 19 and 15 of which have high and medium conservation value, respectively. Our GIS analysis shows substantial overlap of agriculture and urban areas with watersheds given a high or medium conservation value. The Russian River is of particular importance for preventing the extinction and contributing to the recovery of CCC steelhead.

Throughout CCC designated critical habitat, habitat conditions and quality have been degraded by a lack of channel complexity, eroded banks, turbid and contaminated water, low summer flows, 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 fine sediments replacing ideally sized spawning gravel. Elevated water temperature and impaired water quality have further reduced the quality, quantity, and function of the rearing PCE within most streams. High densities of crop farming occur throughout the San Joaquin Basin, the Delta, and along the lower Sacramento River, and 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 likely experience reductions in water quality and prey abundance during allowable pesticide applications adjacent to these systems.

Overall, the likelihood each of the a.i.s will cause adverse effects on the prey base and water quality in CCC designated critical habitat PCEs is high (Table 164). Because use sites are distributed throughout the ESU, we expect a substantial proportion of the CCC designated critical habitat to be affected. Exposure to the stressors of the action is anticipated to produce substantial decreases in prey resources and degradation of water quality. Therefore, NMFS concludes that the proposed action will appreciably reduce the conservation value of CCC designated critical habitat by adversely degrading water quality and reducing salmonid prey.

Table 164. Central California Coast steelhead designated critical habitat

Pesticide	Co-occurrence				Designated critical habitat
	Crop	Non-crop	Developed	Undeveloped	Likelihood to Appreciably reduce conservation values
Diflubenzuron	Yes	Yes	Yes	Yes	High
Fenbutatin oxide	Yes	No	Yes*	No	High
Propargite	Yes	No	Yes*	No	High

[1]Denotes California specific labels for diflubenzuron (CA-970021)

*In developed lands the only labeled use is for nurseries.

California Central Valley (CCV) Steelhead designated critical habitat

California Central Valley (CCV) designated critical habitat is comprised of 67 occupied watersheds, and of those 37 and 18 have high and medium conservation value, respectively. Our GIS analysis indicates tremendous overlap of agricultural land uses with watersheds of high or medium conservation value.

The current condition of CCV steelhead designated critical habitat is degraded and does not function well for ensuring species recovery. Due to degraded water quality, reduced prey resources, and altered physical habitats, the Sacramento-San Joaquin River Delta serves little function for juvenile CCV steelhead rearing and their physiological transition to salt water. Reduced water flow and elevated temperature, especially during the summer months, degrade the condition of the spawning PCE in floodplains and other shallow freshwaters. The rearing PCE is degraded by channelized, leveed, and riprapped river reaches and sloughs in the Sacramento-San Joaquin system. 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 contaminated runoff. Additionally, diflubenzuron can be applied directly to certain aquatic areas. 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. Heavy uses of agricultural pesticides increases the likelihood of negative effects on PCEs and designated critical habitat.

Given these land uses and current conditions in CCV designated critical habitat, freshwater spawning, rearing, and migration PCEs likely experience reductions in water quality and prey abundance during allowable pesticide applications adjacent to these systems. Overall, the likelihood each of the a.i.s will cause adverse effects on the prey base and water quality in CCV designated critical habitat PCEs is high (Table 165). Because use sites are distributed throughout the ESU, we expect a substantial proportion of CCV designated critical habitat to be affected. Exposure to the stressors of the action is anticipated to produce substantial decreases in prey resources and degradation of water quality. Therefore, NMFS concludes that the proposed action will appreciably reduce the conservation value of CCV designated critical habitat by adversely degrading water quality and reducing salmonid prey.

Table 165. California Central Valley steelhead designated critical habitat

Pesticide	Co-occurrence				Designated critical habitat
	Crop	Non-crop	Developed	Undeveloped	Likelihood to Appreciably reduce conservation values
Diflubenzuron	Yes	Yes	Yes	Yes	High
Fenbutatin oxide	Yes	No	Yes*	No	High
Propargite	Yes	No	Yes*	No	High

*In developed lands the only labeled use is for nurseries.

South-Central California Coast (S-CCC) Steelhead designated critical habitat

There are 29 occupied watersheds in South-Central California Coast (S-CCC) designated critical habitat. Of these, 12 have high conservation value and 11 have medium conservation value. Our GIS analysis indicates considerable overlap of agricultural and urban land uses with watersheds with high or medium conservation value.

Throughout S-CCC designated critical habitat, habitat conditions and quality have been degraded by turbid and contaminated water, low summer flows, high water temperatures, multiple contaminants found at toxic levels, and reduced prey resources. 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. Agriculture is the dominant land use in the Salinas River valley and the Pajaro watershed. Dense urban areas are located along river valleys and along the coast, impacting freshwater and estuarine PCEs. Given these land uses and current conditions in S-CCC designated critical habitat, freshwater rearing, migration, and estuarine PCEs likely experience reductions in water quality and prey abundance during allowable pesticide applications adjacent to these systems.

Overall, the likelihood each of the a.i.s will cause adverse effects on the prey base and water quality in S-CCC designated critical habitat PCEs is high (Table 166). Because use sites are distributed throughout the ESU, we expect a substantial proportion of the S-CCC designated critical habitat to be affected. Exposure to the stressors of the action is anticipated to produce substantial decreases in prey resources and degradation of water quality. Therefore, NMFS concludes that the proposed action will appreciably reduce the conservation value of S-CCC designated critical habitat by adversely degrading water quality and reducing salmonid prey.

Table 166. South Central California Coast steelhead designated critical habitat

Pesticide	Co-occurrence				Designated critical habitat
	Crop	Non-crop	Developed	Undeveloped	Likelihood to Appreciably reduce conservation values
Diflubenzuron	Yes	Yes	Yes	Yes	High
Fenbutatin oxide	Yes	No	Yes*	No	High
Propargite	Yes	No	Yes*	No	High

*In developed lands the only labeled use is for nurseries.

Southern California (SC) Steelhead designated critical habitat

Southern California steelhead designated critical habitat is composed of 29 freshwater and estuarine watersheds. Of these, 21 and 5 are of high and medium conservation value, respectively. SC designated critical habitat is at the extreme southern limit of steelhead's range. Our GIS analysis shows overlap of land uses where the three a.i.s have allowed uses with watersheds that have high or medium conservation value.

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. Urban land uses are found throughout this ESU, but overlap with high and medium conservation value watersheds in the northern end of SC habitat. Cultivated crop lands are concentrated in the 4 major river valleys (Santa Maria, Santa Ynez, Ventura, and Santa Clara rivers), and pesticides in runoff contribute to the degradation of freshwater PCEs by reducing prey resources and water quality. Freshwater PCEs are affected by diflubenzuron because it can be used directly on certain aquatic areas.

Overall, the likelihood each of the a.i.s will cause adverse effects on the prey base and water quality in SC designated critical habitat PCEs is high (Table 167). Because use sites are distributed throughout the ESU, we expect a substantial proportion of the SC designated critical habitat to be affected. Exposure to the stressors of the action is anticipated to produce substantial decreases in prey resources and degradation of water quality. Therefore, NMFS concludes that the proposed action will appreciably reduce the conservation value of SC designated critical habitat by adversely degrading water quality and reducing salmonid prey.

Table 167. Southern California steelhead designated critical habitat

Pesticide	Co-occurrence				Designated critical habitat
	Crop	Non-crop	Developed	Undeveloped	Likelihood to Appreciably reduce conservation values
Diflubenzuron	Yes	Yes	Yes	Yes	High
Fenbutatin oxide	Yes	No	Yes*	No	High
Propargite	Yes	No	Yes*	No	High

*In developed lands the only labeled use is for nurseries.

Table 168. Summary of potential for appreciable reductions in conservation value of designated and proposed critical habitat from exposure to the three a.i.s.

Species	ESU/DPS	Diflubenzuron	Fenbutatin-Oxide	Propargite
Chinook	Puget Sound	High	High	High
	Lower Columbia River	High	High	High
	Upper Columbia River Spring - Run	High	High	High
	Snake River Fall - Run	Medium	Medium	Medium
	Snake River Spring/Summer - Run	High	High	High
	Upper Willamette River	High	High	High
	California Coastal	High	High	High
	Central Valley Spring - Run	High	High	High
	Sacramento River Winter - Run	High	High	High
Chum	Hood Canal Summer - Run	Medium	Low	Low
	Columbia River	Medium	Low	Low
Coho	Lower Columbia River	High	High	High
	Oregon Coast	High	Low	Low
	Southern Oregon and Northern California Coast	High	High	High
	Central California Coast	High	High	High
Sockeye	Ozette Lake	Low	Low	Low
	Snake River	Low	Low	Low
Steelhead	Puget Sound	High	High	High
	Lower Columbia River	High	High	High
	Upper Willamette River	High	High	High
	Middle Columbia River	High	High	High
	Upper Columbia River	High	High	High
	Snake River	High	High	High
	Northern California	High	Low	Low
	Central California Coast	High	High	High
	California Central Valley	High	High	High
	South-Central California Coast	High	High	High
	Southern California	High	High	High

14 Conclusion

NMFS held several meetings with EPA, applicants and USDA throughout the Section 7 consultation (summarized in the Consultation History). The meetings increased NMFS understanding of the proposed actions. The meetings also provided applicants an opportunity to review and make changes to pesticide labels. Changes to labels included clarifications such as providing use rates that were not specified, or where label language was not consistent across labels. Other label changes were more substantial in reducing potential loading to salmonid habitats. For example, Chemtura proposed reducing propargite application rate used on walnuts from 4.5 lbs/acre to 3.2 lbs/acre. Other notable changes included: UPI proposed eliminating all aerial applications of fenbutatin oxide. Chemtura proposed eliminating diflubenzuron direct applications to aquatic habitats for mosquito and midge control as well as cancelling use on mushrooms in the four states where Pacific salmonids reside. We incorporated these and other changes offered by the applicants into the Description of the Action. We conducted our analyses based on the resulting action that reflected the label changes.

For each of the three pesticides, we present lines of evidence and their relative strength (Figure 82, Figure 83, and Figure 84). For each line of evidence, we indicate the strength of the relationship by showing one of three types of arrows. A bold arrow indicates a low level of uncertainty and a high level of confidence. A non-bold arrow indicates a moderate level of uncertainty and a moderate level of confidence. A dashed arrow indicates a high level of uncertainty and a low level of confidence. A “yes,” “no,” or a question mark was assigned based on whether population and species effects are anticipated.

Diflubenzuron findings Figure 82

Effects to salmonids from short-term exposures appear unlikely; including effects to salmonid survival and growth. Reductions in salmonid growth from reductions in prey resources appear highly likely. This line of evidence supports that individuals, populations, and species are impacted from diflubenzuron’s effect on prey communities. Exposed populations likely experience reductions in abundance and productivity which translates to reducing reproduction, numbers, or distribution of the ESU/DPS. Impacts to salmonid prey are expected to appreciably reduce conservation values of proposed and designated critical habitat. We found no information on several lines of evidence as indicated by question marks.

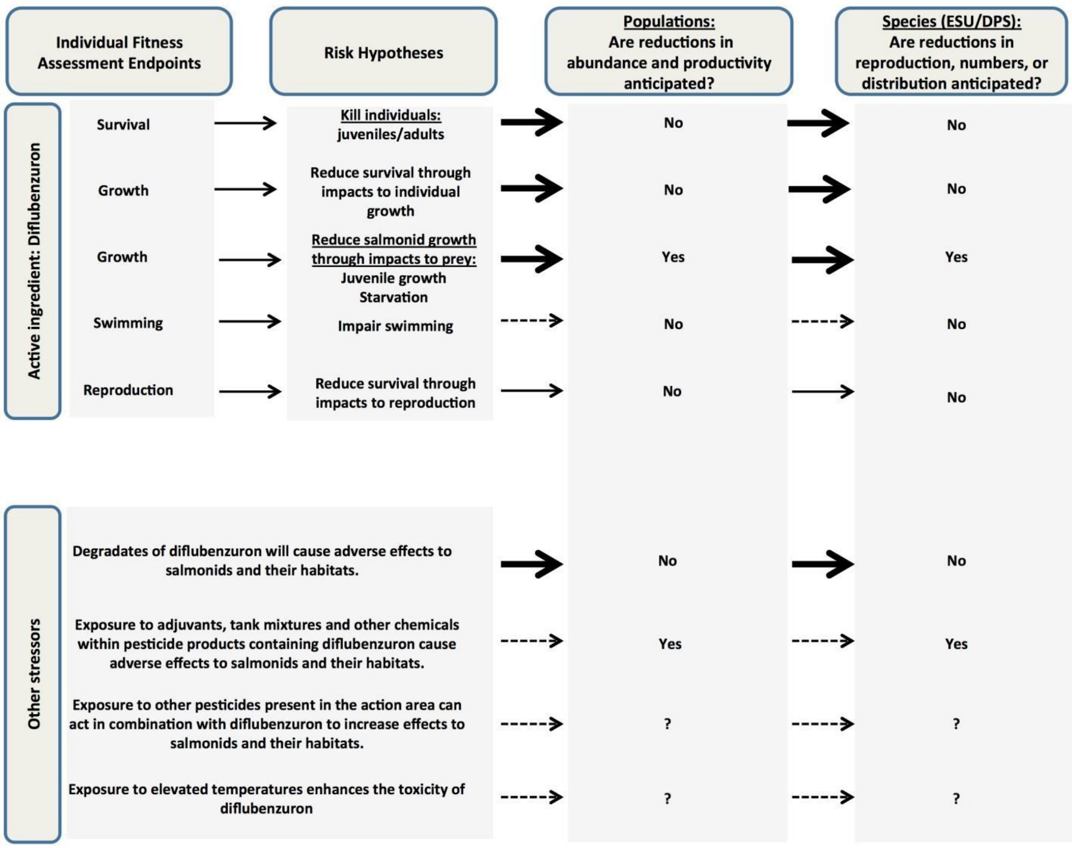

Figure 82. Summary of lines of evidence for diflubenzuron

Fenbutatin oxide Figure 83

Fenbutatin-oxide is extremely and directly toxic to Pacific salmonids and their prey following short term exposures. Robust evidence supported multiple lines of evidence from individual to species levels including effects to survival, growth, and reproduction. We also anticipate that reductions in prey abundance from exposure to fenbutatin oxide are sufficient to appreciably reduce the conservation values of proposed and designated critical habitat. We found no information on several lines of evidence as indicated by question marks.

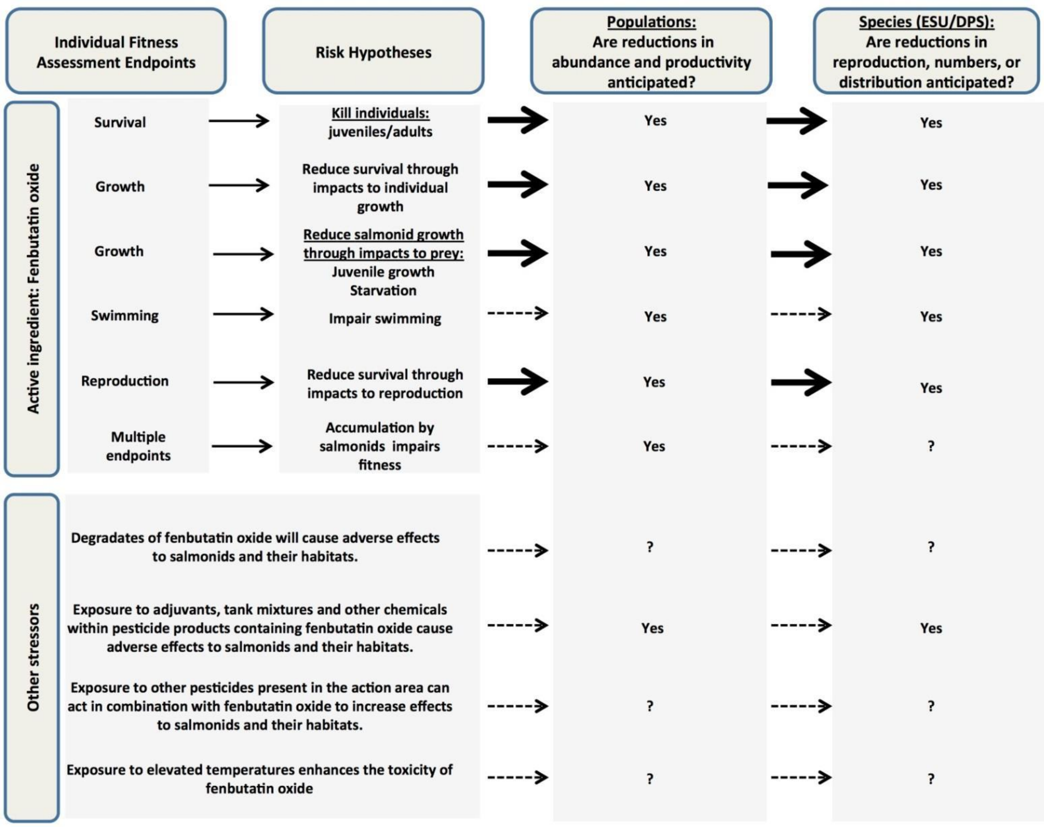

Figure 83. Summary of lines of evidence for fenbutatin-oxide

Propargite Findings (Figure 84)

Multiple lines of evidence were supported by the exposure and response information reviewed in this Opinion. Propargite is very toxic to salmonids and their prey. We anticipate that propargite exposure to authorized uses are sufficient to directly kill salmonids leading to reductions in abundance and productivity of populations that ultimately reduce reproduction, numbers, or distribution at the species level. We anticipate that reductions in prey abundance are sufficient to appreciably reduce the conservation values of proposed and designated critical habitat.

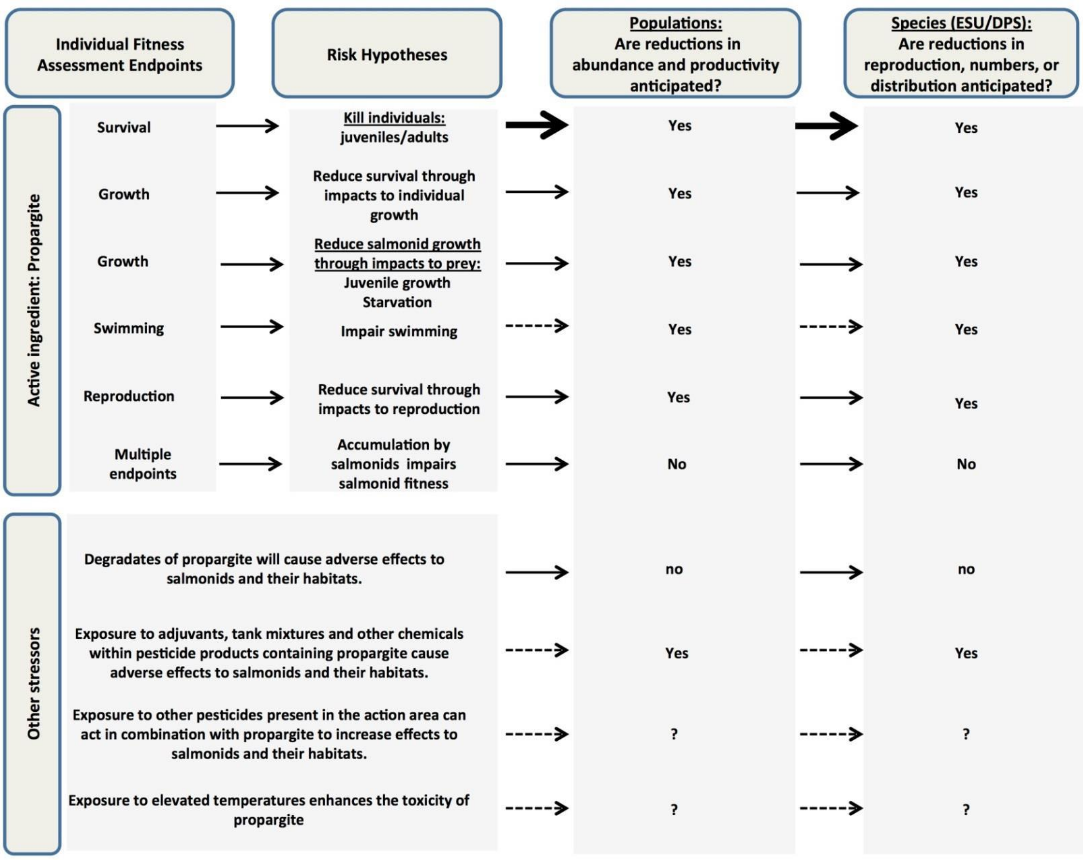

Figure 84. Summary of lines of evidence for propargite

Based on the best available information on potential exposure and effects to each species and designated critical habitat, we determine if the proposed action is likely to jeopardize the species or cause destruction or adversely modify designated critical habitat, respectively. In the *Integration and Synthesis of Effects to Listed Species* section, and *Integration and Synthesis of Effects to Proposed and Designated Critical Habitat* section, we assessed the likelihood of negative effects posed to the survival and recovery of listed Pacific salmonids and their critical habitat as a result of EPA's registration of diflubenzuron, fenbutatin oxide, and propargite. The likelihood of effects assigned to each ESU/DPS for each a.i. reflects NMFS' evaluations of the likelihood that a compound will cause reductions in species viability, and reductions in the conservation values of their critical habitat. We expect diflubenzuron, fenbutatin-oxide, and propargite will have an adverse effect on most listed salmonids and their habitat. For many ESU/DPSs, the effects may be extensive enough to rise to the level of jeopardy and adverse modification, and for some ESU/DPSs the effects may not. Therefore, we find that EPA cannot ensure that its authorization will avoid jeopardy, or will avoid destruction or adverse modification of designated critical habitats. This is primarily a function of the extent of registered use sites in the ESU/DPS that are superimposed on watersheds supporting key populations and their habitat deemed essential for the recovery and eventual delisting of the species. Final jeopardy determinations for each ESU/DPS are summarized in Table 169. Final adverse modification determinations are summarized in Table 170.

Table 169. Species jeopardy determinations

Species	ESU/DPS	Active Ingredient		
		Diflubezuron	Fenbutatin oxide	Propargite
Chinook	Puget Sound	Jeopardy	Jeopardy	Jeopardy
	Lower Columbia River	Jeopardy	Jeopardy	Jeopardy
	Upper Columbia River Spring Run	Jeopardy	Jeopardy	Jeopardy
	Snake River Fall Run	No	No	No
	Snake River Spring/Summer Run	Jeopardy	Jeopardy	Jeopardy
	Upper Willamette River	Jeopardy	Jeopardy	Jeopardy
	California Coastal	Jeopardy	Jeopardy	Jeopardy
	Central Valley Spring Run	Jeopardy	Jeopardy	Jeopardy
	Sacramento River Winter Run	Jeopardy	Jeopardy	Jeopardy
Chum	Hood Canal Summer Run	No	No	No
	Columbia River	No	No	No
Coho	Lower Columbia River	Jeopardy	Jeopardy	Jeopardy
	Oregon Coast	Jeopardy	No	No
	Southern Oregon and Northern California Coast	Jeopardy	Jeopardy	Jeopardy
	Central California	Jeopardy	Jeopardy	Jeopardy
Sockeye	Ozette Lake	No	No	No
	Snake River	No	No	No
Steelhead	Puget Sound	Jeopardy	Jeopardy	Jeopardy
	Lower Columbia River	Jeopardy	Jeopardy	Jeopardy
	Upper Willamette River	Jeopardy	Jeopardy	Jeopardy
	Middle Columbia River	Jeopardy	Jeopardy	Jeopardy
	Upper Columbia River	Jeopardy	Jeopardy	Jeopardy
	Snake River	Jeopardy	Jeopardy	Jeopardy
	Northern California	Jeopardy	No	No
	Central California	Jeopardy	Jeopardy	Jeopardy
	California Central Valley	Jeopardy	Jeopardy	Jeopardy
	South Central California Coast	Jeopardy	Jeopardy	Jeopardy
	Southern California	Jeopardy	Jeopardy	Jeopardy

Table 170. Adverse modification determinations

Species	ESU/DPS	Active ingredient		
		Diflubenzuron	Fenbutatin oxide	Propargite
Chinook	Puget Sound	Adverse modification	Adverse modification	Adverse modification
	Lower Columbia River	Adverse modification	Adverse modification	Adverse modification
	Upper Columbia River Spring - Run	Adverse modification	Adverse modification	Adverse modification
	Snake River Fall - Run	No	No	No
	Snake River Spring/Summer - Run	Adverse modification	Adverse modification	Adverse modification
	Upper Willamette River	Adverse modification	Adverse modification	Adverse modification
	California Coastal	Adverse modification	Adverse modification	Adverse modification
	Central Valley Spring - Run	Adverse modification	Adverse modification	Adverse modification
	Sacramento River Winter - Run	Adverse modification	Adverse modification	Adverse modification
Chum	Hood Canal Summer - Run	No	No	No
	Columbia River	No	No	No
Coho	Lower Columbia River	Adverse modification	Adverse modification	Adverse modification
	Oregon Coast	Adverse modification	No	No
	Southern Oregon and Northern California Coast	Adverse modification	Adverse modification	Adverse modification
	Central California Coast	Adverse modification	Adverse modification	Adverse modification
Sockeye	Ozette Lake	No	No	No
	Snake River	No	No	No
Steelhead	Puget Sound	Adverse modification	Adverse modification	Adverse modification
	Lower Columbia River	Adverse modification	Adverse modification	Adverse modification
	Upper Willamette River	Adverse modification	Adverse modification	Adverse modification
	Middle Columbia River	Adverse modification	Adverse modification	Adverse modification
	Upper Columbia River	Adverse modification	Adverse modification	Adverse modification
	Snake River	Adverse modification	Adverse modification	Adverse modification
	Northern California	Adverse modification	No	No
	Central California Coast	Adverse modification	Adverse modification	Adverse modification
	California Central Valley	Adverse modification	Adverse modification	Adverse modification
	South-Central California Coast	Adverse modification	Adverse modification	Adverse modification
	Southern California	Adverse modification	Adverse modification	Adverse modification

15 Reasonable and Prudent Alternatives

When NMFS concludes that an action is likely to jeopardize listed species or destroy or adversely modify critical habitat, NMFS suggests a reasonable and prudent alternative (RPA) that would allow the action to proceed in compliance with section 7(a)(2) and that can be taken by the action agency and the applicant (ESA Section 7(a)(3)(A)). Joint NMFS and U.S. Fish and Wildlife Service regulations (50 CFR §402.02) implementing section 7 define "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). As noted above, NMFS relies on statutory language to determine adverse modification.

The Services' implementing regulations 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 (50 CFR §402.02). The overarching requirement is that an RPA must be capable of avoiding jeopardizing listed species and adversely modifying critical habitat – all other elements of the definition must be evaluated within this context (*Greenpeace v. NMFS*, 55 F. Supp. 2d 1248, 1268 (W.D. Wa. 1999)). The Services in the preamble to the final section 7 regulations make clear that the overriding consideration is whether a RPA avoids the likelihood of jeopardy. The Services note that the action agency's responsibility "permeates the full range of discretionary authority held by the action agency." Thus, the Services can specify an RPA that involves the maximum exercise of the action agency's authority when the Services deem necessary to avoid the likelihood of jeopardy (51 FR 19926, 19937 (June 3, 1986)).

The other three factors are intended to implement the statutory phrase "can be taken." The third factor, technological and economic feasibility, refers to the ability of the federal agency to implement the RPA: "[t]he requirement that a RPA be 'economically and technologically feasible' only requires that the Corps have the resources and technology necessary to implement the RPA." *In Re: Operation of the Missouri River System Litigation.* 363 F. Supp. 2d 1145, 1161 (D. Minn. 2004), citing *Kandra v. U.S.*, 145 F.Supp. 2d 1192, 1207 (D. Ore.) ("the RPAs must be economically and technically feasible for *the government* to implement."). This regulatory factor was included in the final section 7 implementing regulations in response to a comment, without further explanation or discussion. The ESA contains no requirement for analysis of economic impacts resulting from implementation of a RPA, and the insertion of the phrase "economically feasible" in regulation cannot create this requirement. Any obligation that NMFS "balance the benefit to the species against the economic and technical burden on the industry before approving an RPA would be fundamentally inconsistent with the purposes of the ESA and with case law interpreting the Act." *Greenpeace v. NMFS,* 55 F. Supp. 2d 1248, 1267 (W.D. Wash. 1999). While the Services will defer in most cases to the action agency's expertise as to whether a RPA is reasonable, including whether the RPA is technologically and economically feasible, the Services cannot abdicate their duty to formulate and recommend RPAs (51 FR at 19952). However, the action agency may choose or may be obligated to conduct an economic analysis and to evaluate impacts to interests other than the applicants when it implements a RPA pursuant to its authorities.

In this Opinion, NMFS concluded that EPA's proposed registration of pesticides containing diflubenzuron is likely to jeopardize the continued existence of 23 of the 28 ESUs/DPSs of listed salmonids. We also concluded that EPA's registration of diflubenzuron is likely to adversely modify or destroy designated critical habitat for 23 of 26 ESUs/DPSs; and is likely to adversely modify or destroy proposed designated habitat for 2 of 2 ESUs/DPSs. Additionally, EPA's registrations of fenbutatin oxide and propargite are each likely to jeopardize the continued existence of 21 of the 28 endangered and threatened Pacific salmonid ESUs/DPSs. We also concluded EPA's registration of fenbutatin-oxide and propargite are each likely to destroy or adversely modify designated critical habitat for 21 of the 26 threatened and endangered salmonid ESUs/DPSs, and such registration is likely to adversely modify or destroy proposed designated habitat for 2 of 2 ESUs/DPSs.

NMFS reached these conclusions because predicted concentrations of these three a.i.s are likely to have direct and indirect adverse effects to Pacific salmonids including significant reductions in growth and survival by impairing water quality and salmonid prey production in freshwater rearing, spawning, and migratory habitats, particularly in floodplain habitats[22] and small first and second order streams. NMFS also concluded that the predicted concentrations will have adverse effects to the PCEs of designated critical habitat.

As a result, affected 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 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.

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 chemical compounds may result in greater toxicity; 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 exposure of the stressors of the action to such an extent that effects to species are not anticipated. In the proposed RPA, NMFS does not attempt to ensure there is no take of listed species. NMFS concludes that take will likely 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. The goal of the RPA is to reduce exposure to listed Pacific salmonids to ensure that the action is not likely to jeopardize listed species or destroy or adversely modify critical habitat.

[22] 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).

For each active ingredient, the elements of the RPA selected apply only to those ESUs/DPSs where NMFS has determined that EPA cannot insure that its registration of that a.i. avoids jeopardy or the destruction or adverse modification of critical habitat (Insert tables ref for jeopardy and adverse mod determination from conclusion section). These elements rely upon recognized practices for reducing loading of pesticide products into aquatic habitats.

15.1 Elements of the Reasonable and Prudent Alternative

The RPA is comprised of three elements, of which, at least one must be implemented in its entirety when applying enduse products containing the a.i.s. The RPA must be implemented by EPA within one year of receipt of this Opinion (*i.e.*, June 30 2014), to ensure the registration of the three insecticides 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.

The elements shall be specified on FIFRA labels of all pesticide products containing diflubenzuron, fenbutatin oxide, and propargite. Alternatively, the label could direct pesticide users to the EPA's Endangered Species Protection Program (ESPP) bulletins that specify the elements. 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).

Context and Rationale

In addition to avoiding jeopardy and adverse modification of critical habitat, the following RPA was developed with three intended goals. First, the RPA is intended to reduce loading of harmful pesticide chemicals into salmon habitat. Reduced loading into aquatic systems reduces the likelihood that salmon and/or their critical habitat will be exposed to the stressors of the action. Furthermore, reducing exposure reduces the potential for adverse effects on salmonid health including growth, reproduction, and survival of individuals and populations, as well as adverse effects to critical habitat including areas used for spawning, rearing, and migration. The RPA seeks to reduce loading to the extent that EPA can ensure that jeopardy and adverse modification of critical habitat are avoided while simultaneously allowing applications of registered pesticides.

Second, the RPA aims to incorporate ongoing landowner stewardship efforts in salmonid habitats given those efforts demonstrate reduced loading of the stressors of the action. There are numerous federal, state, and local programs that assist landowners in promoting responsible land management practices and implementing conservation measures that benefit salmonids and critical habitats. Conservation practices including the creation of riparian areas, planting riparian vegetation, shallow wetlands, and conservation buffers have been used to achieve various degrees of habitat protection, species enhancement, and pollution control. If these practices demonstrate reduced loading of the three active ingredients to the extent that jeopardy and adverse modification are avoided, they are acceptable for incorporation into the RPA.

Third, the RPA is intended to protect vulnerable floodplain habitats from the stressors of the action. The RPA is also intended to support current and future restoration and conservation efforts of floodplain habitats, thereby supporting the ultimate recovery of threatened and endangered salmonids. The RPA should be consistent with the emphasis of the ESA to recover protected species, as well as the mission of NOAA Fisheries to manage, conserve, and protect living resources under our jurisdiction. Restoration of salmonid habitats, especially floodplains and other shallow aquatic areas, are essential for salmonid recovery. The more reductions in loading of pesticides into these habitats, the greater the confidence we have that EPA can insure jeopardy and adverse modification of critical habitat are avoided.

Authorized labels state that enduse products containing each of the three active ingredients are toxic to aquatic invertebrates and/or fish. In the *Environmental Hazard* section of pesticide labels, applicators are mandated to keep pesticide enduse products out of aquatic areas from spray drift and runoff and to avoid contaminating water with washwater and rinsate. Based on risk the enduse products comprise to salmonid habitats described in this Opinion, we concur with the need to keep these materials out of aquatic areas that support salmonids and thus structure the RPAs to attain this goal. Surface water monitoring and/or pesticide fate models show that each of the three active ingredients reach salmonid habitats from authorized used at levels where EPA cannot insure that jeopardy and adverse modification of critical habitat are avoided. Therefore, additional economically and technologically feasible restrictions are needed.

We considered site-specific, no-spray buffers as an element of the RPA, however we encountered several aspects that make this approach unfeasible. Ideally, we would be able to apply a no-spray buffer based on site-specific information for each application of the pesticides. Realistically, however, site-specific no-spray buffers require extensive analysis and verification. There may be some aspects of a site that may inform the size of a no-spray buffer such as presence of a functioning riparian area that intercepts both spray drift and runoff. For sites that do not have such an area, further analysis and verification would be needed. Such an analysis includes the following. Prior to application of the stressors of the action site-specific knowledge of aquatic areas potentially contaminated combined with current weather and climactic characteristics are necessary to determine the extent of a no-spray buffer. Aquatic area information would include habitat type, hydrologic parameters (flow rate, depth, volume, width, connectivity, etc.). Monitoring of spray drift before, during, and after application would be necessary to ensure pesticides do not reach the aquatic habitats. Current best available pesticide monitoring practices would include use of spray cards adjacent to aquatic areas paired with surface water sampling verified by laboratory chemical analysis. Pesticide runoff would also need to be monitored following the first and second storm events to ensure off-site transport of pesticides is limited. These measures are not trivial and require expertise in planning, design, execution, analysis, and interpretation all which incur economic resources. EPA would be responsible for ensuring these practices are followed. Overall we found this element to be unfeasible, since it would place an unreasonable technological and/or economic burden on EPA as well as the applicants and end-users.

The three elements below seek to provide flexibility to EPA, applicants, and pesticide end users for use of diflubenzuron, fenbutatin oxide, and propargite.

Element 1.

The tables below represent no-spray buffers required for each of the three active ingredients comprising end-use products (Table 171, Table 172, and Table 173).

Table 171 Required no-spray buffers for aerial and ground applications of end-use products containing diflubenzuron

Aerial applications			Ground applications	
No-spray buffer size (ft)	Application rate (lbs propargite/ acre)		No-spray buffer size (ft)	Application rate (lbs propargite/ acre)
1000	Greater than or equal to 0.125		500	Greater than or equal to 0.3125
500	Less than 0.125		300	Less than 0.3125 and greater than or equal to 0.125
			150	Less than 0.125 and greater than or equal to 0.03125
			75	Less than 0.03125

Table 172 Required no-spray buffers for ground applications of end-use products containing fenbutatin oxide

Aerial applications		Ground applications	
Not applicable, registrant has voluntarily removed this application method from labels.		No-spray buffer size (ft)	Application rate (lbs propargite/ acre)
		500	All authorized rates

Table 173 Required no-spray buffers for aerial and ground applications of end-use products containing propargite

Aerial applications			Ground applications using airblast technologies	
No-spray buffer size (ft)	Application rate (lbs propargite/ acre)		No-spray buffer size (ft)	Application rate (lbs propargite/ acre)
500	Greater than or equal to 2.5		75	Greater than or equal to 2.5
300	Less than 2.5 and greater than or equal to 1.5		50	Less than 2.5
250	Less than 1.5			

<u>Rationale for no-spray buffers</u>:

No-spray buffers are recognized tools to reduce pesticide loading into aquatic habitats *(NRCS 2000)*. NMFS derived no-spray buffers considering all of the qualitative and quantitative information discussed in the Effects of the Proposed Action, including exposure and response data, and the described limitations of their uses and associated uncertainties. EPA's AgDrift model was used to estimate concentrations that would result in habitats of different volumes, at different application rates, and using different application methods. Input parameters for model estimates were consistent with those specified in our exposure analysis (Section 9.1.5.3). These values were compared to both individual and population level toxicological endpoints for the three a.i.s presented earlier in this Opinion. Figure 85Figure 86, and Figure 87 show comparative examples of effects endpoints with exposure estimates for the no-spray buffers in element one. For each no-spray buffer, the range of concentrations predicted based on a range of application rates (based on a.i.) and receiving volumes (2 m wide by from 0.1 m to 1 m deep) is shown.

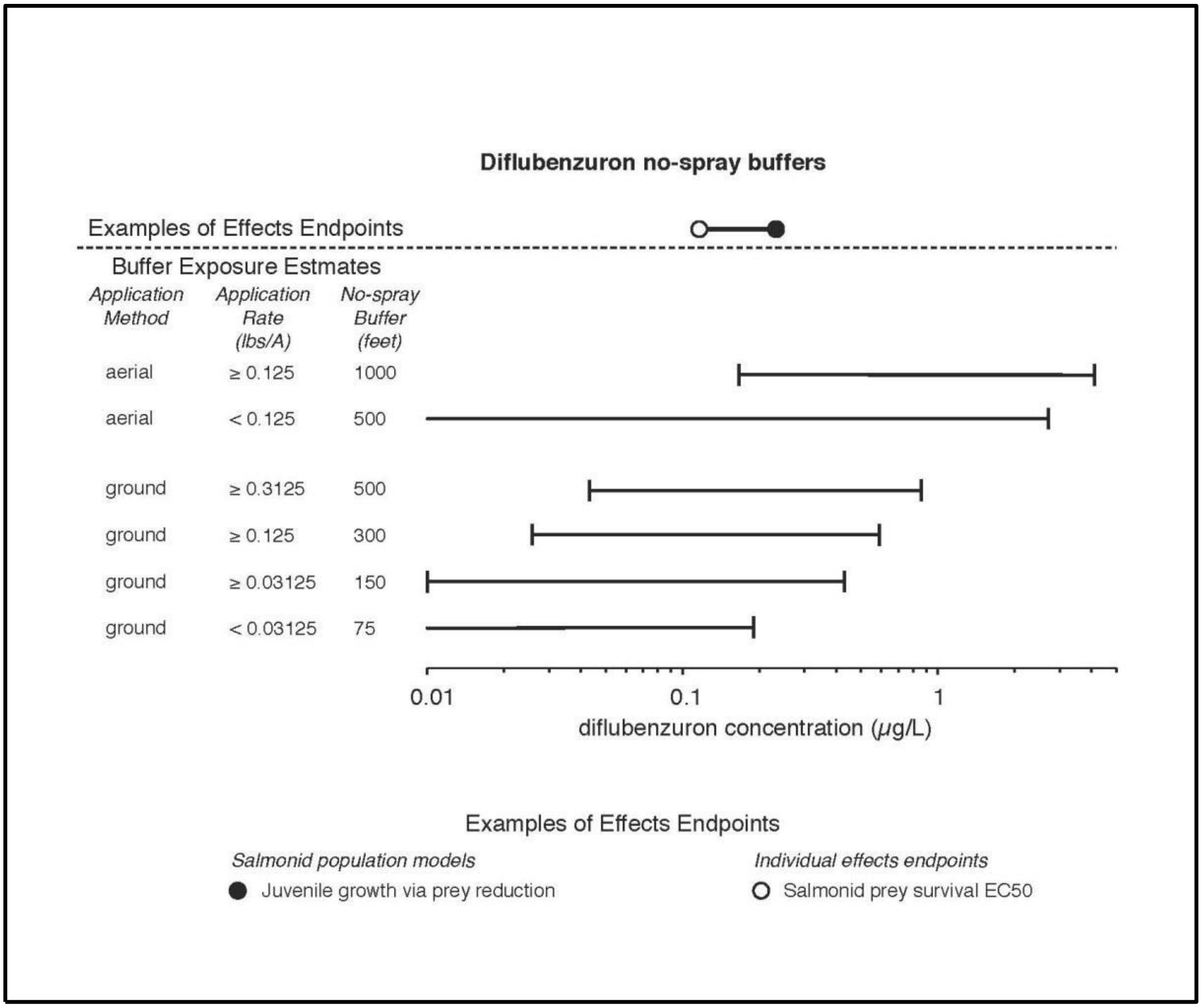

Figure 85. Comparison of diflubenzuron effects endpoints and exposure examples for no-spray buffers.

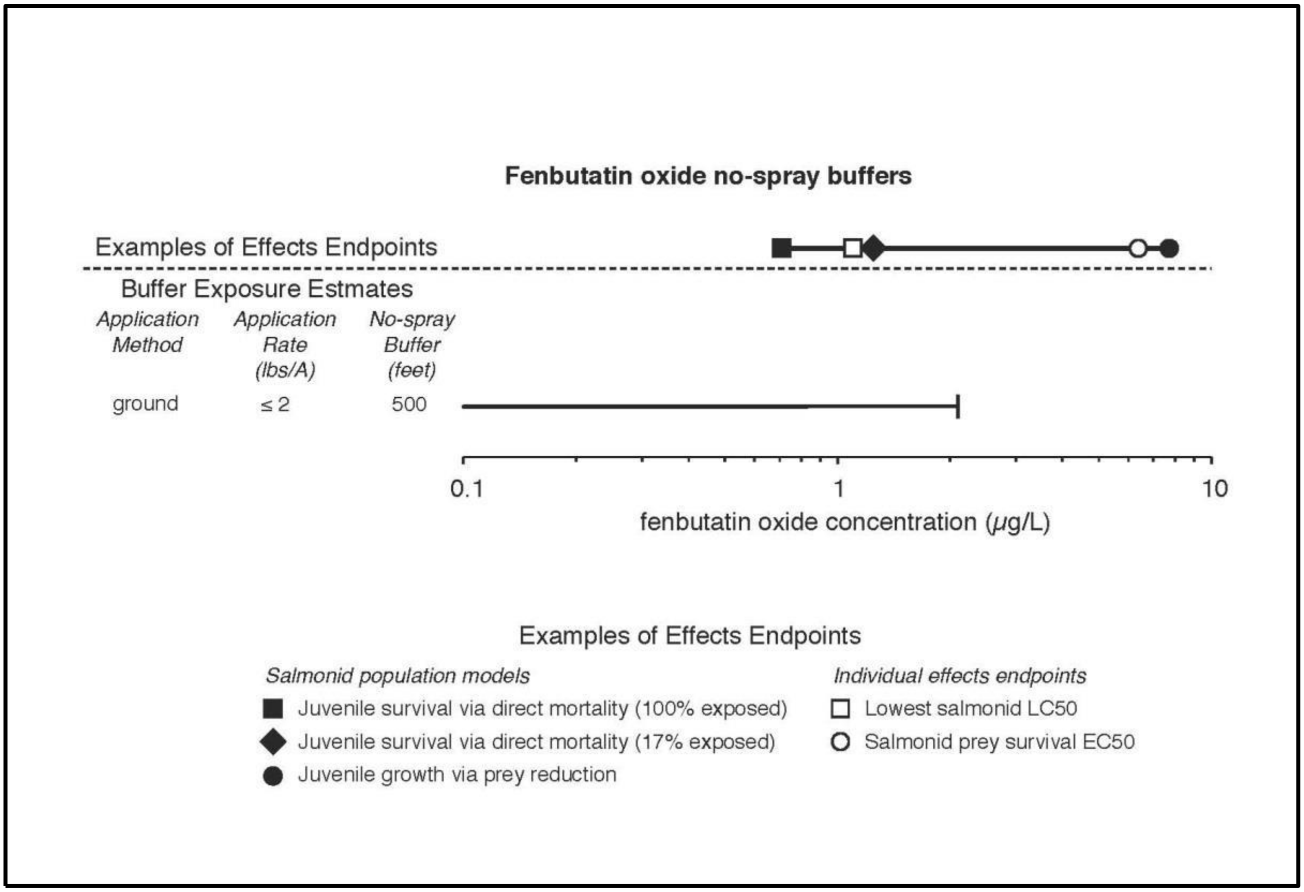

Figure 86. Comparison of fenbutatin oxide effects endpoints and exposure examples for no-spray buffers.

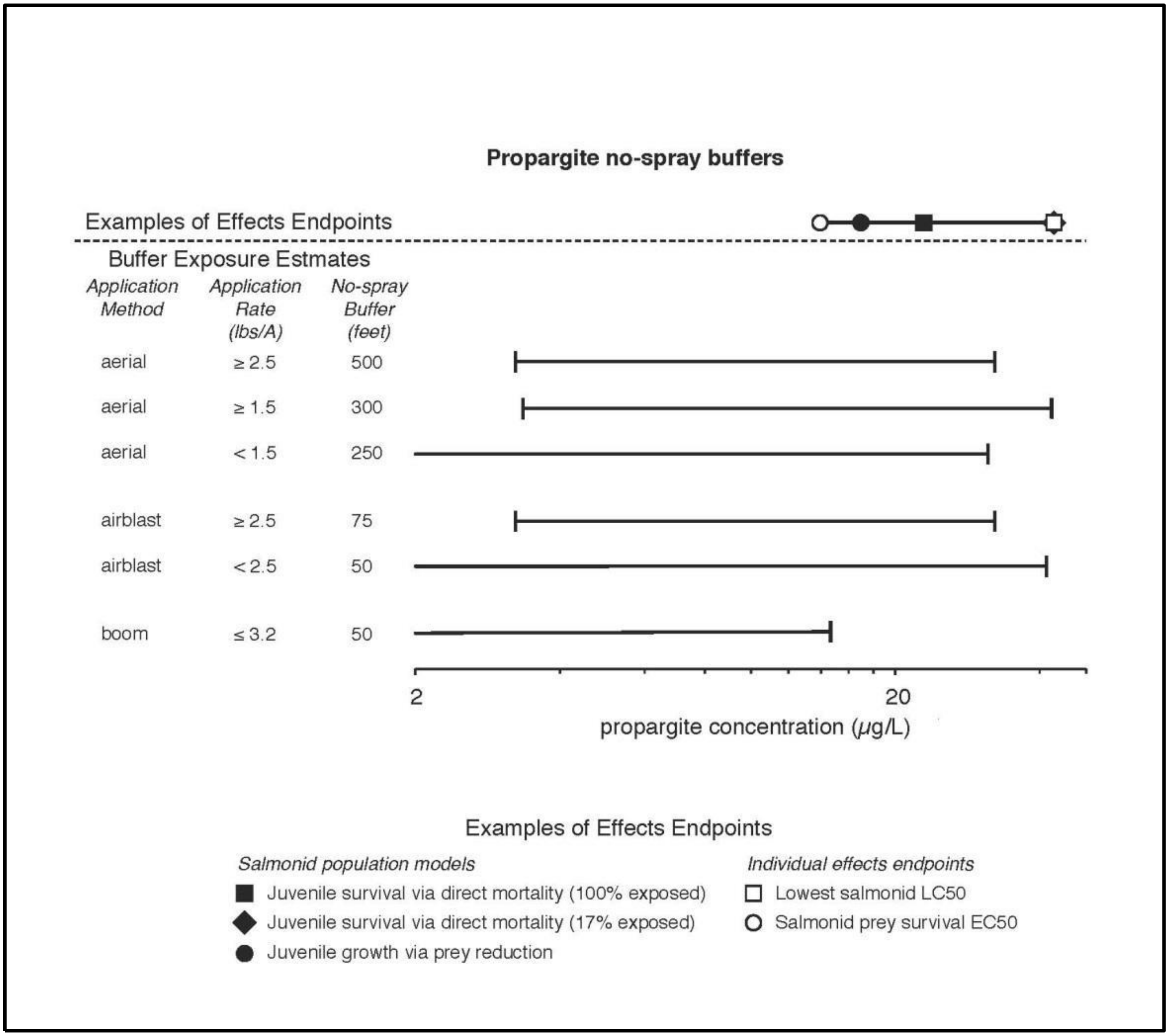

Figure 87. Comparison of propargite effects endpoints and exposure examples for no-spray buffers.

The assignment of no-spray buffers in element 1 weighed the available quantitative and qualitative information depicted as lines-of-evidence. For example, we modeled concentrations in aquatic habitats that ranged in depth from 10 cm to 1 meter to estimate how volume of water in the receiving habitat would affect potential exposure in small streams and floodplain habitats that are important rearing areas for salmonids. We recognize that salmonids use a range of habitats, both larger and smaller than the modeled habitats, which would be predicted to have correspondingly lower or higher concentrations than proved by these estimates. AgDrift values represent predictions for initial average concentrations following a drift event; and we considered that the pesticide concentrations will decline over time due to partitioning, degradation, and dissipation associated with habitat flow or recharge rates. We also considered that the estimates only account for the pesticide active ingredient. They do not account for other stressors that may contribute to increased adverse responses such as other ingredients in the pesticides, adjuvants, and chemical and physical stressors that are present in the baseline habitat.

The degree to which expected exposure to the a.i. overlaps with response endpoints varies somewhat at different buffer sizes and among the three a.i.s. This is due, in part, to qualitative considerations made in assigning the buffers and recognition of inherent uncertainty associated with these estimates. For example, AgDrift estimates associated with aerial application of diflubenzuron at buffers of 500-1000 ft exceed concentrations predicted to cause population level responses associated with reductions in prey. However, we considered that AgDrift has a tendency to overpredict drift at larger buffer distances (Bird et al. 2002). Therefore we expect the predicted concentrations at these buffer distances to occur infrequently. Conversely, greater buffers may be needed than predicted by AgDrift when applications occur closer to salmonid habitat as greater concentrations are (a) more likely to be realized considering drift observed in field trials, and (b) the runoff pathway is more likely to be a contributing pathway of exposure. A greater margin of safety (less overlap of predicted exposure with response endpoints) was warranted for fenbutatin oxide compared to the other a.i.s given differences in environmental fate and effects. AgDrift does not account for potential increases in exposure due to multiple applications of pesticides. Fenbutatin oxide, an organotin, is highly persistent and more readily accumulates in the environment compared to the other two a.i.s. Fenbutatin oxide is also highly toxic to juvenile salmonids and impacts to salmonid survival have direct implications for population level effects. It is important to note the toxicity values presented in the figures above do not address all of the likely effects. For example, the LC50 for propargite represents the median survival response of individual salmonids. We recognize that sensitive individuals will be impacted at lower concentrations reflective of where they are on the dose-response curve. We also note that the examples of effects endpoints presented in the figures are not the sole evidence used to evaluate risk. Other more qualitative lines of evidence were evaluated in the analysis that demonstrates additional risk to listed species (see Figure 82, Figure 83, and Figure 84).

