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

JUN 30 2011

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

Dear Mr. Bradbury:

Enclosed is the National Oceanic Atmospheric Administration National Marine Fisheries Service's (NMFS) biological opinion (Opinion), issued under the authority of section 7(a)(2) of the Endangered Species Act (ESA) of 1973 as amended (16 U.S.C. 1536(a)(2)), on the effects of the U.S. Environmental Protection Agency's (EPA) proposed registration of pesticide products containing the active ingredients 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil, on endangered species, threatened species, and critical habitat that has been designated for those species. This Opinion assesses the effects of all pesticides containing the above listed ingredients on 28 listed Pacific salmonids.

After considering the status of the listed resources, the environmental baseline, and the direct, indirect, and cumulative effects of EPA's proposed action on listed species, NMFS concludes that pesticide products containing triclopyr BEE, diuron, linuron, captan, and chlorothalonil are not likely to jeopardize the continuing existence of any listed Pacific salmonids. NMFS has concluded that 2,4-D is likely to jeopardize the continuing existence of 28 listed Pacific salmonids. NMFS also concludes that the effects of products containing triclopyr BEE, linuron, and captan are not likely to destroy or adversely modify designated critical habitat for listed Pacific salmonids as described in the attached Opinion. Finally, NMFS concludes that the effects of products containing 2,4-D, diuron, and chlorothalonil are likely to destroy or adversely modify designated habitat for some listed Pacific salmonids as described in the attached Opinion. As NMFS has not designated critical habitat for Lower Columbia River coho salmon or Puget Sound steelhead, the Opinion presents no further critical habitat analysis for the these species.

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

If you have questions regarding this Opinion please contact me or Ms. Angela Somma, Chief of our Endangered Species Division at (301) 713-1401.

Sincerely,

for James H. Lecky
Director, Office of Protected Resources

National Marine Fisheries Service

Endangered Species Act Section 7 Consultation

Biological Opinion

Environmental Protection Agency

Registration of Pesticides

2,4-D, Triclopyr BEE, Diuron, Linuron, Captan, and Chlorothalonil

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June 30, 2011

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Agency: United States Environmental Protection Agency

Activities Considered: Authorization of pesticide products (as described by product labels) containing the active ingredients 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil, and their formulations in the United States and its affliated territories.

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

Approved by: Helen Golde for J. H. Lecky

Date: June 30, 2011

Section 7(a)(2) of the Endangered Species Act of 1973, as amended (ESA; 16 U.S.C. §1531 *et seq.*) requires each federal agency to insure that any action they authorize, fund, or carry out is not likely to jeopardize the continued existence of any endangered or threatened species or result in the destruction or adverse modification of critical habitat of such species. When a federal agency's action "may affect" a protected species, that agency is required to consult formally with the National Marine Fisheries Service (NMFS) or the U.S. Fish and Wildlife Service (USFWS), depending upon the endangered species, threatened species, or designated critical habitat that may be affected by the action (50 CFR §402.14(a)). Federal agencies are exempt from this general requirement if they have concluded, with written concurrence from the U.S. Fish and Wildlife Service, NMFS or both, that an action "may affect but is not likely to adversely affect" endangered species, threatened species or designated critical habitat (50 CFR §420.14(b)).

The United States (U.S.) Environmental Protection Agency (EPA) initiated consultation with NMFS on its 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.s) of 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil from August 1, 2003 through December 1, 2004. EPA authorization of pesticide uses are categorized as FIFRA sections 3 (new product registrations), 4 (reregistrations and special review), 18 (emergency use), or 24(c) [Special Local Needs (SLN)]. At that time, EPA determined that uses of pesticide products containing these ingredients "may affect" some, most or all (depending on a.i.) of the 26 Evolutionarily Significant Units (ESUs) of Pacific salmonids listed as endangered or threatened and designated critical habitat for the ESUs. This document represents NMFS' biological opinion (Opinion) on the impacts of EPA's authorizations of pesticide products containing the above-mentioned a.i.s on the listed ESUs, plus on two newly listed salmonids. This is a partial consultation because pursuant to the court's order, EPA sought consultations on only this group of listed species under NMFS' jurisdiction. However, even though the court's order did not address the two more recently listed salmonids, NMFS analyzed the impacts of EPA's action to them because they belong to the same taxon. NMFS analysis requires consideration of the same information. Consultation with NMFS will be completed for registration of each a.i.when EPA makes effect determinations on all remaining species and consults with NMFS as necessary.

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

This Opinion is based on NMFS' review of the package of information the EPA submitted with its 2003 and 2004 requests for formal consultation on the proposed authorizations of the above a.i.s. It also includes our review of recovery plans for listed Pacific salmonids, past and current research and population dynamics modeling efforts, monitoring reports from prior research, Opinions on similar research, published and unpublished scientific information on the biology and ecology of threatened and endangered salmonids in the action area, and other sources of information gathered and evaluated during the consultation on the proposed authorizations of the a.i.s 2,4-D, triclopyr BEE, diuron, linuron, captan and chlorothalonil. 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 and by the registrants identified as applicants by EPA. Finally, NMFs considered comments on the draft RPAs that were provided by EPA, applicants, state agencies, stakeholders, and members of the public.

Background

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

On July 2, 2002, the court ruled that EPA had violated ESA section 7(a)(2) and ordered EPA to initiate interagency consultation and make determinations regarding effects to the salmonids on all 54 a.i.s by December 2004. *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.

The current consultation evaluates 4 herbicides: 2,4-D, triclopyr BEE, diuron and linuron; and 2 fungicides: captan and chlorothalonil. EPA consultations on pesticide products currently focus on their effects to listed Pacific salmonids. EPA consultations remain incomplete until all protected species under NMFS' jurisdiction are covered.

Consultation History

Between August 1, 2003, and December 1, 2004, the EPA transmitted letters to NMFS' Office of Protected Resources (OPR) requesting section 7(a)(2) consultation for the registration of the six a.i's: 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil and detailing their effects determinations on 26 ESUs of Pacific salmonids listed at that time (Puget Sound steelhead and Lower Columbia River coho were not evaluated). In the BE's, and summarized in Table 1, EPA's Office of Pesticide Programs (OPP) determined that the use of 2,4-D may adversely affect 26 ESUs. EPA determined that the continued use of triclopyr BEE may adversely affect 16 ESUs, and is not likely to adversely affect 10 ESUs. For Diuron, EPA determined its continued use may adversely affect 25 ESUs and is not likely to adversely affect Lake Ozette sockeye salmon. EPA determined the use of linuron will have no effect on 19 ESUs, and is not likely to adversely affect 7 ESUs. Considering the fungicides captan and chlorothalonil, EPA determined the continued use of captan will not affect 13 ESUs, is not likely to adversely affect 11 ESUs, but may adversely two ESUs. EPA determined that the continued use of chlorothalonil will have no affect on six ESUs, is not likely to adversely affect 11 ESUs, and may adversely affect nine ESUs

On June 28, 2005, NMFS listed the Lower Columbia River coho salmon ESU as threatened. Given this recent listing, EPA's 2003 and 2004 effects determinations for 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil on listed Pacific salmonids lack an effects determination for the Lower Columbia River coho salmon.

On May 22, 2007, NMFS listed the Puget Sound Steelhead Distinct Population Segment (DPS) as threatened. Given this recent listing, EPA's 2003 and 2004 effects determinations for 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil on listed Pacific salmonids lack an effects determination for the Puget Sound steelhead.

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

On February 11, 2008, NMFS listed the Oregon Coast coho salmon evolutionarily significant unit (ESU) as threatened. This ESU was considered in EPA's Biological Assessments for the six a.i.s.

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

On August 29, 2008, NMFS met with EPA and the applicants for chlorpyrifos, diazinon, and malathion. At that meeting, NMFS asked EPA to identify applicants for this and subsequent pesticide consultations.

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

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

On September 26, 2008, NMFS sent correspondence to EPA regarding the roles of the federal action agency and identified applicants by such agency during formal consultation. NMFS also requested incident reports and label information for subsequent pesticide consultations from EPA. The specified timeline for NMFS' receipt of incident reports and label information for the six a.i.s considered in this Opinion was July 2010.

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

On June 1, 2010, NMFS sent an email to EPA confirming that all current labels for end use products, or if available, a master label that includes all use instructions for all products including 2,4-D, triclopyr BEE, diuron, linuron, captan, or chlorothalonil should be submitted to NMFS in July 2010.

On July 13, 2010, EPA sent a letter to registrants to confirm applicant status and participation in the consultation process for the Opinion covering the herbicides 2,4-D, triclopyr BEE, diuron and linuron; and the fungicides captan and chlorothalonil.

On July 20, 2010, Syngenta (representing GB Biosciences) via email responded to the July 16, 2010 letter from EPA confirming GB Biosciences were certified registrants of chlorothalonil (EPA Reg. No. 50534-7 for technical material), and confirming GB Biosciences (Syngenta) wanted to be considered an applicant and thus participate in the consultation process.