Pesticide droplet size is an important variable that influences how far spray-applied pesticides can drift off site (Bird et al. 2002). We considered label specifications and changes proposed by applicants that restrict droplet size. In some cases it may be possible to reduce the size of no-spray buffers by requiring larger droplet size distributions. However, we are not recommending that approach at this stage of the consultation because further discussions are necessary with EPA and applicants to determine if resulting changes would remain efficacious.

Element 2.

The following no-spray buffers apply to application sites with a maintained ≥30 ft vegetated filter strip of grass or other permanent vegetation designed to remove pesticides and other contaminants in runoff (NRCS 2000).

Table 174 Required no-spray buffers for aerial and ground applications of end-use products containing diflubenzuron [30 ft maintained vegetated filter strip required]

Aerial applications			Ground applications	
No-spray buffer size (ft)	Application rate (lbs propargite/ acre)		No-spray buffer size (ft)	Application rate (lbs propargite/ acre)
750	Greater than or equal to 0.125		375	Greater than or equal to 0.3125
375	Less than 0.125		225	Less than 0.3125 and greater than or equal to 0.125
			110	Less than 0.125 and greater than or equal to 0.03125
			60	Less than 0.03125

Table 175 Required no-spray buffers for ground applications of end-use products containing fenbutatin oxide [30 ft maintained vegetated filter strip required]

No authorized aerial applications		Ground applications	
		No-spray buffer size (ft)	Application rate (lbs propargite/ acre)
		375	All authorized rates

Table 176 Required no-spray buffers for aerial and ground applications of end-use products containing propargite [30 ft maintained vegetated filter strip required]

Aerial applications			Ground applications using airblast technologies	
No-spray buffer size (ft)	Application rate (lbs propargite/ acre)		No-spray buffer size (ft)	Application rate (lbs propargite/ acre)
375	Greater than or equal to 2.5		60	Greater than or equal to 2.5
225	Less than 2.5 and greater than or equal to 1.5		50	Less than 2.5
175	Less than 1.5			

Element 3.

Riparian areas are comprised of native vegetation along rivers, streams, and other water bodies that are adjacent to agricultural and other land types where pesticides are used. Pesticides are known to move from treated agricultural and forested areas via spray drift and surface water runoff into the broader environment, and riparian areas may act to filter runoff and intercept drift thereby reducing loading into off target water bodies. The effective width of a riparian buffer zone for reducing pesticide loading is influenced by many factors including the toxicity of the pesticide active ingredient, habitat characteristics including water depth and flow, weather conditions at the time of application, canopy height and composition, and the type of application system (e.g., aerial vs. ground) (NRCS 2000) . Although these variables are complex and difficult to control, a robust body of research shows that riparian buffers are protective of sensitive aquatic habitats. Reductions in pesticide drift from 75 to 95% up to 30 m (~98.4 feet) downwind occurred with a buffer zone comprised of grass, shrubs, or trees was used (Wolfe et al. 2003). A riparian buffer zone of up to 91.5 m (~300 ft) protected an adjacent stream and pond from aerial applications of chlorothalonil and endosulfan on a Christmas tree plantation (Felsot et al. 2003). Generally, the use of vegetated buffers, coupled with low-drift application methods, reduce drift deposition and runoff into sensitive aquatic habitats adjacent to pesticide use sites.

Riparian areas function as buffers that filter, transformer, and adsorb pesticides and other chemicals. Riparian vegetation slows sediment-laden runoff, and depending on the width and complexity of the area, may deposit or absorb 50 to 100% of sediments as well as the pesticides attached to them (Hawes and Smith 2005). Riparian vegetation may act as a sink by absorbing and degrading pesticides that would otherwise flow into adjacent aquatic habitats. Additionally, certain microbes in the soil associated with the roots of riparian vegetation can degrade pesticides. Another important function of riparian buffers is enhanced infiltration of surface runoff (Dillaha et al. 1989). Riparian vegetation in the buffer zone surrounding a waterbody increases surface roughness and slows overland flows. These slower flows help regulate the volume of water entering rivers and streams, thereby minimizing flood events, scouring of the streambed, and pesticide loading. In addition to reducing pesticide loads, riparian buffers provide many benefits to salmonids and their habitats by increasing shade, reducing water temperatures, increasing inputs of woody debris, increasing inputs of terrestrial insect food items, and reducing flashy water flows.

Riparian areas may substantially reduce pesticide loading negating the need for no-spray buffers. The effectiveness in reducing pesticide loading depends on site specific factors such as dimensions, type, and complexity of the riparian vegetation. By coordinating and collaborating with EPA, USDA NRCS, and others to explore use of riparian areas to reduce loading of pesticides into salmonid habitats, a novel system could be developed to incorporate riparian areas as a tool to reduce pesticide loading. Potentially, riparian areas could be classified and verified by qualified personnel following NRCS protocols to ensure they effectively reduce pesticide loading. If such a system could be designed, land owners with functioning riparian areas would be required to follow a reduced set of no-spray buffers or not have to follow the no-spray buffer requirements outlined in elements 1 and 2.

NMFS has determined that the RPA will enable EPA to proceed with its action in compliance with section 7 of the ESA. NMFS has also determined that the RPA complies with the other regulatory requirements in the Services' implementing regulations.

Consistent with the Intended Purpose of the Action. NMFS has concluded that this RPA is consistent with EPA's purpose of authorizing use of products containing these three a.i.s. EPA can only authorize pesticide use when it does not have an unreasonable adverse effect on the environment. The RPA allows continued authorization of use of these products, but allows it to proceed in a manner consistent with the ESA and FIFRA.

Consistent with the Scope of EPA's Authority. NMFS has concluded that EPA has the authority to authorize the use of ingredients containing these three a.i.s with the limitations recommended in this RPA. EPA has authority to restrict use when such use will cause an unreasonable adverse effect to the environment.

Technological Feasibility. No application and no spray buffers around water bodies are a recognized method of protecting aquatic species from exposure to pesticides. EPA labels contain buffer requirements for other pesticide products.

Economic Feasibility. NMFS has determined that the RPA is economically feasible. As noted above, the requirement is that an RPA is economically feasible to implement, not that its implementation be cost-free. As noted above for technological feasibility, buffers are a commonly used tool to prevent pesticide product from entering waters and riparian zones, and EPA has incorporated buffer requirements in FIFRA labels.

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)).

16 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.

16.1 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 diflubenzuron, fenbutatin-oxide, or propargite, 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 ESA listed 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 diflubenzuron, fenbutatin-oxide, and propargite 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, or if wind conditions exacerbate inputs from drift. 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 3 a.i.s on salmonids includes killing fish directly, reductions in prey leading to starvation or impairing salmonid growth. Impaired growth lends juveniles prone to becoming prey to other fish or avian predators. Impairing feeding ability may also make fish more susceptibility to disease. Thus, we expect some exposed fish will respond to these effects by changing normal behaviors. 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 diflubenzuron, fenbutatin-oxide, and propargite);
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. Minimal 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 these 3 a.i.s with other mixtures or other chemicals in the baseline;
7. Variability in annual land use, crop cover, and pest pressure;
8. Temporal and spatial variability within each ESU, especially at the population-level; and
9. Size and flow variations 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 within the action area attributed to the legal use of diflubenzuron, fenbutatin-oxide, or propargite, 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 are relatively sensitive to pesticides compared to 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 three ingredients, its metabolites, or degradates, if any of the a.i.s is known to have been applied in the vicinity and may reasonably be supposed to have run off or drifted into the affected area, or if surface water samples or pathology indicate lethal levels of the a.i.(s).

NMFS notes that increased monitoring and study of the impact of these pesticides on water quality, particularly water quality in off-channel habitats will inform subsequent consultations and future incidental take statements. Such monitoring and studies will also potentially allow other measures of the extent of take.

16.2 Reasonable and Prudent Measures

The measures described below are non-discretionary measures to avoid or minimize take that must be undertaken by the EPA so that they become binding conditions of any grant or permit issued to the applicant(s), as appropriate, for the exemption in section 7(o)(2) to apply. The EPA has a continuing duty to regulate the activity covered by this incidental take statement. If the EPA (1) fails to assume and implement the terms and conditions or (2) fails to require the applicant(s) to adhere to the terms and conditions of the incidental take statement through enforceable terms that are added to the 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 Office of Protected Resources, ESA Interagency Cooperation Division 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 diflubezuron, fenbutatin-oxide, or propargite; (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 concludes that 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 diflubenzuron, fenbutatin oxide, and propargite by reducing the potential of chemicals to reach salmon-bearing waters;
2. Monitor any incidental take or surrogate measure of take that occurs from the action; and
3. Report annually to NMFS Office of Protected Resources on the monitoring results from the previous year.

16.3 *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 authorize application of pesticide products when wind speeds are greater than or equal to 10 mph.

   b. Do not authorize application of 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 NOAA/National Weather Service, or other similar forecasting service) to occur within 48 hours following application.

   c. 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's Office of Pesticide Programs. 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). Within one year of receipt of this Opinion, EPA shall submit an annual report to NMFS Office of Protected Resources that identifies the total number of fish affected and incident locations.

   d. EPA shall, in close coordination with NMFS Office of Protected Resources, develop and implement an 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 in coordination with NMFS Office of Protected Resources. 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 the three active ingredients. A report summarizing annual monitoring data and including all raw data shall be submitted to NMFS Office of Protected Resources and will summarize annual monitoring data and provide all raw data.

2. a. EPA shall include the following instructions requiring reporting of fish kills either on the labels for all products containing diflubenzuron, fenbutatin oxide, or propargite in ESPP Bulletins:

   NOTICE: Incidents where salmon appear injured or killed as a result of pesticide applications shall be reported to NMFS Office of Protected Resources at 301-713-1401 and EPA's Office of Pesticide Programs. 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. NMFS Office of Protected Resources or Office of Law Enforcement may request the finder to collect specimens or take other measures to ensure that evidence intrinsic to the specimen is preserved.

   b. EPA shall report to NMFS Office of Protected Resources any incidences regarding diflubenzuron, fenbutatin oxide, or propargite 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.

16.4 Conservation Recommendations

Section 7(a) (1) of the ESA directs federal agencies to use their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of endangered and threatened species. Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of a proposed action on listed species or critical habitat, to help implement recovery plans, or to develop information.

The following conservation recommendations would provide information for future consultations involving future authorizations of pesticide a.i.s that may affect listed species:

1. Conduct mixture toxicity analysis in screening-level and endangered species biological evaluations;
2. Develop models to estimate pesticide concentrations in flood plain habitats; and
3. Develop models to estimate pesticide concentrations in aquatic habitats associated with non-agricultural applications, particularly in residential and industrial environments.
4. Work with other appropriate federal agencies to determine efficacy of riparian area management methods in reducing pesticide loading from authorized uses especially the types of vegetation and width of riparian areas needed.

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 Office of Protected Resources of any conservation recommendations it implements in the final action.

16.5 *Reinitiation Notice*

This concludes formal consultation on the EPA's proposed registration of pesticide products containing diflubenzuron, fenbutatin oxide, and propargite 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. NMFS' analysis and conclusions assume that EPA's action includes the label changes proposed by the applicants and incorporated in our consideration of the *Proposed Action*. If those changes are not made within 12 months, EPA should contact NMFS to discuss reinitiation. If reinitiation of consultation appears warranted due to one or more of the above circumstances, EPA must contact NMFS Office of Protected Resources, ESA Interagency Cooperation Division. 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 Opinion. 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. It is recommended that EPA request reinitiation with sufficient time prior to reaching 15 years to allow sufficient time to consult and to prevent lapse of coverage for the active ingredients in this Opinion.

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17 Appendix 1: Abbreviations / Acronyms

7-DADMax	7-day average of the daily maximum
ACA	Alternative Conservation Agreement
AChE	acetylcholinesterase
a.i.	active ingredient
APEs	alkylphenol ethoxylates
APHIS	U.S. Department of Agriculture Animal Plant and Health Inspection Service
BE	Biological Evaluation
BEAD	Biological and Economic Analysis Division
BLM	Bureau of Land Management
BMP	Best Management Practices
BOR	Bureau of Reclamation
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
µg/L	micrograms per liter
UCR	Upper Columbia River
USFS	United States Forest Service
USC	United States Code

USFWS	United States Fish and Wildlife Service
USGS	United States Geological Survey
UWR	Upper Willamette River
VOC	Volatile Organic Compounds
VSP	Viable Salmonid Population
WDFW	Washington Department of Fish and Wildlife
WLCRTRT	Willamette/Lower Columbia River Technical Review Team
WQS	Water Quality Standards
WWTIT	Western Washington Treaty Indian Tribes
WWTP	Wastewater Treatment Plant
YOY	Young of year

18 Appendix 2: Glossary

303(d) waters	Section 303 of the federal Clean Water Act requires states to prepare a list of all surface waters in the state for which beneficial uses – such as drinking, recreation, aquatic habitat, and industrial use - are impaired by pollutants. These are water quality limited estuaries, lakes, and streams that do not meet the state's surface water quality standards and are not expected to improve within the next two years. After water bodies are put on the 303(d) list they enter into a Total Maximum Daily Load Clean Up Plan.
Active ingredient	The component(s) that kills or otherwise affects the pest. A.i.s are always listed on the label (FIFRA 2(a)).
Adulticide	A compound that kills the adult life stage of the pest insect.
Anadromous Fish	Species that are hatched in freshwater migrate to and mature in salt water and return to freshwater to spawn.
Adjuvant	A compound that aides the operation or improves the effectiveness of a pesticide.
Alevin	Life history stage of a salmonid immediately after hatching and before the yolk-sac is absorbed. Alevins usually remain buried in the gravel in or near the egg nest (redd) until their yolk sac is absorbed when they swim up and enter the water column.
Anadromy	The life history pattern that features egg incubation and early juvenile development in freshwater migration to sea water for adult development, and a return to freshwater for spawning.

Assessment Endpoint	Explicit expression of the actual ecological value that is to be protected (*e.g.*, growth of juvenile salmonids).
Bioaccumulation	Accumulation through the food chain (*i.e.*, consumption of food, water/sediment) or direct water and/or sediment exposure.
Bioconcentration	Uptake of a chemical across membranes, generally used in reference to waterborne exposures.
Biomagnification	Transfer of chemicals via the food chain through two or more trophic levels as a result of bioconcentration and bioaccumulation.
Core population	Salmonid populations within an ESU or DPS that historically were the most productive and that best represent the historical genetic diversity (genetic legacy) of the ESU or DPS. These populations are deemed essential in recovery plans for the long-term recovery and delisting of the species.
Degradates	New compounds formed by the transformation of a pesticide by chemical or biological reactions.
Dependent Populations (DPs)	Populations with a substantial likelihood of going extinct within a 100-year time period in isolation due to smaller population size, but receive sufficient immigration to alter their dynamics and reduce extinction risk.
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 for an significant to the remainder of its species based on factors such as physical, behavioral, or genetic characteristics, it occupies an unusual or unique ecological setting, or its loss would represent a significant gap in the species' range.

Escapement	The number of fish that survive to reach the spawning grounds or hatcheries. The escapement plus the number of fish removed by harvest form the total run size.
Evolutionarily Significant Unit	A group of Pacific salmon or steelhead trout that is (1) substantially reproductively isolated from other conspecific units and (2) represent an important component of the evolutionary legacy of the species.
Fall Chinook 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.
Functionally Independent Populations (FIPs)	Populations with a high likelihood of persisting over 100-year time scales due to their population size and relatively independent dynamics (i.e. negligible influence of migrants from neighboring populations on extinction risk).

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.
Potentially Independent Populations (PIPs)	Populations with a high likelihood of persisting in isolation over 100-year time scales due to large population size, but were likely too strongly influenced by immigration from other population to exhibit independent dynamics.
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, and that the Services believes would avoid the likelihood of jeopardizing the continued existence of the listed species or the destruction or adverse modification of designated critical habitat.
Redd	A nest constructed by female salmonids in streambed gravels where eggs are deposited and fertilization occurs.

Riparian zone	Zone 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 and valley bottoms that support riparian vegetation.
Risk	The probability of harm from actual or predicted concentrations of a chemical in the aquatic environment – a scientific judgment.
Salmonid	Fish of the family *Salmonidae*, including salmon, trout, chars, grayling, and whitefish. In general usage, the term usually refers to salmon, trout, and chars.
SASSI	A cooperative program by WDFW and WWTIT to inventory and evaluate the status of Pacific salmonids in Washington State. The SASSI report is a series of publications from this program.
Semelparous	The condition in an individual organism of reproducing only once in a lifetime.
Smolt	A juvenile salmon or steelhead migrating to the ocean and undergoing physiological changes to adapt from freshwater to a saltwater environment.
Sublethal	Below the concentration that directly causes death. Exposure to sublethal concentrations of a material may produce less obvious effect on behavior, biochemical, and/or physiological function of the organism often leading to indirect death.
Surfactant	A substance that reduces the interfacial or surface tension of a system or a surface-active substance.

Synergism	A phenomenon in which the toxicity of a mixture of chemicals is greater than that which would be expected from a simple summation of the toxicities of the individual chemicals present in the mixture.
Technical Grade Active Ingredient (TGAI)	Pure or almost pure active ingredient. Available to formulators. Most toxicology data are developed with the TGAI. The percent AI is listed on all labels.
Technical Recovery Teams (TRT)	Teams convened by NOAA Fisheries to develop technical products related to recovery planning. TRTs are complemented by planning forums unique to specific states, tribes, or reigns, which use TRT and other technical products to identify recovery actions.
Teratogenic	Effects produced during gestation that evidence themselves as altered structural or functional processes in offspring.
Total Maximum 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.

19 Appendix 3: Toxicity of Eleven Pesticides to Embryonic Zebrafish

Appendix 3
Toxicity of Eleven Pesticides to Embryonic Zebrafish

November 2011

Project Summary

The Northwest Fisheries Science Center conducted an experiment requested by NOAA's Office of Protected Resources in support of a Biological Opinion regarding the toxicity of various pesticides to endangered salmon species. The experiment detailed here investigated the effects of eleven pesticides on developing zebrafish (*Danio rerio)*, a species that is widely used as a toxicological model for other fish species. Zebrafish are a useful model species because the early ontogeny of zebrafish is rapid and well documented (Kimmel et al., 1995) and their features are easily observed through translucent chorions and bodies. In this experiment, embryonic zebrafish were exposed to oryzalin, trifluralin, prometryn, pendimethalin, fenbutatin oxide, thiobencarb, propargite, metolachlor, 1,3-dichloropropene, bromoxynil and diflubenzuron in 5-day static-renewal exposures. Toxicity endpoints included mortality, developmental abnormalities, and body length on the final day of the experiment. Three of the chemicals tested, prometryn, fenbutatin oxide, and diflubenzuron, did not produce an adverse effect on zebrafish survival, morphology or length at the tested concentrations. The pesticides trifluralin, pendimethalin and thiobencarb increased the rate of abnormality in developing zebrafish without appreciably increasing the rate of mortality at the concentrations tested. Fish lengths were significantly smaller following exposure to oryzalin, bromoxynil, trifluralin, pendimethalin, thiobencarb, propargite, metolachlor and 1,3-dichloropropene.

Methods

Fish: Zebrafish (*D. rerio*) embryos were obtained from a colony maintained at the Northwest Fisheries Science Center according to standard operating procedures (Linbo, 2009). Male and female zebrafish were combined in spawning tanks and eggs were collected at the beginning of the next light cycle, approximately one hour after the spawning event. Embryos were housed in a temperature-controlled incubator at 28.5 °C for the duration of the experiment.

Pesticide stock solutions: Pesticides were obtained in pure form from Chem Service, Inc. (West Chester, Pennsylvania). Pesticide stock solutions were made in acetone and stored under dark conditions at 4 °C. A working solution composed of stock solution and water from the zebrafish colony (system water) was mixed fresh at the start of each day, and subsequent exposure concentrations serially diluted. The maximum acetone concentration for any exposure was 0.1%. The highest pesticide concentration of each compound tested was generally the reported rainbow trout or zebrafish 96-hr LC_{50} value (the concentration lethal to 50% of the test organisms). The highest exposure concentration of 1,3-dichloropropene was 100 times lower than the reported LC_{50} value because of observed developmental effects, while exposure concentrations of diflubenzuron were lower due to low solubility in acetone.

Pesticide exposures: Normally developing zebrafish embryos at 1.5-2.5 hpf (hours post-fertilization) were selected and placed in 60 mm acetone-washed glass Petri dishes with 10 ml of pesticide solution. Individual dishes contained 15 embryos and each exposure concentration was tested in triplicate. Exposures were conducted in batches comprised of one or two pesticides, water controls, and 0.1% acetone controls. Exposure solutions were renewed every 24 hours. Dead embryos were removed from the dishes each day to prevent fungal growth and contamination.

Anatomical screening and measurement of fish body length: Embryos were scored every 24 hr for mortality and abnormalities through 5 dpf (days post-fertilization). See Table 2 for a description of the observed developmental abnormalities. Daily anatomical screenings were performed using a Nikon-SMZ-800 stereomicroscope with a diascopic base (Meridian Instruments, Seattle, Washington). Only surviving fish were screened for anatomical abnormalities. At 5 dpf, the embryos were anesthetized with tricaine methanesulfonate (MS-222; Sigma-Aldrich, St. Louis, Missouri) to measure body length. All surviving embryos from each exposure dish were simultaneously photographed using a Spot RT digital camera (Diagnostic Instruments, Inc., Sterling Heights, Michigan) mounted on a stereomicroscope. Length was measured from the anterior tip of the mouth along the notochord to the posterior tip of the notochord, and quantified using ImageJ software (available online at http://rsbweb.nih.gov/ij/).

Statistical tests: Length was the only parameter explicitly tested. Lengths of control fish were compared using a two-factor ANOVA comparing type (water and acetone) and batch, and showed a significant result of batch only. Subsequent analyses of exposures compared the average of three dishes (n = 3) to their corresponding batch controls. Differences in embryo lengths between concentrations of a given pesticide were tested using one-way ANOVAs with a Tukey HSD post hoc (Tables 3-13).

Results

Chemical-specific mortality and abnormality data, as well as their respective controls, are presented in Figures 1-11. Both water and acetone controls showed consistently low rates of both mortality and abnormality. We found that 3 pesticides (prometryn, fenbutatin oxide and diflubenzuron) showed no increases in mortality or abnormality as well as no significant differences in embryo length. Three additional chemicals (trifluralin, pendimethalin and thiobencarb) produced higher rates of abnormalities and significantly shorter embryos at the highest exposure concentration without increasing mortality. While the remaining pesticides (oryzalin, bromoxynil, propargite, metolachlor, and 1,3-dichloropropene) produced significantly shorter embryos at various exposure concentrations with no effect on mortality or abnormality, there was no clear dose-dependent trend. Whether there is a biological consequence to these shorter lengths at the concentrations tested here is a subject for further investigation.

Table 1. Nominal concentrations of pesticides used in exposures and rainbow trout LC_{50} values.

Compound Name	Type	Exposure Concentrations (μg/l)	Rainbow Trout LC_{50} values (μg/l)
Oryzalin	Herbicide	3, 30, 300, 3000	3260
Trifluralin	Herbicide	0.05, 0.5, 5, 50	50
Prometryn	Herbicide	0.9, 9, 90, 900	2900
Pendimethalin	Herbicide	0.15, 1.5, 15, 150	138
Fenbutatin oxide	Insecticide	0.01, 0.1, 1, 10	10
Thiobencarb	Herbicide	0.8, 8, 80, 800	790
Propargite	Insecticide	0.15, 1.5, 15, 150	<168
Metolachlor	Herbicide	0.3, 3, 30, 300	300
1,3-Dichloropropene	Insecticide	0.03, 0.3, 0.3, 3	270
Bromoxynil	Herbicide	0.05, 0.5, 5, 50	41
Diflubenzuron	Insecticide/Fungicide	2, 20, 200, 2000	72000

Table 2. Abnormalities observed during zebrafish embryo exposures.

Abnormality	Description
Edema	Accumulation of excess fluid in any one of the following cavities: heart, yolk sac, yolk extension, eyes.
Unhatched	Failure to hatch at 5 dpf.
Curved	Curvature of the tail dorsally in the sagittal plane such that a line drawn from the posterior tip of the notochord to the mouth of the fish would yield a gap between the line and body.
Lethargic	An inability to maintain an upright posture and/or inactivity.
Deformed fins	The absence or improper formation of fin tissue.
Deformed tail	A notable shortening of the tail or improper notochord development.
Bent	A bend in the body or tail of the embryo in the coronal plane.

Oryzalin

Oryzalin exposure did not impact developing zebrafish in a dose-dependent manner. Mortality was the highest (20%) at 30 μg/l, but declined to 8.9% at 3000 μg/l. Abnormality was the highest at 3000 μg/l (17.1%), but was also elevated at 3 μg/l (16.2%). The most common abnormality observed was edema.

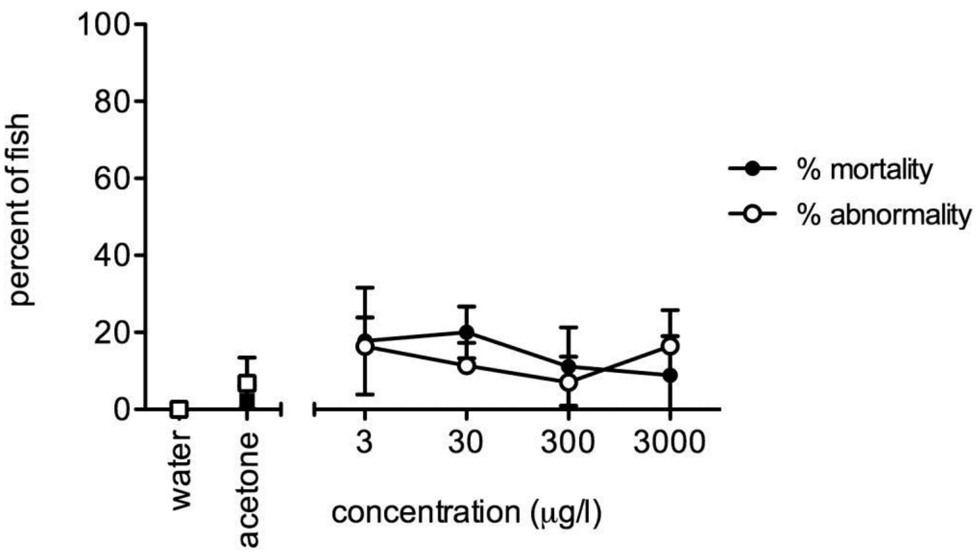


Figure 1: Percent mortality and abnormality observed in control and oryzalin-exposed zebrafish. Symbols are means (n = 3) ± SD.

Table 3: Average length of fish exposed to oryzalin and controls (n = 3 dishes). There was a significant effect of oryzalin (One-way ANOVA, *p* < 0.0001). * Indicates treatment significantly different than controls (Tukey HSD, *p* < 0.01).

Treatment (μg/l)	Average length ± SD (mm)
Water control	4.49 ± 0.02
0.1% acetone	4.50 ± 0.02
3	4.53 ± 0.05
30	4.48 ± 0.02
300	4.51 ± 0.05
3000	4.27 ± 0.02*

Bromoxynil

Bromoxynil exposure did not cause an increase in mortality or abnormality in developing zebrafish. The highest rate of abnormality (6.7%) was observed at 0.05 μg/l and 50 μg/l. Mortality occurred the most frequently at 0.5 μg/l and 5 μg/l at a rate of 2.2%.

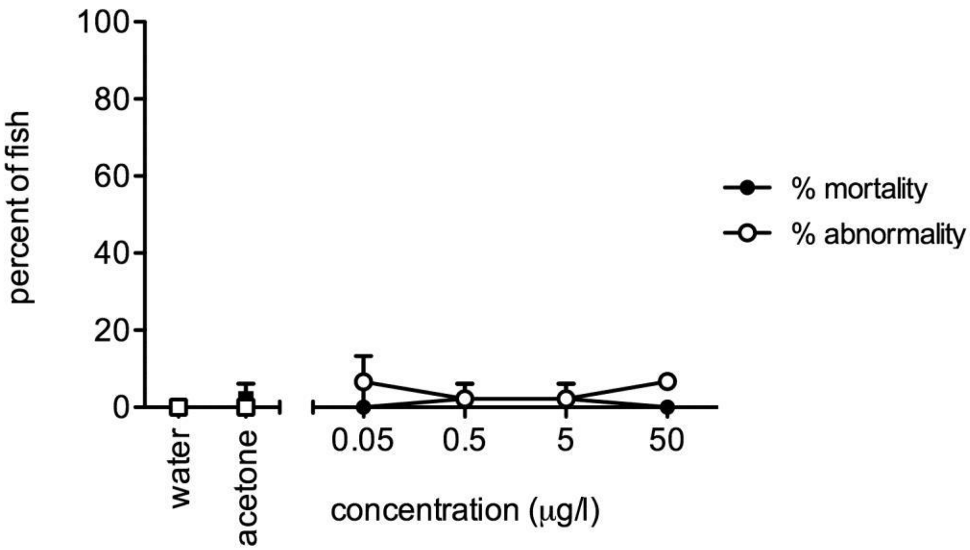


Figure 2. Percent mortality and abnormality in controls and zebrafish exposed to bromoxynil. Symbols are means (n = 3) ± SD.

Table 4: Average length of fish exposed to bromoxynil and controls (n = 3 dishes). There was a significant effect of bromoxynil (One-way ANOVA, $p < 0.0001$). * Indicates treatment significantly different than controls (Tukey HSD, $p < 0.05$).

Treatment (μg/l)	Average length ± SD (mm)
Water control	4.20 ± 0.04
0.1% acetone	4.06 ± 0.02
0.05	3.97 ± 0.03*
0.5	4.08 ± 0.01

5	4.05 ± 0.04
50	4.13 ± 0.06

Trifluralin

Exposure to trifluralin caused significant abnormalities at the highest dose tested (50 μg/l). The rate of abnormality at this dose was 95.3%, and the most common abnormality noted was lethargy, characterized by the absence of active swimming and a tendency to lose upright posture. Mortality was the greatest (22.2%) at 0.5 μg/l.

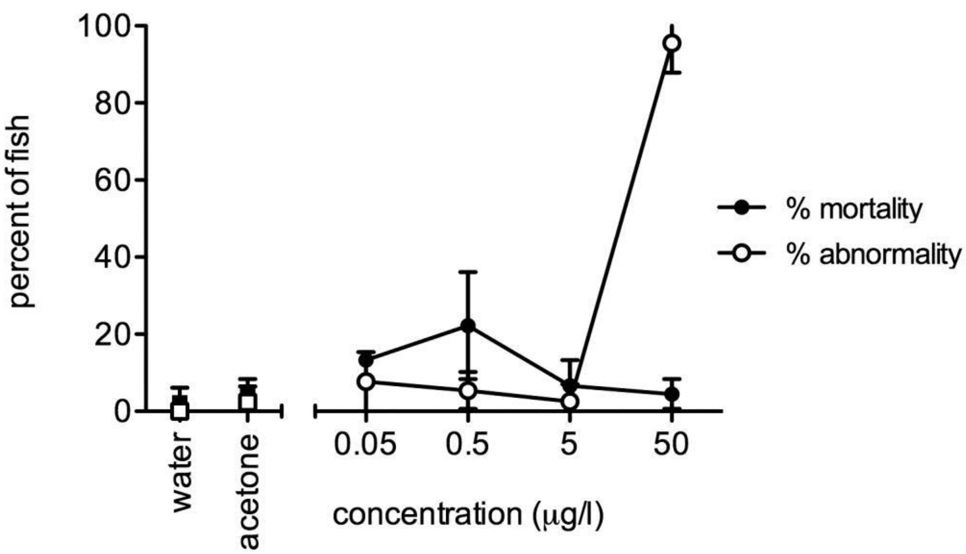


Figure 3. Percent mortality and abnormality of controls and zebrafish exposed to trifluralin. Symbols are means (n = 3) ± SD.

Table 5: Average lengths of fish exposed to trifluralin and controls (n = 3 dishes). There was a significant effect of trifluralin (One-way ANOVA, $p < 0.0001$). * Indicates treatment significantly different than controls (Tukey HSD, $p < 0.01$).

Treatment (μg/l)	Average length ± 1 SD (mm)
Water control	4.02 ± 0.01
0.1% acetone	4.11 ± 0.07

0.05	4.05 ± 0.02
0.5	4.11 ± 0.07
5	4.01 ± 0.07
50	3.59 ± 0.03*

Prometryn

Prometryn exposure did not adversely affect either the rate of abnormality or mortality in developing zebrafish. The highest rate of morality observed was at 9 µg/l (4.4%), and the highest rate of abnormality was at 0.9 µg/l and 900 µg/l (2.3%).

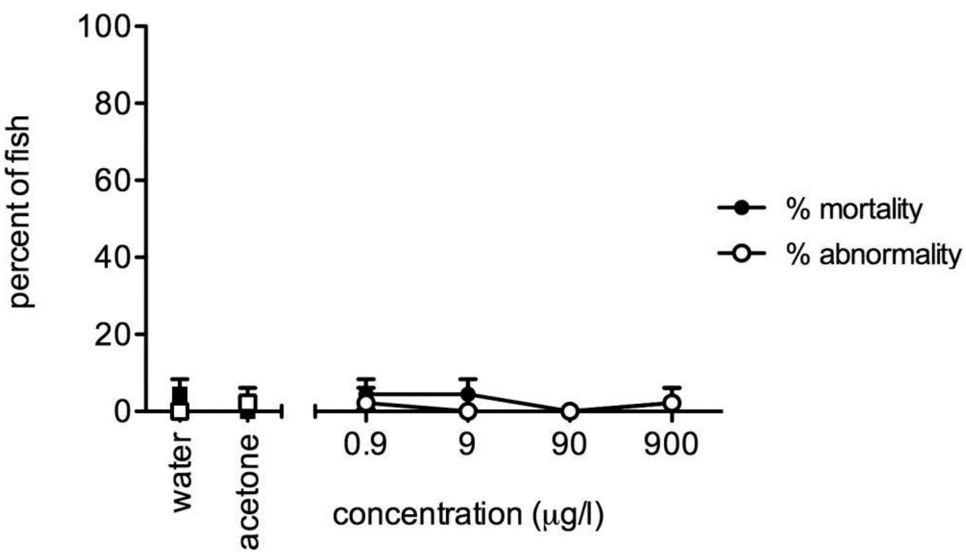


Figure 4. Percent mortality and abnormality of controls and prometryn exposed fish. Symbols are means (n = 3) ± SD.

Table 6: Average lengths of fish exposed to prometryn and controls (n = 3 dishes). Exposure to prometryn did not significantly affect fish length (One-way ANOVA, p > 0.05).

Treatment (µg/l)	Average length ± SD (mm)
Water control	3.85 ± 0.06

0.1% acetone	3.96 ± 0.03
0.9	3.96 ± 0.06
9	3.95 ± 0.01
90	3.97 ± 0.03
900	3.85 ± 0.02

Pendimethalin

Embryos exposed to 150 μg/l of pendimethalin developed a significant amount (100%) of abnormalities. Abnormal embryos were lethargic and struggled to swim. The highest rate of mortality (11.1%) was noted at 15 μg/l.

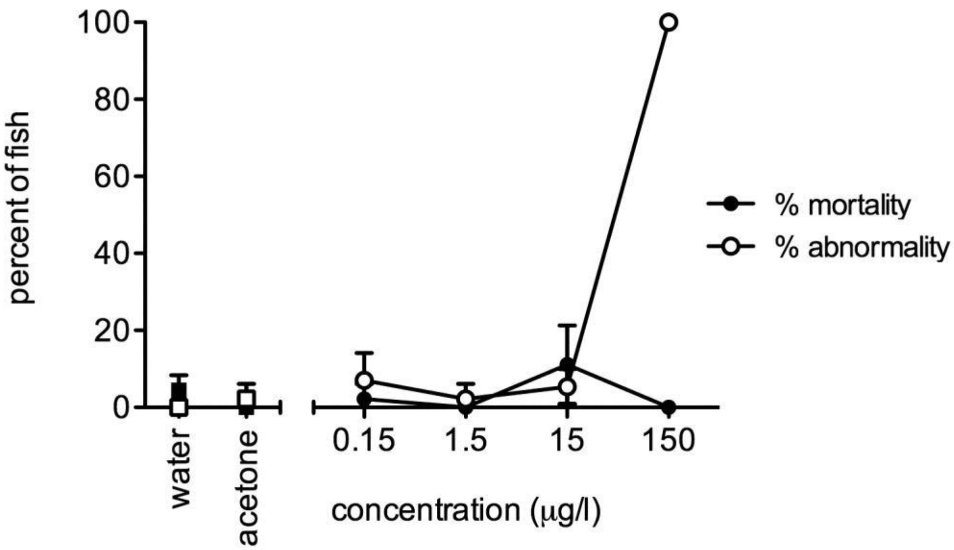


Figure 5. Percent mortality and abnormality of controls and fish exposed to pendimethalin. Symbols are means (n = 3) ± SD.

Table 7: Average lengths of fish exposed to pendimethalin and controls (n = 3 dishes). Pendimethalin exposure significantly impacted the length of larvae (One-way ANOVA, *p* <0.001). * Indicates treatment significantly different than controls (Tukey HSD, *p* < 0.01).

Treatment (μg/l)	Average length ± SD (mm)
Water control	3.85 ± 0.06
0.1% acetone	3.96 ± 0.03
0.15	3.98 ± 0.04
1.5	3.97 ± 0.03
15	3.94 ± 0.06

150	3.59 ± 0.03*

Fenbutatin oxide

Fenbutatin oxide did not cause a dose-dependent change in mortality or abnormality. Mortality occurred the most frequently at 10 μg/l (28.9%). Abnormality on the other hand was highest at 0.1 μg/l (26.3%), and declined at higher concentrations.

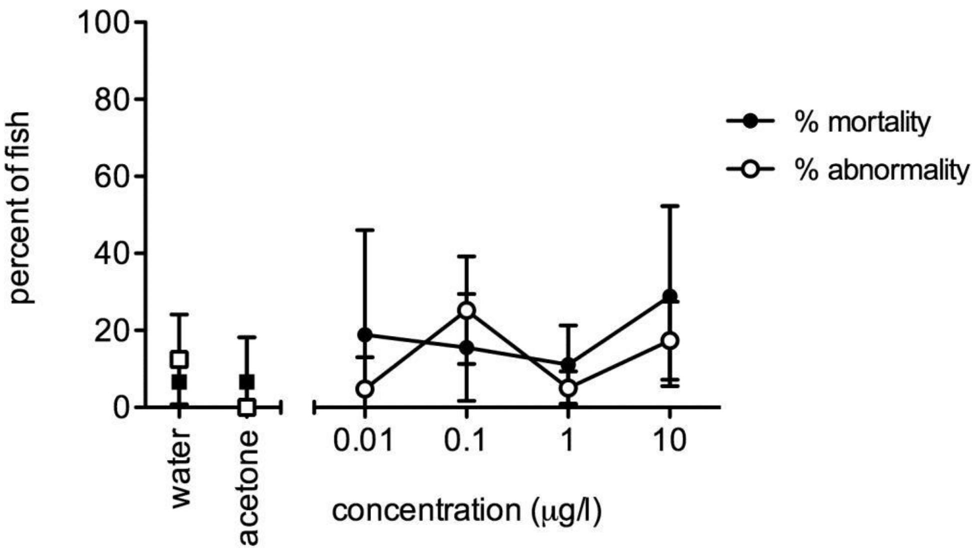


Figure 6. Percent mortality and abnormality of controls and fish exposed to fenbutatin oxide. Symbols are means (n = 3) ± SD.

Table 8: Average lengths of fish exposed to fenbutatin oxide and controls (n = 3 dishes). Fenbutatin oxide exposure did not affect the length of fish (One-way ANOVA, p > 0.05).

Treatment (μg/l)	Average length ± SD (mm)
Water control	3.90 ± 0.06
0.1% acetone	3.93 ± 0.01
0.01	3.91 ± 0.03
0.1	3.88 ± 0.06
1	3.91 ± 0.04
10	3.87 ± 0.02

Thiobencarb

Exposing developing zebrafish to thiobencarb produced abnormalities in 100% of the embryos at 800 μg/l. The 5-dpf larvae behaved abnormally with erratic swimming patterns. Mortality at 800 μg/l was 13.3%.

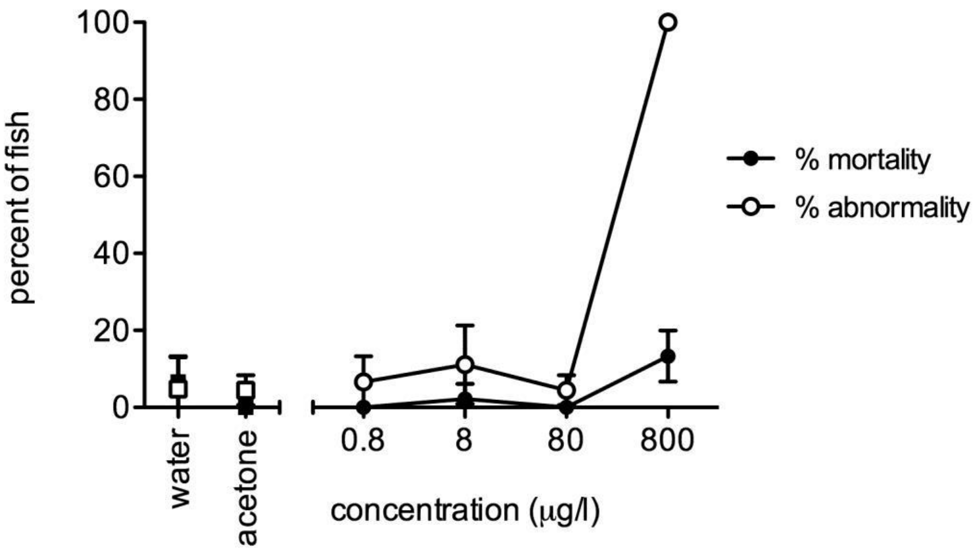


Figure 7. Percent mortality and abnormality observed in controls and fish exposed to thiobencarb. Symbols are means (n = 3) ± SD.

Table 9: Average lengths of fish exposed to thiobencarb and controls (n = 3 dishes). There was a significant effect of thiobencarb (One-way ANOVA, *p*<0.0001). * Indicates treatment significantly different than controls (Tukey HSD, *p* < 0.01).

Treatment (μg/l)	Average length ± SD (mm)
Water control	3.92 ± 0.04
0.1% acetone	3.99 ± 0.03
0.8	3.91 ± 0.03
8	3.87 ±0.04
80	3.91 ± 0.03

800	3.69 ± 0.07*

Propargite

Zebrafish embryos exposed to propargite did not show increased rates of mortality or abnormality. The highest rate of mortality (4.4 %) was observed at 0.15 µg/l and 1.5 µg/l. Embryos had the greatest number of abnormalities (13.6%) at 150 µg/l.

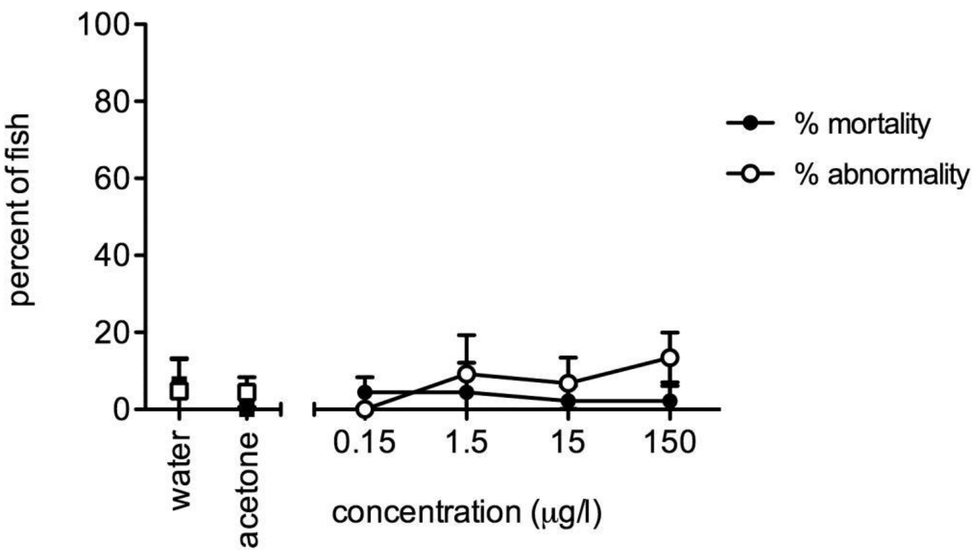


Figure 8. Percent mortality and abnormality in controls and fish exposed to propargite. Symbols are means (n = 3) ± SD.

Table 10: Average lengths of fish exposed to propargite and controls (n = 3 dishes). Propargite produced significant effects (One-way ANOVA, *p*= 0.005). * Indicates treatment significantly different than controls (Tukey HSD, *p* < 0.01).

Treatment (µg/l)	Average length ± SD (mm)
Water control	3.92 ± 0.04
0.1% acetone	3.99 ± 0.03
0.15	3.95 ± 0.04
1.5	3.92 ± 0.04
15	3.94 ± 0.02
150	3.83 ± 0.01*

Metolachlor

Exposure to metolachlor did not alter zebrafish mortality, although a higher rate (28.6%) of abnormality was observed at 300 μg/l. The most frequent abnormality noted was a failure to hatch by 5 dpf.

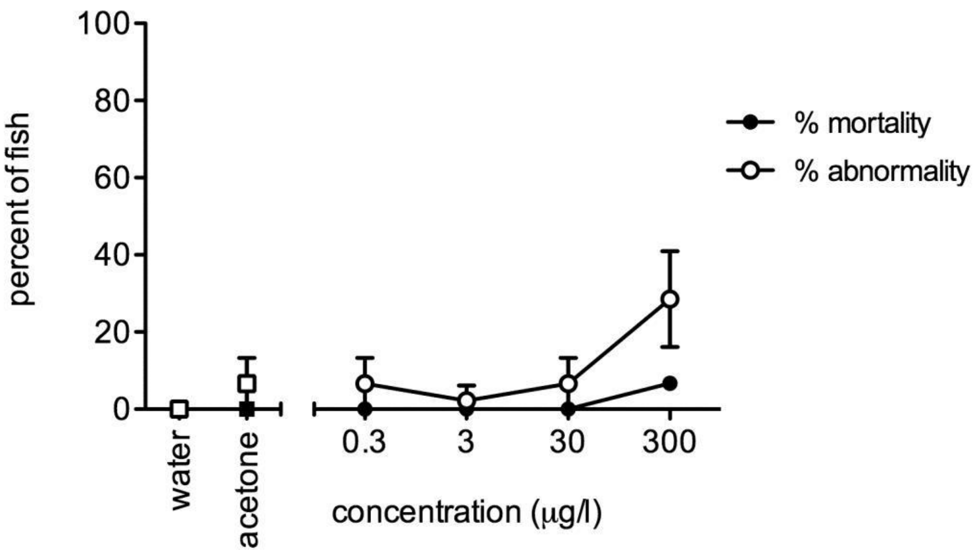


Figure 9. Percent mortality and abnormality of zebrafish exposed to metolachlor and controls. Symbols are means (n = 3) ± SD.

Table 11: **Average lengths of fish exposed to metolachlor and controls (n = 3 dishes). There was a significant effect of metolachlor (One-way ANOVA, $p < 0.0001$). * Indicates treatment significantly different than controls (Tukey HSD, $p < 0.01$).**

Treatment (μg/l)	Average length ± SD (mm)
Water control	4.42 ± 0.03
0.1% acetone	4.37 ± 0.03
0.3	4.24 ± 0.06*
3	4.40 ± 0.05
30	4.23 ± 0.05*
300	4.18 ± 0.05*

1,3-Dichloropropene

Exposure to 1,3-dichloropropene caused an increase in abnormality and mortality in developing zebrafish, but not in a dose dependent manner. The highest rate of mortality (28.9%) occurred at 0.3 μg/l, and declined at higher concentrations. The highest rate of abnormality (37.5%) was observed at 3 μg/l. The rate of abnormality remained between 28.1% and 37.5% for all exposure concentrations and the most commonly observed abnormality was failure to hatch by 5dpf.

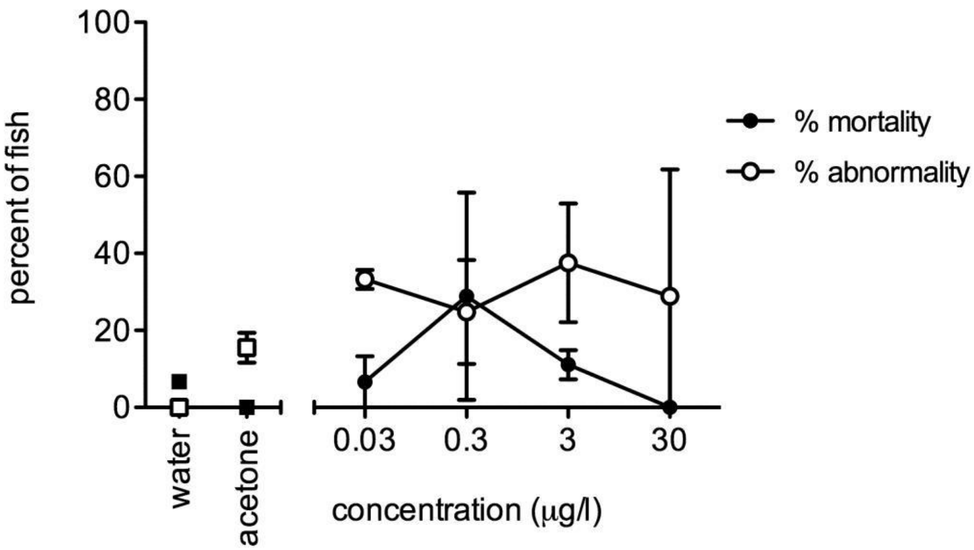


Figure 10. Percent mortality and abnormality observed in fish exposed to 1,3-dichloropropene and controls. Symbols are means (n = 3) ± SD.

Table 12. Average lengths of fish exposed to 1,3-dichloropropene and controls (n = 3 dishes). There was a significant effect of 1,3-dichloropropene (One-way ANOVA, *p* <0.001). * Indicates treatment significantly different than controls (Tukey HSD, *p* < 0.01).

Treatment (μg/l)	Average length ± SD (mm)
Water control	4.46 ± 0.03
0.1% acetone	4.34 ± 0.03
0.03	4.32 ± 0.04
0.3	4.14 ± 0.05*
3	4.28 ± 0.08
30	4.27 ± 0.06

Diflubenzuron

Diflubenzuron did not influence zebrafish mortality or abnormality. The highest rate of abnormality (6.8%) was observed at 20 μg/l, and the highest rate of morality (4.4%) was observed at 2 μg/l. However, it is important to note that diflubenzuron was difficult to work with because of its low solubility in acetone (6.5 g/l). The most concentrated stock solution of diflubenzuron we were able to make was 2 g/l. Diflubenzuron appeared to remain in solution after dosing the exposure dishes, however after 24hrs, the highest exposure concentration dishes (2000 μg/l) had visible floating particles. Thus, without using alternative methodologies (e.g. DMSO as the carrier), we are not confident about accurate dosing for this compound.