On July 23, 2010, NMFS received grower-provided use information data from the Washington State Department of Agriculture (supplied by the USDA's National Agricultural Statistics Service (NASS) on the known use of 2,4-D, diuron, linuron, captan, and chlorothalonil in Washington State during the 2009 growing season for a few commodities.

On July 27, 2010, Syngenta forwarded the July 20, 2010 email noted above to NMFS to verify with NMFS that Syngenta were involved as an applicant for the consultation.

On July 30, 2010, NMFS received notification from Tessenderlo Kerley, Inc. (in response to the July 13, 2010 letter sent by EPA, noted above) that they were registrants for linuron and desired applicant status for the consultation with EPA for this active ingredient.

On August 2, 2010, NMFS received notification from Dow AgroSciences LLC (in response to the July 13, 2010 letter sent by EPA, noted above) that they were registrants for triclopyr BEE and desired applicant status for the consultation with EPA for this active ingredient.

On August 2, 2010, NMFS received information from Syngenta regarding the fungicide chlorothalonil.

On August 3, 2010, NMFS received notifications from Dow AgroSciences LLC, PBI-Gordon Corporation, Atanor S.A., and AgroGor Corporation (in response to the July 13, 2010 letter sent by EPA, noted above) that they were registrants for 2,4-D and desired applicant status for the consultation with EPA for this active ingredient.

On August 3, 2010, NMFS received notifications from Arysta Lifescience North America and Mahkteshim Chemical LTD (in response to the July 13, 2010 letter sent by EPA, noted above) that they were registrants for captan and desired applicant status for the consultation with EPA for this active ingredient.

On August 4, 2010, in a letter dated August 3, 2010, NMFS received notification from Albaugh, Inc (in response to the July 13, 2010 letter sent by EPA, noted above) that they were registrants for 2,4-D and desired applicant status for the consultation with EPA for this active ingredient.

On August 4, 2010, NMFS received notification from NuFarm (in response to the July 13, 2010 letter sent by EPA, noted above) that they were registrants for both triclopyr

BEE and 2,4-D, and desired applicant status for the consultation with EPA for these active ingredients.

On August 9, 2010, via email, NMFS asked EPA to clarify information related to the Red-legged frog (RLF) biological assessment for 2,4-D. In that BA, EPA used a master label approach. NMFS wanted confirmation this approach is accurate and reflects what is in the individual labels. If this was the case, NMFS could get started more quickly on the analysis for the 4th Opinion. NMFS also asked EPA to confirm if there are or are not registrants whose labels do not conform to the master label used in the RLF BA. EPA responded on the same day via email with an attached file of the most recent master label for 2,4-D (dated June 20, 2005). EPA indicated the products will reflect the master label by September 30, 2010. In addition EPA indicated they would send ten special 24(c) labels on a CD sent in the mail. The master label and the 24(c) labels package of product labels would be complete for 2,4-D.

On August 10, 2010, in a letter dated August 5, 2010, NMFS received notification from Albaugh, Inc (in response to the July 13, 2010 letter sent by EPA, noted above) that they were registrants for triclopyr BEE and desired applicant status for the consultation with EPA for this active ingredient.

On August 11, 2010, NMFS confirmed with EPA that Syngenta had requested a meeting as applicants for the purposes of the consultation process for this Opinion. NMFS expressed a desire to hold all applicant meetings early in September, 2010.

On August 11, 2010, NMFS received via email the 24(c) label for 2, 4-D registered in California, Oregon and Washington. NMFS asked for the Section 3 labels associated with the 24(c) labels and additional information concerning labels CA040027 and OR-10016 (this label was shown as expired on 12/31/2009). NMFS wanted to know if it had or was going to be renewed (NMFS received clarification on this issue from EPA on September 16, 2010 via email).

On August 11, 2010, NMFS received notice from EPA via email that two compact discs were sent; and with the email transmittal, two special local needs labels for captan were attached. NMFS was still awaiting label information from EPA on the other active ingredients being considered for this Opinion.

On August 12, 2010, NMFS received notification from DuPont De Nemours and Co. (in response to the July 13, 2010 letter sent by EPA, noted above) that they were registrants for diuron and desired applicant status for the consultation with EPA for this active ingredient.

On August 18, 2010, via email EPA asked NMFS how they would like to proceed with the applicant meetings. For chemicals with more than one applicant NMFS was asked if we wanted to ask the applicants if they are willing to meet together with NFS as opposed to meeting individually with NMFS. On August 20, 2010, NMFS responded by email to EPA that NMFS would prefer to consolidate the meetings to the extent possible. EPA responded later this same day that they would try to proceed in the manner NMFS preferred.