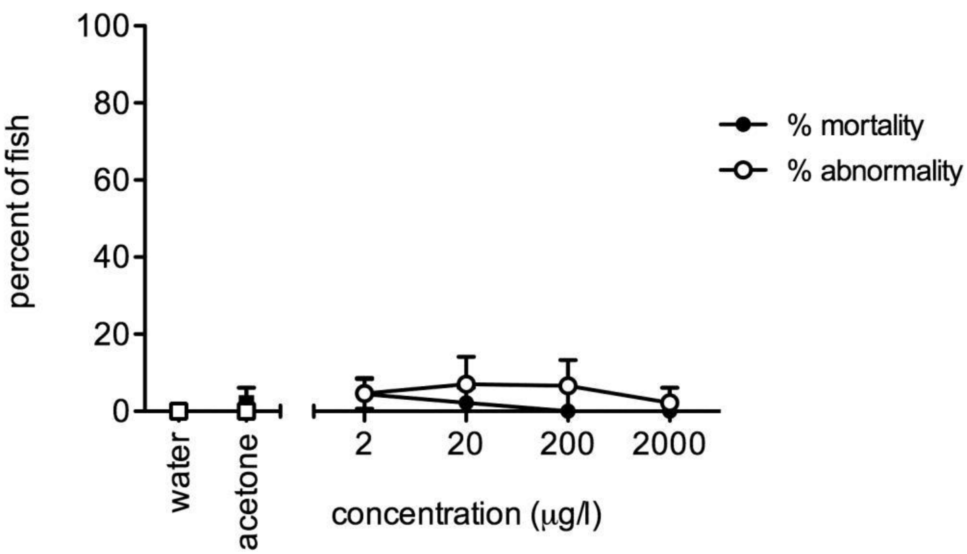


Figure 11. Percent mortality and abnormality observed in control fish and fish exposed to diflubenzuron. Symbols are means (n = 3) ± SD.

Table 13: Average lengths of fish exposed to diflubenzuron and controls (n = 3 dishes). There was not a significant effect of diflubenzuron on fish length (One-way ANOVA, p > 0.05).

Treatment (μg/l)	Average length ± SD (mm)
Water control	4.20 ± 0.04
0.1% acetone	4.06 ± 0.02
2	4.06 ± 0.03
20	4.88 ± 0.06

200	4.98 ± 0.03
2000	4.08 ± 0.03

References

Kimmel, CB, WW Ballard, SR Kimmel, B Ullman and TF Schilling. 1995. Stages of embryonic development of the zebrafish. *Developmental Dynamics* 203(3):253-310.

Linbo, TL. 2009. Zebrafish (*Danio rerio*) husbandry and colony maintenance at the Northwest Fisheries Science Center. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-NWFSC-100, 62p.

20 Appendix 4: 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 that 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 did consider differences in state regulations and SLN registrations, as well as general cropping trends for different basins.

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.

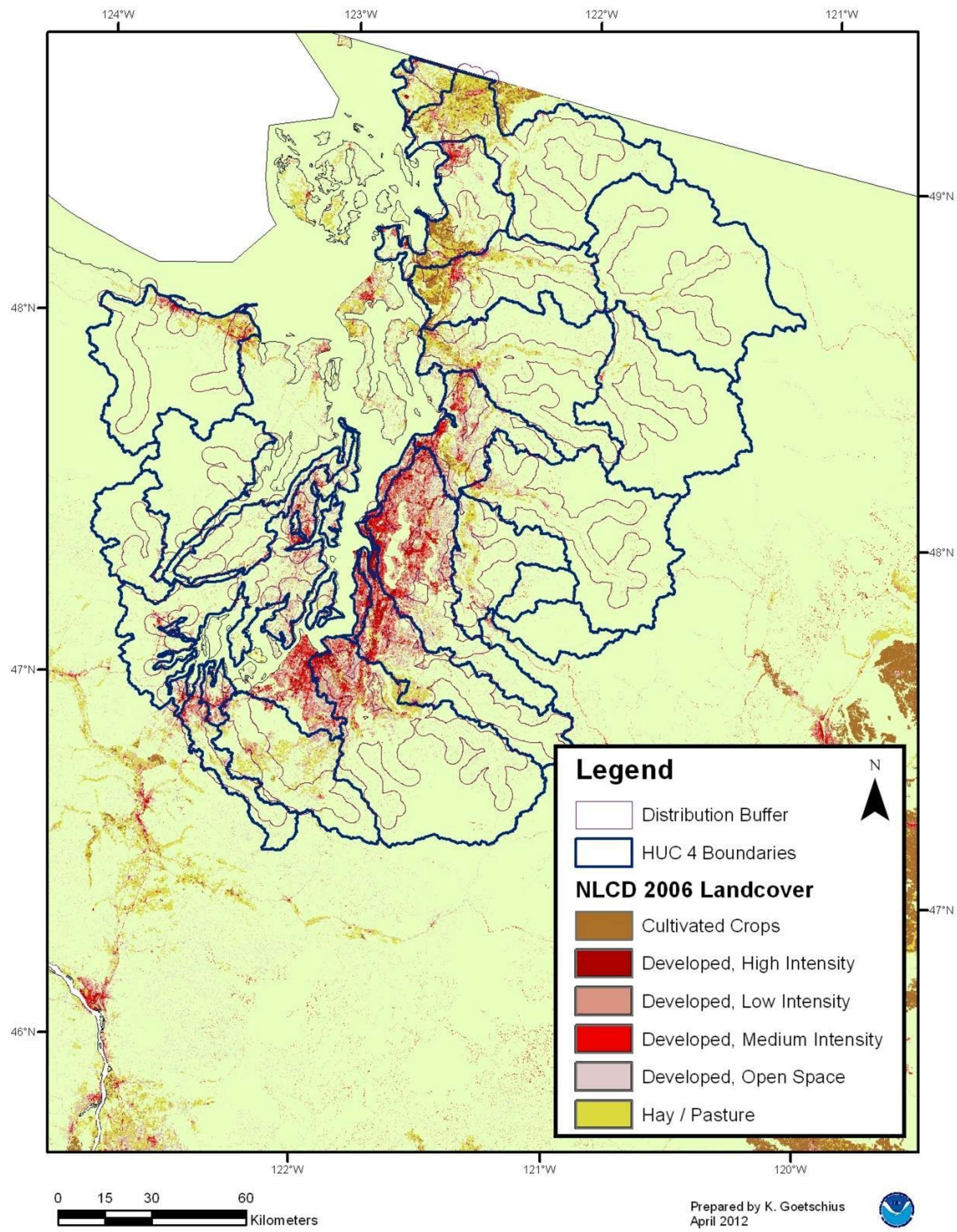
Once co-occurrence is determined via GIS for each a.i., we evaluated the spatial and temporal extent of potential exposure for the ESU/DPS, given the life history of the species. In many cases, fish may be in the system for prolonged periods of time, and there is generally no specific seasonal restriction 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. Other important factors that were taken into consideration include location of federal land, railroad lines, and electrical transmission lines.

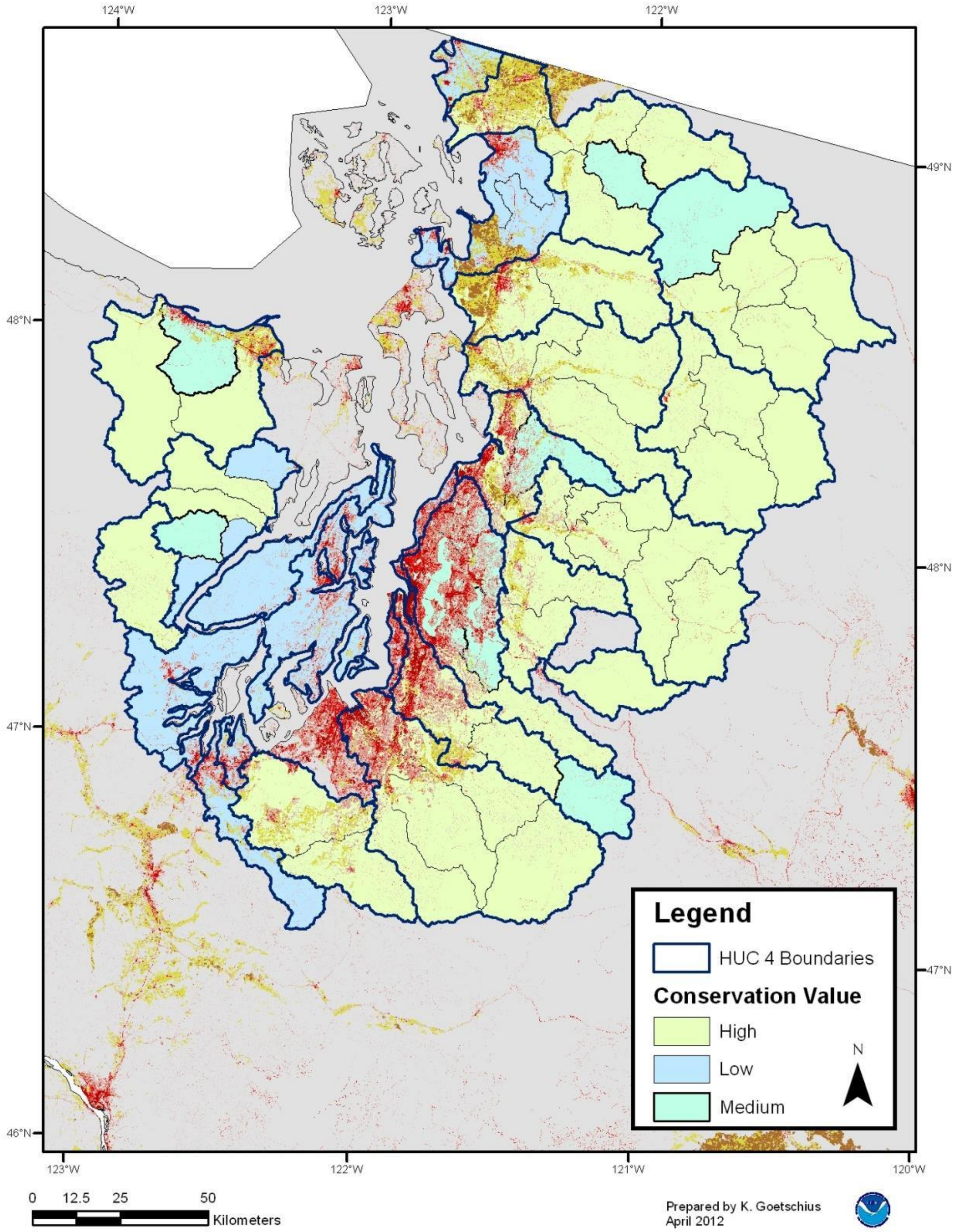
To illustrate the co-occurrence analysis process, this appendix includes two maps for each ESU/DPS. The first map shows the range of the ESU with each HUC 4 outlined in blue, the 2.5 km buffer in burgundy and relevant categories from the 2006 NLCD land use layer. This map aided in the Species analyses. 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. Lower Columbia River Coho (proposed)
10. Oregon Coast Coho
11. Puget Sound Steelhead (proposed)
12. Lower Columbia River Steelhead
13. Middle Columbia River Steelhead
14. Upper Columbia River Steelhead
15. Upper Willamette River Steelhead
16. Snake River Steelhead
17. Northern California Steelhead
18. Central California Coast Steelhead
19. California Central Valley Steelhead
20. South-Central California Coast Steelhead
21. Southern California Steelhead

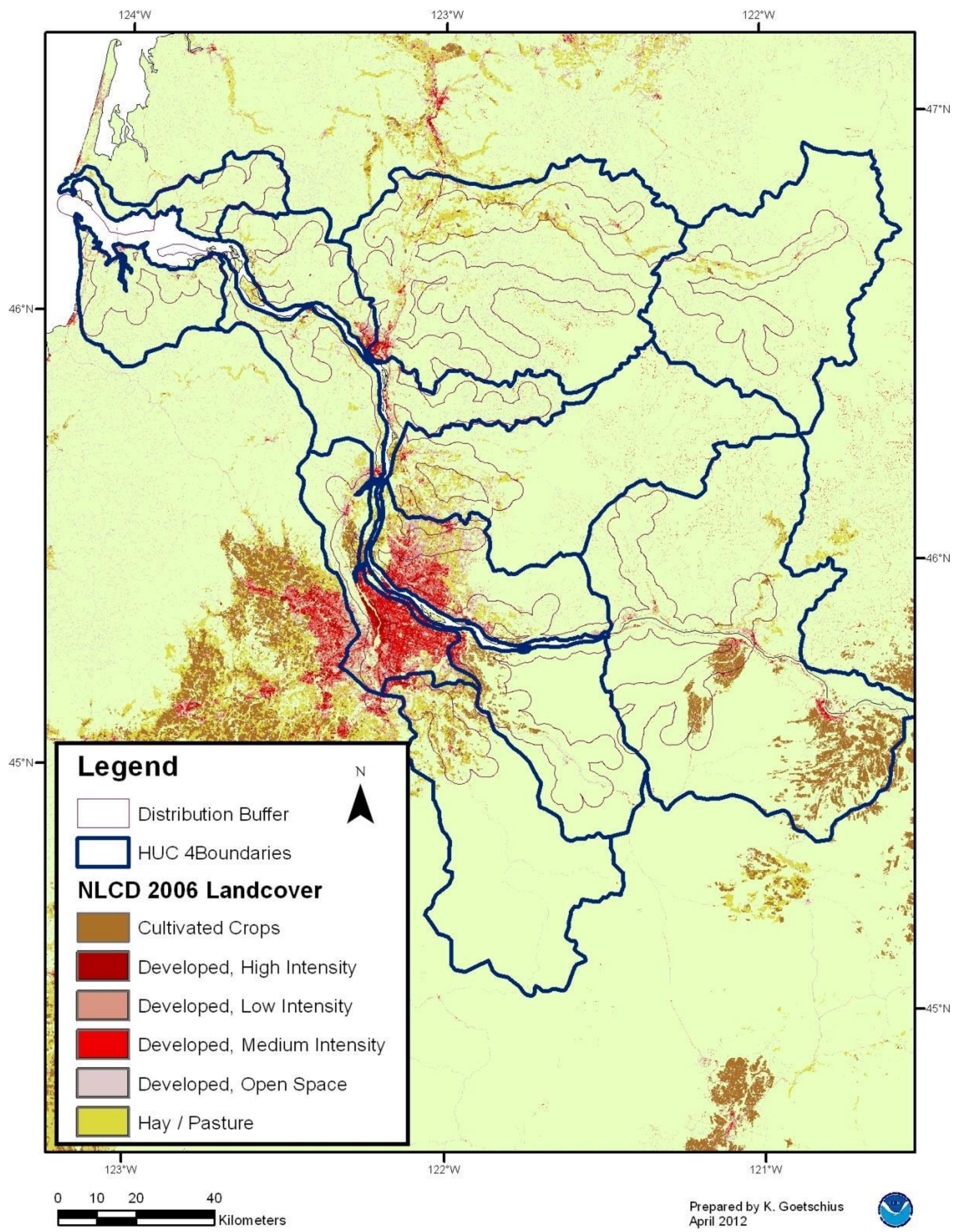
Puget Sound Chinook ESU

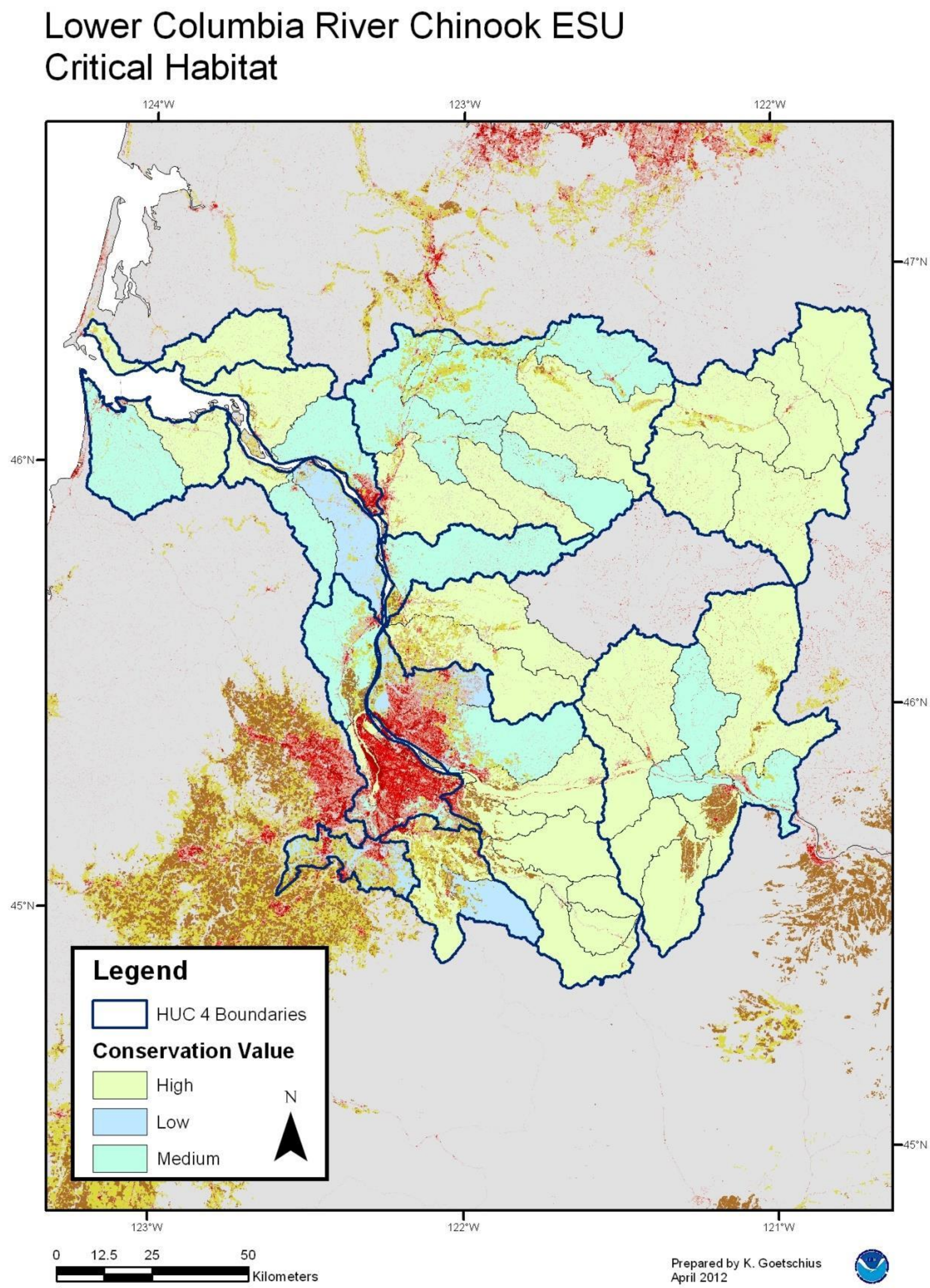


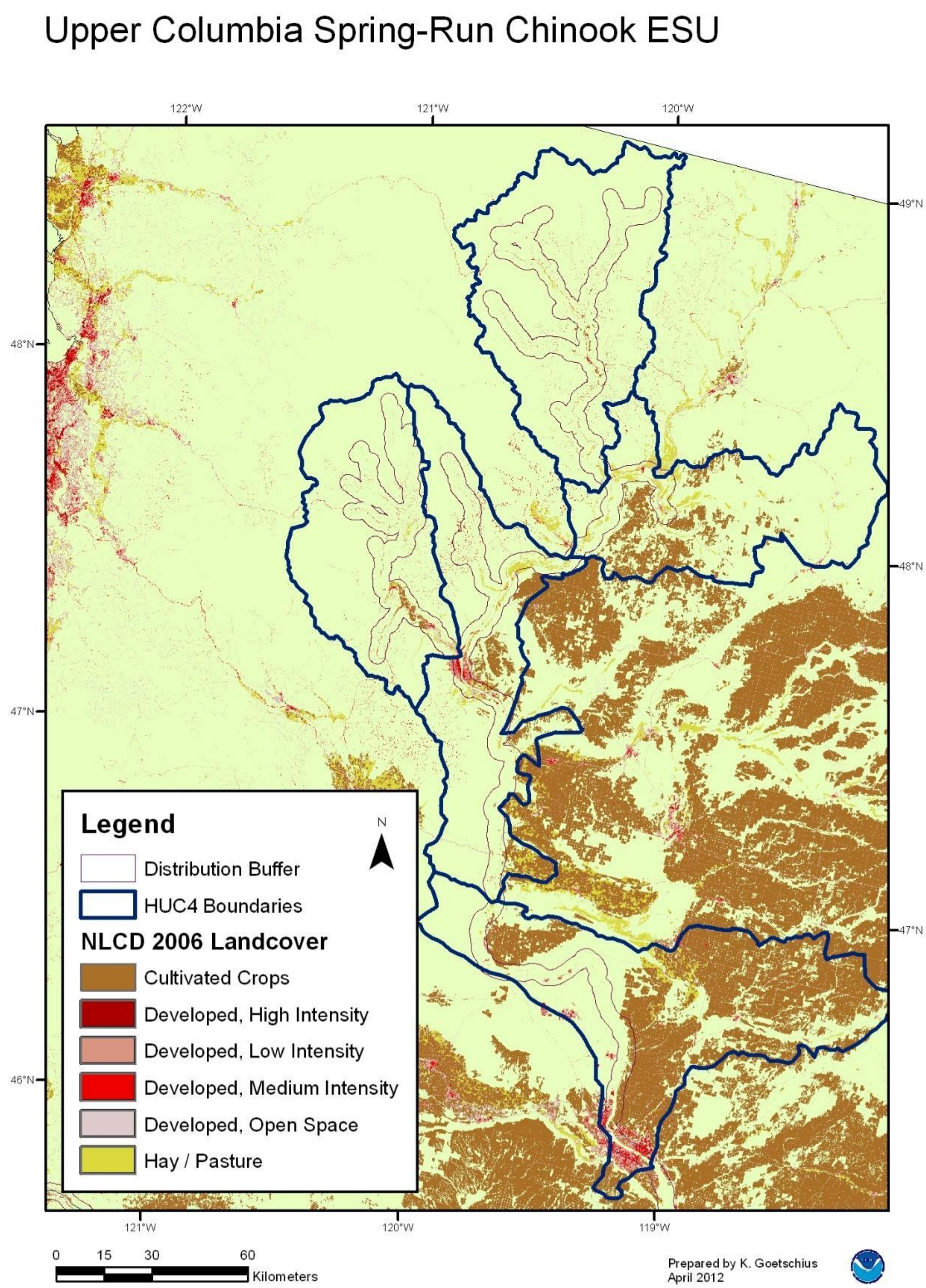
Puget Sound Chinook ESU
Critical Habbitat



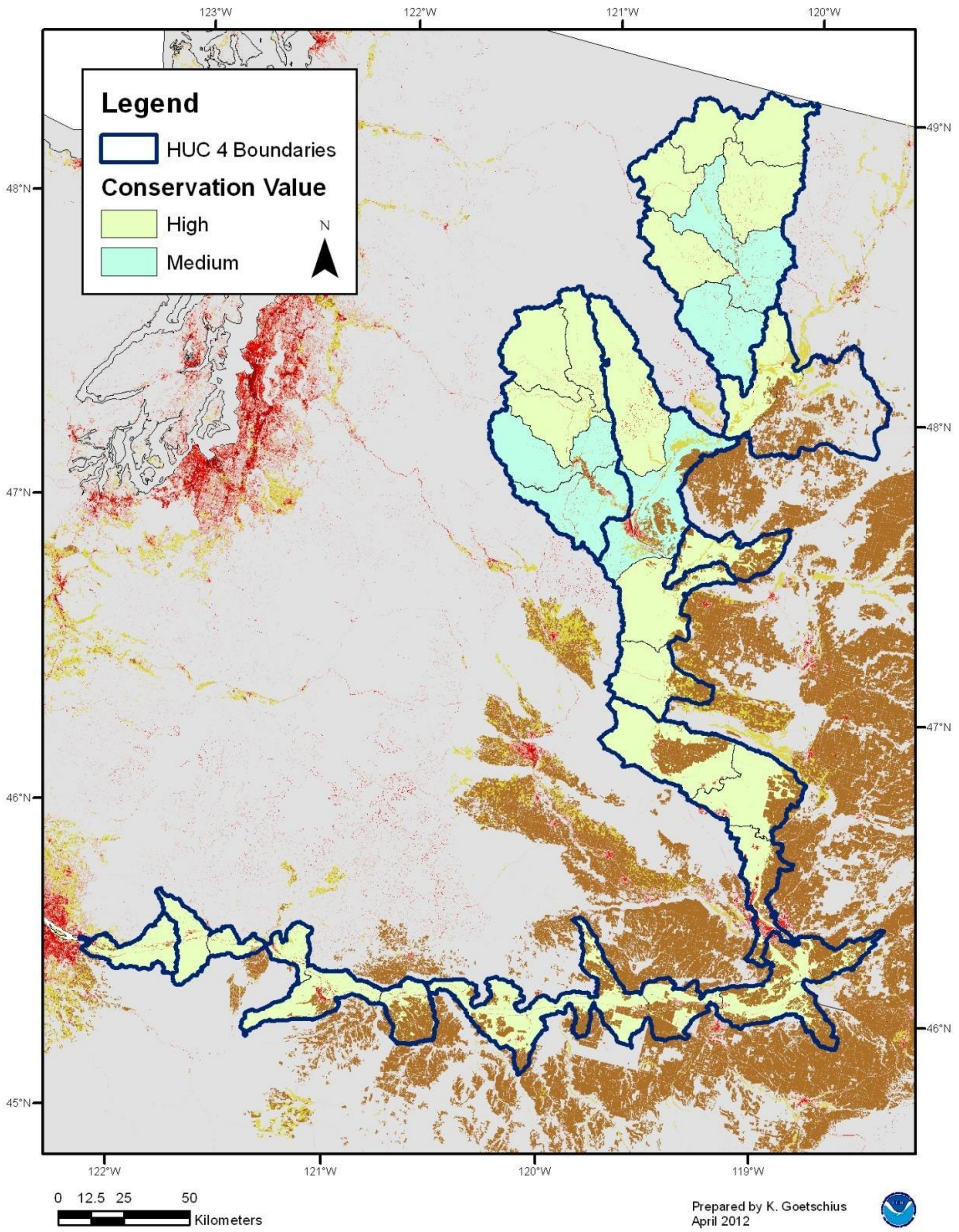
Lower Columbia River Chinook ESU



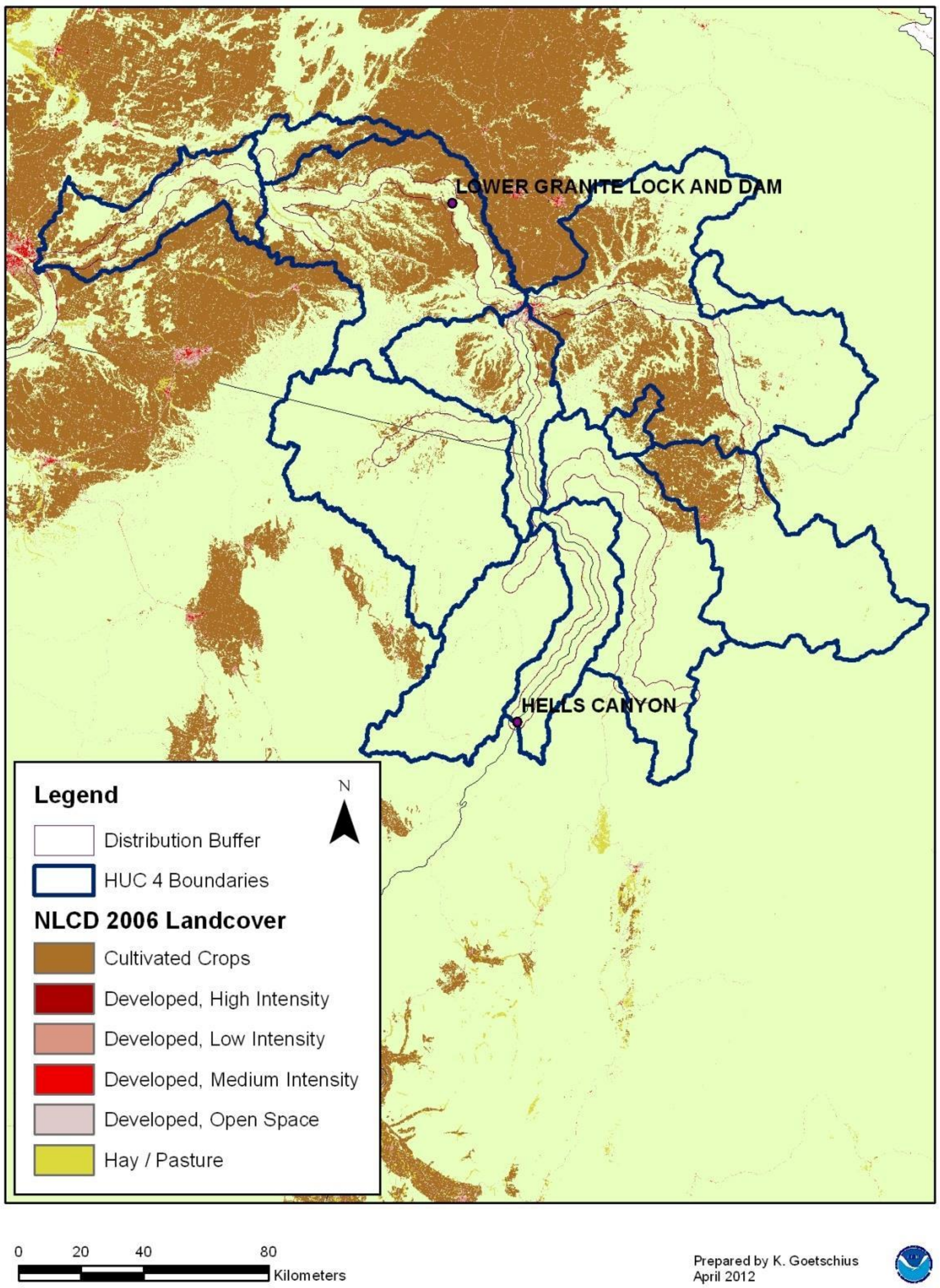

Lower Columbia River Chinook ESU
Critical Habitat
124°W
123°W
122°W
47°N
46°N
45°N
121°W
Legend
HUC 4 Boundaries
Conservation Value
High
Low
Medium
N
0 12.5 25 50
Kilometers
Prepared by K. Goetschius
April 2012

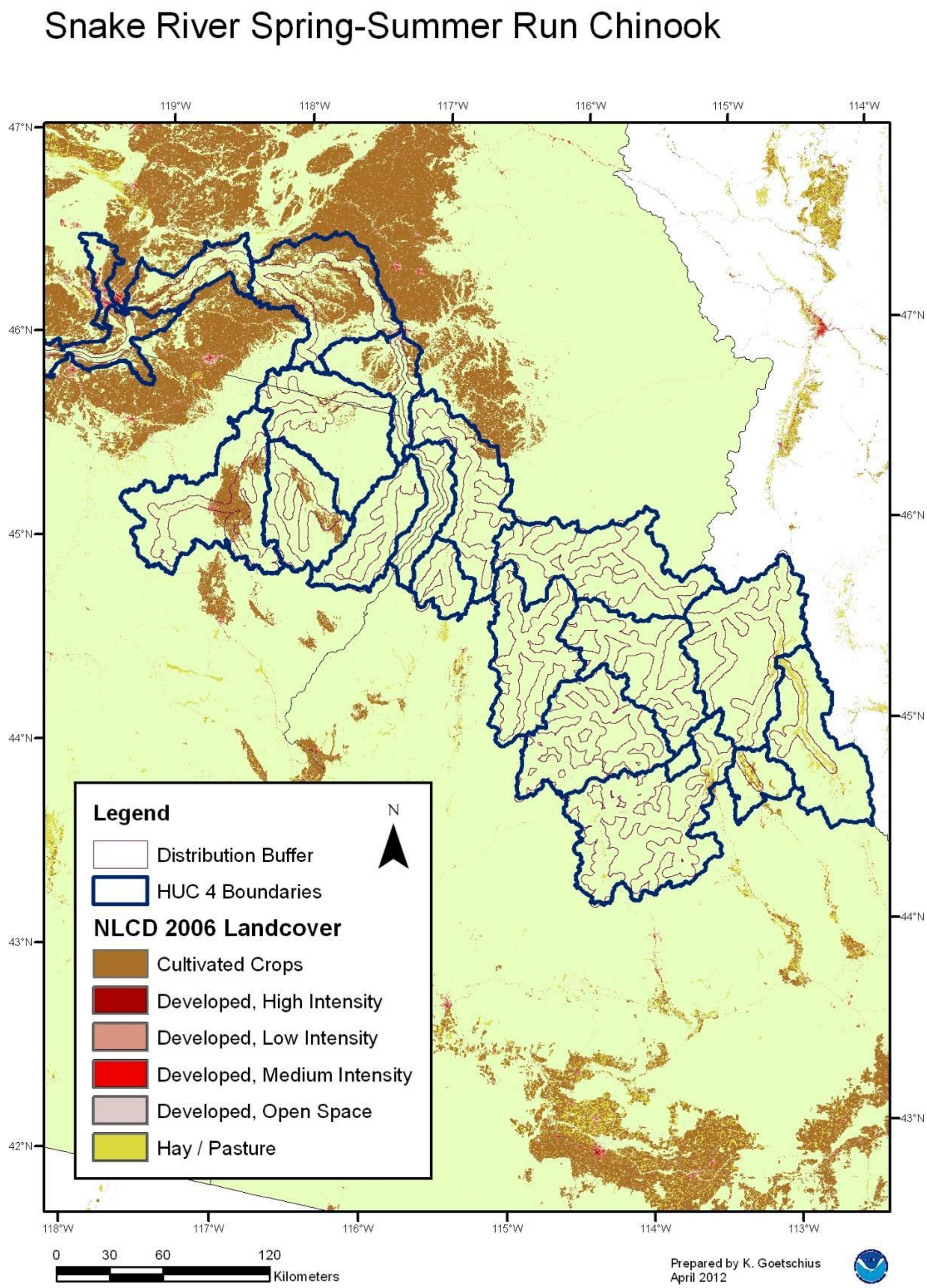

Upper Columbia Spring-Run Chinook ESU
122°W
121°W
120°W
49°N
48°N
47°N
46°N
119°W
Legend
N
Distribution Buffer
HUC4 Boundaries
NLCD 2006 Landcover
Cultivated Crops
Developed, High Intensity
Developed, Low Intensity
Developed, Medium Intensity
Developed, Open Space
Hay / Pasture
0 15 30 60
Kilometers
Prepared by K. Goetschius
April 2012

Upper Coulmbia River Chinook ESU
Critical Habitat

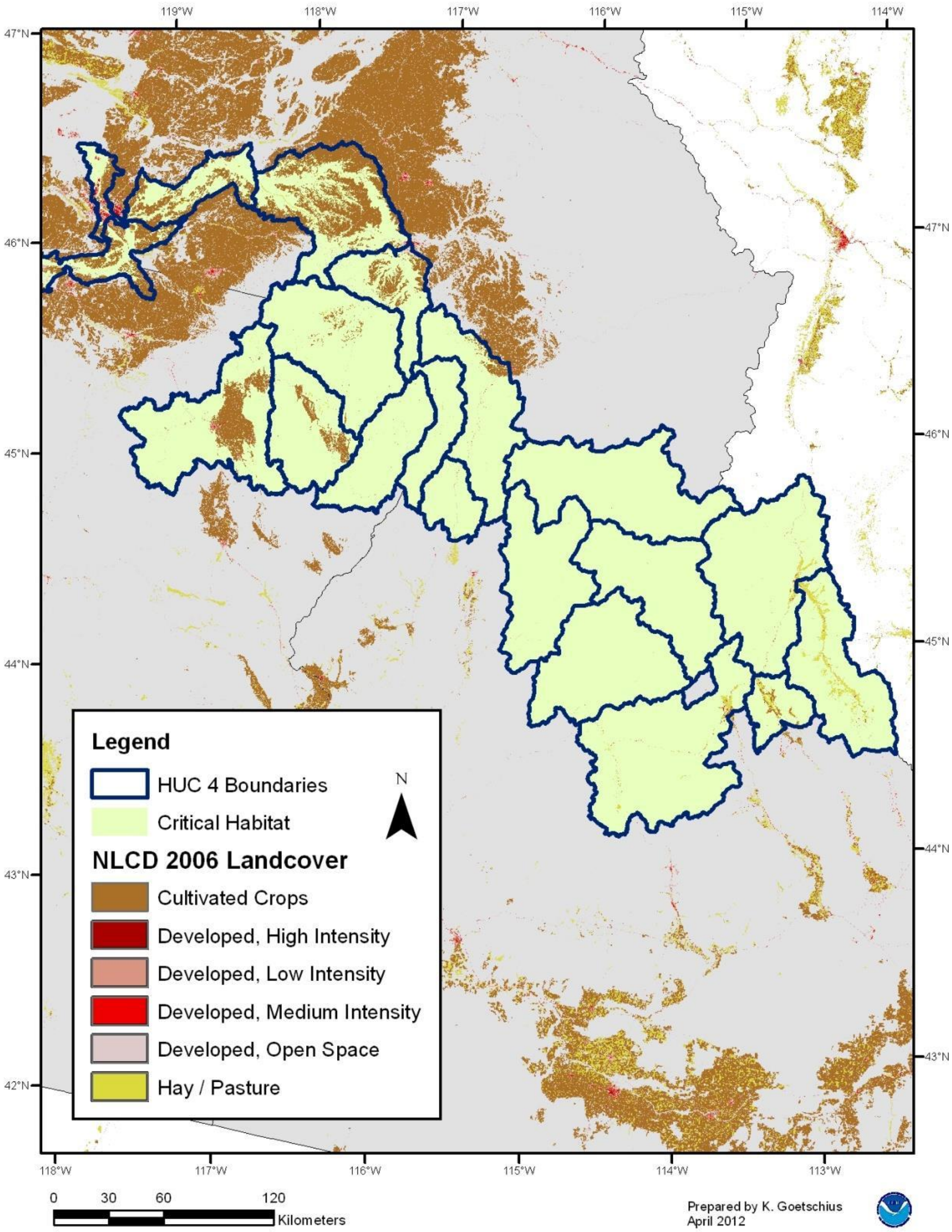


Snake River Fall Run Chinook ESU

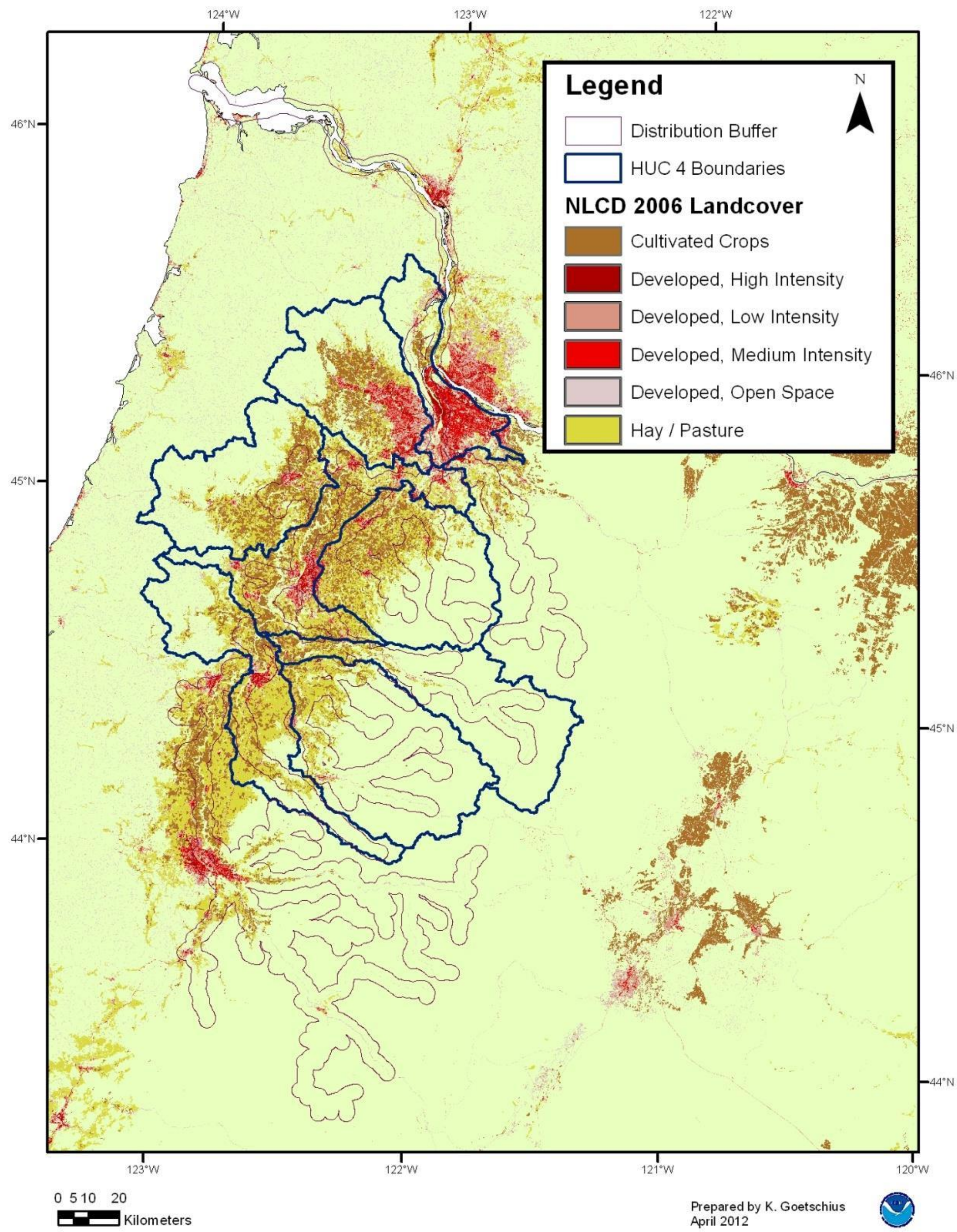


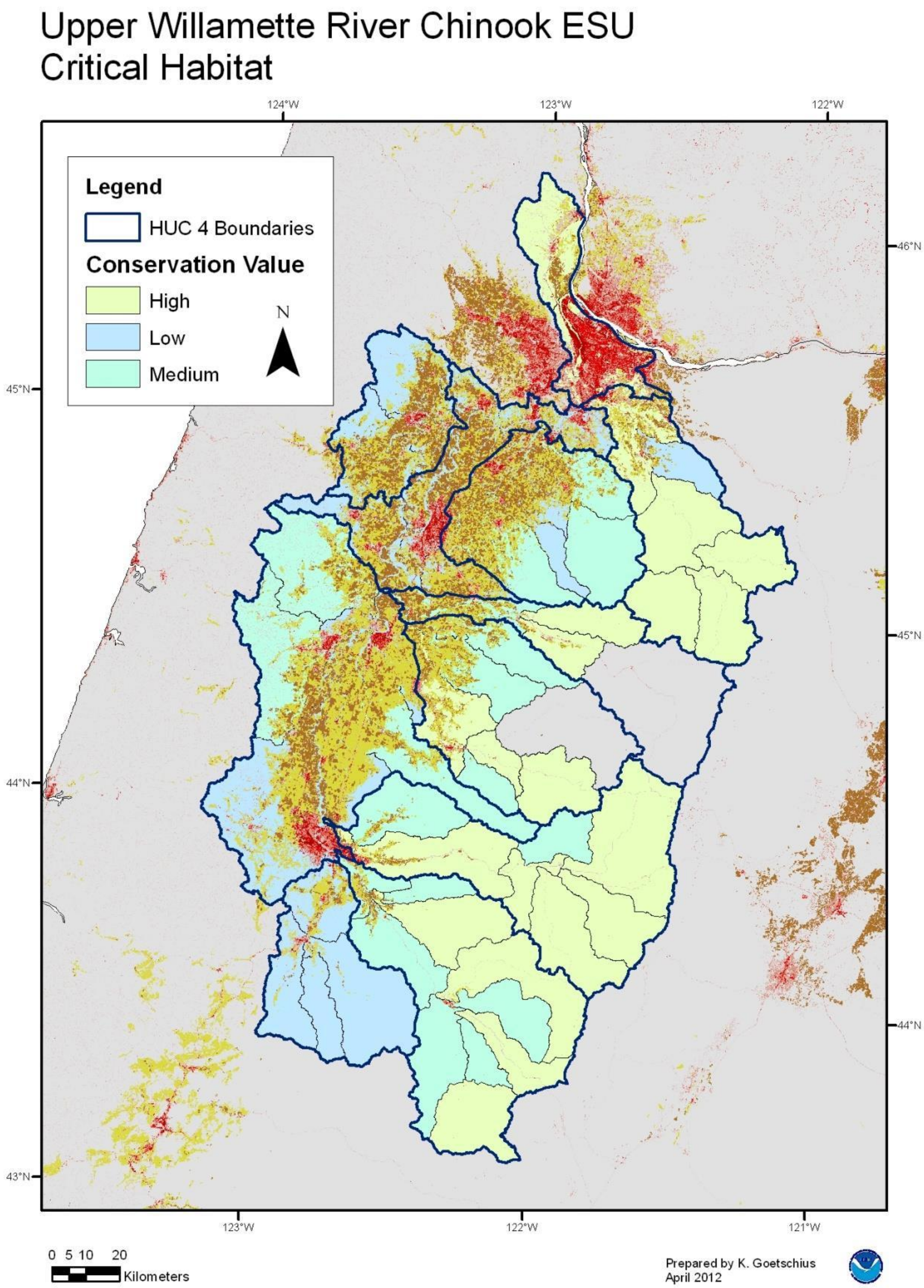

Snake River Spring-Summer Run Chinook
Legend
Distribution Buffer
HUC 4 Boundaries
NLCD 2006 Landcover
Cultivated Crops
Developed, High Intensity
Developed, Low Intensity
Developed, Medium Intensity
Developed, Open Space
Hay / Pasture
N
0 30 60 120
Kilometers
Prepared by K. Goetschius
April 2012

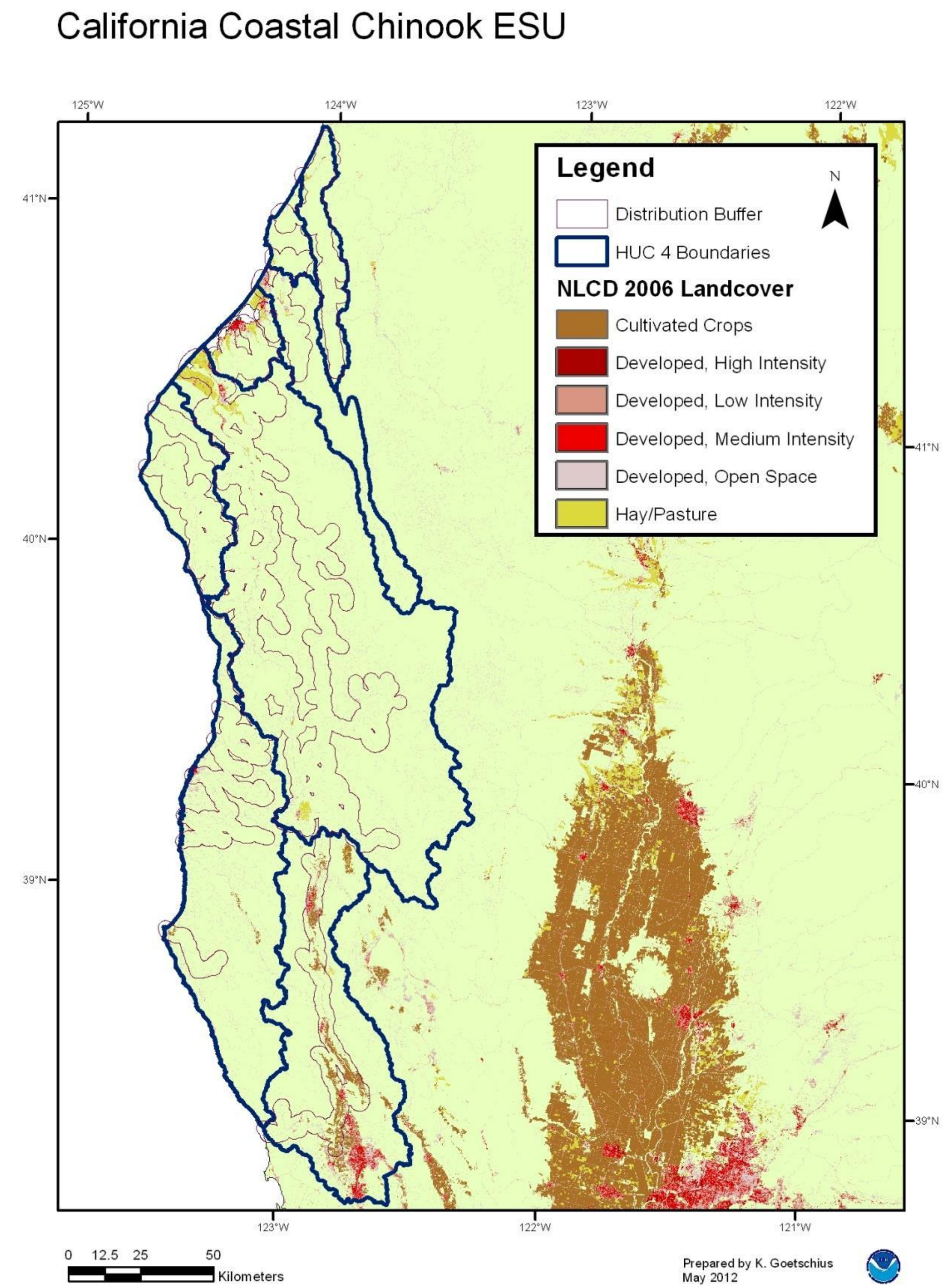
Snake River Spring-Summer Run Chinook Critical Habitat