On August 19, 2010, NMFS received via email from EPA most recent stamped (approved) linuron product labels and a list of the products giving the registration number, product and company name, percent active ingredient, and label stamp date. EPA informed NMFS with this email that they were still assembling the labels for diuron and chlorothalonil.

On August 19, 2010, via a separate email from EPA, NMFS was notified that EPA had received word from GB Biosciences (Syngenta) on when they can meet on chlorothalonil. EPA also informed NMFS they were waiting on Dupont's reply regarding a meeting to discuss diuron.

On August 20, 2010, NMFS received via email notice from the Chemical Review Manager for triclopyr in EPA's Office of Pesticide Program's Pesticide Re-evaluation

Division that they were attempting to set up applicant meetings between NMFS, OPP, and applicants Albaugh Inc. and Nufarm Americas Inc.

On August 25, 2010, EPA confirmed via email to NMFS the applicant meeting to discuss triclopyr was scheduled for September 23, 2010.

On August 25, 2010, NMFS received letters from NuFarm and Dow indicating that as applicants for the consultation, their initial submission of information and data was being transmitted under a joint effort with several applicants through the "Industry Task Force II on 2,4-D Research Data." A third letter was also received on this day from the 2,4-D Task Force submitting data for the consultation. In this letter they also asked for a joint meeting with EPA and NMFS.

On August 26, 2010, NMFS confirmed via email with EPA that the initial meetings with the applicants involved in the consultation were to introduce the applicants to the ESA Section 7(a)(2) consultation process, and to describe to them their role in the process. In this email, NMFS requested EPA that the applicants provide materials explaining unique application methods or uses for their chemicals if applicable. Also, NMFS informed EPA that several applicant letters were received regarding the meetings, including the 2,4-D Task Force (this letter was attached to the email to EPA), who proposed a joint meeting with the triclopyr applicants. NMFS confirmed that NMFS was amenable to this proposal.

On August 30, 2010, NMFS received two packages via Federal Express. NuFarm sent hard copies of two triclopyr BEE lables. NuFarm suggested these labels should b used in lieu of labels dating back to 2004. The second package was from AGRO-GOR and PBI/GORDON. Each sent identical letters dated August 25, 2010 (in same FedEx envelope) that their initial submissions of information are being transmitted through the 2,4-D Task Force

On August 31, 2010, NMFS received notification via email that E.I. dupont de Nemours and Company ("DuPont") wanted to be included as applicants and agreed to submit information to EPA and to NMFS for consideration during consultation in the development of this Opinion. This email from DuPont was in reaction to EPA's letter dated July 13, 2010, asking if DuPont wished to participate in the consultation process. This email included 13 attachments.

On September 1, 2010, NMFS received a background report on chlorothalonil from Syngenta in advance of the September 22, 2010, applicant meeting. This report was forwarded to NMFS via email from EPA.

On September 3, 2010, NMFS received the diuron labels from EPA via UPS.

On September 7, 2010, NMFS received an email from EPA stating that EPA was still checking on the chlorothalonil labels and will get those to us in the following week.

On September 7, 2010, NMFS received additional background materials from Syngenta via email, on chlorothalonil, for review prior to the September 22, 2010 applicant meeting. Syngenta also sent labels to NMFS to review for the consultation process for this Opinion.

September 16, 2010, NMFS received requested information on 2,4-D labels via email from EPA per the August 11, 2010, request noted above.

On September 17, 2010, via email NMFS sent a two-page request to EPA to clarify linuron labels and uses. NMFS was later copied on an EPA email passing the questions on internally for response.

On September 17, 2010, NMFS received from EPA via email background information for linuron from Tessenderlo Kerley, Inc (TKI), an applicant.

On September 21, 2010, NMFS, EPA and applicant TKI met and shared information for this consultation.   At this meeting NMFS explained the consultation process and the role of applicants in this process. TKI provided materials on linuron to EPA and NMFS for consideration in the consultation, and the development of the Opinion.

On September 22, 2010, NMFS, EPA and applicants BG Biosciences/Syngenta and Drexel met to discuss the fungicides captan and chlorothalonil. At this meeting NMFS explained the consultation process and the role of applicants in this process. The applicants provided materials on captan and chlorothalonil to EPA and NMFS for consideration in the consultation, and the development of the Opinion.

On September 22, 2010, NMFS received via email from EPA, two files on captan referenced in the applicant meeting earlier in the day by the Captan Task Force.

On September 23, 2010, NMFS met with EPA and applicants: 2,4-D Task Force, DOW, Nufarm, Albaugh Inc. to discuss the herbicides 2,4-D and triclopyr BEE. Earlier in the day, NMFS received via email an advanced copy of DOW's presentations on 2,4-D and triclopyr BEE. At this meeting NMFS explained the consultation process and the role of applicants in this process. The applicants provided materials on 2,4-D and triclopyr BEE to EPA and NMFS for consideration in the consultation, and the development of the Opinion.