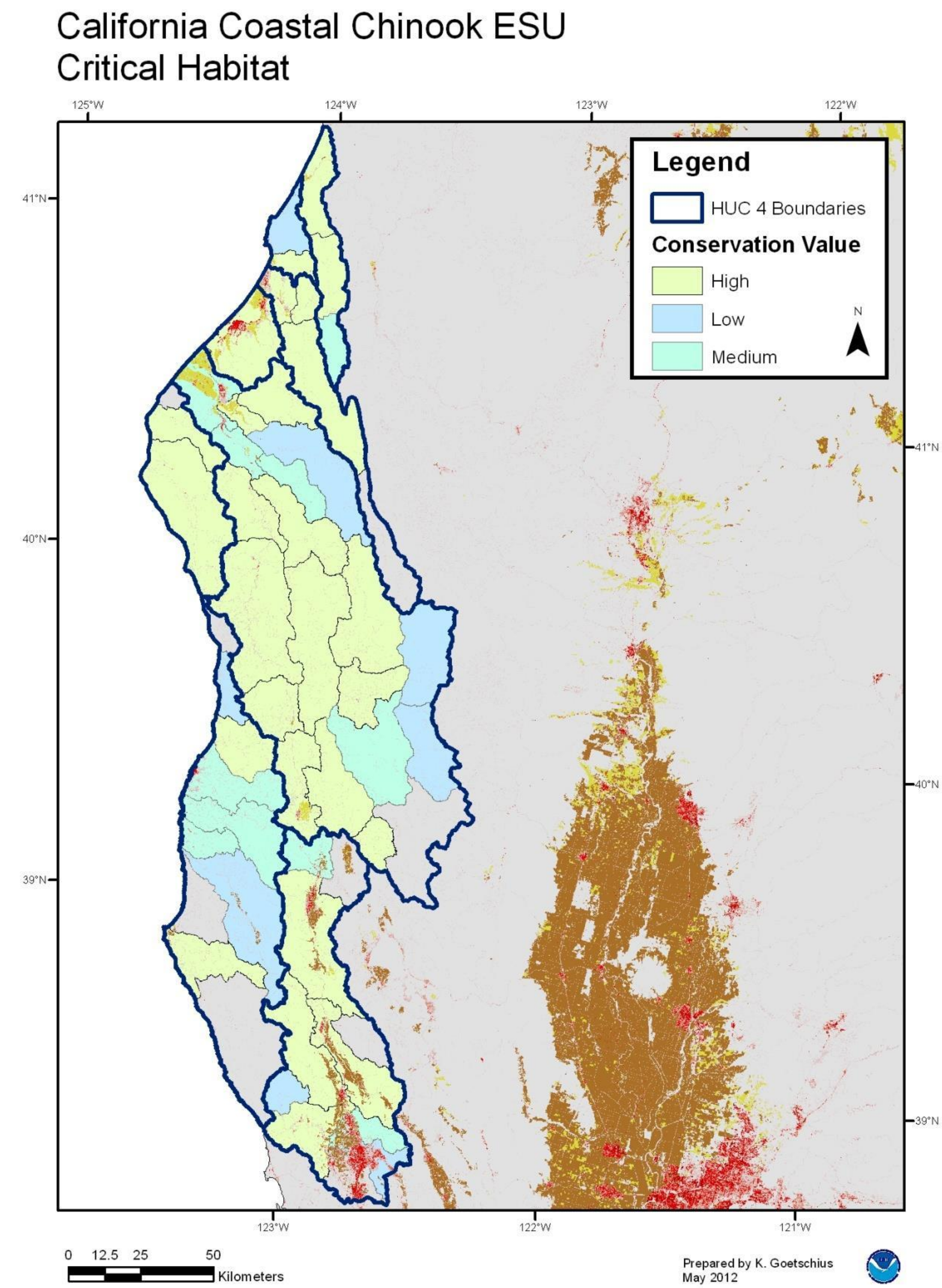


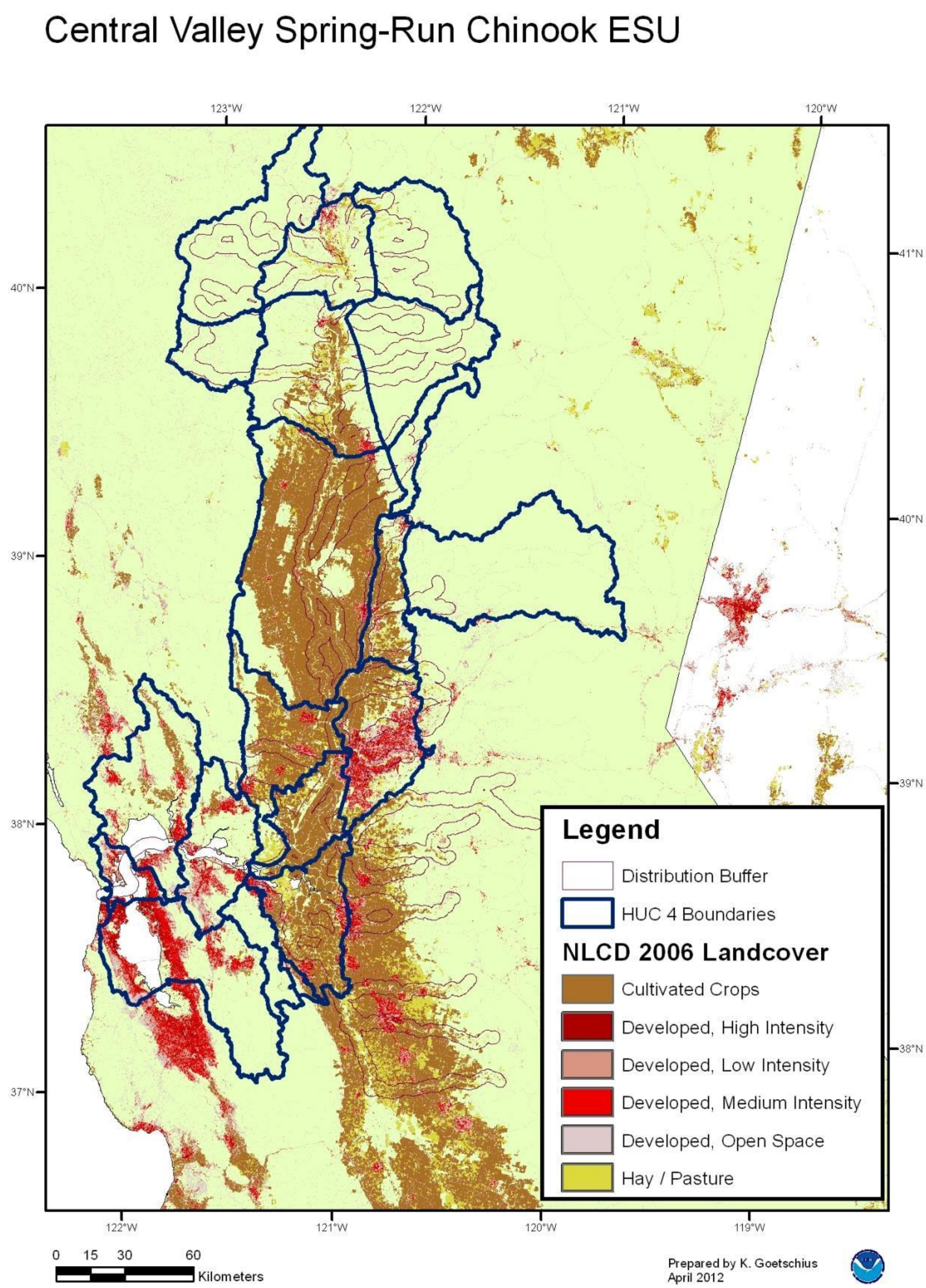
Upper Willamette River Chinook ESU




Upper Willamette River Chinook ESU
Critical Habitat
Legend
HUC 4 Boundaries
Conservation Value
High
Low
Medium
N
124°W
123°W
122°W
121°W
46°N
45°N
44°N
43°N
0 5 10 20
Kilometers
Prepared by K. Goetschius
April 2012

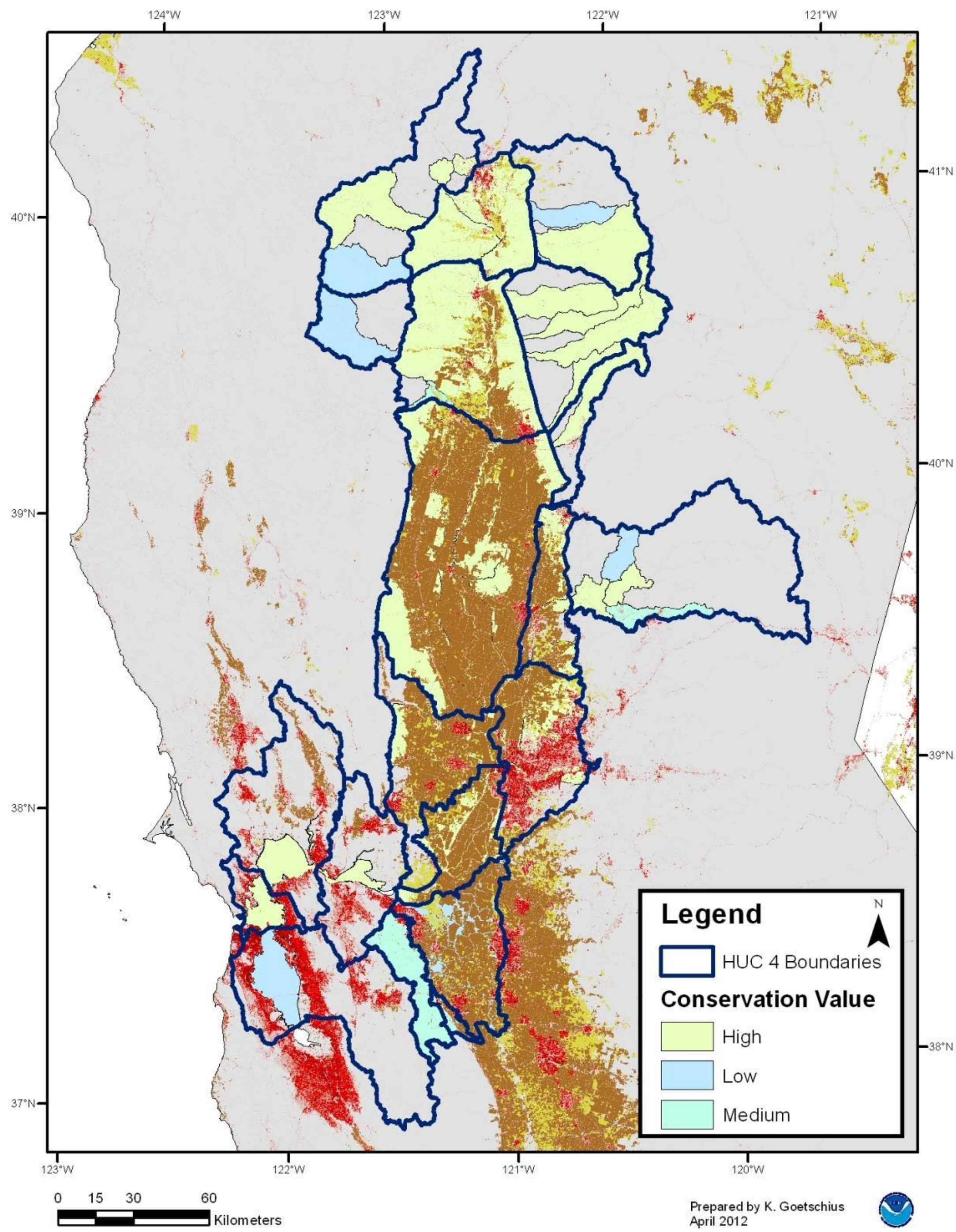

California Coastal Chinook ESU
Legend
N
Distribution Buffer
HUC 4 Boundaries
NLCD 2006 Landcover
Cultivated Crops
Developed, High Intensity
Developed, Low Intensity
Developed, Medium Intensity
Developed, Open Space
Hay/Pasture
125°W
124°W
123°W
122°W
121°W
41°N
40°N
39°N
0 12.5 25 50
Kilometers
Prepared by K. Goetschius
May 2012


California Coastal Chinook ESU
Critical Habitat
125°W
124°W
123°W
122°W
41°N
40°N
39°N
121°W
Legend
HUC 4 Boundaries
Conservation Value
High
Low
Medium
N
0 12.5 25 50
Kilometers
Prepared by K. Goetschius
May 2012



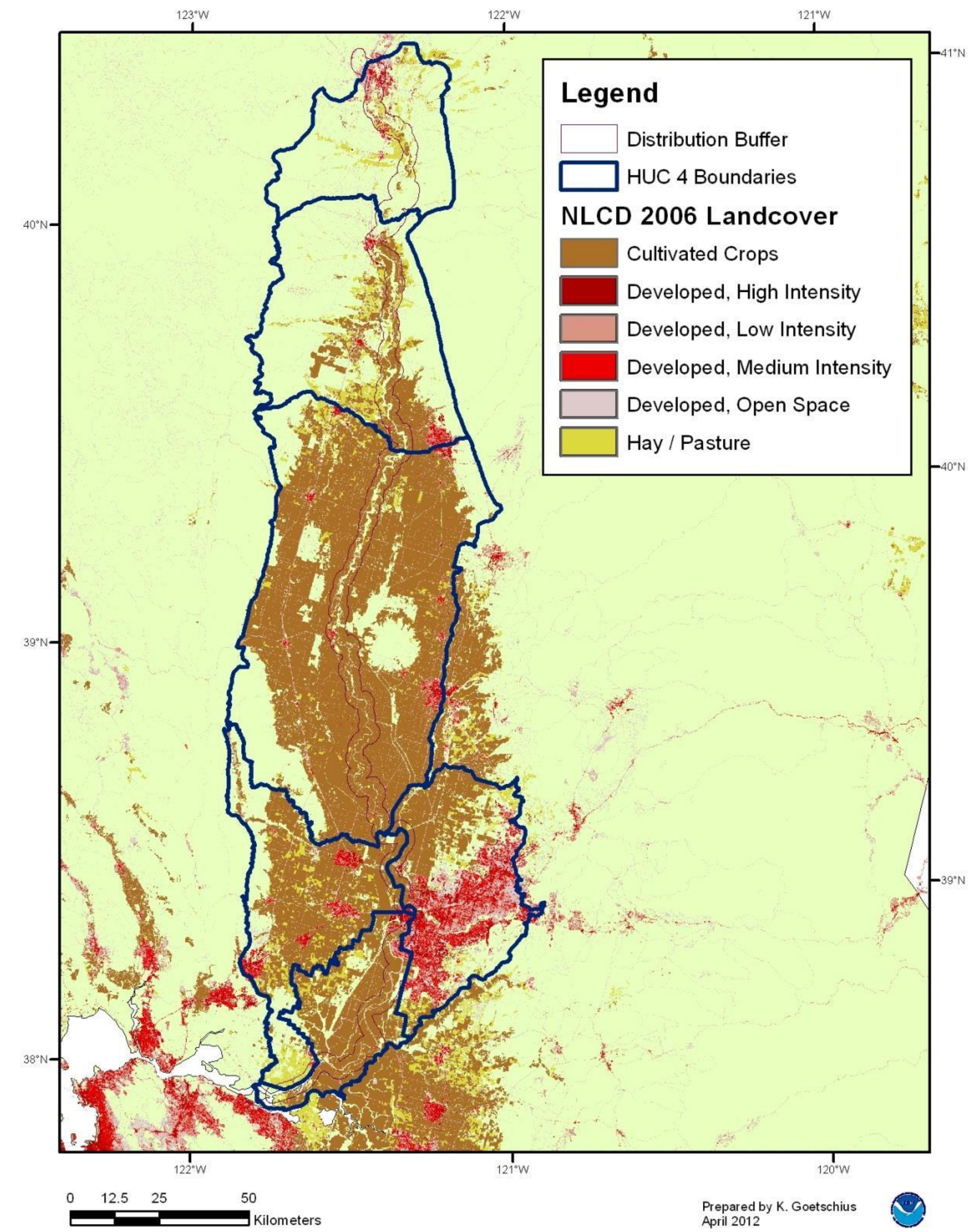
Central Valley Spring-Run Chinook ESU
123°W
122°W
121°W
120°W
41°N
40°N
39°N
38°N
37°N
122°W
121°W
120°W
119°W
Legend
Distribution Buffer
HUC 4 Boundaries
NLCD 2006 Landcover
Cultivated Crops
Developed, High Intensity
Developed, Low Intensity
Developed, Medium Intensity
Developed, Open Space
Hay / Pasture
0 15 30 60
Kilometers
Prepared by K. Goetschius
April 2012

Central Valley Spring-Run Chinook ESU
Critical Habitat

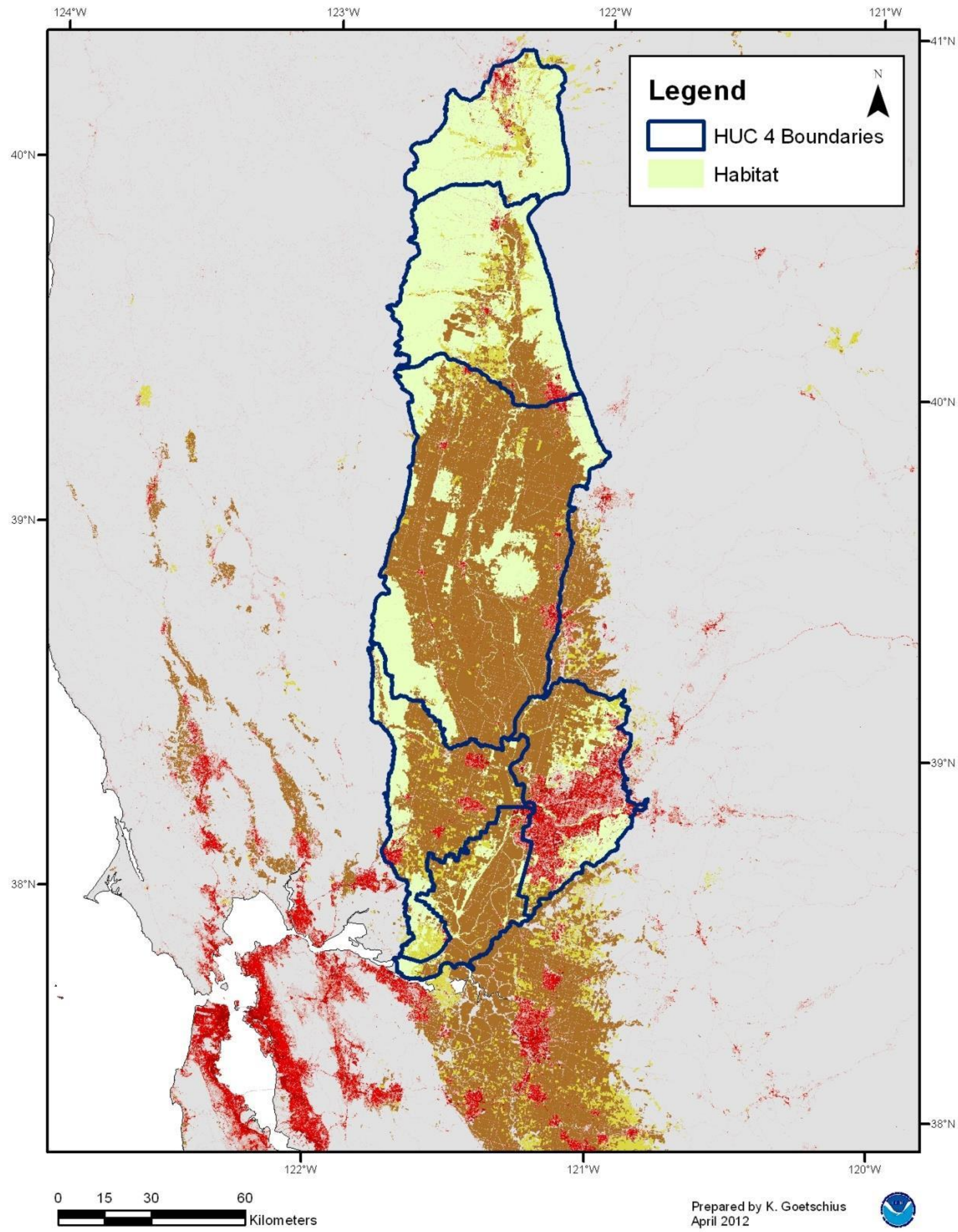


Prepared by K. Goetschius
April 2012

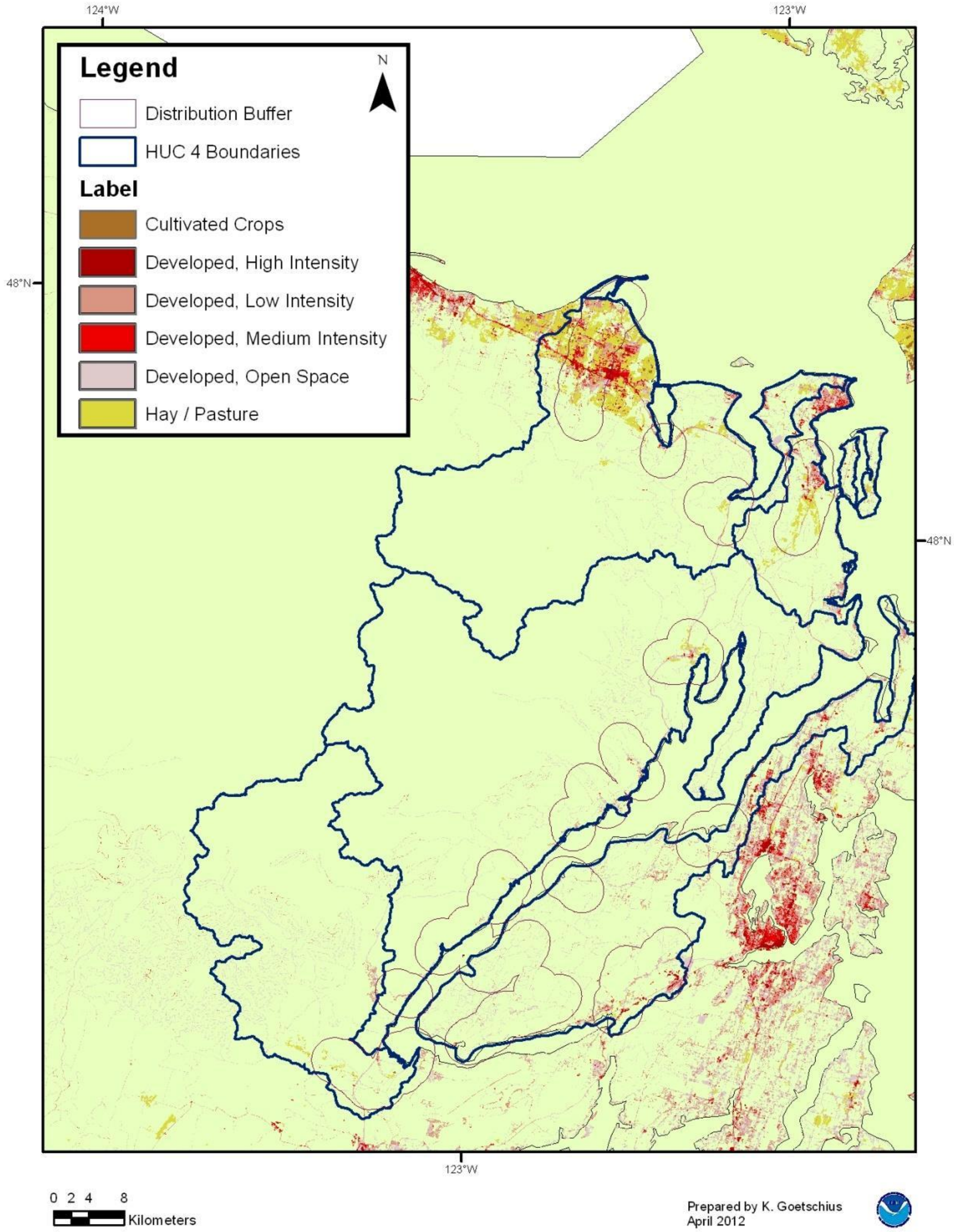
Sacramento River Winter Run Chinook ESU



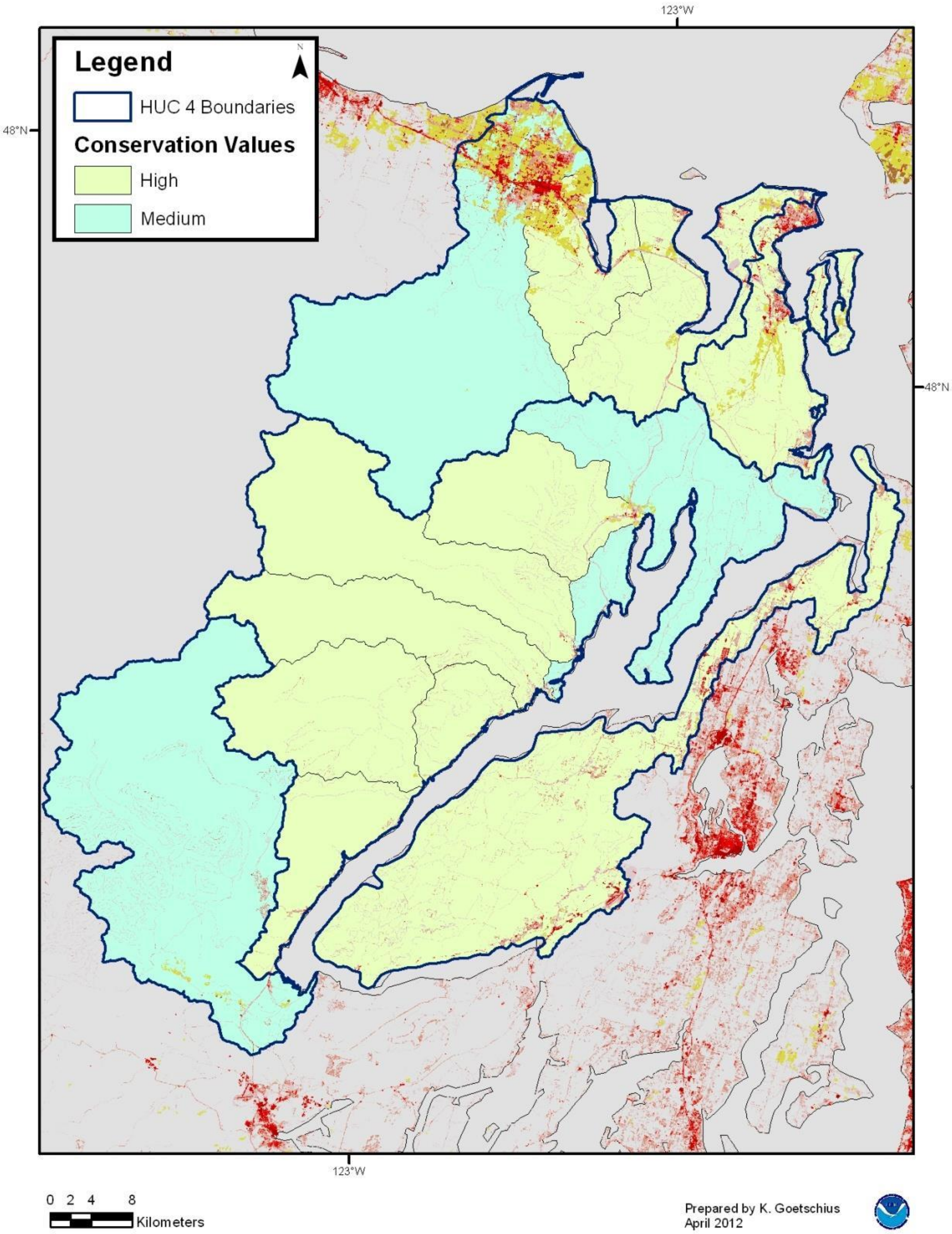
Sacramento River Winter Run Chinook ESU
Critical Habitat

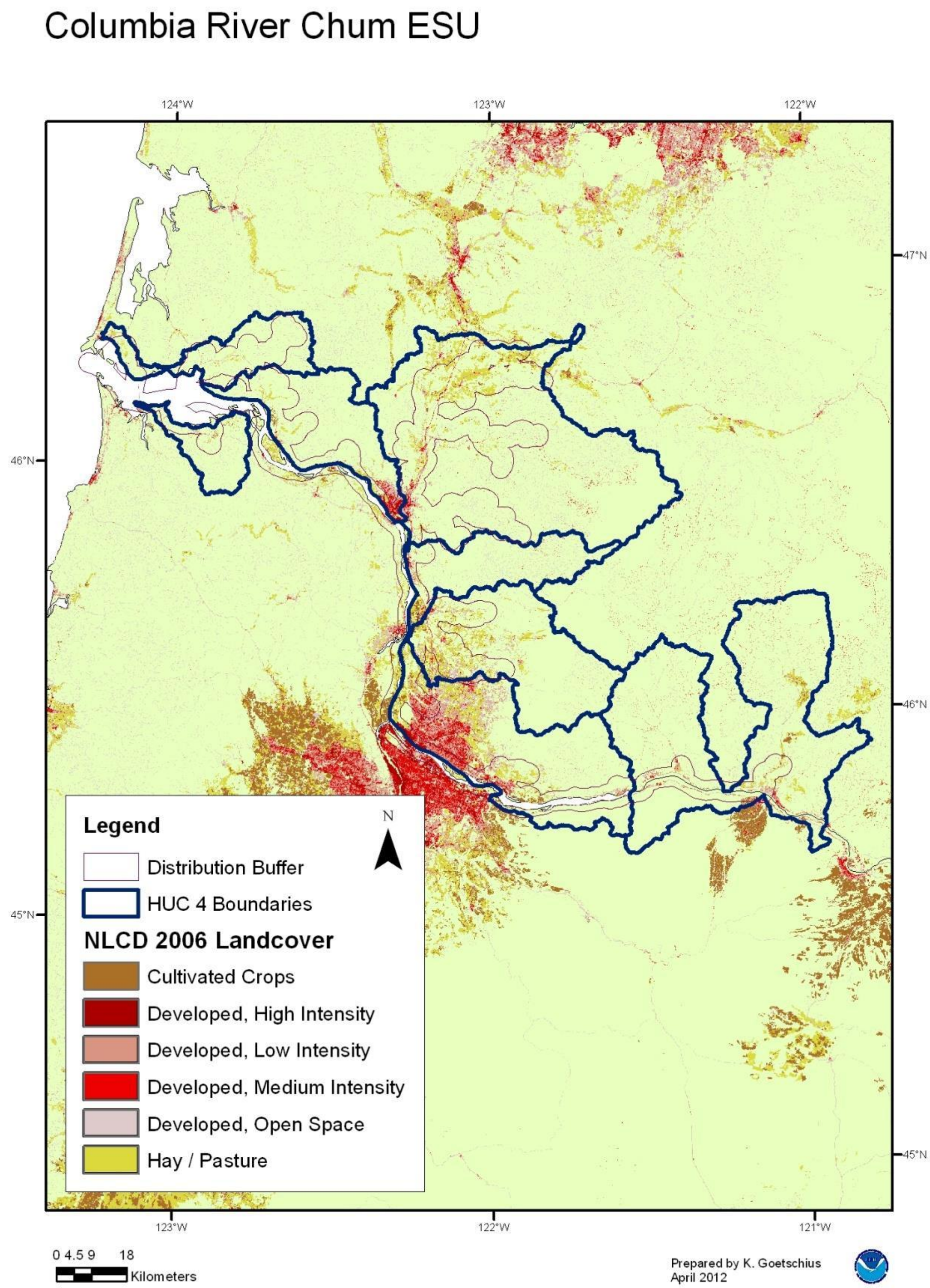


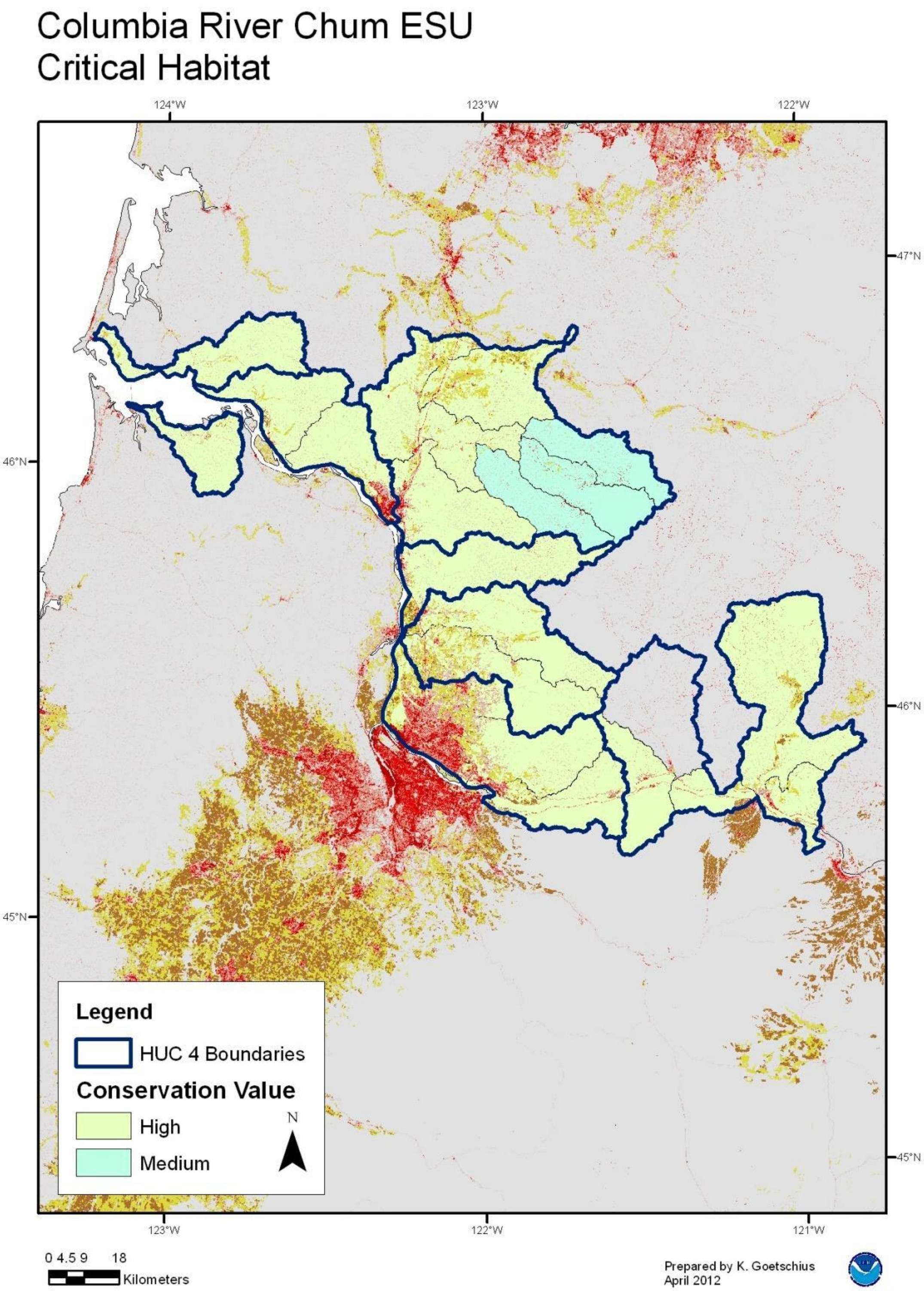
Hood Canal Summer-Run Chum ESU



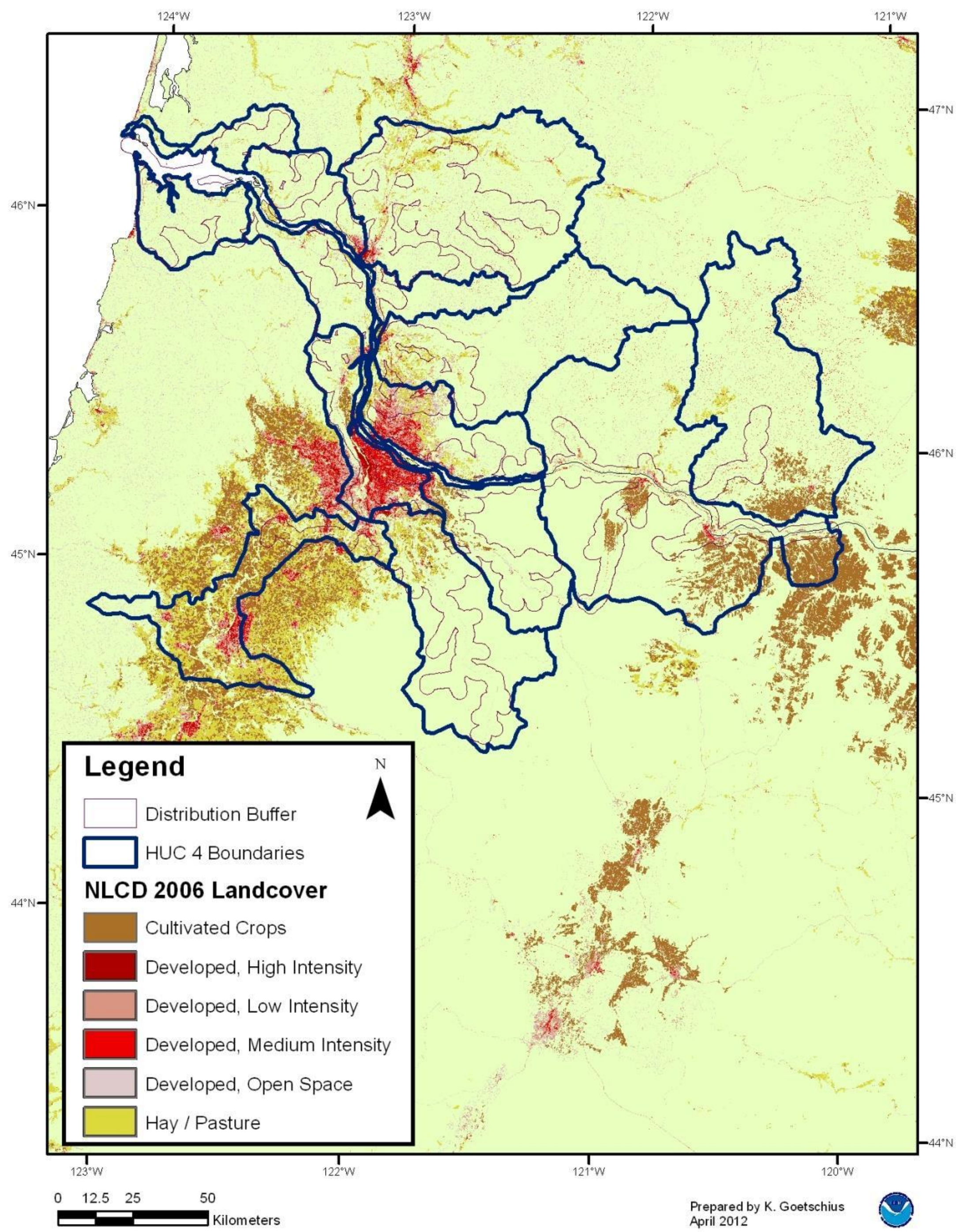
Hood Canal Summer-Run Chum ESU Critical Habitat

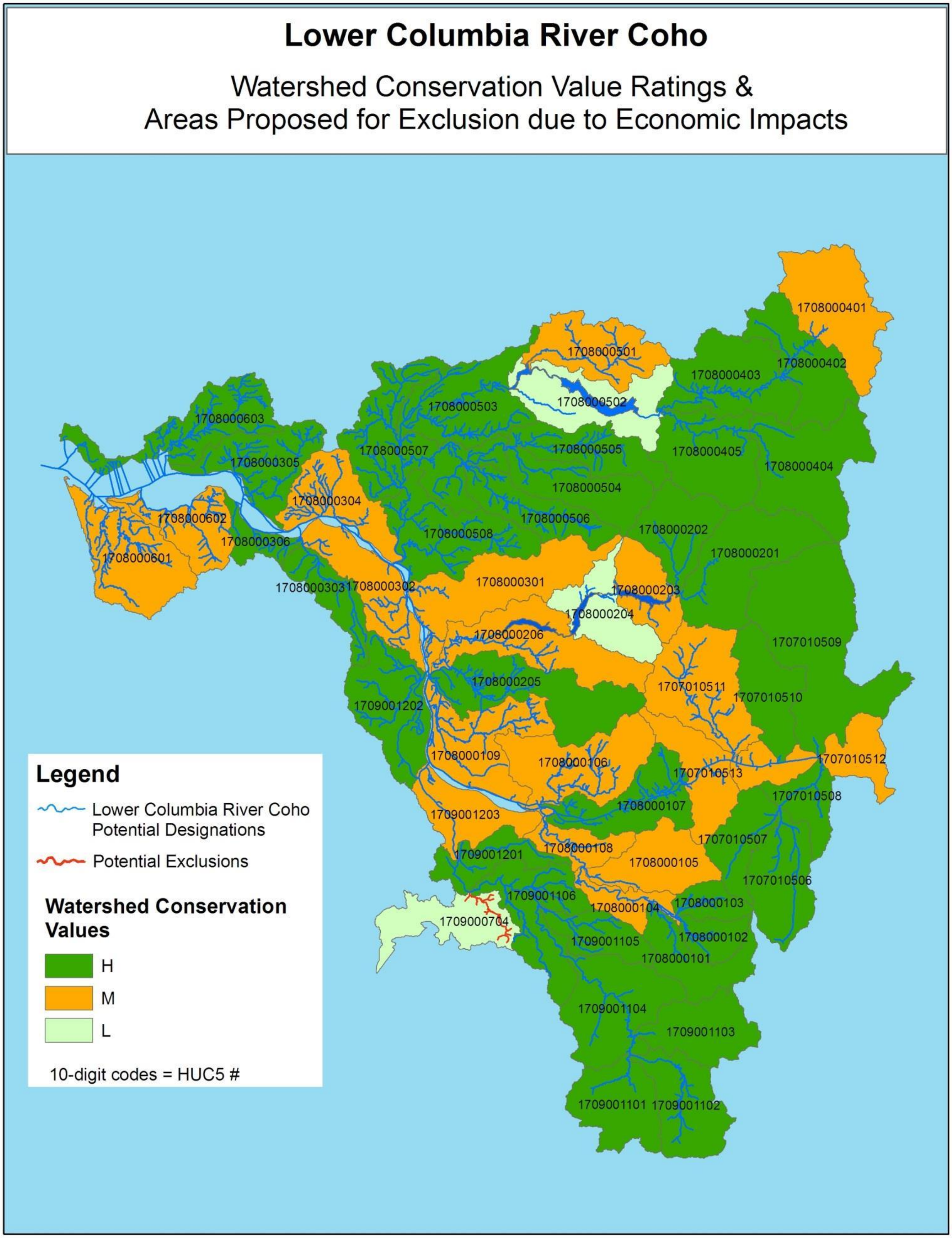



Columbia River Chum ESU
124°W
123°W
122°W
47°N
46°N
45°N
121°W
Legend
Distribution Buffer
HUC 4 Boundaries
NLCD 2006 Landcover
Cultivated Crops
Developed, High Intensity
Developed, Low Intensity
Developed, Medium Intensity
Developed, Open Space
Hay / Pasture
N
0 4.5 9 18
Kilometers
Prepared by K. Goetschius
April 2012


Columbia River Chum ESU
Critical Habitat
124°W
123°W
122°W
47°N
46°N
45°N
121°W
Legend
HUC 4 Boundaries
Conservation Value
High
Medium
N
0 4.5 9 18
Kilometers
Prepared by K. Goetschius
April 2012

Lower Columbia River Coho ESU





Lower Columbia River Coho
Watershed Conservation Value Ratings &
Areas Proposed for Exclusion due to Economic Impacts
Legend
Lower Columbia River Coho Potential Designations
Potential Exclusions
Watershed Conservation Values
H
M
L
10-digit codes = HUC5 #
1708000401
1708000501
1708000402
1708000403
1708000502
1708000503
1708000603
1708000507
1708000505
1708000405
1708000305
1708000404
1708000504
1708000304
1708000602
1708000506
1708000202
1708000306
1708000508
1708000201
1708000601
1708000303
1708000302
1708000301
1708000203
1708000204
1708000206
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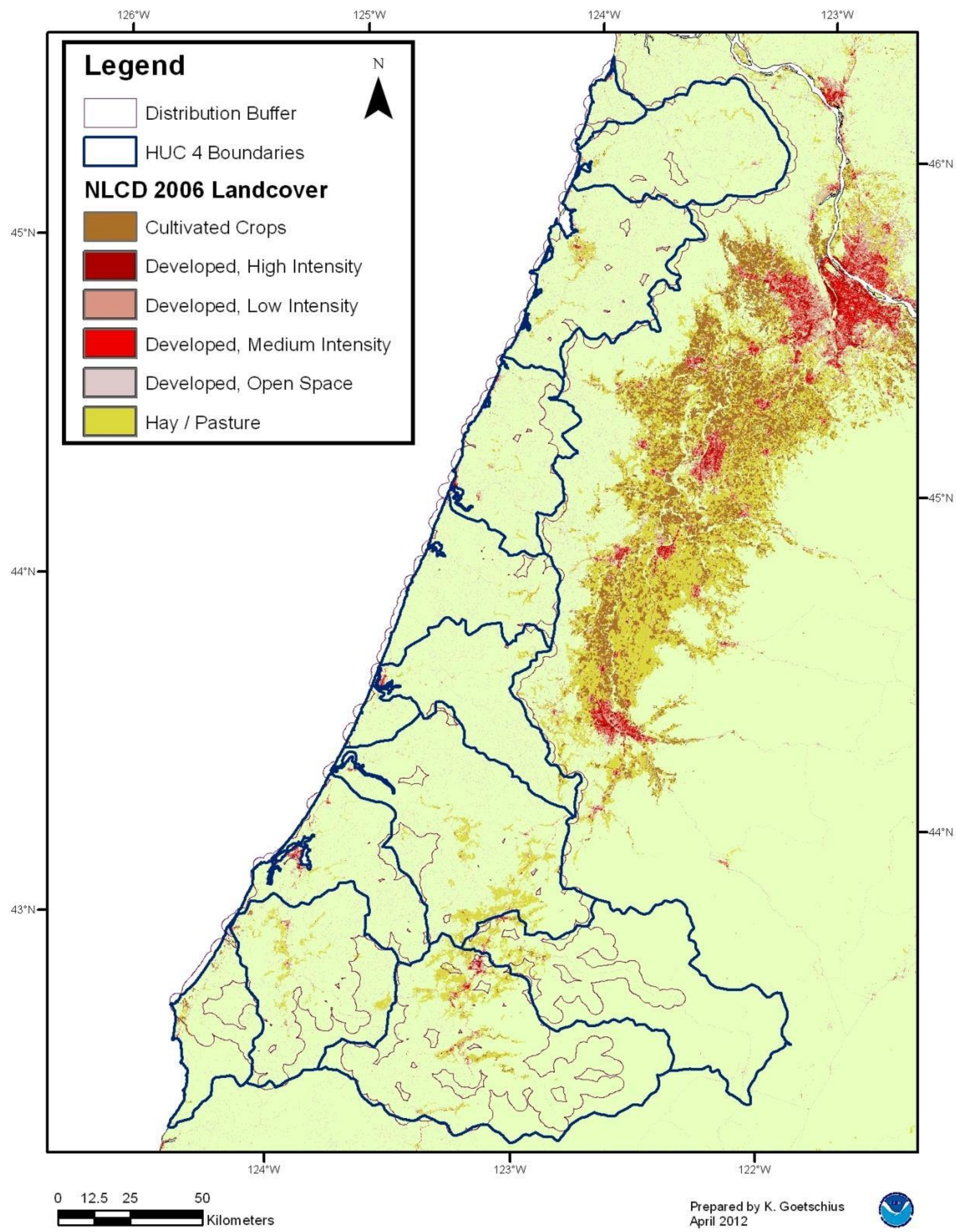
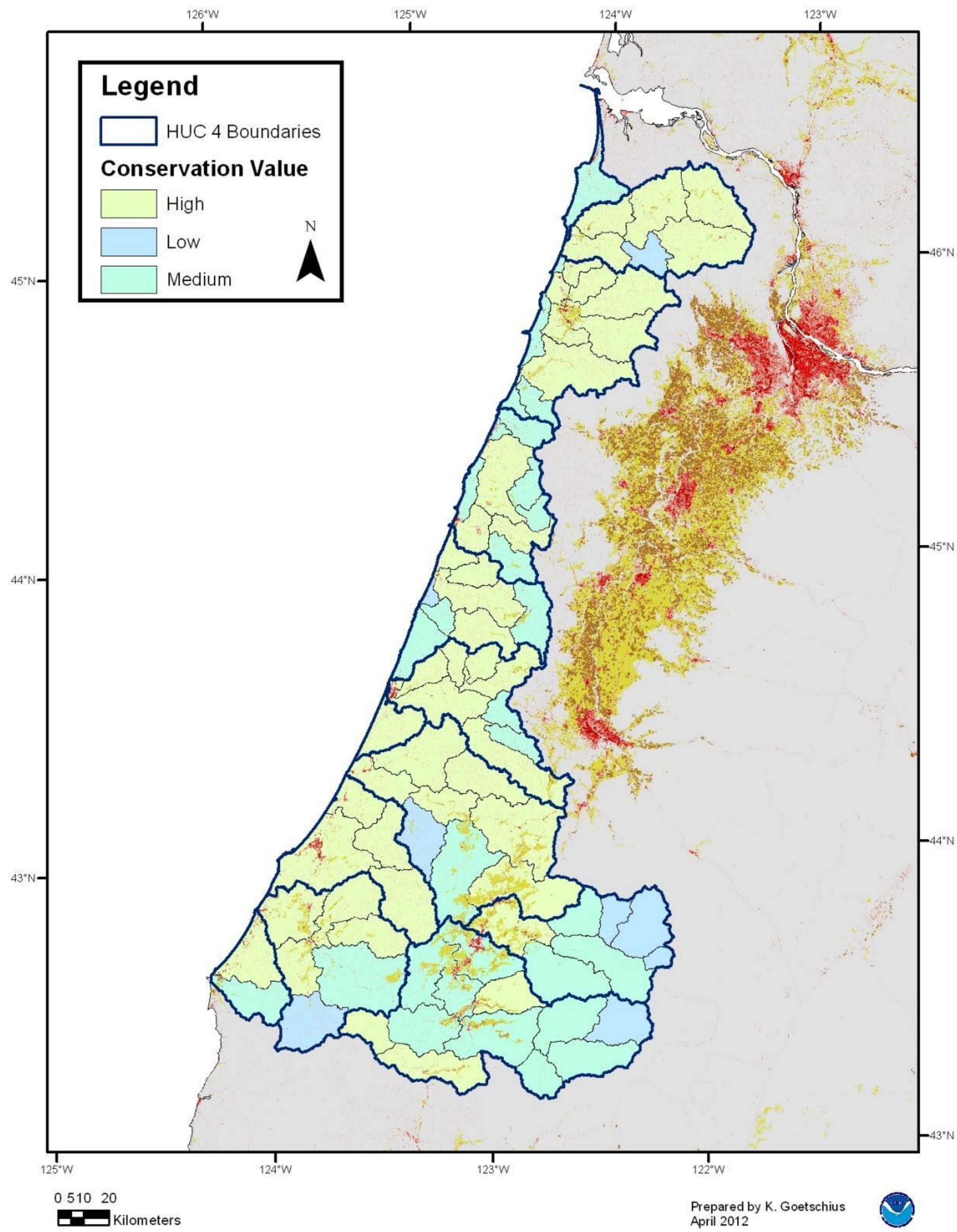
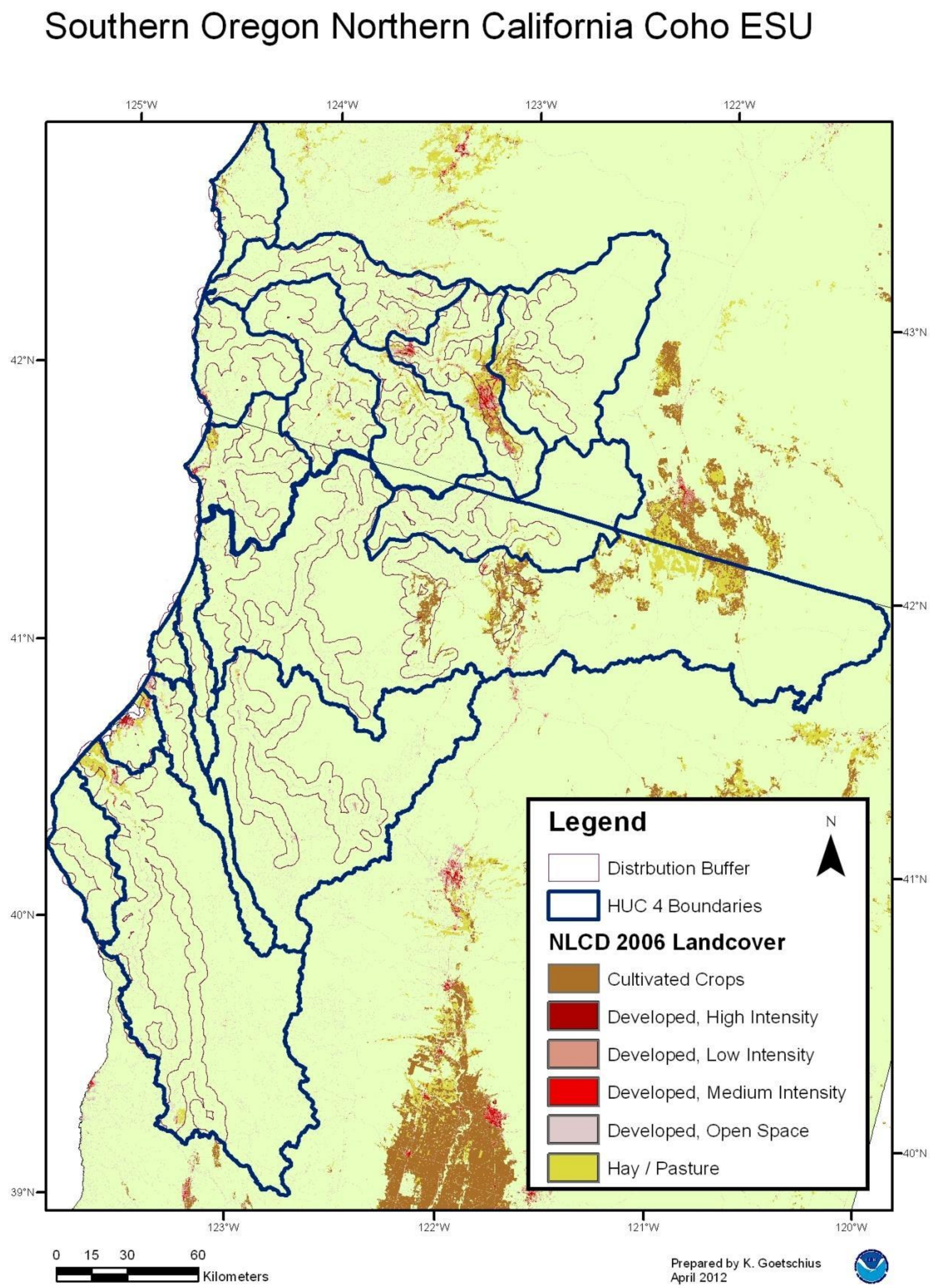
1708000205
1707010511
1707010510
1709001202
1708000109
1708000106
1707010512
1707010513
1707010508
1708000107
1709001203
1707010507
1709001201
1708000108
1708000105
1707010506
1709001106
1708000104
1708000103
1709000704
1708000102
1709001105
1708000101
1709001104
1709001103
1709001101
1709001102

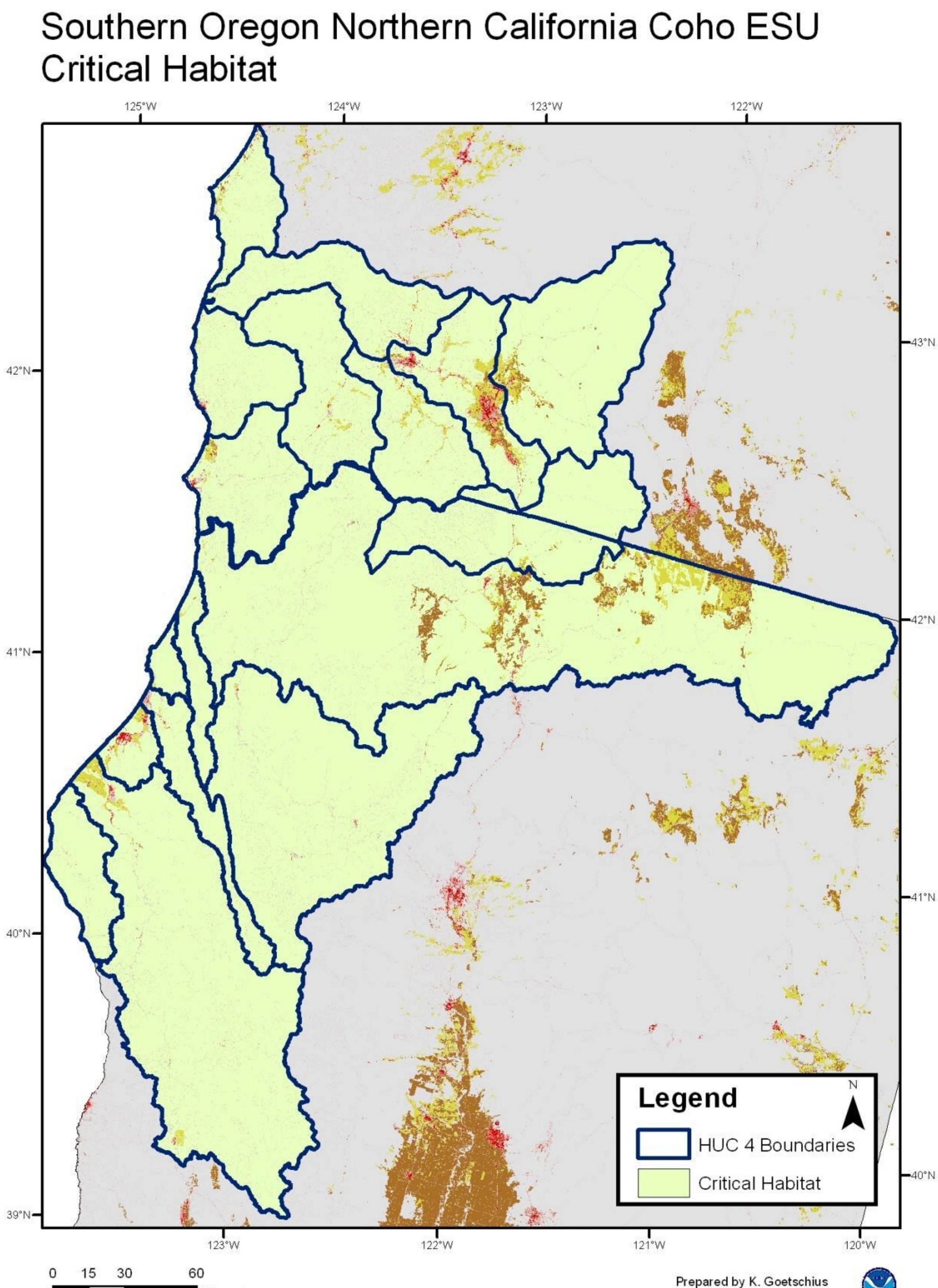
Oregon Coast Coho ESU



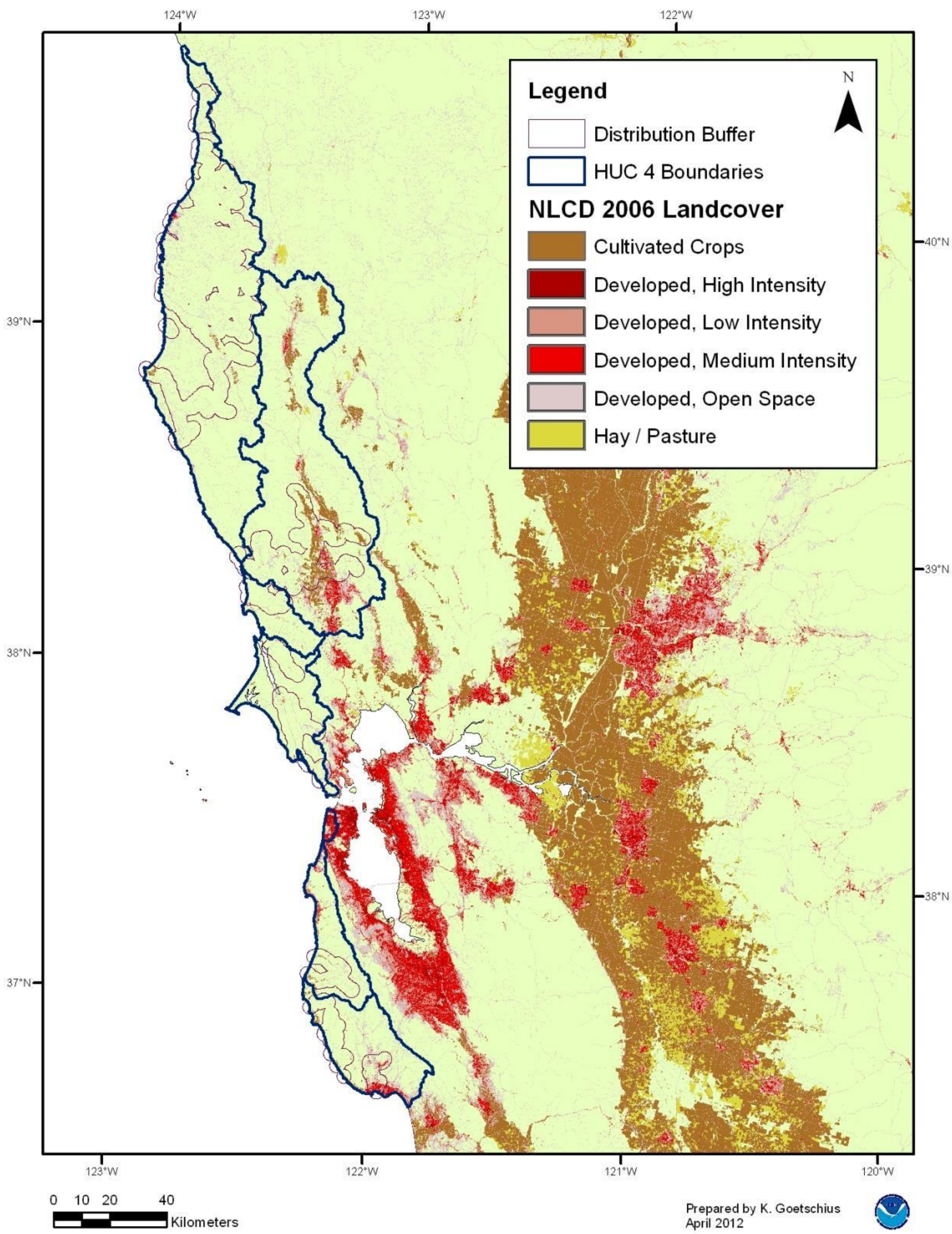
Oregon Coast Coho ESU

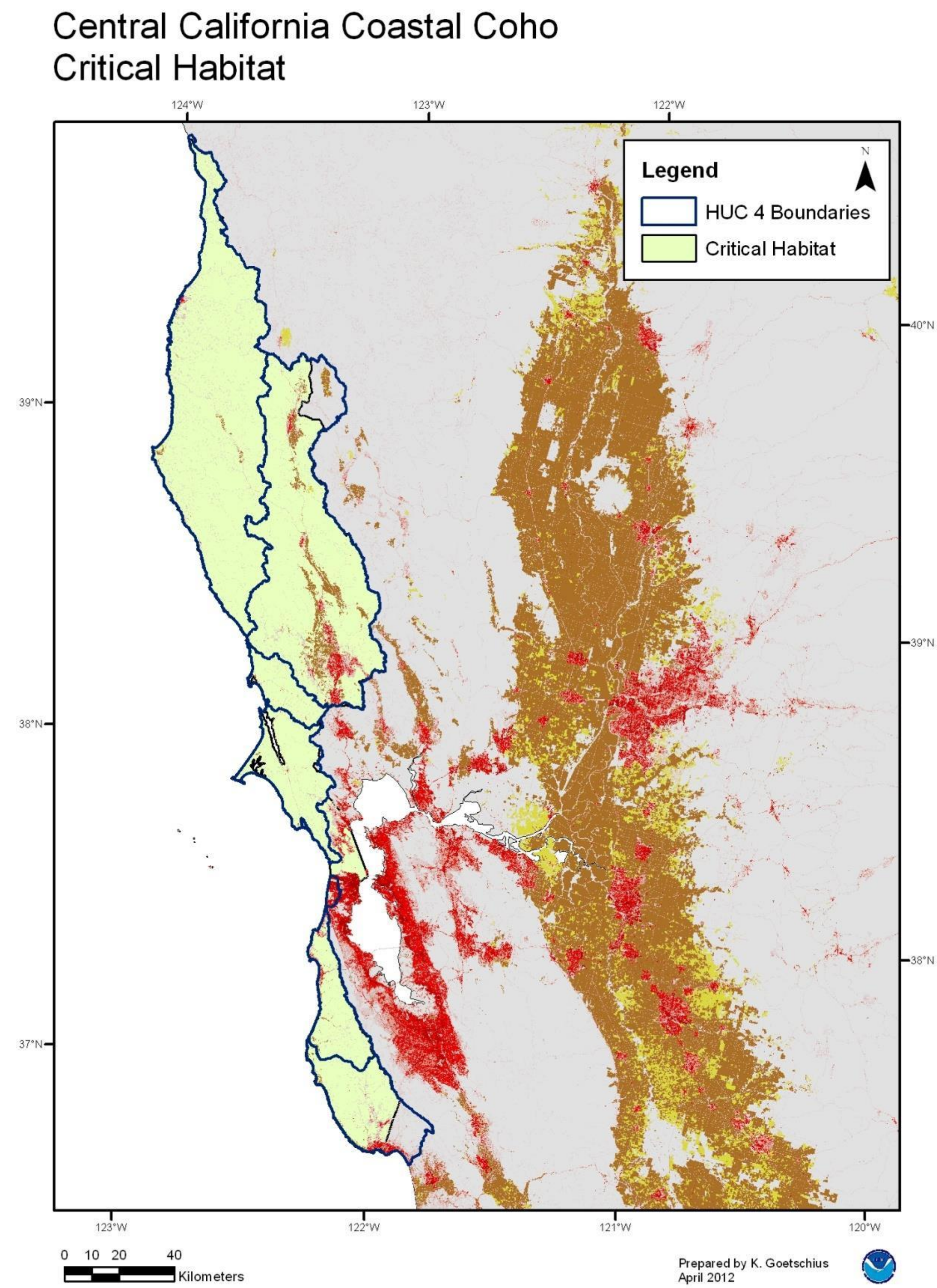


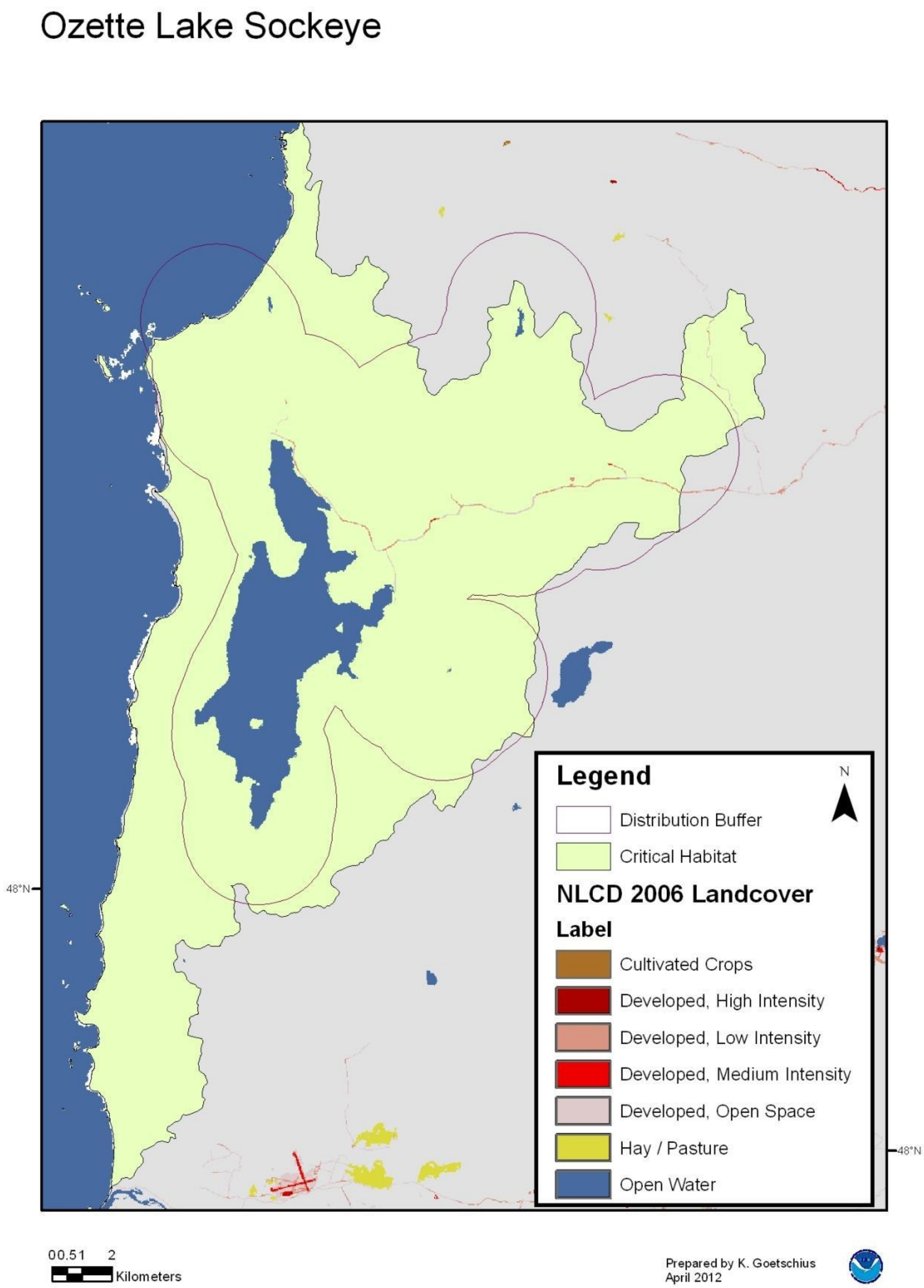

Southern Oregon Northern California Coho ESU
125°W
124°W
123°W
122°W
43°N
42°N
41°N
40°N
39°N
123°W
122°W
121°W
120°W
Legend
N
Distrbution Buffer
HUC 4 Boundaries
NLCD 2006 Landcover
Cultivated Crops
Developed, High Intensity
Developed, Low Intensity
Developed, Medium Intensity
Developed, Open Space
Hay / Pasture
0 15 30 60
Kilometers
Prepared by K. Goetschius
April 2012


Southern Oregon Northern California Coho ESU
Critical Habitat
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
Critical Habitat
0 15 30 60
Kilometers
Prepared by K. Goetschius
April 2012

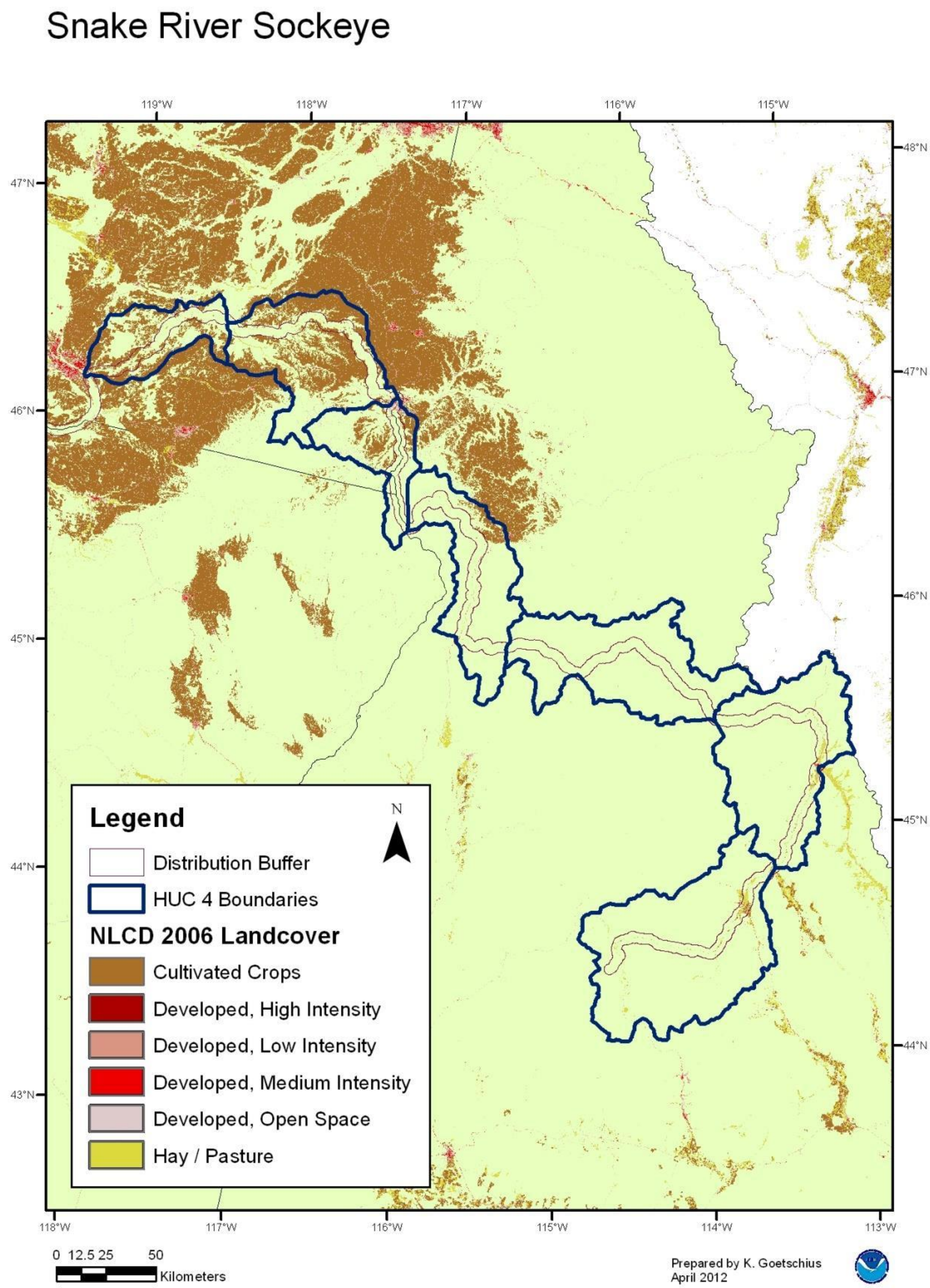
Central California Coastal Coho

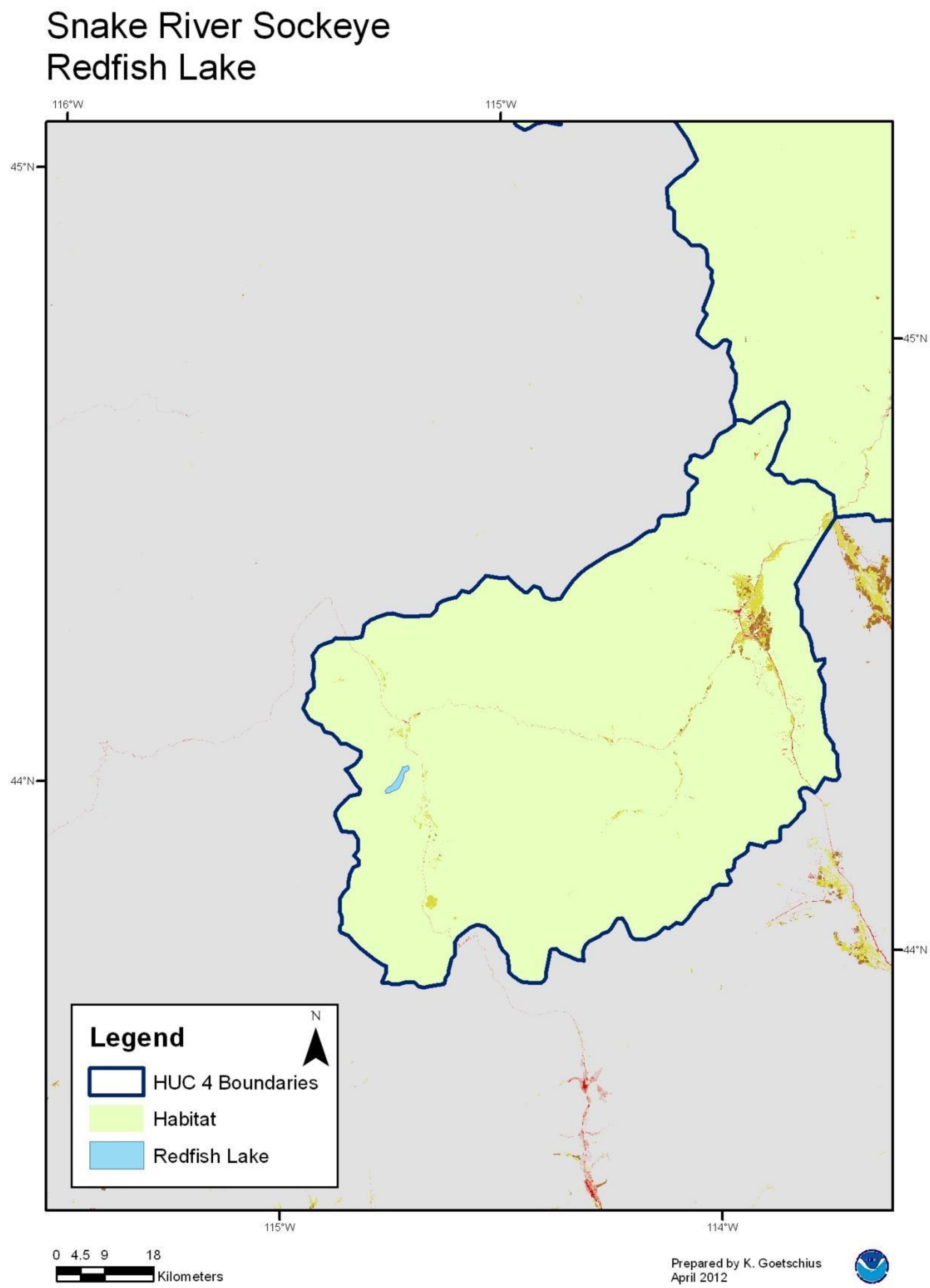



Central California Coastal Coho
Critical Habitat
Legend
HUC 4 Boundaries
Critical Habitat
N
124°W
123°W
122°W
121°W
120°W
40°N
39°N
38°N
37°N
0 10 20 40
Kilometers
Prepared by K. Goetschius
April 2012

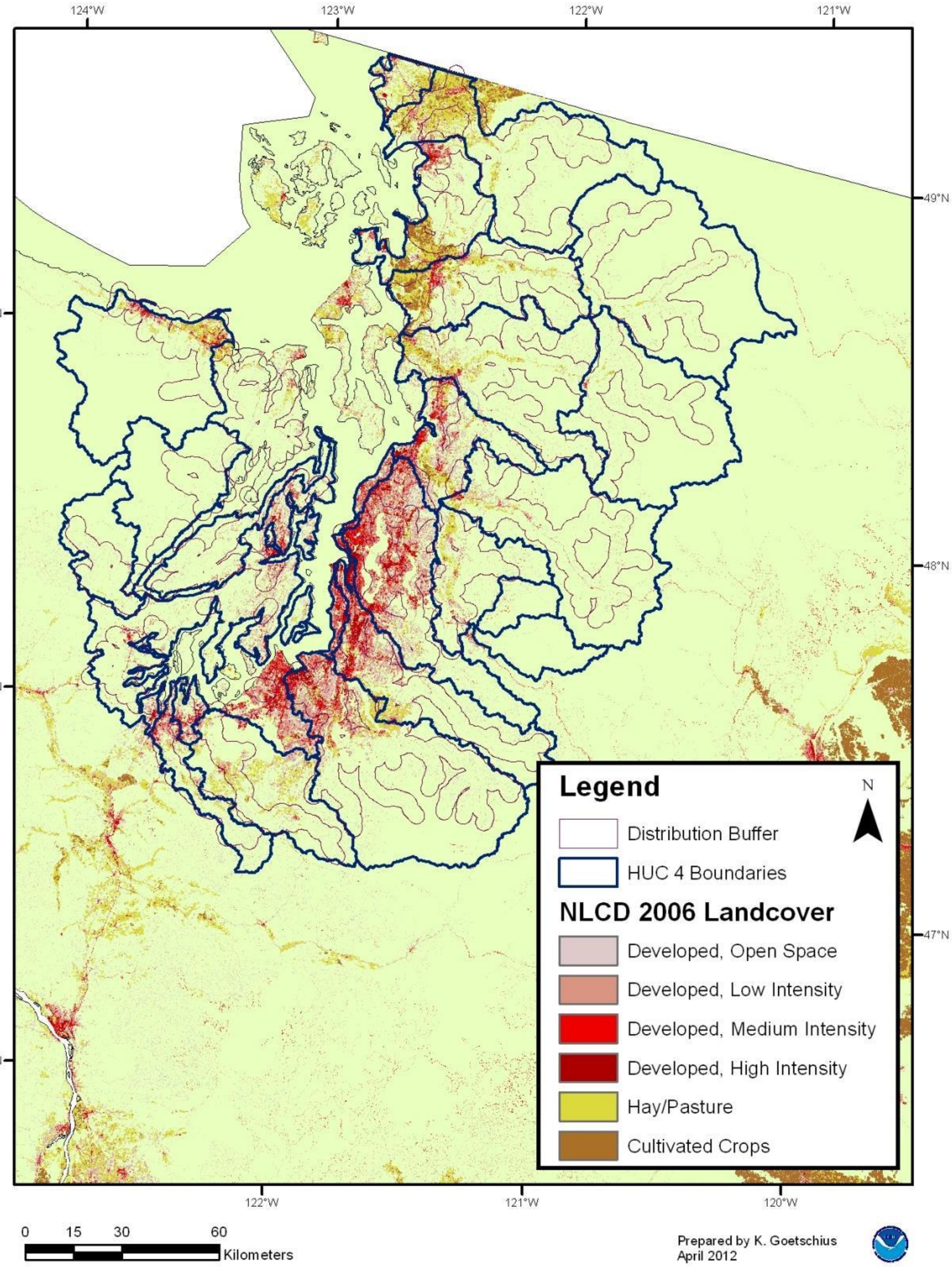


Ozette Lake Sockeye
48°N
48°N
Legend
N
Distribution Buffer
Critical Habitat
NLCD 2006 Landcover
Label
Cultivated Crops
Developed, High Intensity
Developed, Low Intensity
Developed, Medium Intensity
Developed, Open Space
Hay / Pasture
Open Water
0 0.5 1 2
Kilometers
Prepared by K. Goetschius
April 2012


Snake River Sockeye
119°W
118°W
117°W
116°W
115°W
48°N
47°N
46°N
45°N
44°N
43°N
113°W
114°W
Legend
N
Distribution Buffer
HUC 4 Boundaries
NLCD 2006 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
April 2012

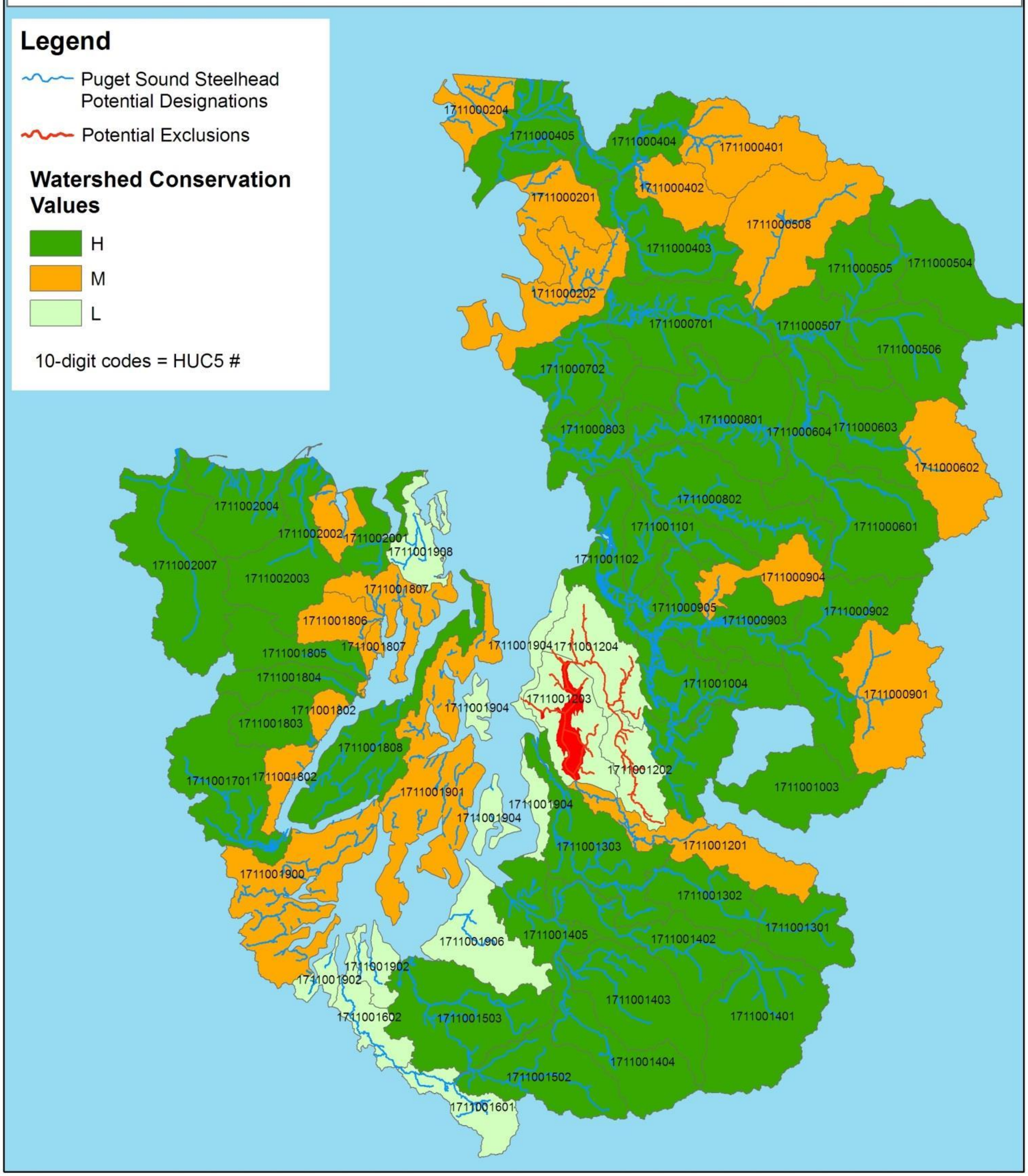

Snake River Sockeye
Redfish Lake
116°W
115°W
45°N
44°N
114°W
Legend
N
HUC 4 Boundaries
Habitat
Redfish Lake
0 4.5 9 18
Kilometers
Prepared by K. Goetschius
April 2012

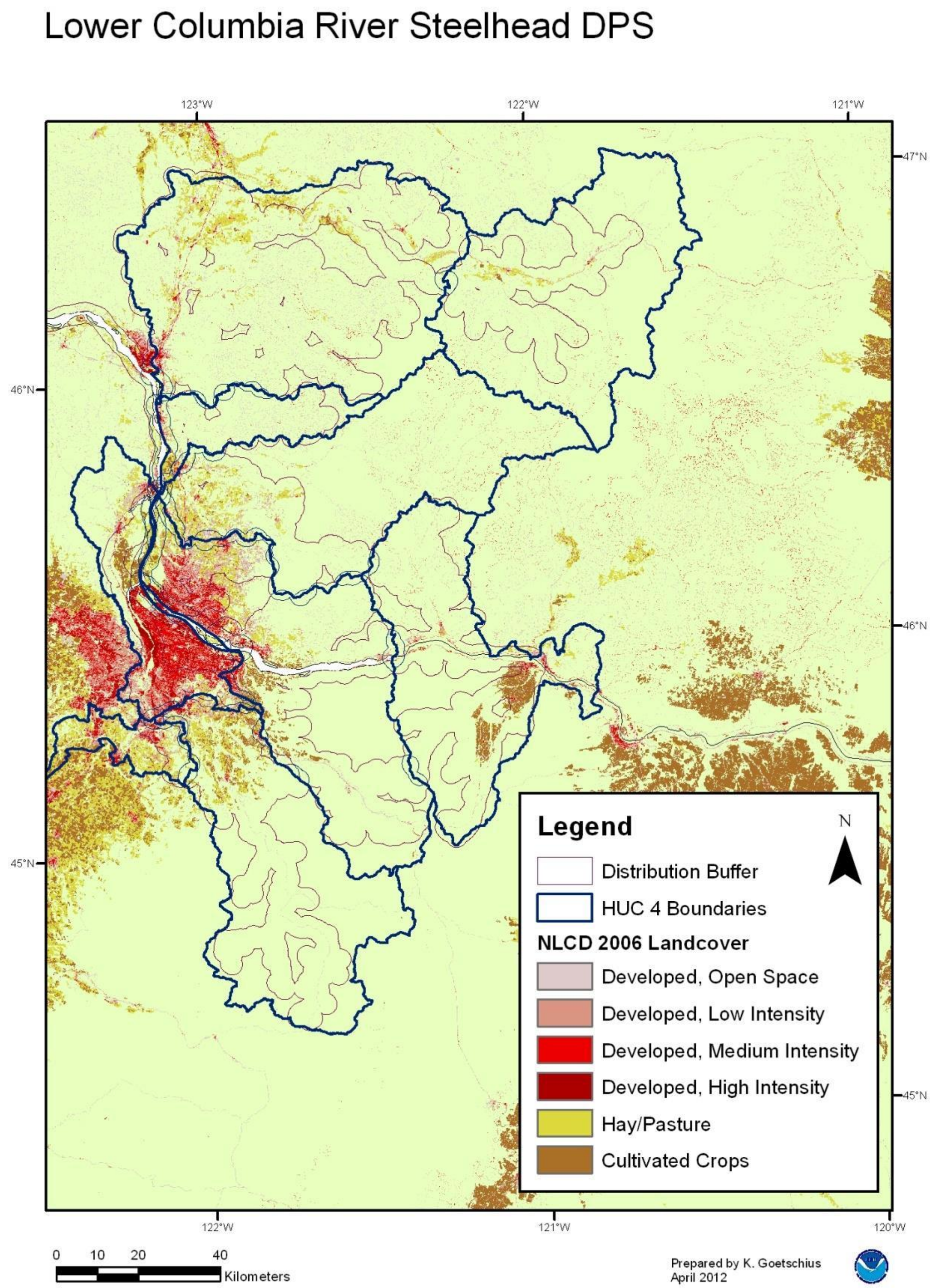
Puget Sound Steelhead DPS

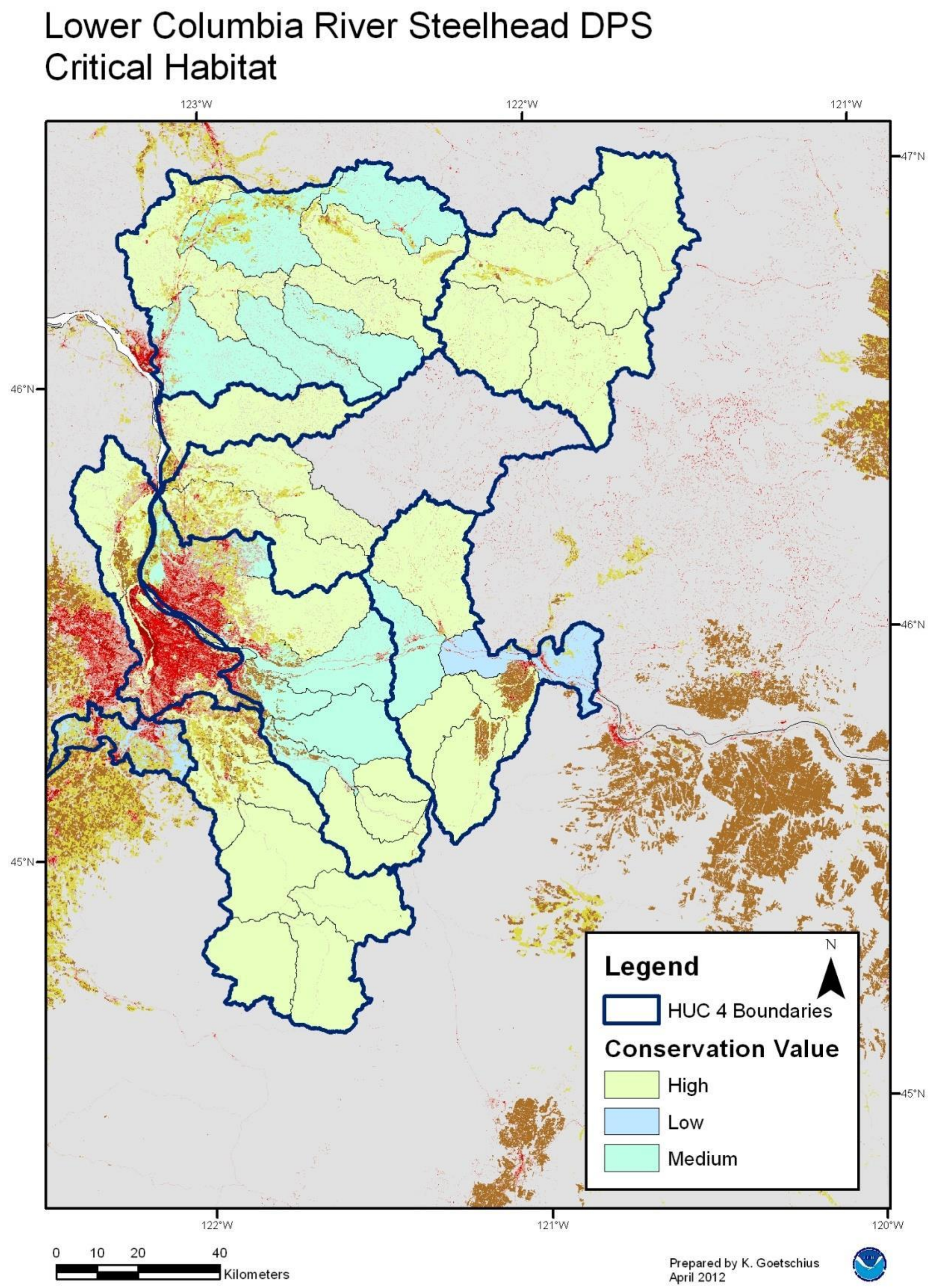


Puget Sound Steelhead

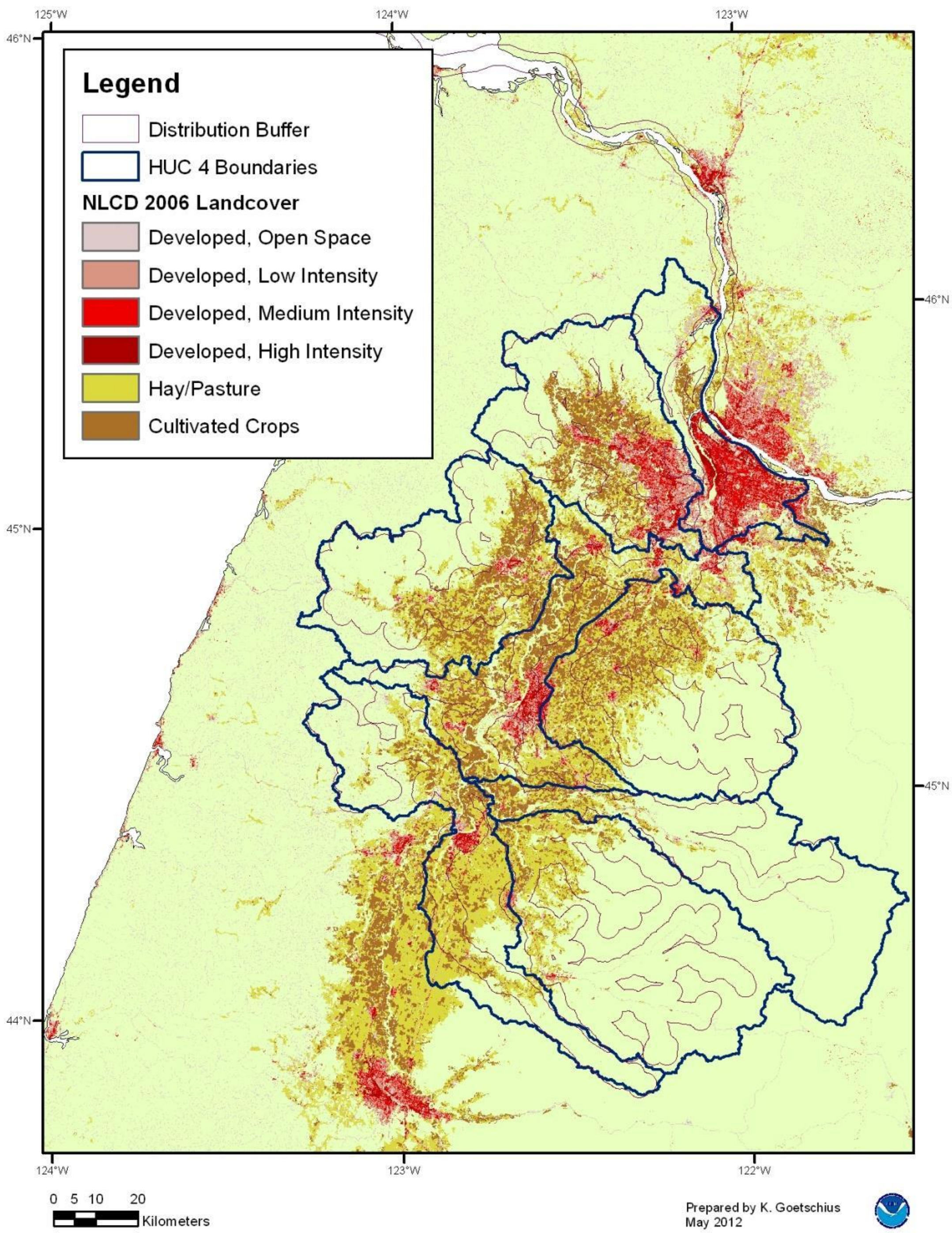
Watershed Conservation Value Ratings &
Areas Proposed for Exclusion due to Economic Impacts

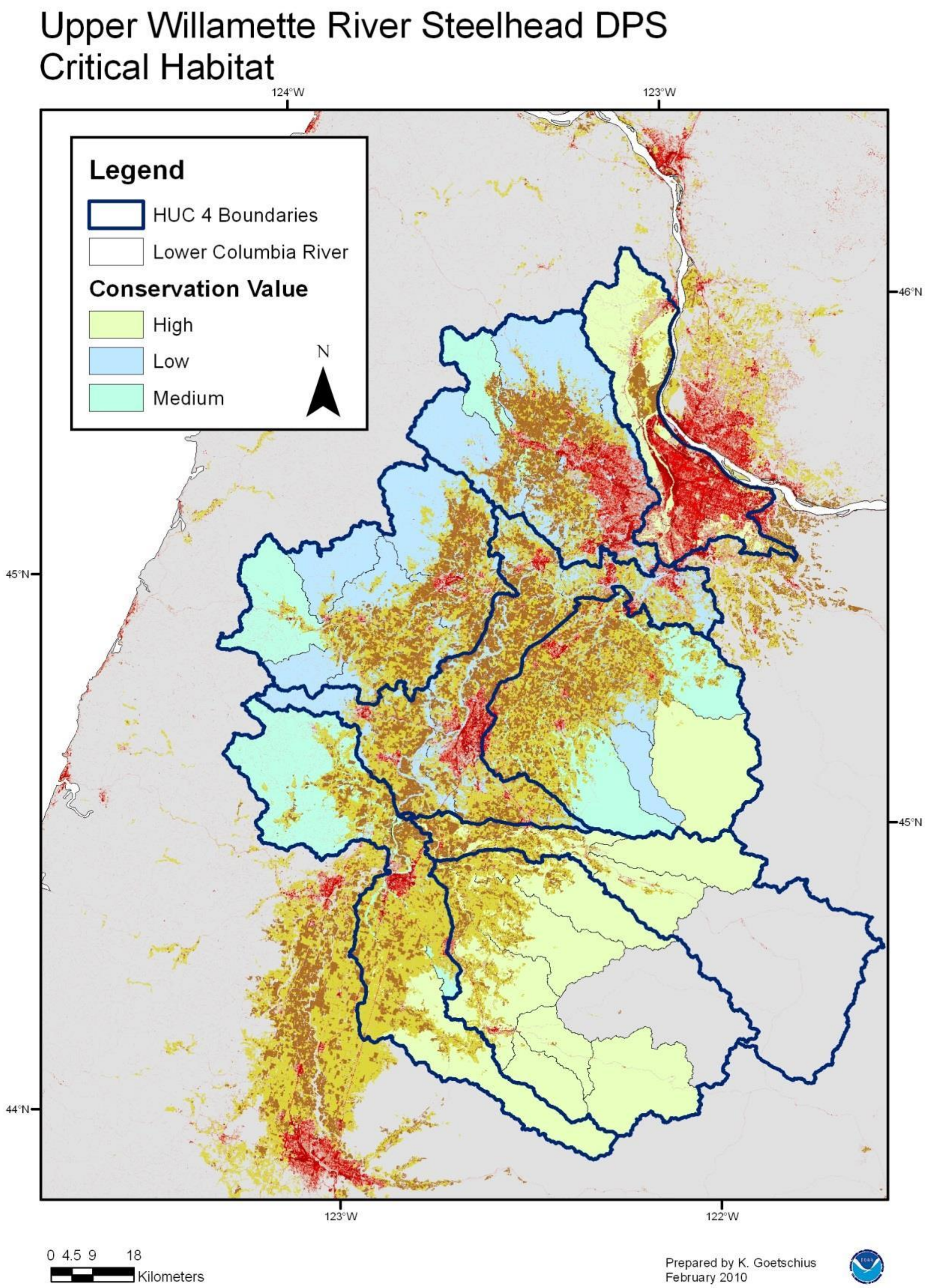


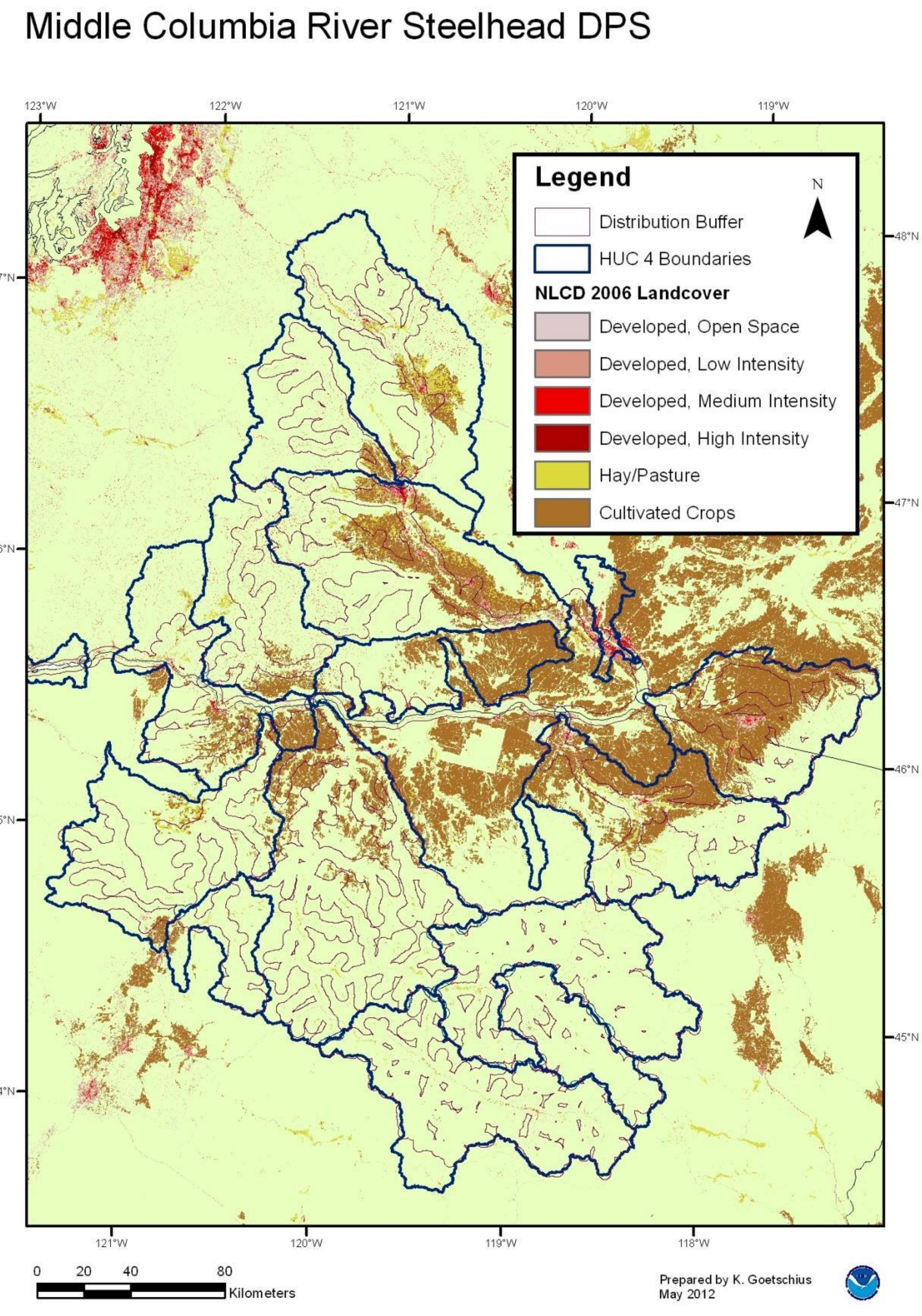

Lower Columbia River Steelhead DPS
123°W
122°W
121°W
47°N
46°N
46°N
45°N
45°N
122°W
121°W
120°W
Legend
N
Distribution Buffer
HUC 4 Boundaries
NLCD 2006 Landcover
Developed, Open Space
Developed, Low Intensity
Developed, Medium Intensity
Developed, High Intensity
Hay/Pasture
Cultivated Crops
0 10 20 40
Kilometers
Prepared by K. Goetschius
April 2012