On September 24, 2010, NMFS provided EPA via email the presentation NMFS gave at each of the applicant meetings held earlier in the same week.

On September 29, 2010, NMFS, EPA and DuPont (applicant for the herbicide diruon) met and shared information for this consultation. Earlier in the day, NMFS received via email from EPA advanced copies of DuPont's power-point presentations on diuron. At this meeting NMFS explained the consultation process and the role of applicants in this process. The applicant provided materials on diuron to EPA and NMFS for consideration in the consultation, and the development of the Opinion.

On September 29, 2010, NMFS received via email from EPA electronic versions of hand-outs provided at the September 23rd meeting for those who could only attend via phone.

On September 29, 2010, NMFS received contact information via email from EPA for the PMRA drift model and how EPA's Office of Pesticide Program's uses it.

On October 1, 2010, via email NMFS requested additional information from EPA on 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil. In particular, NMFS highlighted the need to consider the potential impact of all stressors associated with the federal action to listed species and their designated critical habitat. NMFS asked for additional information on "inert" and "other" ingredients approved for use in end-use pesticide products known to be toxic to aquatic organisms. NMFS explained to EPA that inert and other ingredients are considered as potential stressors and are part of the action that NMFS must evaluate. To date NMFS had not received complete composition information (list of all ingredients and percentage of formulation) for end-use products EPA is proposing to authorize under FIFRA. NMFS reminded EPA that several of the staff involved in the consultation are annually recertified to receive classified business information (CBI). NMFS therefore requested EPA to provide NMFS with complete composition information for all of the end-use products which contain 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil to adequately complete the consultation. NMFS requested this information be received by October 31, 2010.

On October 8, 2010, NMFS received via email from EPA two documents on triclopyr BEE that were referenced during the applicant meeting on September 23, 2010.

On October 11, 2010, NMFS received via email from a representative of the Captan Task Force, a power point presentation given at the applicant meeting held with EPA and NMFS on September 22, 2010.

On October 13, 2010, the Department of Justice (DOJ), representing NMFS, submitted a request to plaintiffs for a 90-day extension to complete the Opinion. The deadline extension would allow for NMFS to complete the final biological opinion by April 30, 2011, instead of January 31, 2011 as required by the settlement agreement. DOJ also inquired whether a single Opinion could cover all of the pesticides instead of completing three separate Opinions.

On October 13, 2010, NMFS and EPA received from Dow AgroSciences LLC, three files of data collected from California Pesticide Use Reporting Information on 2,4-D and triclopyr BEE.

On October 13, 2010, NMFS received via email a power point presentation and notes from Syngenta from the September 22, 2010, applicant meeting noted above.

On October 18, 2010, NMFS requested additional information from EPA, via email, where a few more 24C labels were missing the corresponding section 3 label.

On October 20, 2010, NMFS received information on missing section 3 labels requested on October 18, 2010.

On October 25, 2010, plaintiffs respond to DOJ agreeing to the October 13, 2010 request for a 90-day extension, and to NMFS' covering all six pesticides in one Opinion.

On October 26, 2010, NMFS notified EPA via email that the plaintiffs agreed to a 90-day extension for the Opinion and agreed to a flexible approach to batching the chemicals into one Opinion. NMFS noted that the 90-day extension had not yet been approved by the Court.

On October 27, 2010, NMFS received via email information from Washington Department of Ecology on the use of 2,4-D in Washington State for aquatic weed control.

On October 29, 2010, the U. S. District Court approved the 90-day extension to complete the Opinion, and allowed flexibility in the number of Opinions NMFS issued to complete for the batch of six chemicals under consultation.

On November 1, 2010, NMFS requested via email additional information from EPA on one of the chlorothalonil labels that appeared to have conflicting information between mixture ratios and use ratios. This needed to be cleared up in order to determine maximum seasonal use rates associated with the products in question.

On November 1, 2010, NMFS received additional information via email from the Washington State Department of Ecology on 2,4-D aquatic applications for Washington.

On November 5, 2010, NMFS received a response and clarifying information via email from EPA on questions raised in NMFS' request for additional information on November 1, 2010, noted above.

On November 11, 2010, NMFS received various state restrictions on pesticide use compiled by DuPont for the diuron consultation.

On November 16, 2010, NMFS requested via email additional information from EPA on two captan labels (CA-020017 and WA-940026).