Lower Columbia River Steelhead DPS
Critical Habitat
123°W
122°W
121°W
47°N
46°N
45°N
120°W
Legend
HUC 4 Boundaries
Conservation Value
High
Low
Medium
N
0 10 20 40
Kilometers
Prepared by K. Goetschius
April 2012

Upper Willamette River Steelhead DPS

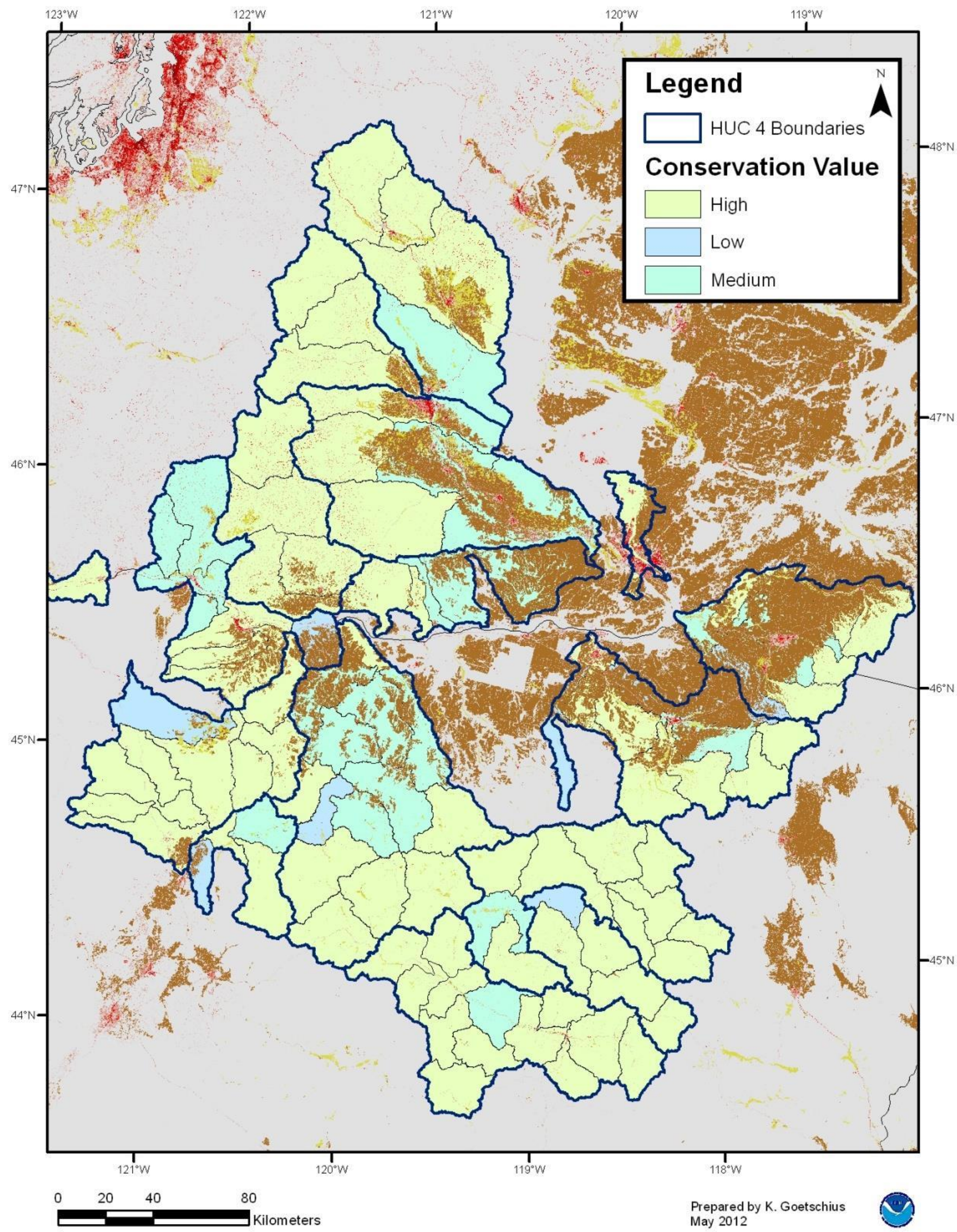



Upper Willamette River Steelhead DPS
Critical Habitat
124°W
123°W
Legend
HUC 4 Boundaries
Lower Columbia River
Conservation Value
High
Low
Medium
N
46°N
45°N
45°N
44°N
123°W
122°W
0 4.5 9 18
Kilometers
Prepared by K. Goetschius
February 2010

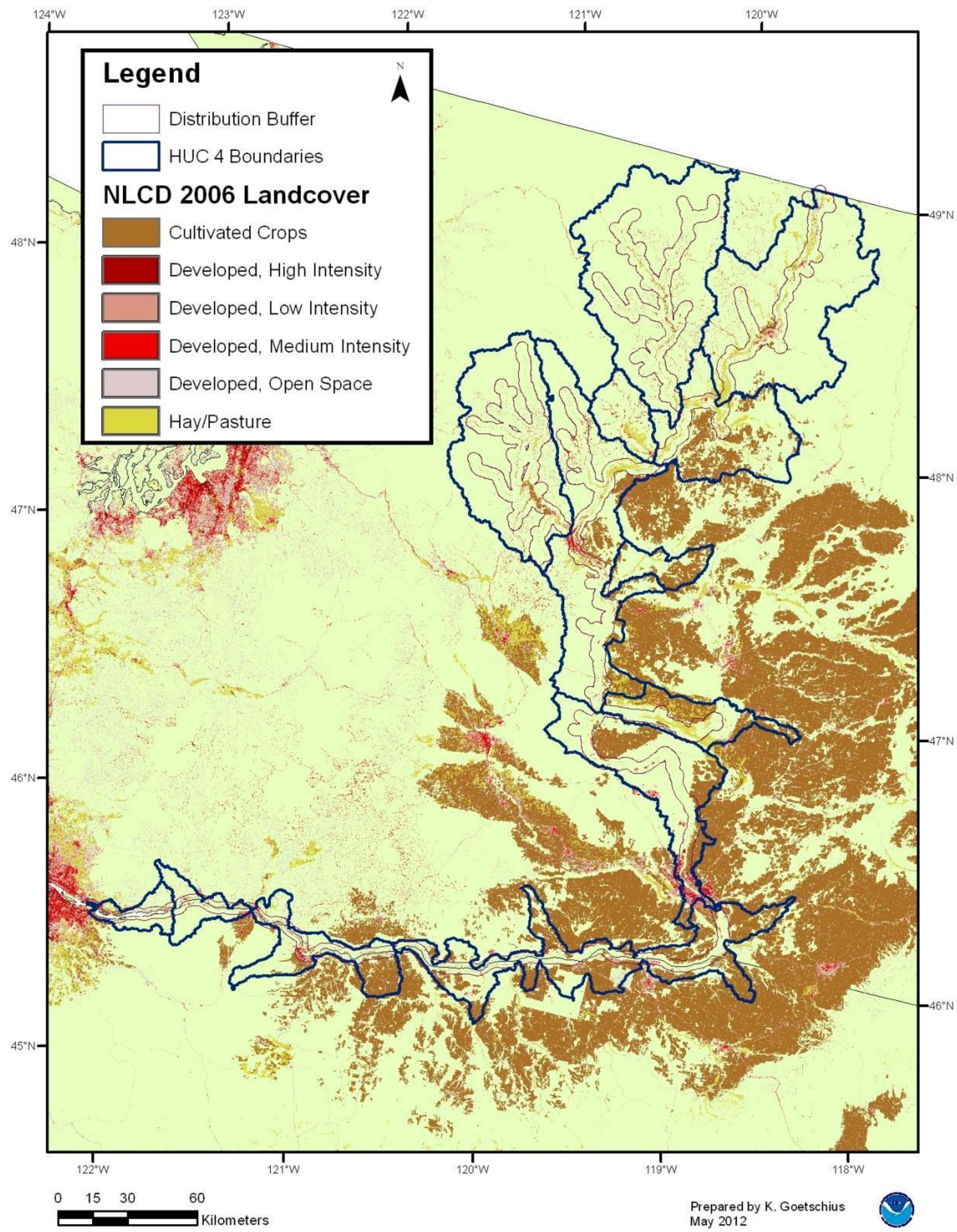


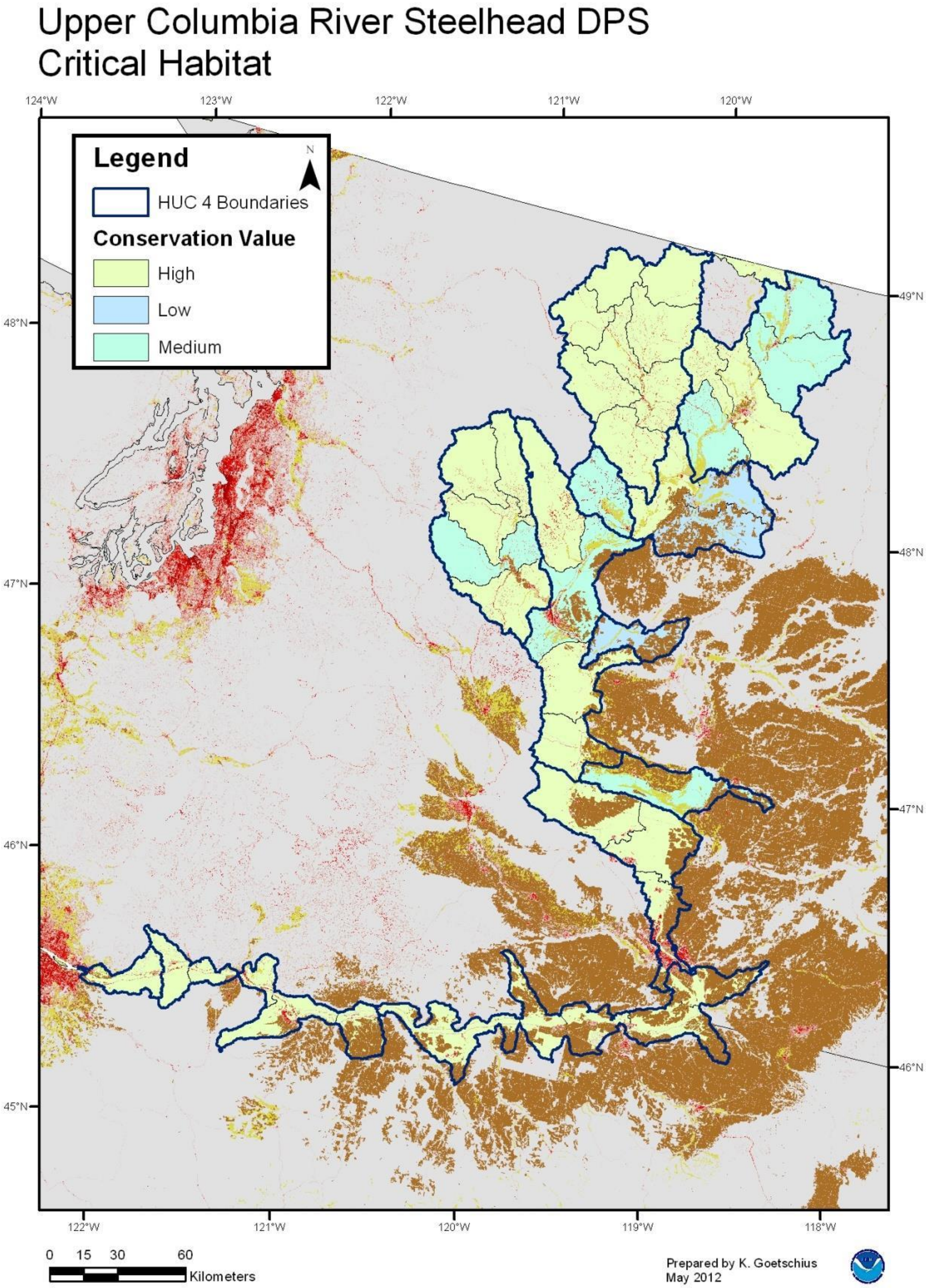
Middle Columbia River Steelhead DPS
Legend
Distribution Buffer
HUC 4 Boundaries
NLCD 2006 Landcover
Developed, Open Space
Developed, Low Intensity
Developed, Medium Intensity
Developed, High Intensity
Hay/Pasture
Cultivated Crops
N
123°W
122°W
121°W
120°W
119°W
118°W
48°N
47°N
46°N
45°N
44°N
0 20 40 80
Kilometers
Prepared by K. Goetschius
May 2012

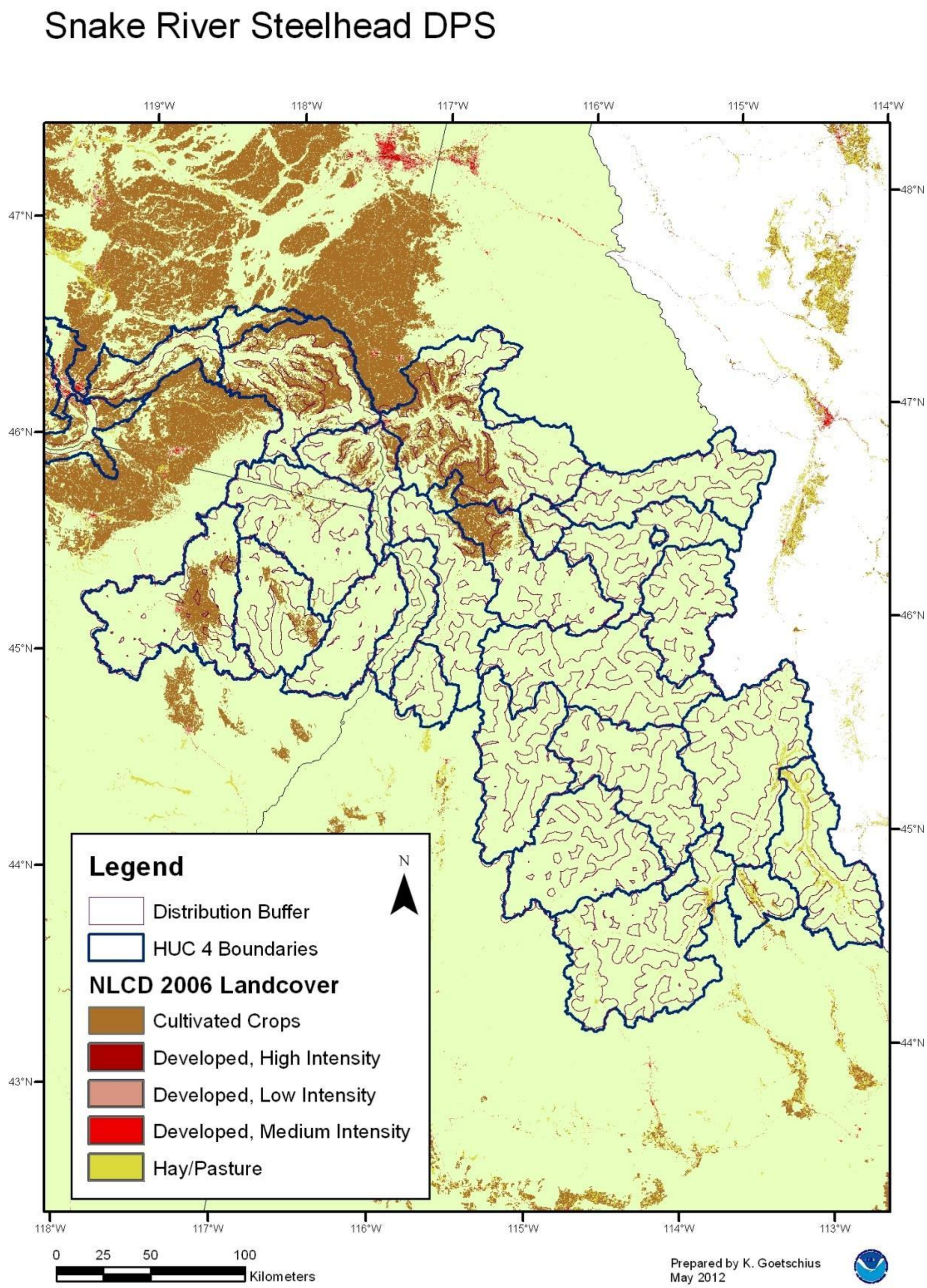
Middle Columbia River Steelhead DPS
Critical Habitat

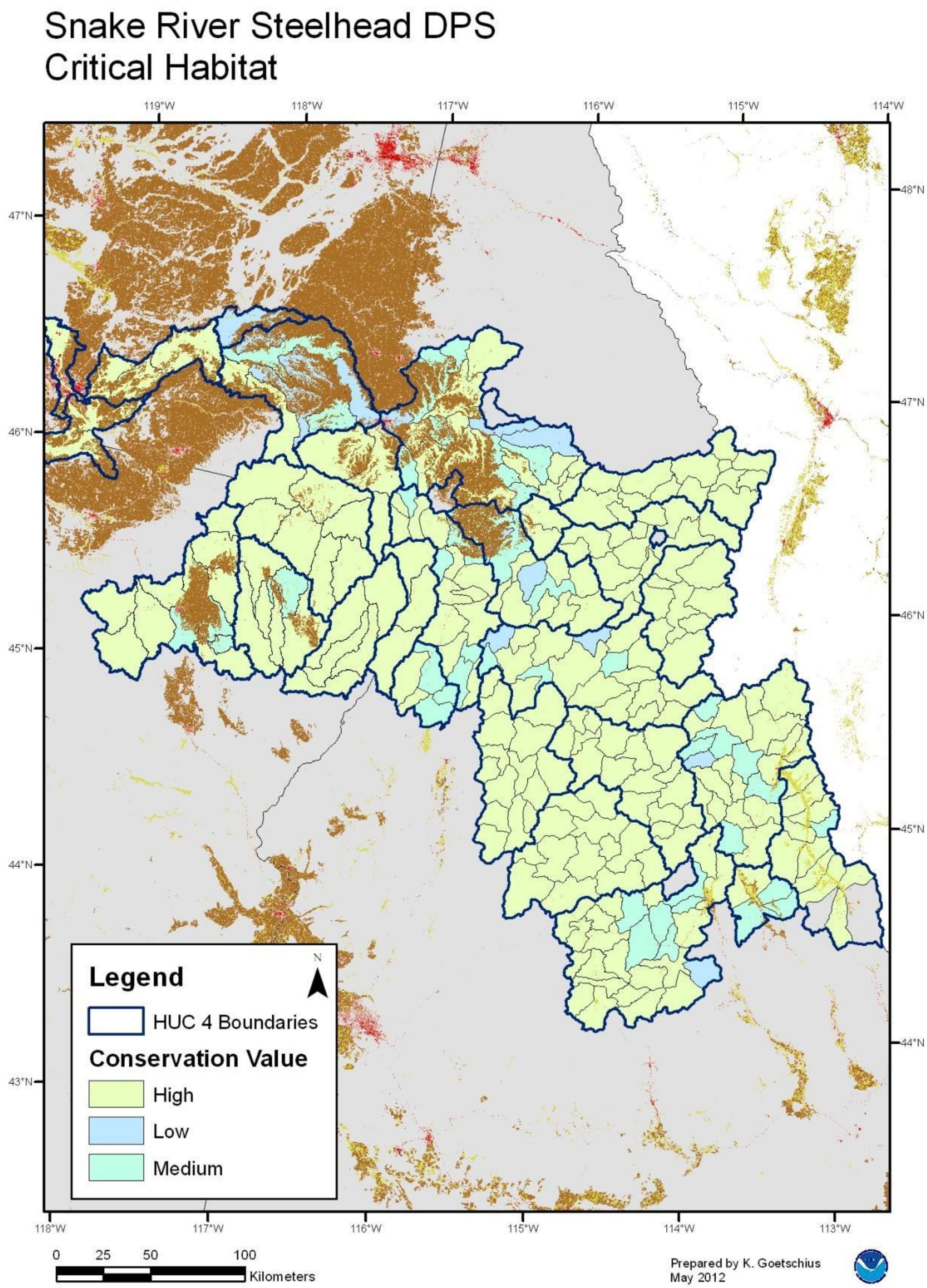


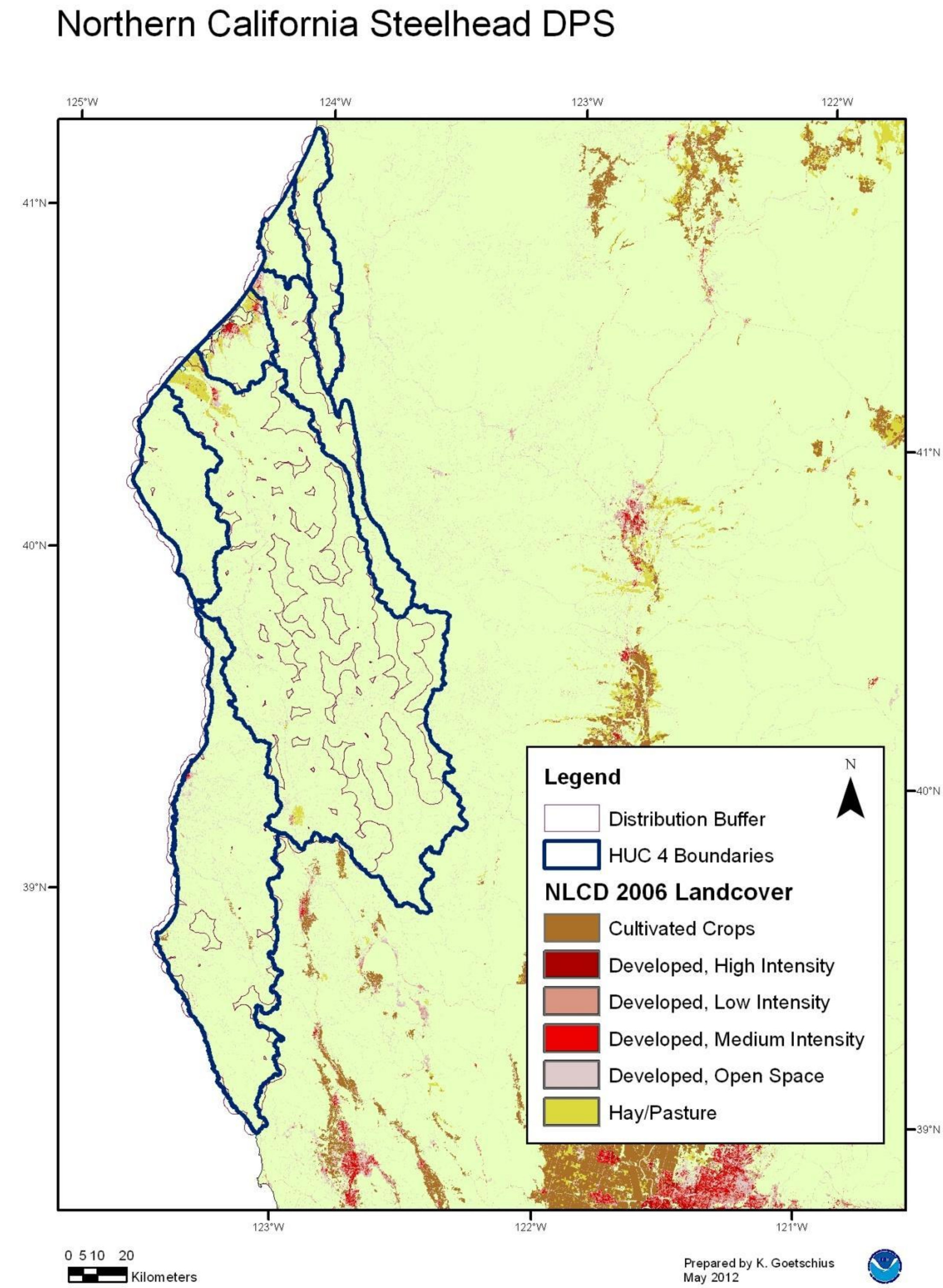
Upper Columbia River Steelhead DPS



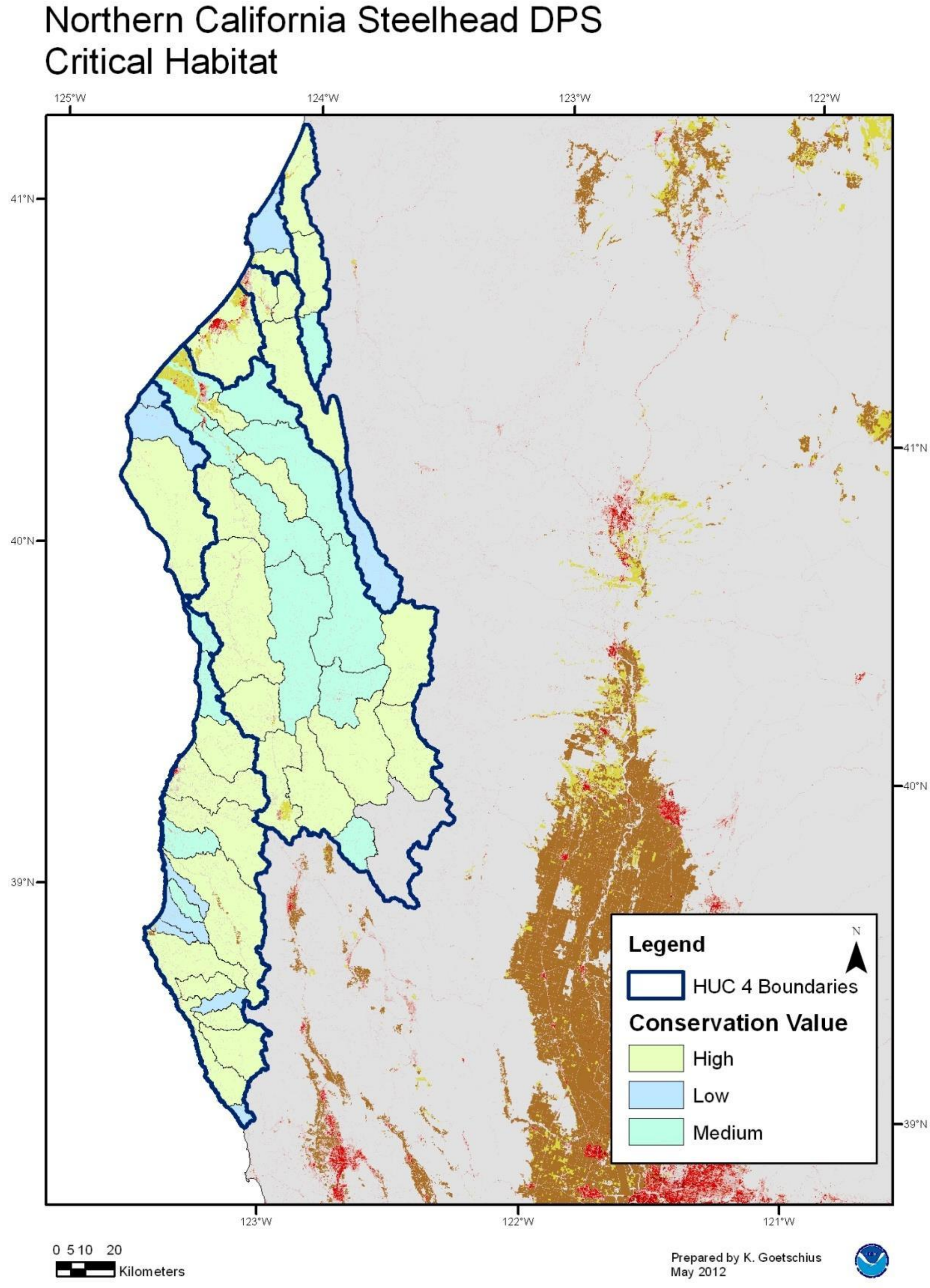

Upper Columbia River Steelhead DPS
Critical Habitat
Legend
HUC 4 Boundaries
Conservation Value
High
Low
Medium
N
124°W
123°W
122°W
121°W
120°W
119°W
118°W
49°N
48°N
47°N
46°N
45°N
0 15 30 60
Kilometers
Prepared by K. Goetschius
May 2012

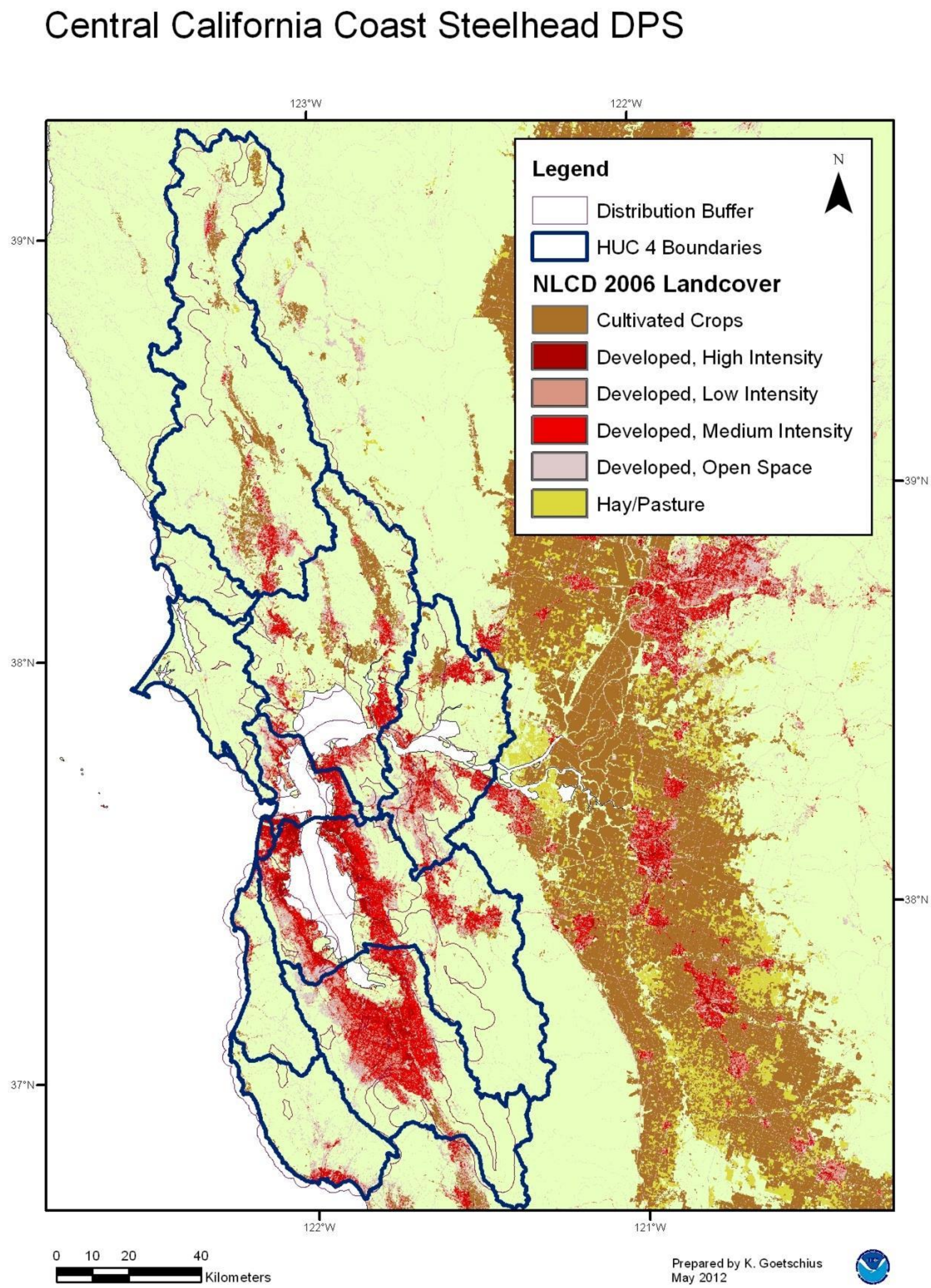

Snake River Steelhead DPS
119°W
118°W
117°W
116°W
115°W
114°W
48°N
47°N
46°N
45°N
44°N
43°N
113°W
Legend
N
Distribution Buffer
HUC 4 Boundaries
NLCD 2006 Landcover
Cultivated Crops
Developed, High Intensity
Developed, Low Intensity
Developed, Medium Intensity
Hay/Pasture
0 25 50 100
Kilometers
Prepared by K. Goetschius
May 2012

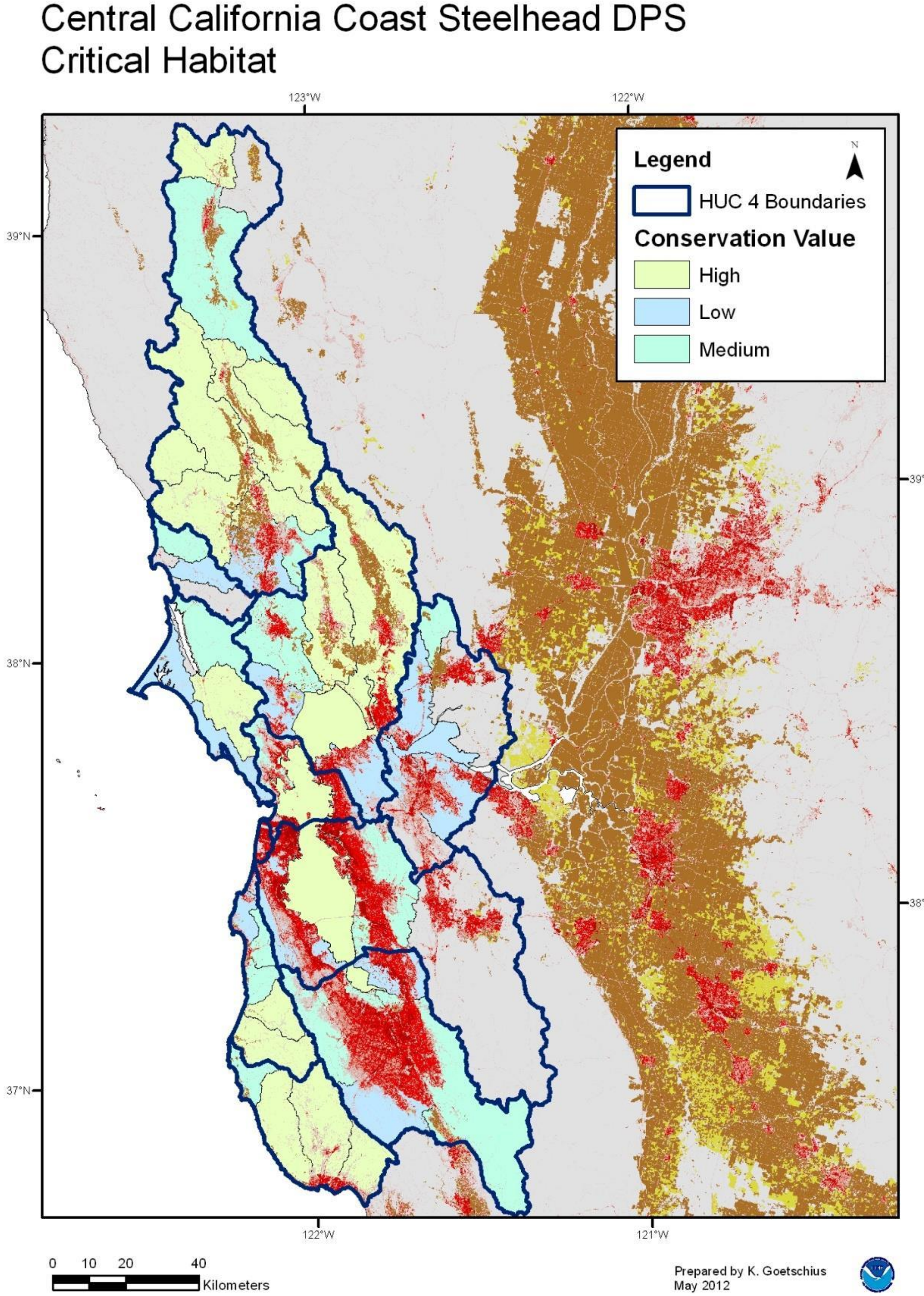

Snake River Steelhead DPS
Critical Habitat
119°W
118°W
117°W
116°W
115°W
114°W
113°W
48°N
47°N
46°N
45°N
44°N
43°N
Legend
N
HUC 4 Boundaries
Conservation Value
High
Low
Medium
0 25 50 100
Kilometers
Prepared by K. Goetschius
May 2012

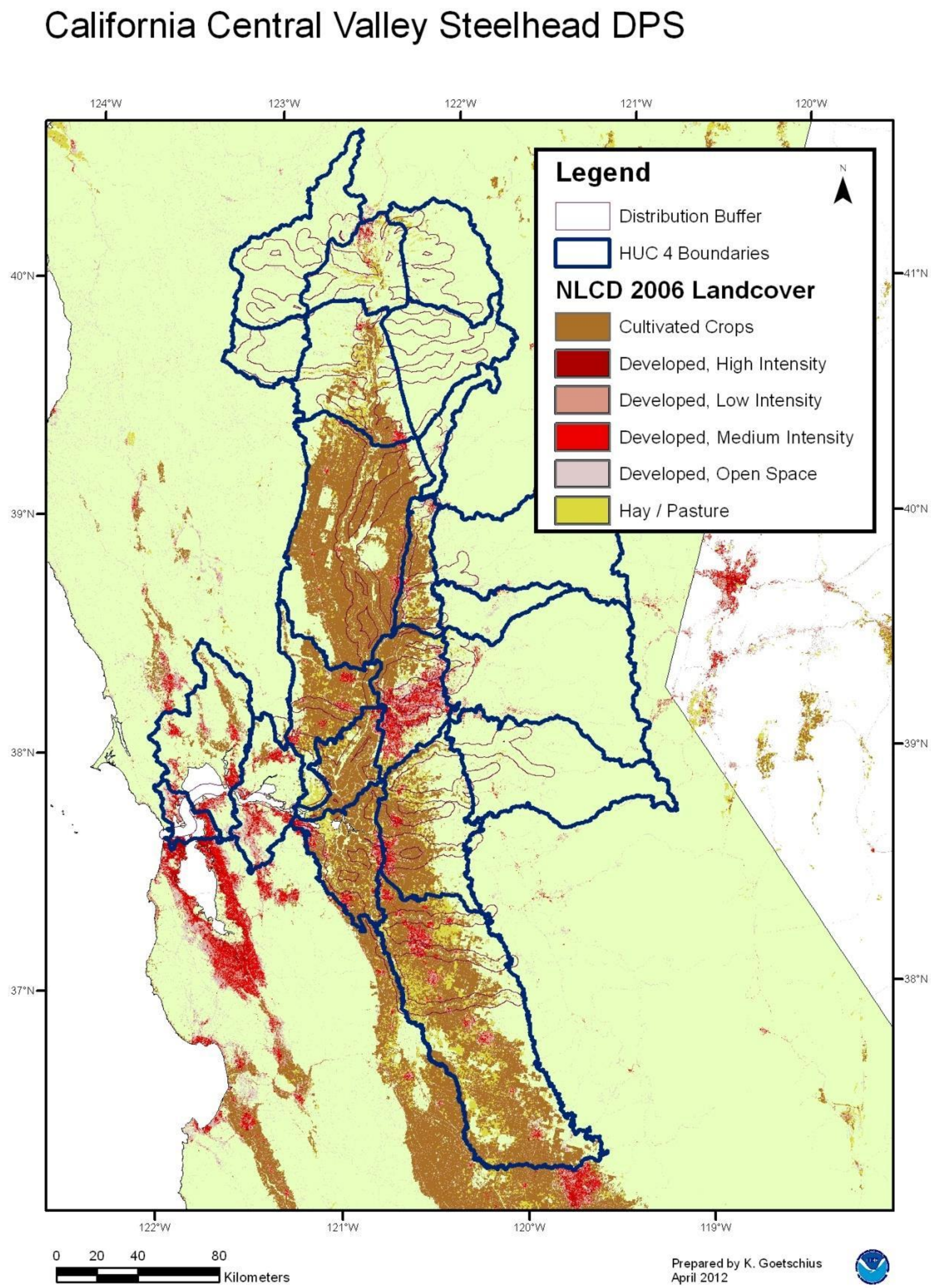


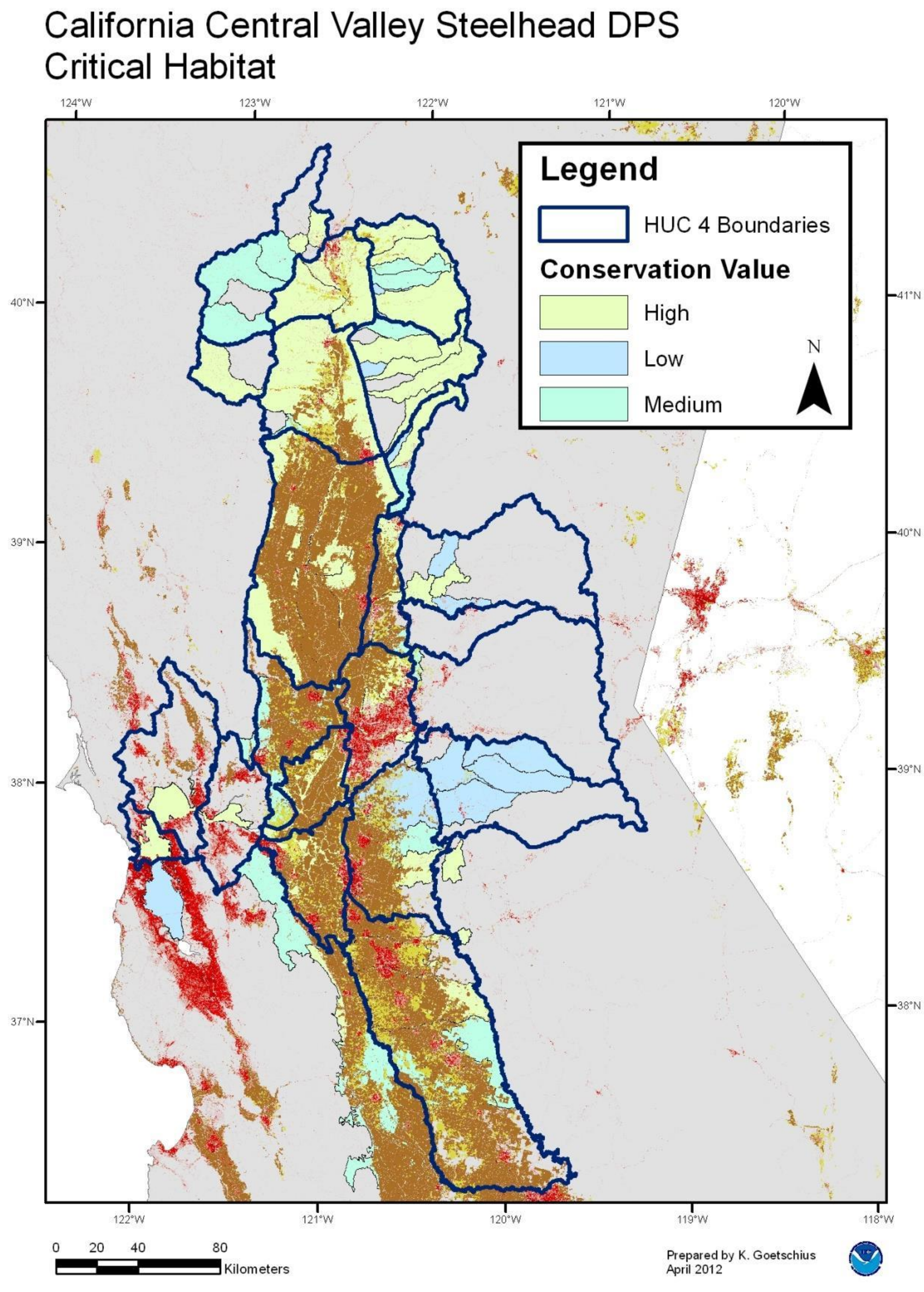
Northern California Steelhead DPS
125°W
124°W
123°W
122°W
121°W
41°N
40°N
39°N
Legend
N
Distribution Buffer
HUC 4 Boundaries
NLCD 2006 Landcover
Cultivated Crops
Developed, High Intensity
Developed, Low Intensity
Developed, Medium Intensity
Developed, Open Space
Hay/Pasture
0 5 10 20
Kilometers
Prepared by K. Goetschius
May 2012

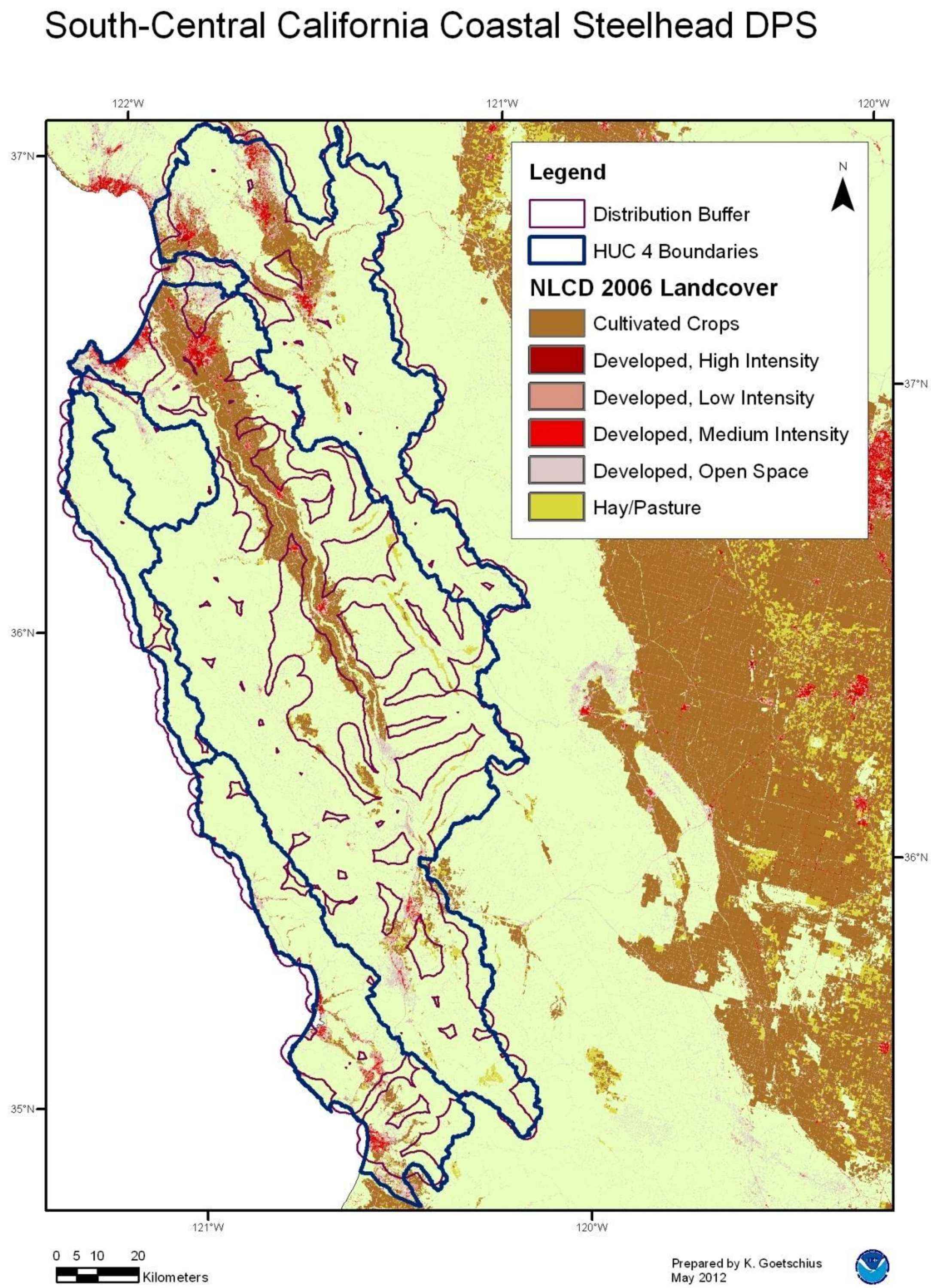

Northern California Steelhead DPS
Critical Habitat
125°W
124°W
123°W
122°W
41°N
40°N
39°N
121°W
Legend
HUC 4 Boundaries
Conservation Value
High
Low
Medium
N
0 5 10 20
Kilometers
Prepared by K. Goetschius
May 2012