On November 18, 2010, NMFS received a response from EPA via email regarding additional information on one of the two captan labels. EPA stated that information on the other label (WA-940026) would be sent soon.

On November 18, 2010, NMFS requested an additional 24(c) label that was not provided by EPA (EPA Reg. No. 51036-166).

On November 19, 2010, NMFS received a response to the November 18 request noted above from EPA. NMFS was informed the CA SLN references an old registration number. EPA Reg. No. 51036-166 was transferred to Arysta in 2006 -- thus the registration number for the product changed to EPA Reg. No. 6630-234. EPA provided this label information as an attached file. EPA informed NMFS that they were checking to see if the CA SLN is still active and would let NMFS know of its status within the next few days.

On November 29, 2010, NMFS received via email from DuPont a report sent previously to EPA for "86-5 compliance" regarding consultation on diuron.

On December 6, 2010, NMFS phoned EPA to get an update on when NMFS might expect responses to questions regarding linuron label statements and uses first requested on September 17, 2010.

On December 12, 2010, NMFS received via FedEx a CD from applicant Syngenta additional new data and information for the consultation on chlorothalonil.

On December 13, 2010, NMFS received via email from EPA responses to questions about linuron labels, along with additional labels not previously provided. This information is important in understanding the scope of the proposed action and in determining any possible effects of the action to listed salmon and steelhead and their designated critical habitat.

In response to the information provided by EPA on December 13, 2010, NMFS sought additional clarifying information on linuron in an email request sent on December 15, 2010.

On December 21, 2010, NMFS received an email from EPA asking if NMFS had received a CD containing additional information on chlorothalonil from Syngenta. NMFS responded that same day stating that the CD had arrived along with a transmittal letter dated December 15, 2010.

On December 21, 2010, NMFS sent an email to EPA requesting additional chemical fate information regarding captan.

On January 3, 2011, NMFS received two email responses from EPA, with numerous attachments, to the questions and information requests sent on Decenmber 21, 2010.

On January 4, 2011, NMFS received an email from EPA inquiring about the availability of a draft Opinion. EPA was interested in scheduling meetings with the applicants to discuss the draft.

On February 14, 2011, NMFS received via FedEx a letter and CD from Syngenta Crop Protection, LLC. Syngenta provided additional information to EPA and NMFS on non-crop uses of chlorothalonil. This information was received too late to consider in time for the release of the first draft Biological Opinion issued on March 1, 2011, but was considered in detail prior to the release of the second draft Biological Opinion issued on May 13, 2011.

On February 27, 2011, NMFS received via email additional information from Syngenta for NMFS and EPA to consider in our consultation. The information pertained to a drinking water assessment for the IR-4 registration of chlorothalonil and its specific degradation product for new uses on bulb vegetables, bushberries, and low growing berries.

On March 1, 2011, NMFS delivered via weblink a first draft of this Opinion with transmittal letter covering the 6 a.i.s, 2,4-D, tricloypr BEE, diuron, linuron, captan, and chlorothalonil.

On March 7, 2011, NMFS began reviewing comments posted on the EPA Docket in response to the March 1, 2011 draft Biological Opinion. In addition to input from the general public, several State agencies provided useful commentary on the RPAs. NMFS considered the comments and issues raised and made appropriate changes to the Opinion prior to the release of the second draft on May 13, 2011.

On March 9, 2011, NMFS held three separate meetings with individual applicants and EPA to discuss the first draft Opinion and to receive initial comments from the applicants. NMFS informed each of the applicants that they could provide written comments to NMFS by April 15, 2011.

On March 10, 2011, NMFS had two separate meetings with separate applicants and EPA to discuss the first draft Opinion and to receive initial comments from the applicants. NMFS informed each of the applicants they could provide written comments to NMFS by April 15, 2011.

On March 17, 2011, NMFS had a meeting with additional applicants and EPA to discuss the first draft Opinion and to receive initial comments from the applicants. NMFS informed the applicants they could provide written comments to NMFS by April 15, 2011.

On March 23, 2011, NMFS teleconferenced with EPA to discuss the draft RPAs and RPMs.

On March 29, 2011, Department of Justice filed a stipulation with the court requesting a 60 day extension, until June 30, 2011, for completion of the biological opinion, to allow for release of a second draft opinion and more time for comment. Plaintiffs had agreed to the extension, which the court approved on April 1, 2011.

On April 4, 2011, NMFS received via email comments from Dow (triclopyr BEE) in response to the first draft Biological Opinion released March 1, 2011. NMFS considered the comments and issues raised and made appropriate changes to the Opinion prior to the release of the second draft on May 13, 2011.

On April 5, 2011, NMFS received via email comments from Syngenta (chlorothalonil) in response to the first draft Biological Opinion released March 1, 2011. NMFS considered the comments and issues raised and made appropriate changes to the Opinion prior to the release of the second draft on May 13, 2011.

On April 11, 2011, NMFS received via separate emails comments from Drexel (diuron), and MANA (diuron) in response to the first draft Biological Opinion released March 1, 2011. NMFS considered the comments and issues raised and made appropriate changes to the Opinion prior to the release of the second draft on May 13, 2011.

On April 12, 2011, NMFS received via email comments from DuPont (diuron) in response to the first draft Biological Opinion released on March 1, 2011. NMFS considered the comments and issues raised and made appropriate changes to the Opinion prior to the release of the second draft on May 13, 2011.

On April 14, 2011, NMFS received comments via FedEx from the 2,4-D Task Force in response to the first draft Biological Opinion released on March 1, 2011. NMFS considered the comments and issues raised and made appropriate changes to the Opinion prior to the release of the second draft on May 13, 2011.

On April 15, 2011, NMFS received comments from applicant TKI (linuron) in response to the first draft Biological Opinion issued on March 1, 2011. NMFS considered the comments and issues raised and made appropriate changes prior to the release of the second draft on May 13, 2011.

On April 19, 2011 NMFS received written comments from EPA on the March 1, 2011 draft Opinion. NMFS considered the comments and issues raised and made appropriate changes to the Opinion prior to the release of the second draft on May 13, 2011.

On April 26, 2011 NMFS received a copy of the updated Drinking Water Assessment for chlorothalonil that Syngenta had referenced during the March 10 meeting (EPA, 2010).

The assement used revised values for environmental fate properties, including an aquatic half-life much lower than the one used in the salmonids BE. The updated fate information was a significant factor in revising the chlorothalonil determinations in the May 13 draft.

On May 11, 2011, EPA Office of Pesticide Programs announced the pending release of the second draft of this Opinion seeking comments by June 3, 2011 on the revised Reasonable and Prudent Measures and Reasonable and Prudent Alternatives.

On May 13, 2011, NMFS delievered via weblink a second draft Opinion to EPA covering the 6 a.i.s, 2,4-D, tricloypr BEE, diuron, linuron, captan, and chlorothalonil. This draft included revisions to the jeopardy and adverse modification determinations. The changes in the chlorothalonil determinations were based on further analysis of turf use data NMFS received on February 14, as well as the revised environmental fate parameters from the Drinking Water Assessment received on April 26, 2010. Revisions to the 2,4-D adverse modification determinations were based on additional analysis of use patterns of aquatic applications and uses related to restoration activities. With this transmittal, NMFS asked EPA and applicants to provide comments on the second draft by June 13, 2011. NMFS also offered to meet with EPA and applicants to discuss the second draft.

On May 14, 2011, NMFS began reviewing the comments submitted to the EPA docket in response to the second draft Biological Opinion. NMFS considered the comments and issues raised and made appropriate changes to the Opinion prior to the release of the final draft on June 30, 2011.

On May 26, 2011 NMFS received via email applicant input from Dow AgroSciences for triclopyr BEE in response to the May 13, 2011 draft Biological Opinion. NMFS considered the comments and issues raised and made appropriate changes to the Opinion prior to the release of the final draft on June 30, 2011.

On May 31, 2011, NMFS met with applicants and EPA to discuss the changes in the second draft and to receive preliminary comments on the second draft.

On June 3, 2011, NMFS received separate emails from applicants: Syngenta (chlorothalonil), and DuPont (diuron) commenting on the May 13, 2011 draft Biological Opinion. NMFS considered the comments and issues raised and made appropriate changes to the Opinion prior to the release of the final draft on June 30, 2011.

On June 7, 2011, NMFS received via email applicant input from the Captan Task Force in response to the May 13, 2011 draft Biological Opinion. NMFS considered the comments and issues raised and made appropriate changes to the Opinion prior to the release of the final draft on June 30, 2011.

On June 9, 2011, NMFS met with additional applicants and EPA to discuss the changes in the second draft and to receive preliminary comments on the second draft.

On June 13, 2011, NMFS received emails from applicants: 2,4-D Task Force, DuPont (diuron), and Syngenta (chlorothalonil), commenting on the May 13, 2011 draft Biological Opinion. NMFS considered the comments and issues raised and made appropriate changes to the Opinion prior to the release of the final draft on June 30, 2011.

On June 14, 2011, NMFS received comments from EPA on the May 13, 2011 draft Biological Opinion via email. NMFS has considered EPA's comments and issues raised prior to completing the final draft.