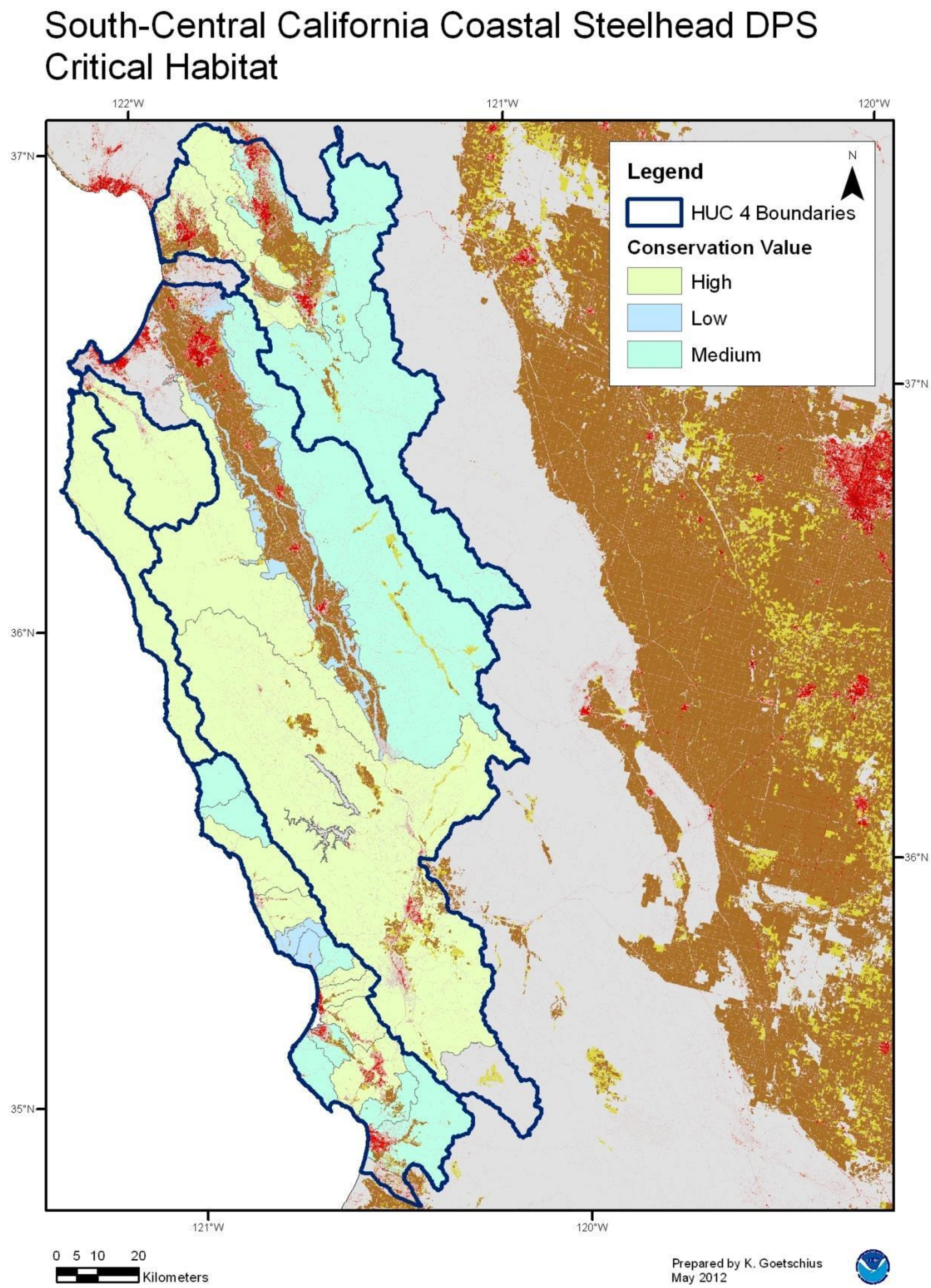

Central California Coast Steelhead DPS
Legend
N
Distribution Buffer
HUC 4 Boundaries
NLCD 2006 Landcover
Cultivated Crops
Developed, High Intensity
Developed, Low Intensity
Developed, Medium Intensity
Developed, Open Space
Hay/Pasture
123°W
122°W
39°N
38°N
37°N
121°W
0 10 20 40
Kilometers
Prepared by K. Goetschius
May 2012

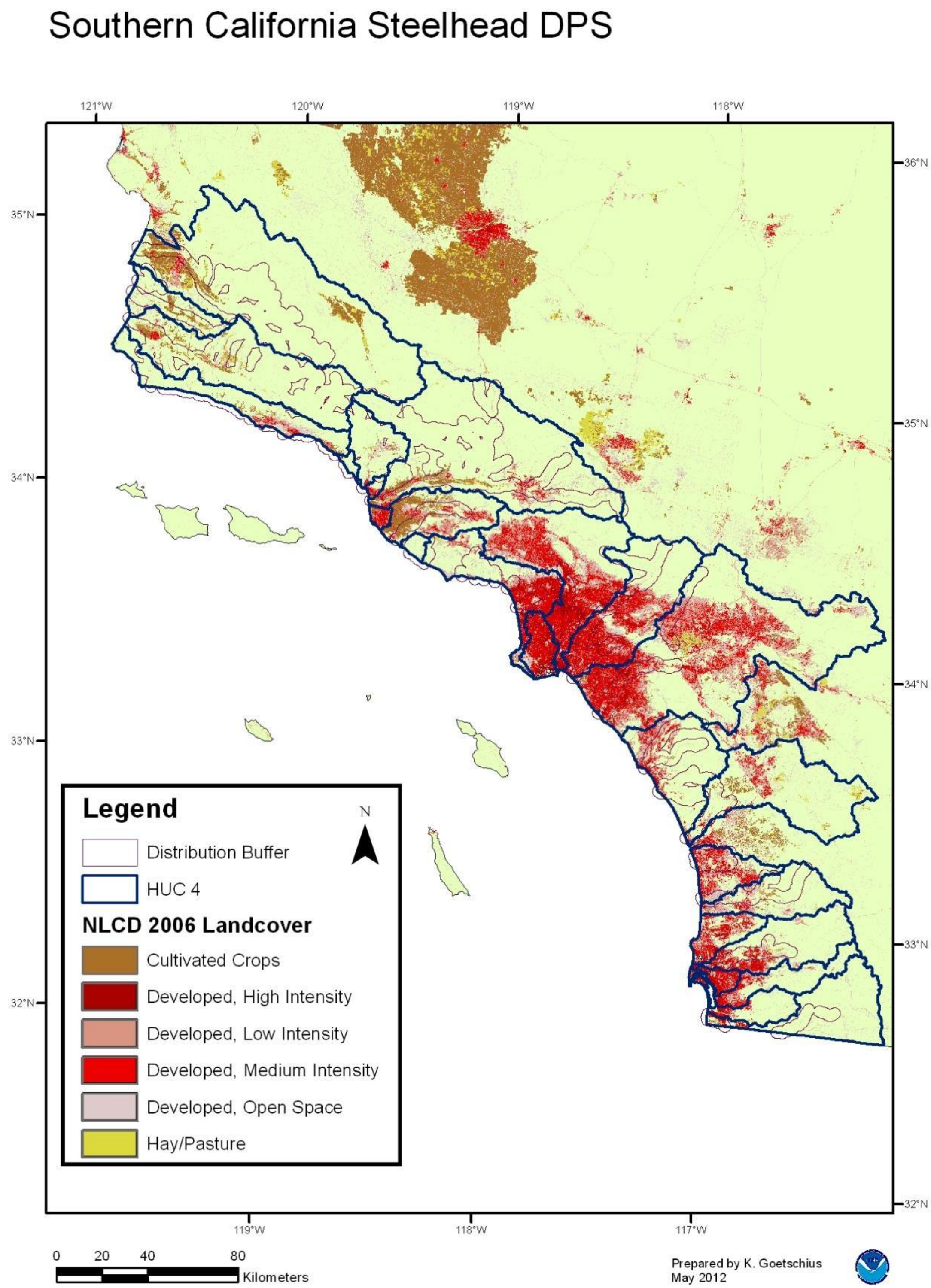

Central California Coast Steelhead DPS
Critical Habitat
123°W
122°W
39°N
38°N
37°N
121°W
Legend
N
HUC 4 Boundaries
Conservation Value
High
Low
Medium
0 10 20 40
Kilometers
Prepared by K. Goetschius
May 2012

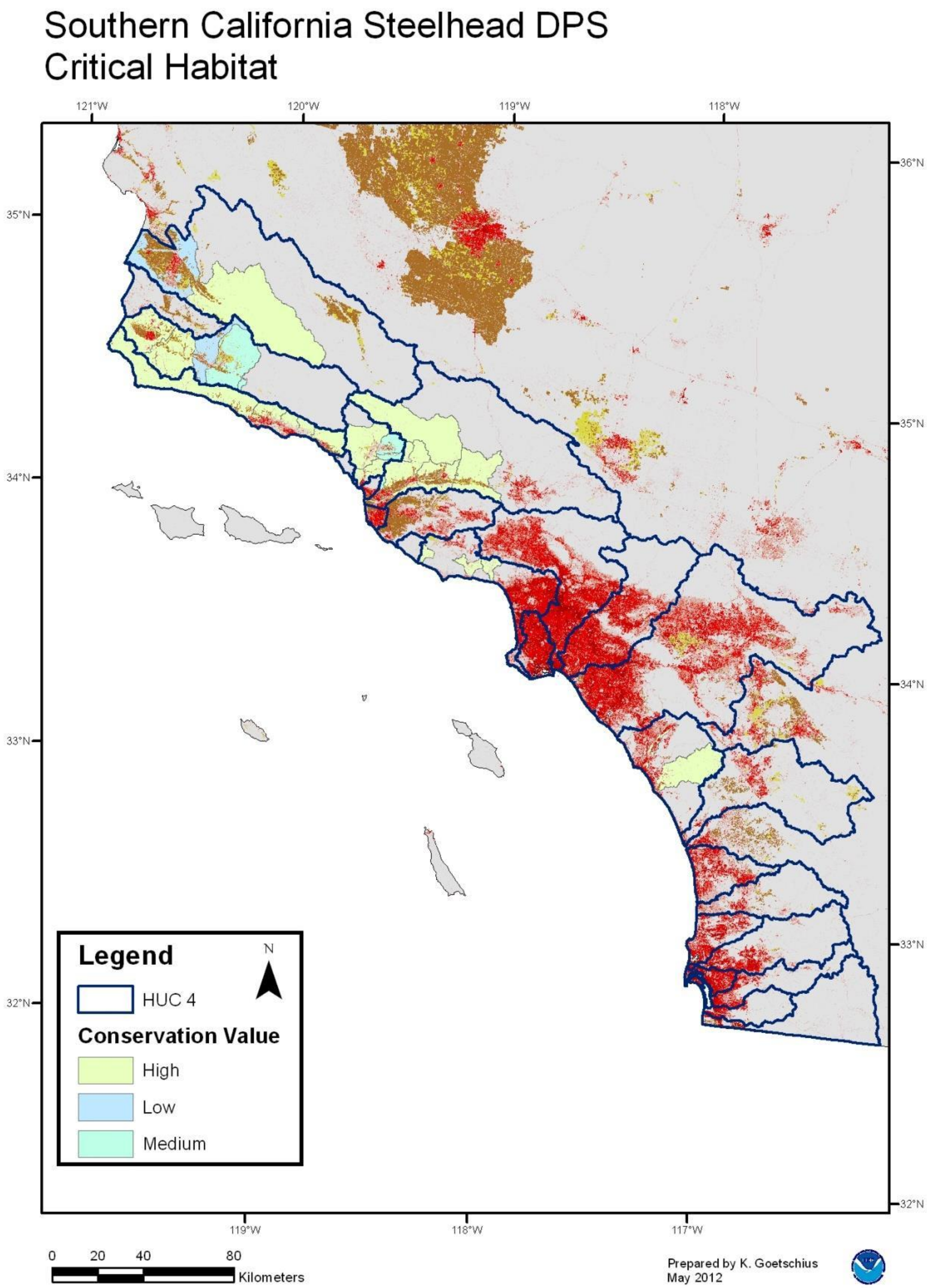

California Central Valley Steelhead DPS
124°W
123°W
122°W
121°W
120°W
40°N
39°N
38°N
37°N
41°N
40°N
39°N
38°N
122°W
121°W
120°W
119°W
Legend
N
Distribution Buffer
HUC 4 Boundaries
NLCD 2006 Landcover
Cultivated Crops
Developed, High Intensity
Developed, Low Intensity
Developed, Medium Intensity
Developed, Open Space
Hay / Pasture
0 20 40 80
Kilometers
Prepared by K. Goetschius
April 2012


California Central Valley Steelhead DPS
Critical Habitat
124°W
123°W
122°W
121°W
120°W
40°N
39°N
38°N
37°N
41°N
40°N
39°N
38°N
122°W
121°W
120°W
119°W
118°W
Legend
HUC 4 Boundaries
Conservation Value
High
Low
Medium
N
0 20 40 80
Kilometers
Prepared by K. Goetschius
April 2012


South-Central California Coastal Steelhead DPS
122°W
121°W
120°W
37°N
36°N
35°N
Legend
N
Distribution Buffer
HUC 4 Boundaries
NLCD 2006 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
May 2012


South-Central California Coastal Steelhead DPS
Critical Habitat
122°W
121°W
120°W
37°N
36°N
35°N
Legend
N
HUC 4 Boundaries
Conservation Value
High
Low
Medium
0 5 10 20
Kilometers
Prepared by K. Goetschius
May 2012


Southern California Steelhead DPS
121°W
120°W
119°W
118°W
117°W
36°N
35°N
34°N
33°N
32°N
Legend
Distribution Buffer
HUC 4
NLCD 2006 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
May 2012


Southern California Steelhead DPS
Critical Habitat
121°W
120°W
119°W
118°W
117°W
36°N
35°N
34°N
33°N
32°N
Legend
N
HUC 4
Conservation Value
High
Low
Medium
0 20 40 80
Kilometers
Prepared by K. Goetschius
May 2012

21 Appendix 5: Temporal Distribution of ESA listed Pacific Coast Salmon and Steelhead

An important part of determining if listed salmonids will be exposed to pesticides is determining if they are actually present in the system at the same time that pesticides are present. Pesticides may enter water bodies via several routes including runoff from a treated area near the stream, spray drift from a treated area near the stream, groundwater interchange, transport from a treated area upstream, partitioning from contaminated sediment, or atmospheric deposition. The importance of each of these pathways depends greatly on physicochemical properties of the a.i. and the method of application. Land use, soil types, and geography within the ESUs/DPSs are also factors.

The tables in this appendix provide presence/absence information on various salmonid life stages for each of the ESUs/DPSs across the course of a calendar year. Shaded boxes indicate the life stage is expected to be present, while unshaded boxes indicate the life stage is not expected to be present. This information was collated from a number of sources by OPR staff. It represents a generalized annual run-timing, and there may be some variations on a local scale or in a particular year. However, one important conclusion we drew from this analysis is that *in most systems, some sensitive life stage is present year-round.*

Chinook Salmon

Puget Sound Chinook (spring/summer, fall combined)

<table>
<tr><td>Life History phase</td><td>Jan</td><td>Feb</td><td>Mar</td><td>Apr</td><td>May</td><td>Jun</td><td>Jul</td><td>Aug</td><td>Sep</td><td>Oct</td><td>Nov</td><td>Dec</td></tr>
<tr><td>Entering Fresh Water
(adults/jacks)</td><td></td><td></td><td></td><td colspan="8">Present</td><td></td></tr>
<tr><td>Spawning</td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="6">Present</td></tr>
<tr><td>Incubation (eggs)</td><td colspan="2">Present</td><td></td><td></td><td></td><td></td><td colspan="6">Present</td></tr>
<tr><td>Emergence
(alevin to fry phase)</td><td colspan="4">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="2">Present</td></tr>
<tr><td>Rearing and migration
(juveniles)</td><td colspan="12">Present</td></tr>
</table>

Lower Columbia River Chinook

<table>
<tr><td>Life History phase</td><td>Jan</td><td>Feb</td><td>Mar</td><td>Apr</td><td>May</td><td>Jun</td><td>Jul</td><td>Aug</td><td>Sep</td><td>Oct</td><td>Nov</td><td>Dec</td></tr>
<tr><td>Entering Fresh Water
(adults/jacks)</td><td></td><td></td><td colspan="10">Present</td></tr>
<tr><td>Spawning</td><td>Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="5">Present</td></tr>
<tr><td>Incubation (eggs)</td><td colspan="3">Present</td><td></td><td></td><td></td><td></td><td colspan="5">Present</td></tr>
<tr><td>Emergence
(alevin to fry phases</td><td colspan="4">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Rearing and migration
(juveniles)</td><td colspan="12">Present</td></tr>
</table>

Upper Columbia River Spring-run Chinook

<table>
<tr><td>Life History phase</td><td>Jan</td><td>Feb</td><td>Mar</td><td>Apr</td><td>May</td><td>Jun</td><td>Jul</td><td>Aug</td><td>Sep</td><td>Oct</td><td>Nov</td><td>Dec</td></tr>
<tr><td>Entering Fresh Water
(adults/jacks)</td><td></td><td></td><td colspan="7">Present</td><td></td><td></td><td></td></tr>
<tr><td>Spawning</td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="3">Present</td><td></td><td></td><td></td></tr>
<tr><td>Incubation (eggs)</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="5">Present</td></tr>
<tr><td>Emergence
(alevin to fry phases)</td><td colspan="4">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="2">Present</td></tr>
<tr><td>Rearing and migration
(juveniles)</td><td colspan="12">Present</td></tr>
</table>

Snake River Fall Run Chinook

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)								Present				
Spawning										Present		
Incubation (eggs)	Present									Present		
Emergence (alevin to fry phases	Present											Present
Rearing and migration (juveniles)	Present											

Snake River Spring/Summer Run Chinook

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)			Present									
Spawning								Present				
Incubation (eggs)								Present				
Emergence (alevin to fry phases	Present									Present		
Rearing and migration (juveniles)	Present											

Upper Willamette River Chinook

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)					Present							
Spawning								Present				
Incubation (eggs)									Present			
Emergence (alevin to fry phases)	Present										Present	
Rearing and migration (juveniles)	Present											

California Coastal Chinook

<table>
<tr><td>Life History phase</td><td>Jan</td><td>Feb</td><td>Mar</td><td>Apr</td><td>May</td><td>Jun</td><td>Jul</td><td>Aug</td><td>Sep</td><td>Oct</td><td>Nov</td><td>Dec</td></tr>
<tr><td>Entering Fresh Water (adults/jacks)</td><td>Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="3">Present</td></tr>
<tr><td>Spawning</td><td>Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="2">Present</td></tr>
<tr><td>Incubation (eggs)</td><td colspan="3">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="2">Present</td></tr>
<tr><td>Emergence (alevin to fry phases)</td><td></td><td colspan="2">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Rearing and migration (juveniles)</td><td></td><td colspan="5">Present</td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
</table>

Central Valley Spring-run Chinook

<table>
<tr><td>Life History phase</td><td>Jan</td><td>Feb</td><td>Mar</td><td>Apr</td><td>May</td><td>Jun</td><td>Jul</td><td>Aug</td><td>Sep</td><td>Oct</td><td>Nov</td><td>Dec</td></tr>
<tr><td>Entering Fresh Water (adults/jacks)</td><td></td><td></td><td colspan="5">Present</td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Spawning</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="3">Present</td><td></td><td></td></tr>
<tr><td>Incubation (eggs)</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="5">Present</td></tr>
<tr><td>Emergence (alevin to fry phases)</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="2">Present</td></tr>
<tr><td>Rearing and migration (juveniles)</td><td colspan="12">Present</td></tr>
</table>

Sacramento River Winter-run Chinook

<table>
<tr><td>Life History phase</td><td>Jan</td><td>Feb</td><td>Mar</td><td>Apr</td><td>May</td><td>Jun</td><td>Jul</td><td>Aug</td><td>Sep</td><td>Oct</td><td>Nov</td><td>Dec</td></tr>
<tr><td>Entering Fresh Water (adults/jacks)</td><td colspan="6">Present</td><td></td><td></td><td></td><td></td><td colspan="2">Present</td></tr>
<tr><td>Spawning</td><td></td><td></td><td></td><td colspan="5">Present</td><td></td><td></td><td></td><td></td></tr>
<tr><td>Incubation (eggs)</td><td></td><td></td><td></td><td colspan="6">Present</td><td></td><td></td><td></td></tr>
<tr><td>Emergence (alevin to fry phases</td><td></td><td></td><td></td><td></td><td></td><td colspan="4">Present</td><td></td><td></td><td></td></tr>
<tr><td>Rearing and migration (juveniles)</td><td colspan="3">Present</td><td></td><td></td><td></td><td></td><td colspan="5">Present</td></tr>
</table>

Chum Salmon

Hood Canal Summer-run

<table>
<tr><td>Life History phase</td><td>Jan</td><td>Feb</td><td>Mar</td><td>Apr</td><td>May</td><td>Jun</td><td>Jul</td><td>Aug</td><td>Sep</td><td>Oct</td><td>Nov</td><td>Dec</td></tr>
<tr><td>Entering Fresh Water (adults/jacks)</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="3">Present</td><td></td><td></td></tr>
<tr><td>Spawning</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="2">Present</td><td></td><td></td></tr>
<tr><td>Incubation (eggs)</td><td colspan="2">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="4">Present</td></tr>
<tr><td>Emergence (alevin to fry phases)</td><td></td><td colspan="4">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Rearing and migration (juveniles)</td><td></td><td colspan="5">Present</td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
</table>

Columbia River Chum

<table>
<tr><td>Life History phase</td><td>Jan</td><td>Feb</td><td>Mar</td><td>Apr</td><td>May</td><td>Jun</td><td>Jul</td><td>Aug</td><td>Sep</td><td>Oct</td><td>Nov</td><td>Dec</td></tr>
<tr><td>Entering Fresh Water (adults/jacks)</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="4">Present</td></tr>
<tr><td>Spawning</td><td>Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="2">Present</td></tr>
<tr><td>Incubation (eggs)</td><td colspan="3">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="2">Present</td></tr>
<tr><td>Emergence (alevin to fry phases)</td><td></td><td colspan="3">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Rearing and migration (juveniles)</td><td></td><td colspan="4">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
</table>

Coho Salmon

Lower Columbia River Coho

<table>
<tr><td>Life History phase</td><td>Jan</td><td>Feb</td><td>Mar</td><td>Apr</td><td>May</td><td>Jun</td><td>Jul</td><td>Aug</td><td>Sep</td><td>Oct</td><td>Nov</td><td>Dec</td></tr>
<tr><td>Entering Fresh Water (adults/jacks)</td><td colspan="2">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="4">Present</td></tr>
<tr><td>Spawning</td><td colspan="4">Present</td><td></td><td></td><td></td><td></td><td></td><td colspan="3">Present</td></tr>
<tr><td>Incubation (eggs)</td><td colspan="4">Present</td><td></td><td></td><td></td><td></td><td></td><td colspan="3">Present</td></tr>
<tr><td>Emergence (alevin to fry phases)</td><td></td><td></td><td colspan="5">Present</td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Rearing and migration (juveniles)</td><td colspan="12">Present</td></tr>
</table>

Oregon Coast Coho

<table>
<tr><td>Life History phase</td><td>Jan</td><td>Feb</td><td>Mar</td><td>Apr</td><td>May</td><td>Jun</td><td>Jul</td><td>Aug</td><td>Sep</td><td>Oct</td><td>Nov</td><td>Dec</td></tr>
<tr><td>Entering Fresh Water (adults/jacks)</td><td>Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="3">Present</td></tr>
<tr><td>Spawning</td><td>Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="3">Present</td></tr>
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<tr><td>Rearing and migration (juveniles)</td><td colspan="12">Present</td></tr>
</table>

Southern Oregon / North California Coast Coho

<table>
<tr><td>Life History phase</td><td>Jan</td><td>Feb</td><td>Mar</td><td>Apr</td><td>May</td><td>Jun</td><td>Jul</td><td>Aug</td><td>Sep</td><td>Oct</td><td>Nov</td><td>Dec</td></tr>
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<tr><td>Rearing and migration (juveniles)</td><td colspan="12">Present</td></tr>
</table>

Central California Coast Coho

<table>
<tr><td>Life History phase</td><td>Jan</td><td>Feb</td><td>Mar</td><td>Apr</td><td>May</td><td>Jun</td><td>Jul</td><td>Aug</td><td>Sep</td><td>Oct</td><td>Nov</td><td>Dec</td></tr>
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<tr><td>Rearing and migration (juveniles)</td><td colspan="12">Present</td></tr>
</table>

Sockeye Salmon

Ozette Lake Sockeye

<table>
<tr><td>Life History phase</td><td>Jan</td><td>Feb</td><td>Mar</td><td>Apr</td><td>May</td><td>Jun</td><td>Jul</td><td>Aug</td><td>Sep</td><td>Oct</td><td>Nov</td><td>Dec</td></tr>
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<tr><td>Rearing and migration (juveniles)</td><td colspan="12">Present</td></tr>
</table>

Snake River Sockeye Salmon

<table>
<tr><td>Life History phase</td><td>Jan</td><td>Feb</td><td>Mar</td><td>Apr</td><td>May</td><td>Jun</td><td>Jul</td><td>Aug</td><td>Sep</td><td>Oct</td><td>Nov</td><td>Dec</td></tr>
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<tr><td>Rearing and migration (juveniles)</td><td colspan="12">Present</td></tr>
</table>

Steelhead

Puget Sound Steelhead (winter/summer runs)

<table>
<tr><td>Life History phase</td><td>Jan</td><td>Feb</td><td>Mar</td><td>Apr</td><td>May</td><td>Jun</td><td>Jul</td><td>Aug</td><td>Sep</td><td>Oct</td><td>Nov</td><td>Dec</td></tr>
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<tr><td>Incubation (eggs)</td><td></td><td colspan="7">Present</td><td></td><td></td><td></td><td></td></tr>
<tr><td>Emergence (alevin to fry phases)</td><td></td><td></td><td></td><td colspan="6">Present</td><td></td><td></td><td></td></tr>
<tr><td>Rearing and migration (juveniles)</td><td colspan="12">Present</td></tr>
</table>

Lower Columbia River Steelhead (winter/summer runs)

<table>
<tr><td>Life History phase</td><td>Jan</td><td>Feb</td><td>Mar</td><td>Apr</td><td>May</td><td>Jun</td><td>Jul</td><td>Aug</td><td>Sep</td><td>Oct</td><td>Nov</td><td>Dec</td></tr>
<tr><td>Entering Fresh Water (adults/jacks)</td><td colspan="12">Present</td></tr>
<tr><td>Spawning</td><td></td><td></td><td colspan="4">Present</td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Incubation (eggs)</td><td></td><td></td><td colspan="5">Present</td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Emergence (alevin to fry phases</td><td></td><td></td><td></td><td></td><td colspan="4">Present</td><td></td><td></td><td></td><td></td></tr>
<tr><td>Rearing and migration (juveniles)</td><td colspan="12">Present</td></tr>
</table>

Upper Willamette River Steelhead

<table>
<tr><td>Life History phase</td><td>Jan</td><td>Feb</td><td>Mar</td><td>Apr</td><td>May</td><td>Jun</td><td>Jul</td><td>Aug</td><td>Sep</td><td>Oct</td><td>Nov</td><td>Dec</td></tr>
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<tr><td>Rearing and migration (juveniles)</td><td colspan="12">Present</td></tr>
</table>

Middle Columbia River Steelhead

<table>
<tr><th>Life History phase</th><th>Jan</th><th>Feb</th><th>Mar</th><th>Apr</th><th>May</th><th>Jun</th><th>Jul</th><th>Aug</th><th>Sep</th><th>Oct</th><th>Nov</th><th>Dec</th></tr>
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Upper Columbia River Steelhead

<table>
<tr><th>Life History phase</th><th>Jan</th><th>Feb</th><th>Mar</th><th>Apr</th><th>May</th><th>Jun</th><th>Jul</th><th>Aug</th><th>Sep</th><th>Oct</th><th>Nov</th><th>Dec</th></tr>
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<tr><td>Rearing and migration (juveniles)</td><td colspan="12">Present</td></tr>
</table>

Snake River Basin Steelhead

<table>
<tr><th>Life History phase</th><th>Jan</th><th>Feb</th><th>Mar</th><th>Apr</th><th>May</th><th>Jun</th><th>Jul</th><th>Aug</th><th>Sep</th><th>Oct</th><th>Nov</th><th>Dec</th></tr>
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<tr><td>Rearing and migration (juveniles)</td><td colspan="12">Present</td></tr>
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Northern California Steelhead

<table>
<tr><td>Life History phase</td><td>Jan</td><td>Feb</td><td>Mar</td><td>Apr</td><td>May</td><td>Jun</td><td>Jul</td><td>Aug</td><td>Sep</td><td>Oct</td><td>Nov</td><td>Dec</td></tr>
<tr><td>Entering Fresh Water (adults/jacks)</td><td colspan="2">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="2">Present</td></tr>
<tr><td>Spawning</td><td colspan="2">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td>Present</td></tr>
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Central California Coast Steelhead

<table>
<tr><td>Life History phase</td><td>Jan</td><td>Feb</td><td>Mar</td><td>Apr</td><td>May</td><td>Jun</td><td>Jul</td><td>Aug</td><td>Sep</td><td>Oct</td><td>Nov</td><td>Dec</td></tr>
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California Central Valley Steelhead

<table>
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<tr><td>Rearing and migration (juveniles)</td><td colspan="12">Present</td></tr>
</table>

South- Central California Coast Steelhead

<table>
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Southern California Steelhead

<table>
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<tr><td>Spawning</td><td></td><td></td><td></td><td colspan="2">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
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<tr><td>Rearing and migration (juveniles)</td><td colspan="12">Present</td></tr>
</table>

22 Appendix 6: Washington State Department of Agriculture Propargite and Fenbutatin-oxide Use Summaries

STATE OF WASHINGTON
DEPARTMENT OF AGRICULTURE
P.O. Box 42560 • Olympia, Washington 98504-2560 • (360) 902-1800

PROPARGITE USE SUMMARY

Common Trade Names: Comite, Omite

Use Type: insecticide

Chemical Class: unknown

CAS Number: 1582-09-8

Washington State Use Practices – WSDA Data[1]												
Crop Name	2011 WSDA ESU Crop Acres [2]	Application Date		Lbs AI per Acre		# of Apps		% Acres Treated		App. Method	Total Lbs AI Applied	
		Begin	End	Min	Max	Min	Max	Min	Max		Min	Max
mint	22,359	03/31	11/30	1.64	1.64	1.0	4.0	1.0	24.0	ground	367	35,202
potato[3]	109,542	05/15	10/01	2.05	2.05	1.0	1.0	1.0	1.0	ground	2,246	2,246

[1] This data was collected by WSDA staff through phone interviews and meetings with growers. The data is a profile of "typical" pesticide use.
[2] Washington State Department of Agriculture (WSDA) land use geo-database, with crop acreage clipped to the bounds of the known salmonid ESU acres.
[3] This potato data is exclusive to russet production in the Columbia Basin, so the ESU acreage number provided does not include the 10,000 acres of specialty potatoes grown in western Washington.

STATE OF WASHINGTON
DEPARTMENT OF AGRICULTURE
P.O. Box 42560 • Olympia, Washington 98504-2560 • (360) 902-1800

FENBUTATIN-OXIDE USE SUMMARY

Common Trade Names: Vendex, Hexakis

Use Type: insecticide

Chemical Class: organotin, heavy metal

CAS Number: 13356-08-6

Washington State Use Practices – NASS Data[1]

Crop Name	2011 WSDA ESU Crop Acres [2]	Application Date		Lbs AI per Acre		# of Apps		% Acres Treated		App. Method	Total Lbs AI Applied	
		Begin	End	Min	Max	Min	Max	Min	Max		Min	Max
apple	155,831	03/15	07/01	1.0	1.0	1.0	1.0	1.0	1.0	Air blast	1,558	1,558
pear	21,327	05/01	08/31	1.0	1.0	1.0	1.0	2.0	2.0	Air blast	427	427

[1] This data has been supplied to WSDA by the USDA's National Agricultural Statistics Service (NASS). Each data point provided by NASS is a compilation of grower-provided data collected directly from pesticide use records.

[2] Washington State Department of Agriculture (WSDA) land use geo-database, with crop acreage clipped to the bounds of the known salmonid ESU acres.

Washington State Use Practices – WSDA Data[1]												
Crop Name	**2011 WSDA ESU Crop Acres [2]**	**Application Date**		**Lbs AI per Acre**		**# of Apps**		**% Acres Treated**		**App. Method**	**Total Lbs AI Applied**	
		Begin	**End**	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**		**Min**	**Max**
caneberry	12,810	05/31	08/15	1.0	1.0	1.0	1.0	20.0	20.0	ground	2,562	2,562
strawberry	1,513	06/01	09/30	1.0	1.0	1.0	1.0	10.0	10.0	ground	151	151

[1] This data was collected by WSDA staff through phone interviews and meetings with growers. The data is a profile of "typical" pesticide use and supplements NASS data on minor crops in Washington State.

[2] Washington State Department of Agriculture (WSDA) land use geo-database, with crop acreage clipped to the bounds of the known salmonid ESU acres.

FENBUTATIN-OXIDE USE SUMMARY

Washington State Department of Agriculture

January 23, 2013

References:

Database

- 2011 Washington State registered pesticide labels
- U.S. Department of Agriculture National Agricultural Statistics Service, 2003 Apple Chemical Use Survey
- U.S. Department of Agriculture National Agriculture Statistics Service, 2003 Pear Chemical Use Survey

Meeting

- Northwest Washington Berry Growers. November 16, 2009.

Web site

- ExToxNet Pesticide Information Profiles: http://ace.orst.edu/info/extoxnet/pips/pips.html
- Greenbook Product Directory: http://www.greenbook.net/
- U.S. Department of Agriculture National Agricultural Statistics Service - Agricultural Chemical Use Database: http://www.pestmanagement.info/nass/

Fenbutatin-oxide registered uses without available pesticide use data

Cherry, Christmas tree, eggplant, grape, nectarine, nursery, nursery (greenhouse), ornamental, ornamental (greenhouse), ornamental bulb, ornamental deciduous/shade tree, ornamental flower, ornamental flower (greenhouse), ornamental rose, ornamental shrub, ornamental tree, peach, plum, prune, walnut

23 APPENDIX 7: Population Modeling

Population Modeling

Introduction

To assess the potential for adverse impacts of the pesticides on Pacific salmon populations, two models were developed that explicitly link pesticide exposures to population-level impacts.

A growth model was constructed that estimated the population-level impacts of increased juvenile size-dependent mortality resulting from reduced growth due to decreases in prey abundance from exposures to pesticides. These models excluded direct effects of the pesticide exposures on salmon and focused on the population-level outcomes resulting from decreased juvenile growth due to an annual exposure of salmon prey to a pesticide. The population-level impact was determined using a size-dependent change in first year survival in a salmon life-history matrix.

An acute toxicity model was constructed that estimated the population-level impacts of juvenile mortality resulting from lethal exposures to the pesticides. These models excluded sublethal and indirect effects of the pesticide exposures and focused on the population-level outcomes resulting from increased mortality due to a single, annual exposure of subyearlings to a pesticide. The lethal impact was implemented as a change in first year survival in a salmon life-history matrix.

The 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 (McElhany et al. 2000, 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 models were developed to serve as a means to assess the potential effects on ESA-listed salmon populations from exposure to pesticides. Assessing the results from different pesticide exposure scenarios relative to a control (i.e. unexposed) scenario can indicate the potential for pesticide exposures to lead to changes in the survival of individual subyearling salmon either indirectly due to reduced growth (growth model) or directly (acute mortality model). While models were constructed for different salmon species and life-history strategies, they were not specific to a particular salmon population. For many salmon populations, the necessary demographic data are unavailable. Additionally, while there may be differences in some demographic rates among individual populations, the similarity of response within a life-history strategy is well established (Winemiller and Rose 1992; Schaaf et al. 1987, Schaaf et al. 1993). Generic models, therefore, are still useful to estimate risk across populations that share a life history strategy. Consequently, changes in salmon population dynamics as indicated by percent change in a population's intrinsic rate of increase (λ) 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.

Growth Model

Insecticides have 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 toxic concentrations of insecticides for as little as minutes or hours, but even a brief exposure can result in a massive flux or "spike" of dead or dying invertebrates from the benthos into the water column and a subsequent depletion of populations that can take months or even years to recover (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 inhibiting pesticide) and fenvalerate (another commonly used insecticide), eight of eleven abundant invertebrate species disappeared initially and the remaining three were reduced in abundance (Liess and Schulz 1999). Recovery of most species occurred within 11 months (Liess and Schulz 1999), however reductions like these, in both the benthos and the water column (drift), can last long enough to affect higher trophic levels. For example, monthly and bi-weekly applications of the chitin-inhibiting insecticide diflubenzuron to pond mesocosms resulted in direct reductions (fivefold) of invertebrate abundances and biomass, and thus indirectly resulted in a 50% reduction in the weight of predators (juvenile bluegill) (Boyle et al. 1996).

Long-term changes in invertebrate densities and community composition from pesticide exposure would likely result in reductions in prey for juvenile salmon. Because pesticides are often more toxic to invertebrates than vertebrates, concentrations of pesticides that may not be lethal for fish may be sufficient to indirectly affect salmon via reductions in their prey (Peterson et al. 2001). 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 inhibiting insecticides (Liess and Schulz 1999, Schulz et al. 2002) and with diflubenzuron (Boyle et al. 1996), suggesting indirect effects to salmon via prey availability may be similar across various classes of insecticides.

One likely biological consequence of reduced 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 subyearling 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 availability of prey items and related reduced somatic growth, such as an insecticide exposure, could result in decreased first-year survival and, thus, reduce population productivity.

Using a set of life-history matrix models, changes to the size of subyearling salmon from exposure to 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 (Figure 1) 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 and see below). 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 the populations of all the species. While there may be differences in some demographic rates among individual populations, the similarity of response within a life-history strategy is well established (Winemiller and Rose 1992; Schaaf et al. 1987, Schaaf et al. 1993).

The growth model consists of two parts, an organismal portion and a population portion. The organismal portion of the model links insecticide exposure to reduced prey abundance 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 7.12.0 (R2011a) (The MathWorks, Inc. Natick, MA).

Organismal Portion

For the organismal model a relationship between prey availability and somatic growth of salmonid fingerlings was developed using a series of relationships between pesticide exposure, prey abundance, food ration, and somatic growth rate (Figure 2). 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.

The models allow exposures that can include multiple pesticides over various time pulses. 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). The relative concentration for each day was summed across all the pulses to result in a total concentration for each day. Sigmoidal dose-response relationships, at steady-state, between each single pesticide exposure and relative prey abundance are modeled using specific EC50s and slopes (Figure 2). The sigmoid slope used in the calculation with 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 effects of pesticide exposure on feeding are incorporated by multiplying the relative prey abundance following exposure by the control ration to determine the ration available for exposed fish. The ration for all fish would be equal to control ration if there were no pesticide exposure.

Next, additional empirical data (e.g. Weatherley and Gill 1995) defined the relationship between ration and somatic growth rate (Figure 2D). 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. Combining the relationship between ration and somatic growth rate (Figure 2D) with the timecourse of prey abundance (Figure 2C) produces a relationship representing somatic growth rate over time (Figure 2E), which is then used to model individual growth rate and size over time incorporating the impact of pesticide exposure on prey abundance.

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 (Nelson et al. 2004). 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.

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 2B). 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. The floor abundance is separate from, 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 Figure 2.

Figure 2A uses a step function:

time < start; exposure = 0

start ≤ time ≤ end; exposure = exposure concentration(s)

time > end; exposure = 0.

Figure 2B uses a sigmoid function:

y = bottom + (top – bottom)/(1 + (exposure concentration/EC50)^slope).

y = prey abundance, top = Pc (in this case 1), bottom = Pf.

For Figure 2C, an exposure pulse would result in a 1-day spike followed by a decline to the impacted level based upon the prey toxicity. Even during exposures resulting in low prey toxicity, some 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

Figures 2D use a linear function (the point-slope form of a line):

y = m*(x – x1) + y1.

m = Mgr, x1 = Rc, and y1 = Gc.

Figure 2E is generated by using the output of Figure 2C for a given time as the input for 2D. Performing this series of computations across multiple days produces the entire relationship in 2E.

Population Portion

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 control survival ϕ value when Δlength equals zero.

Equation D: $\Delta\text{length} = \text{fish length(mm)} - \text{mean length(mm)}$

Equation S: Survival $\phi = (e^{(\alpha+(0.0329*\Delta\text{length}))}) / (1 + e^{(\alpha+(0.0329*\Delta\text{length}))})$

Randomly selecting length values from the normal distribution calculated from the organismal model size output 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 for each model iteration (year). Ocean conditions, freshwater habitat, fishing pressure, and marine resource availability were assumed constant and density independent.

Threatened and endangered species are, by definition, at historically low population abundances. Assumptions of density-independent demographic processes are therefore appropriate, and also in accord with previous modeling by NMFS to support the management and recovery of ESA-listed salmonids (e.g., Hinrichsen, 2002, McElhany et al. 2000, McClure et al. 2003). For the majority of ESA-listed stocks, information on watershed or sub-basin carrying capacity, juvenile density, and related factors are lacking. Without additional information on the type, intensity, form of the density-survival relationship, as well as the carrying capacity for each stock, it is inappropriate to utilize density-dependent models for assessing the viability of a salmon population (McElhany et al. 2000).

Also, when estimating risks to wild populations, the ESA requires that resource agencies minimize the likelihood of Type II errors – i.e., predicting no impacts to listed species when in fact such impacts exist. Density-dependent compensation is unlikely to be a factor for subyearling salmon survival and abundance within relatively rare populations. Moreover, these dynamics are population-specific, with each population having different habitats, densities, and carrying capacities. Assuming density-dependent compensation across populations where this is negligible or nonexistent is a form of a Type II error. In the absence of direct data to the contrary, a presumption of density-independent dynamics gives the benefit of the doubt to listed species, as mandated by the ESA.

In the model an individual fish experiences an exposure scenario once as a subyearling (during its first spring/summer) and never again. The pesticide exposure is assumed to occur annually. During each year's exposure, all subyearlings within a given population are assumed to have their prey base exposed to the pesticide. The model integrates this as every brood class being exposed as subyearlings and that the vital demographic rates of the transition matrix are continually impacted in the same manner.

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. It is reported as the mean percent change in lambda and standard deviation to normalize the values and allow comparison across populations.

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 7.12.0 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, subyearling salmon are assumed to be exposed once each year. 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 (similar to that used for Figure 2B) 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 3. 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 Outputs

The outputs for all scenarios of both the growth model and acute model are shown in the main text.

While strong trends in effects were seen 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.

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]
prey floor	0.20[9]	n/a	0.07
prey recovery rate	0.01[10]	n/a	0.15
control prey drift	1.0[4]	0.05	2.15
somatic growth rate (Gc)	1.3[11]	0.06[5]	2.49
potential ration	5[8]	0.56[8]	-0.14
growth vs. ration slope (Mgr)	0.35[6]	0.02[6]	-0.14
initial weight	1 gram[7]	0.1[7]	0.99

[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 Brett et al. 1969

[6] data from Brett et al. 1969 have no variability (ration was the independent variable) so a variability of 1% was selected to introduce some variability

[7] consistent with field-collected data for juvenile Chinook (Nelson et al. 2004)

[8] incorporates fish feeding behavior and activity and related variability

[9] estimated from Van den Brink et al. 1996

[10] derived from Ward et al. 1995, Van den Brink et al. 1996, Colville et al. 2008

[11] 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. 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

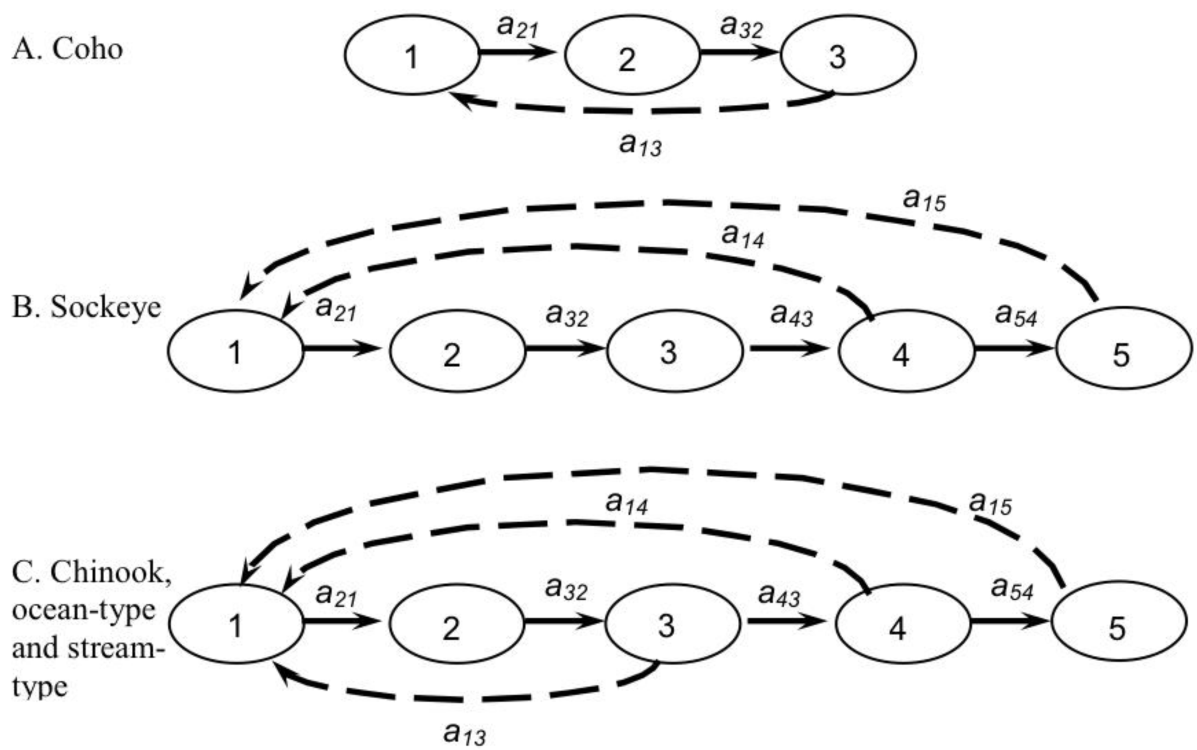
[1] Value calculated from data in Healy and Heard 1984, Fast et al. 1988, Beckman et al. 2000, Knudsen et al. 2006

[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

Figure 1.



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 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.

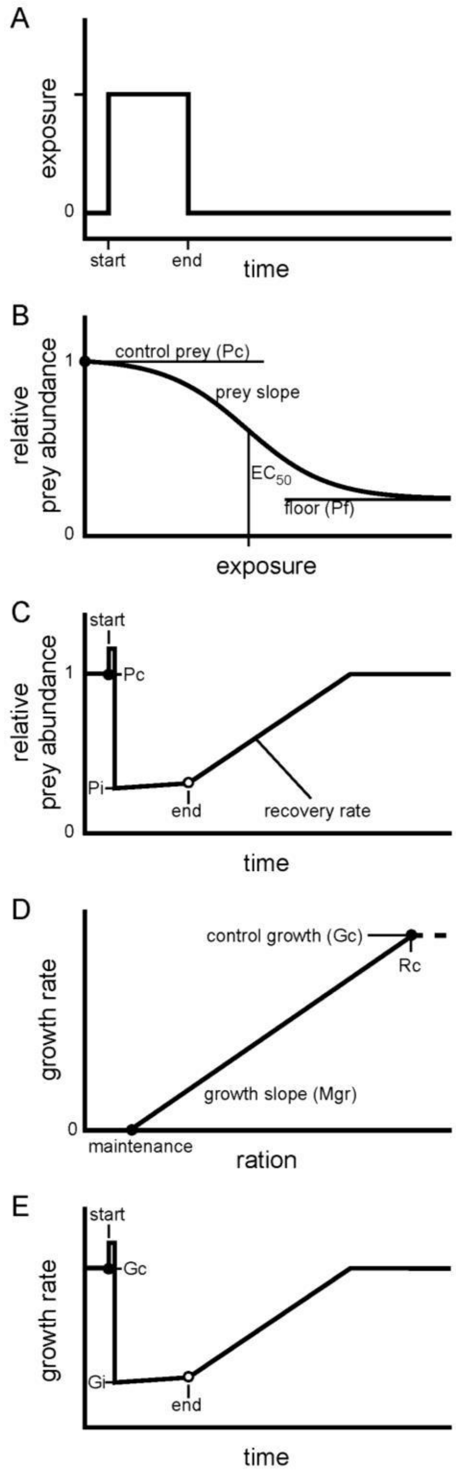

A
exposure
0
start
end
time
B
relative
prey abundance
1
0
control prey (Pc)
prey slope
EC50
floor (Pf)
exposure
C
start
1
Pc
Pi
0
end
recovery rate
relative
prey abundance
time
D
growth rate
0
control growth (Gc)
Rc
growth slope (Mgr)
maintenance
ration
E
start
Gc
Gi
end
growth rate
time

Figure 2: Relationships used to link pesticide exposure to the availability of prey. See text for details. A) Representation of a single exposure to constant level of 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), sigmoidal 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 decrease to the level of impact (based on B) or the floor, whichever is greater. During exposures, recovery can begin at the constant prey influx rate multiplied by the current level of toxicity. After exposure, recovery to control prey is at the constant rate of influx from upstream habitats. D) A linear model was used to relate ration to growth rate using a line passing through the control conditions and through the maintenance condition with a slope denoted by Mgr. E) Timecourse for effect of exposure to an insecticide on growth rate produced by combining C & D. For each time point, ration is determined by multiplying the control ration (Rc) by the relative prey abundance and then used to calculate growth rate.

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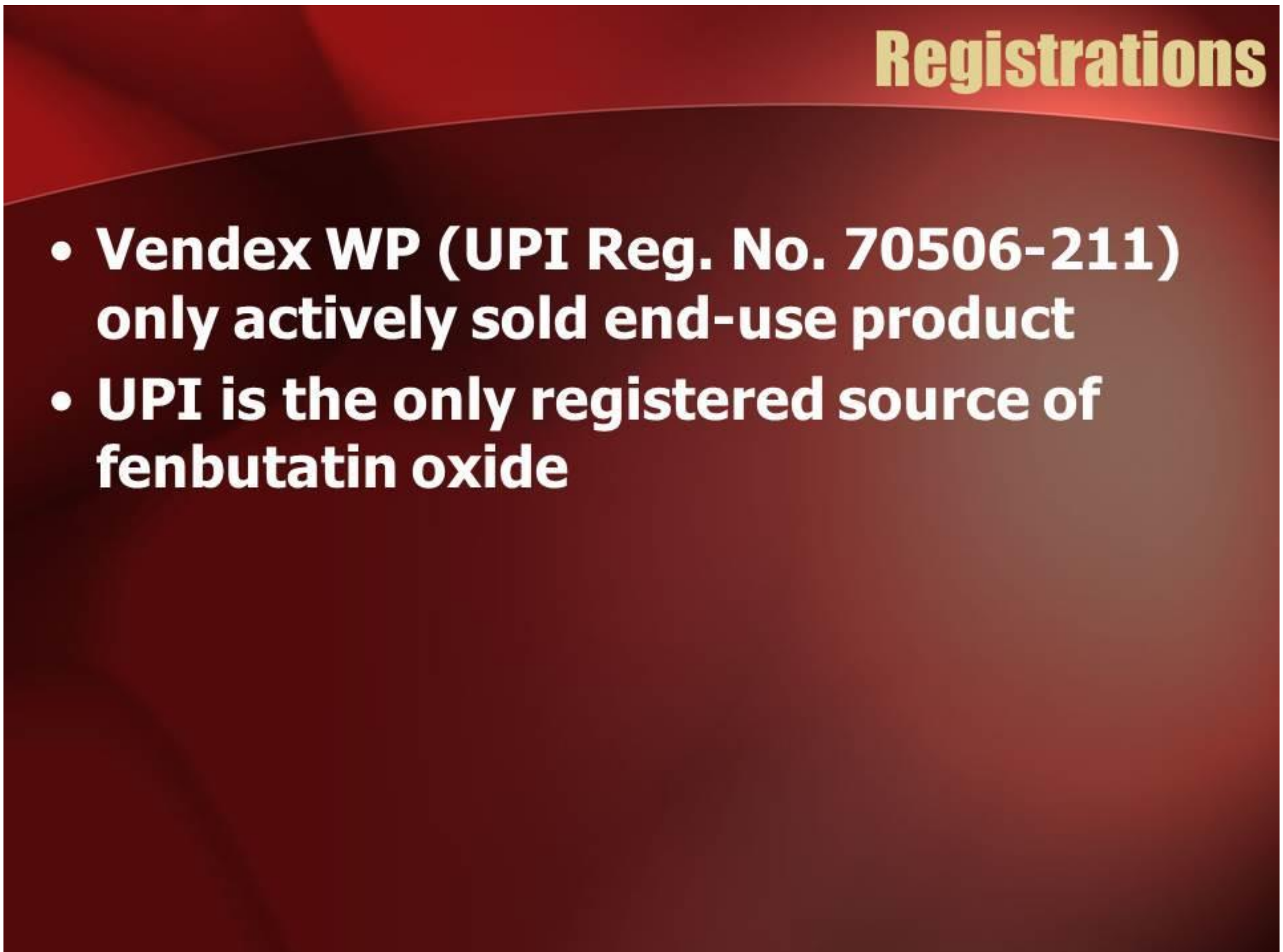
24 Appendix 8: Proposed Voluntary Label Mitigations for Fenbutatin-oxide by United Phosporous Inc. (UPI)

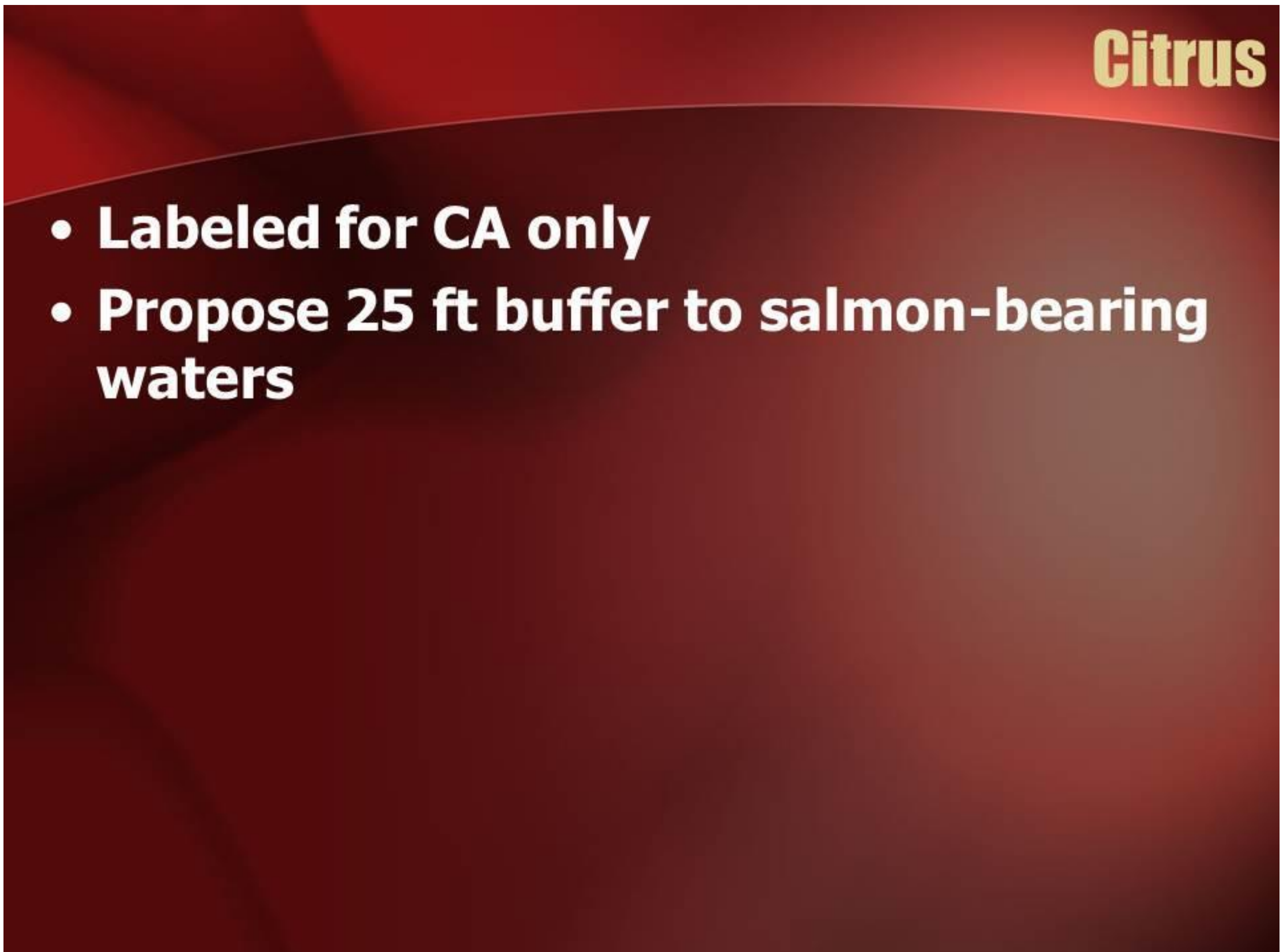
The following PowerPoint slide presentation was provided to NMFS by UPI during a January 16, 2013 meeting between NMFS, UPI, EPA and USDA.

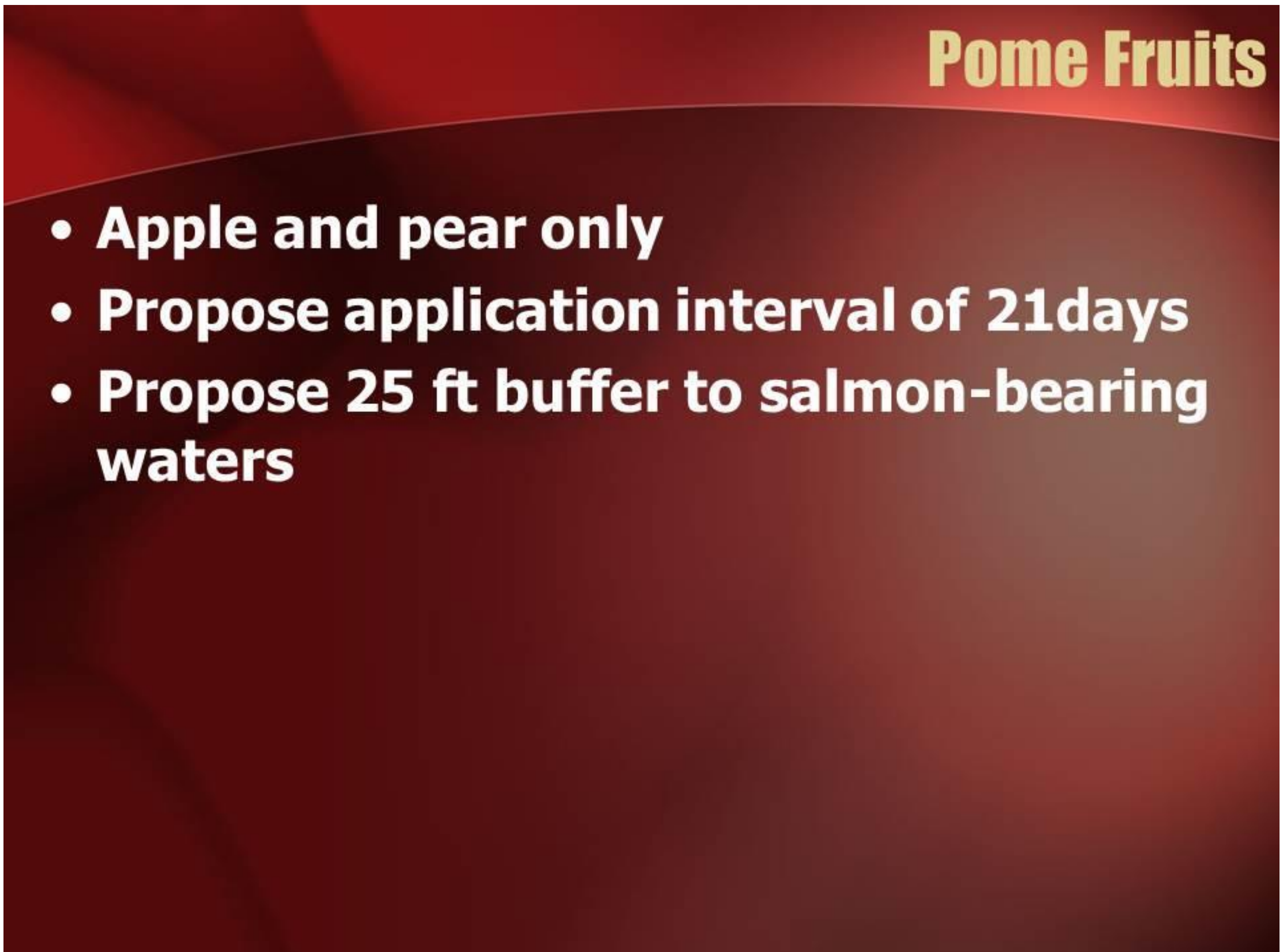

Fenbutatin Oxide Biological Opinion
Proposed Voluntary Label Mitigations
United Phosphorus, Inc.
Jan. 16, 2013

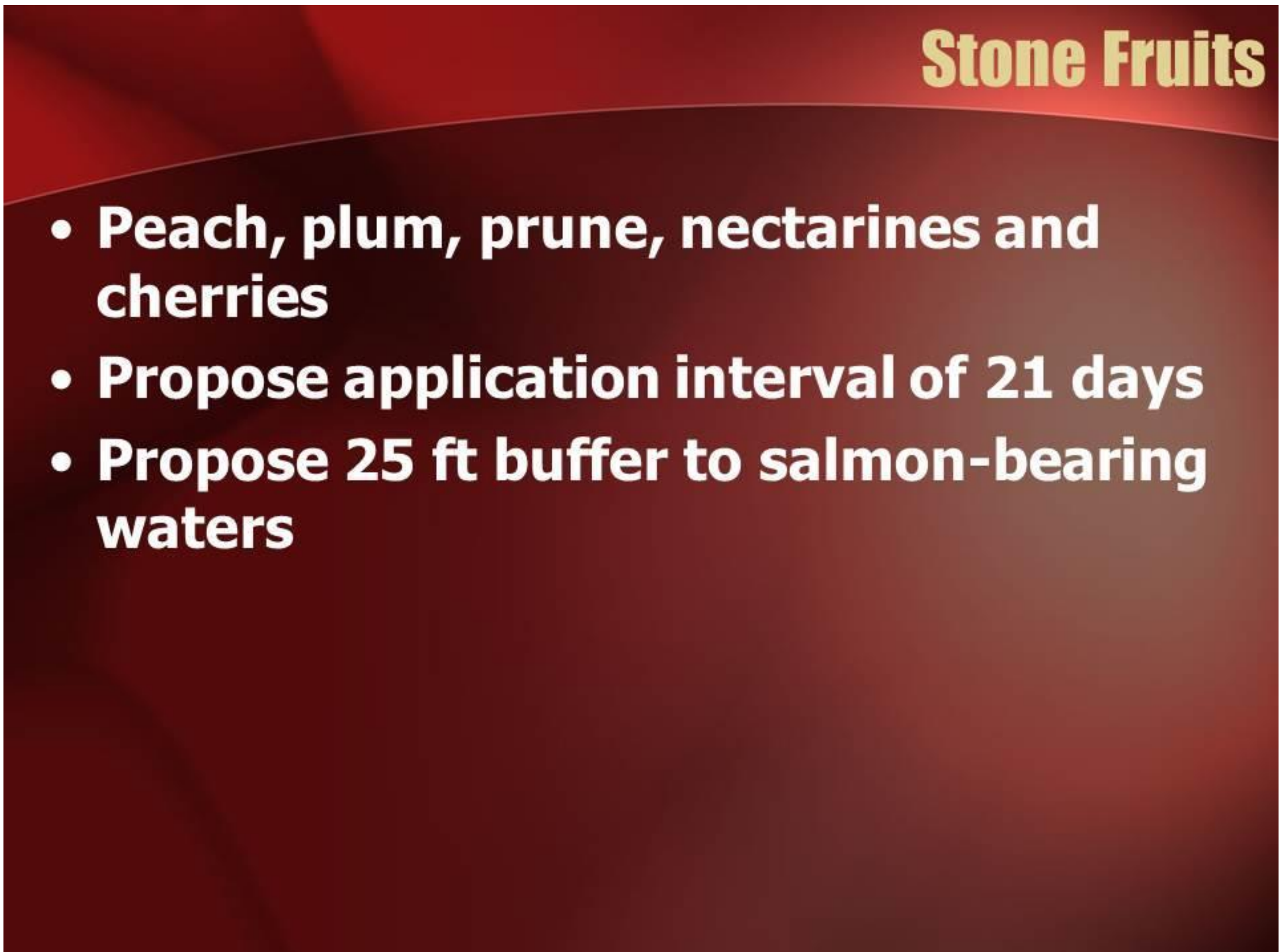

Registrations
• Vendex WP (UPI Reg. No. 70506-211) only actively sold end-use product
• UPI is the only registered source of fenbutatin oxide

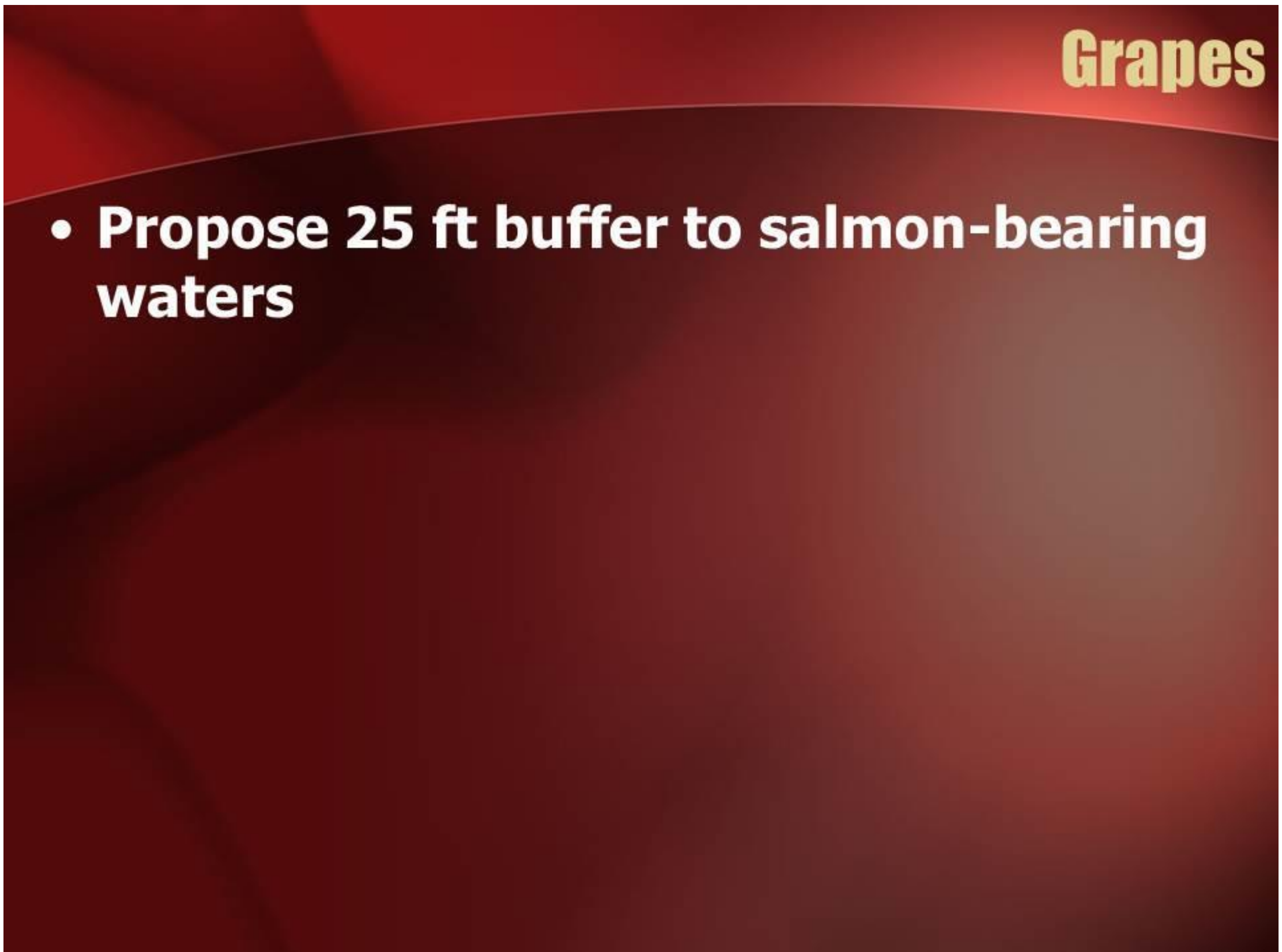

General Proposed Restrictions
• UPI is willing to limit technical label for formulation into products incorporating proposed mitigations
– Ensures SePro (EPA Reg. No. 67690-40) would need to implement mitigations if using UPI as a source
• Prohibit aerial application in CA, ID, WA and OR
• Considered vegetative buffer but not feasible because of new CA Food Safety legislation

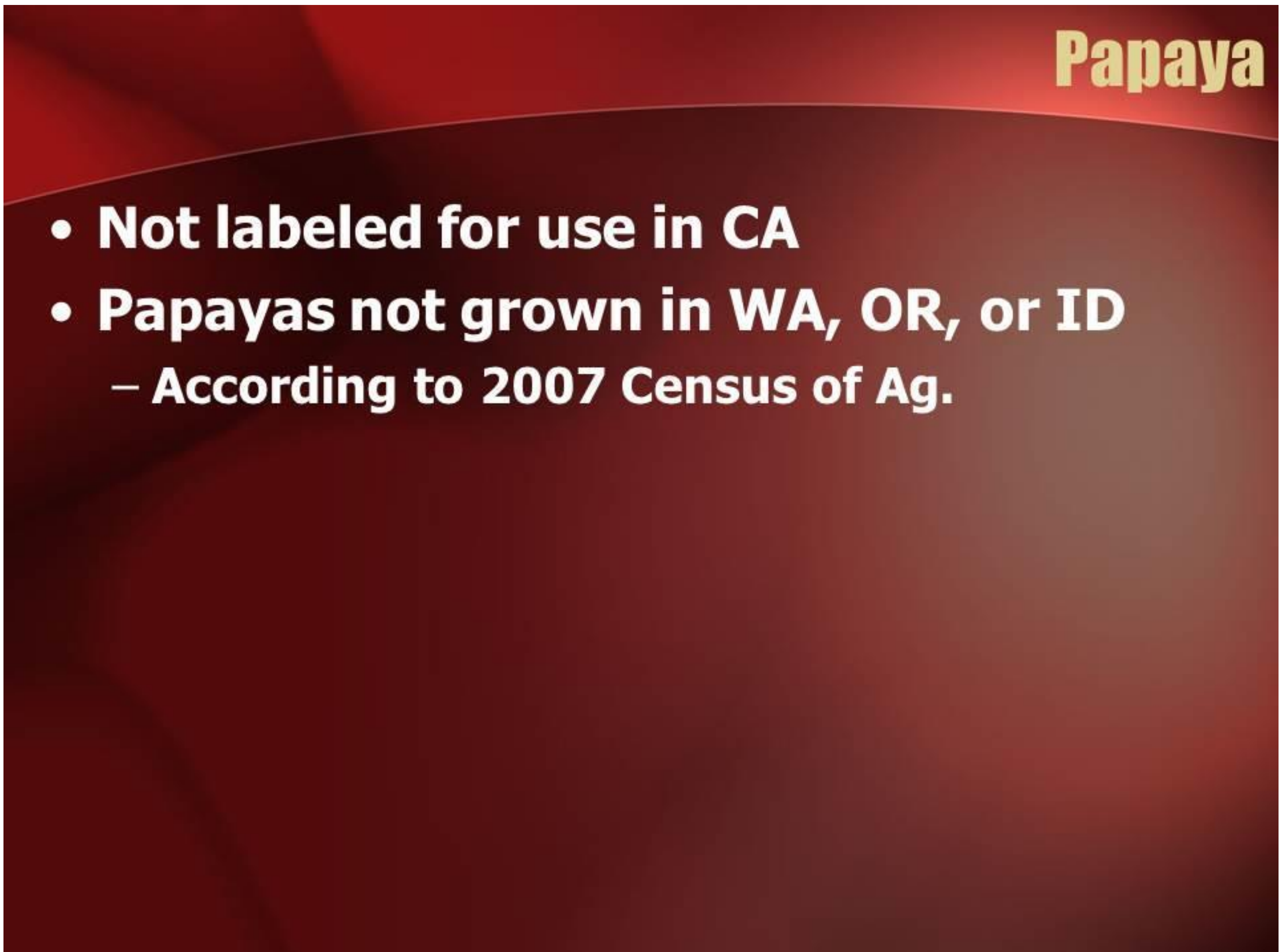

Current Uses
• Citrus
• Pome Fruits
• Stone Fruits
• Grapes
• Papaya
• Tree Nuts
• Strawberries
• Eggplant
• Raspberries
• Christmas Trees
• Ornamentals (Commercial and Landscape)

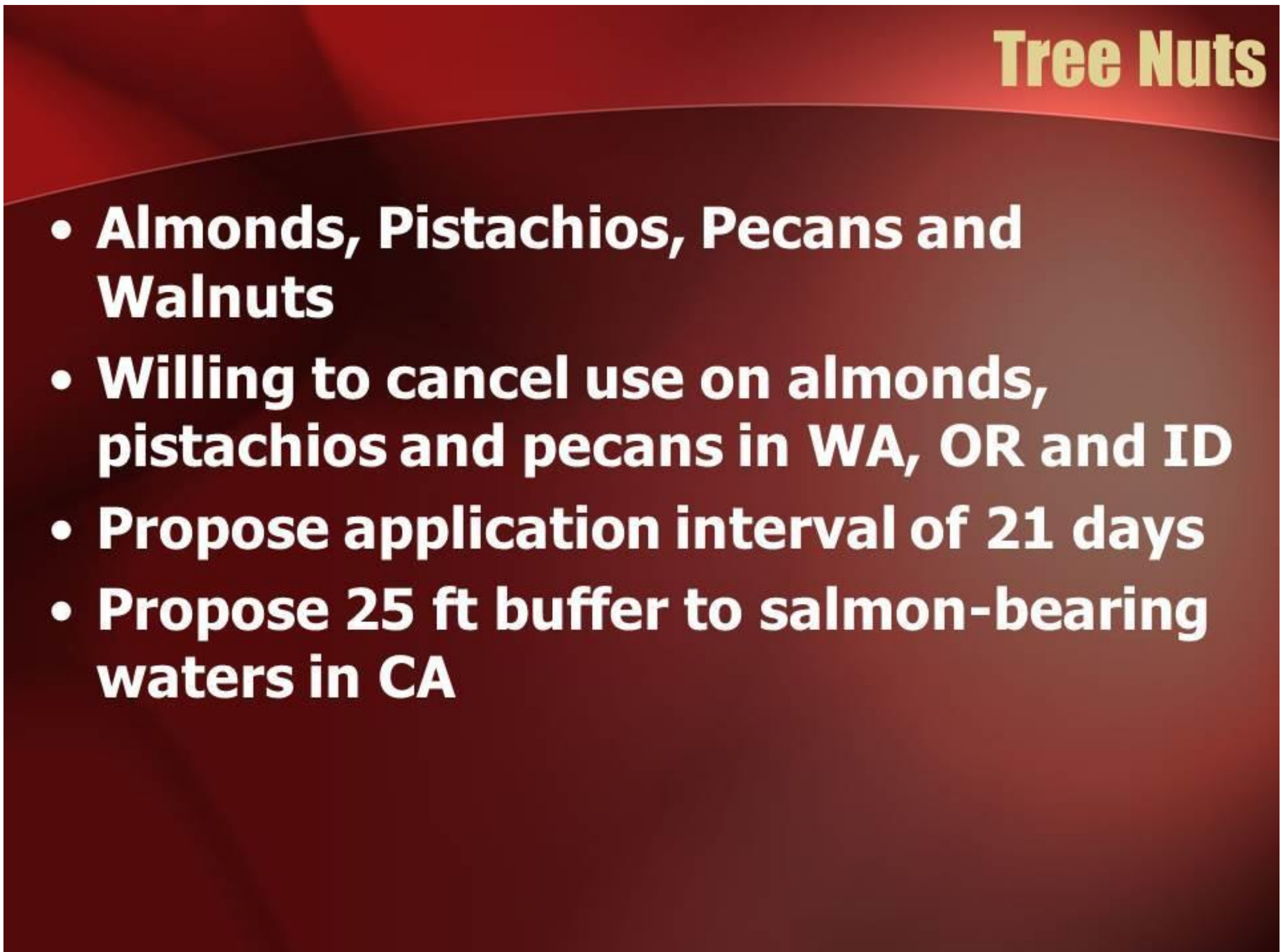

Citrus
• Labeled for CA only
• Propose 25 ft buffer to salmon-bearing waters

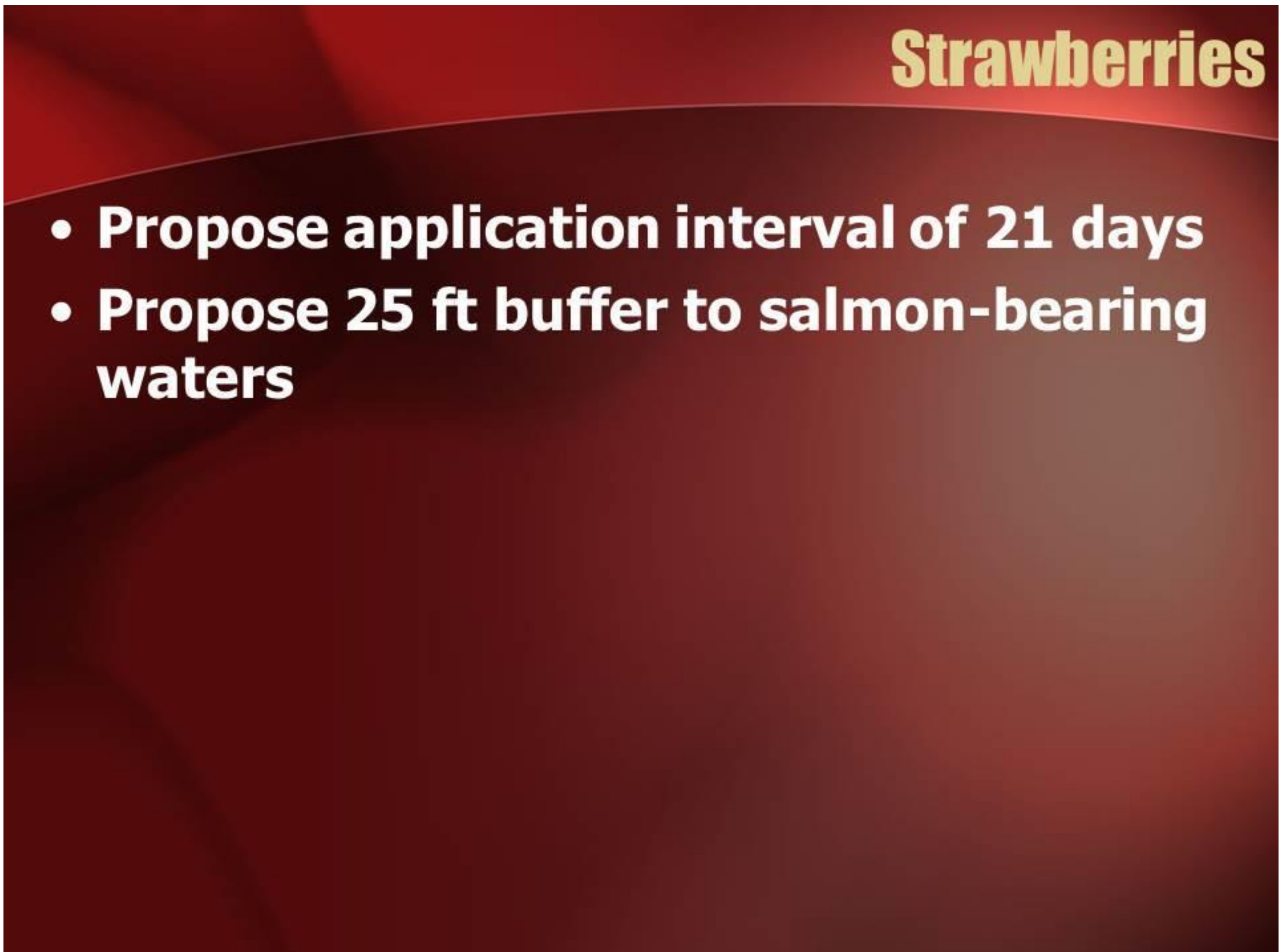

Pome Fruits
• Apple and pear only
• Propose application interval of 21days
• Propose 25 ft buffer to salmon-bearing waters

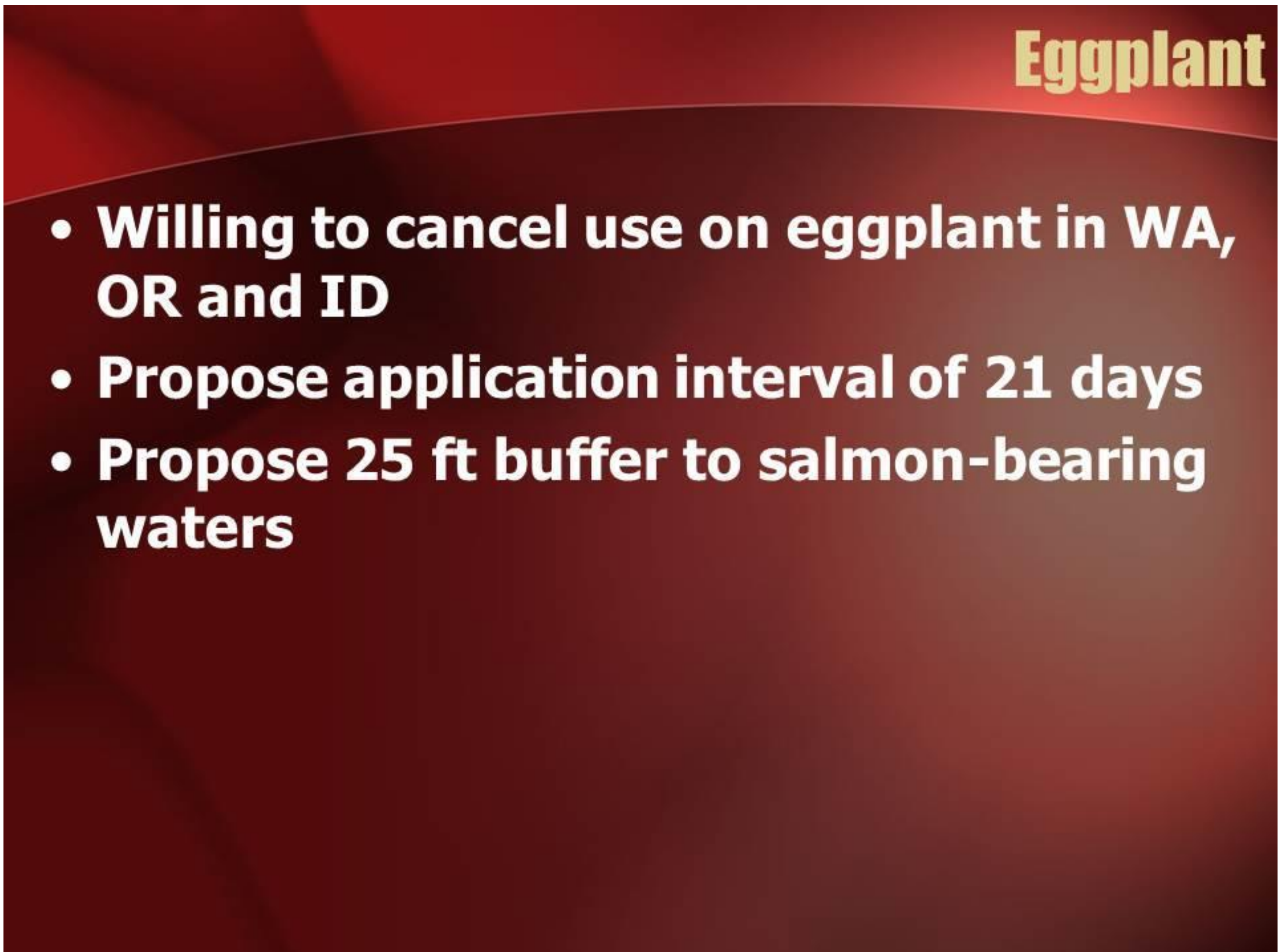

Stone Fruits
• Peach, plum, prune, nectarines and cherries
• Propose application interval of 21 days
• Propose 25 ft buffer to salmon-bearing waters

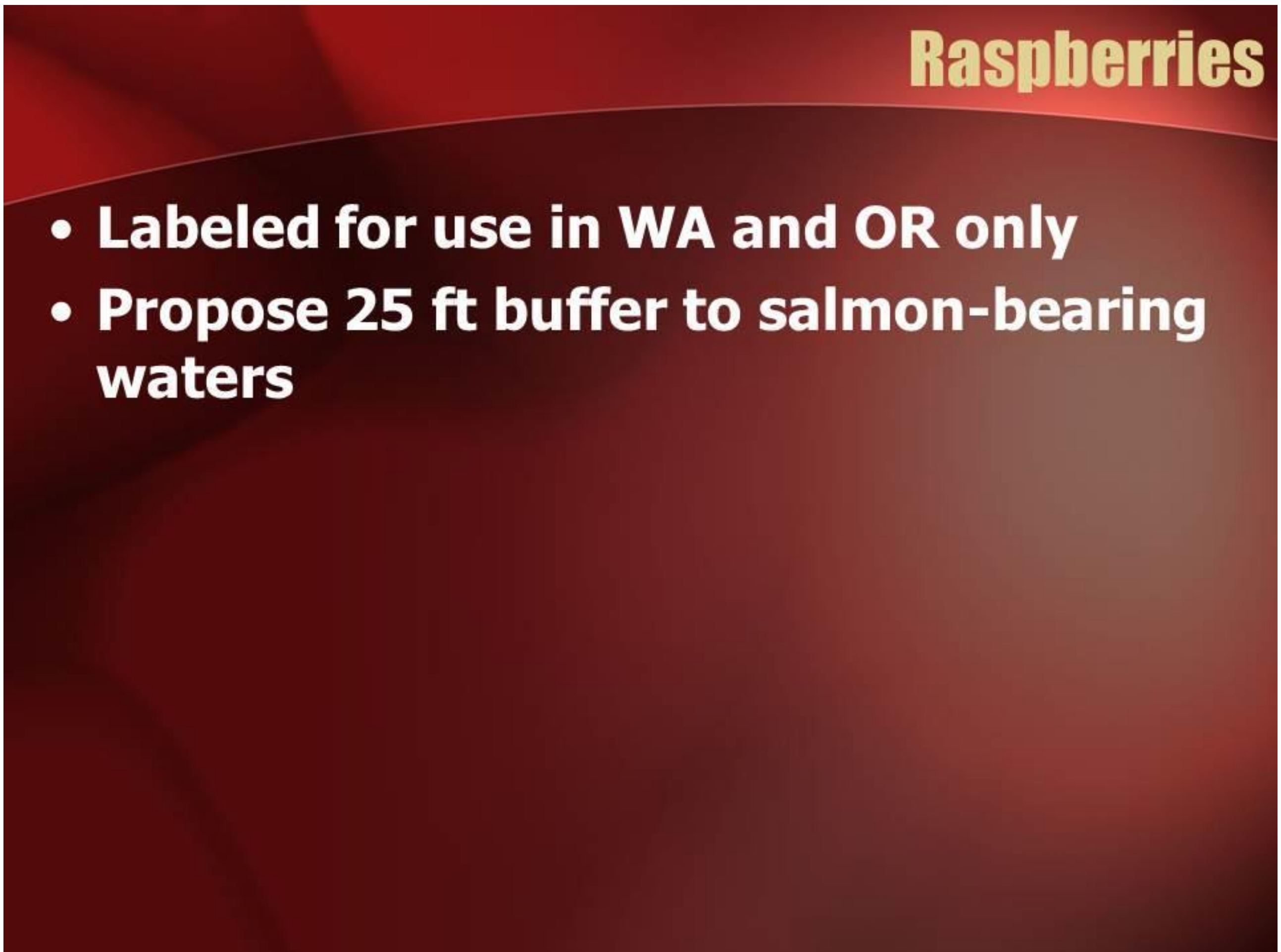

Grapes
• Propose 25 ft buffer to salmon-bearing waters

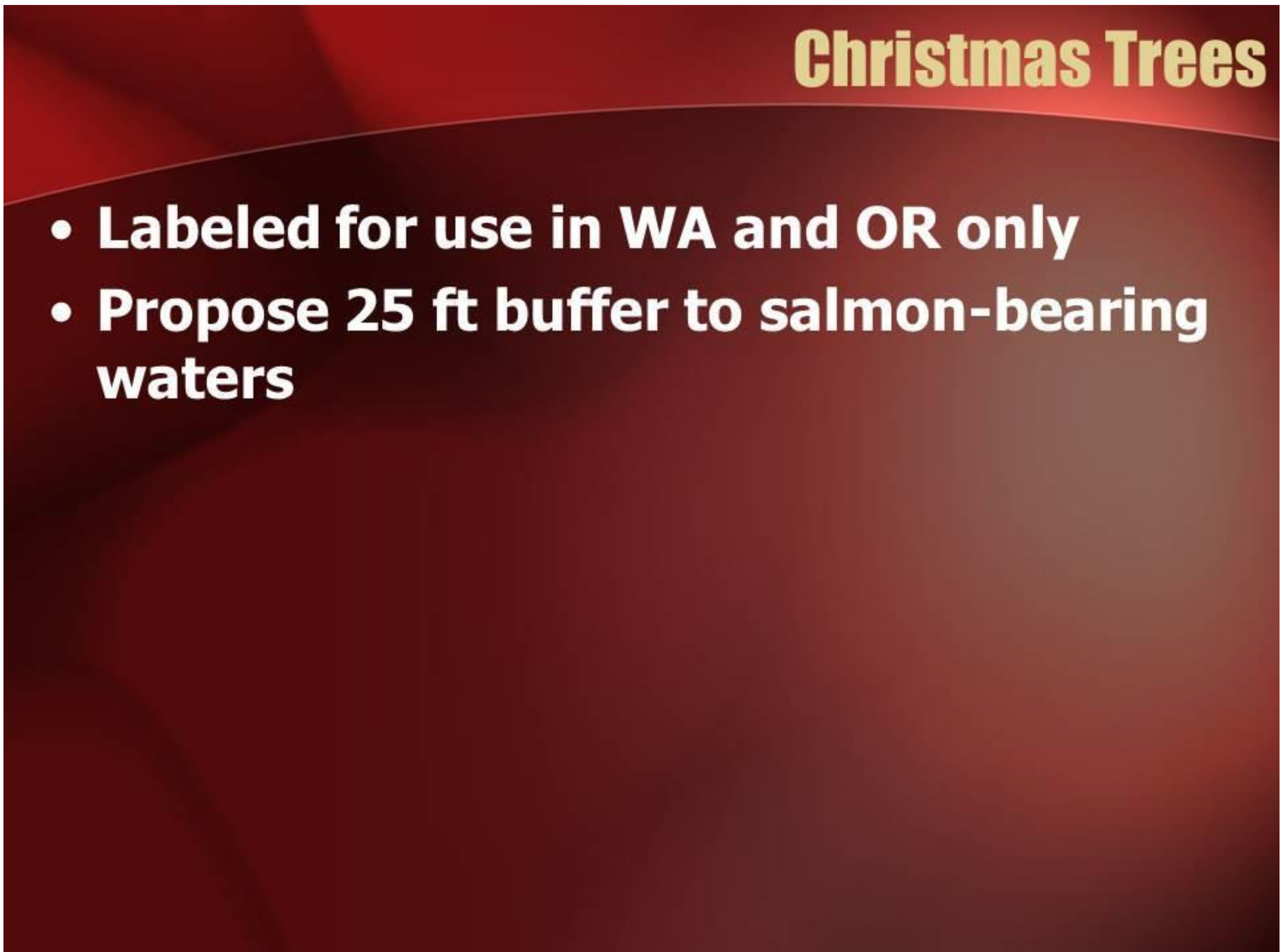

Papaya
• Not labeled for use in CA
• Papayas not grown in WA, OR, or ID
– According to 2007 Census of Ag.

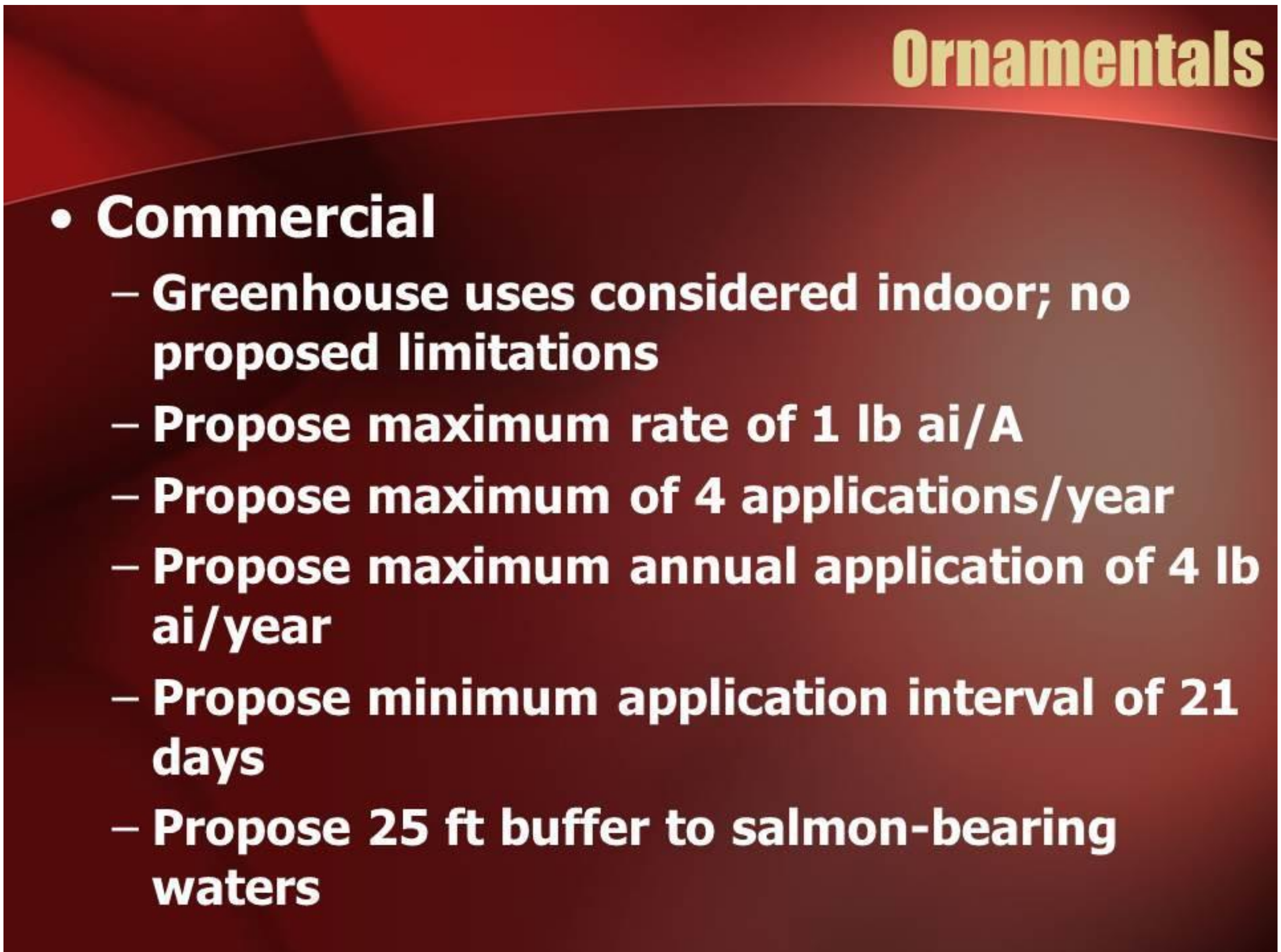

Tree Nuts
• Almonds, Pistachios, Pecans and Walnuts
• Willing to cancel use on almonds, pistachios and pecans in WA, OR and ID
• Propose application interval of 21 days
• Propose 25 ft buffer to salmon-bearing waters in CA

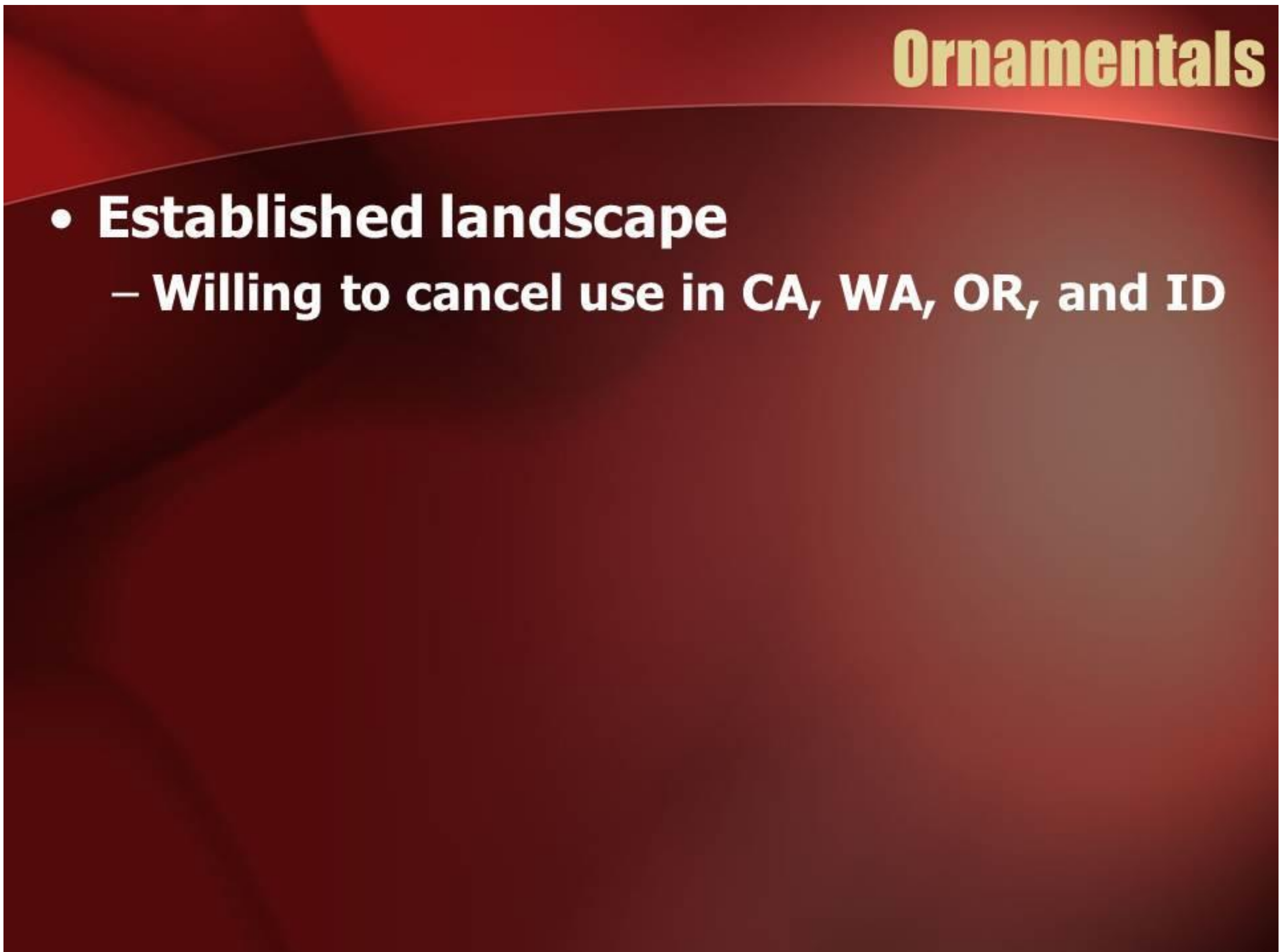

Strawberries
• Propose application interval of 21 days
• Propose 25 ft buffer to salmon-bearing waters


Eggplant
• Willing to cancel use on eggplant in WA, OR and ID
• Propose application interval of 21 days
• Propose 25 ft buffer to salmon-bearing waters


Raspberries
• Labeled for use in WA and OR only
• Propose 25 ft buffer to salmon-bearing waters


Christmas Trees
• Labeled for use in WA and OR only
• Propose 25 ft buffer to salmon-bearing waters


Ornamentals
• Commercial
– Greenhouse uses considered indoor; no proposed limitations
– Propose maximum rate of 1 lb ai/A
– Propose maximum of 4 applications/year
– Propose maximum annual application of 4 lb ai/year
– Propose minimum application interval of 21 days
– Propose 25 ft buffer to salmon-bearing waters


Ornamentals
• Established landscape
– Willing to cancel use in CA, WA, OR, and ID