On June 30, 2011, NMFS issued the final draft Biological Opinion covering EPA's proposed re-registration of 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil.

Species Addressed in the BEs

EPA's BEs considered the effects of pesticides containing the six a.i.s to 26 species of listed Pacific salmonids and their designated critical habitat (EPA, 2003a, 2003b, 2003c, 2004a, 2004b, 2004f). Two listed species, the Lower Columbia River coho and the Puget Sound steelhead, were not considered in the BEs. For each a.i. considered in this opinion, EPA determined that its registration would affect at least some ESUs or DPSs. (Table 1). With the exception of linuron, EPA determined that registration of each a.i. may adversely affect at least one ESU or DPS. With the exception of 2, 4-D, EPA determined that the registration of each a.i. may affect but was not likely to adversely affect (NLAA) at least one ESU or DPS. Based on the analysis in this opinion, NMFS does not concur with any of the NLAA determinations made by EPA for these six registrations. When an action agency concludes its action will not affect any listed species or critical habitat, then no section 7 consultation is necessary (USFWS, & NMFS 1998). However, when an action may adversely affect listed species or designated critical habitat, NMFS conducts a formal consultation to determine whether that action is likely to jeopardize listed species or destroy or adversely modify critical habitat and issues a biological opinion with those determinations. NMFS conducted formal consultation and issues this biological opinion because EPA concluded for five of the a.i.s that registration may adversely affect some or all listed Pacific anadromous salmonids and their designated critical habitat. NMFS did not concur with any of EPA's "NLAA" determinationsfor linuron and has determined that linuron may adversely affect some ESUs. Once NMFS enters into formal consultation it considers all species and critical habitat affected. In this Opinion, NMFS will analyze the impacts to all ESUs/DPSs of Pacific salmonids present in the action area, including those salmonid species identified by EPA as being unaffected and including the two species of salmonid listed after EPA provided its BEs to NMFS.

Table 1. EPA's effects determinations.

Species	ESU	Herbicides				Fungicides	
		2,4-D	Triclopyr BEE	Diuron	Linuron	Captan	Chlorothalonil
Chinook	Puget Sound	may affect	may affect	may affect	no effect	NLAA	may affect
	Lower Columbia River	may affect	may affect	may affect	no effect	NLAA	NLAA
	Upper Columbia River Spring - Run	may affect	may affect	may affect	NLAA	may affect	may affect
	Snake River Fall - Run	may affect	may affect	may affect	NLAA	NLAA	may affect
	Snake River Spring/Summer - Run	may affect	may affect	may affect	NLAA	NLAA	may affect
	Upper Willamette River	may affect	may affect	may affect	no effect	NLAA	may affect
	California Coastal	may affect	NLAA	may affect	no effect	no effect	NLAA
	Central Valley Spring - Run	may affect	NLAA	may affect	no effect	no effect	NLAA
	Sacramento River Winter - Run	may affect	NLAA	may affect	no effect	no effect	NLAA
Chum	Hood Canal Summer - Run	may affect	may affect	may affect	no effect	no effect	no effect
	Columbia River	may affect	may affect	may affect	no effect	no effect	no effect
Coho	Lower Columbia River	*not evaluated*	*not evaluated*	*not evaluated*	*not evaluated*	*not evaluated*	*not evaluated*
	Oregon Coast	may affect	may affect	may affect	no effect	no effect	no effect
	Southern Oregon and Northern California Coast	may affect	NLAA	may affect	no effect	no effect	NLAA
	Central California Coast	may affect	NLAA	may affect	no effect	no effect	NLAA
Sockeye	Ozette Lake	may affect	may affect	NLAA	no effect	no effect	no effect
	Snake River	may affect	may affect	no effect	no effect	may affect	no effect
Steelhead	Puget Sound	*not evaluated*	*not evaluated*	*not evaluated*	*not evaluated*	*not evaluated*	*not evaluated*
	Lower Columbia River	may affect	may affect	may affect	no effect	no effect	NLAA
	Upper Willamette River	may affect	may affect	may affect	no effect	NLAA	may affect
	Middle Columbia River	may affect	may affect	may affect	NLAA	NLAA	may affect
	Upper Columbia River	may affect	may affect	may affect	NLAA	NLAA	may affect
	Snake River	may affect	may affect	may affect	no effect	NLAA	may affect
	Northern California	may affect	NLAA	may affect	no effect	no effect	no effect
	Central California Coast	may affect	NLAA	may affect	no effect	no effect	NLAA
	California Central Valley	may affect	NLAA	may affect	no effect	no effect	NLAA
	South-Central California Coast	may affect	NLAA	may affect	NLAA	NLAA	NLAA
	Southern California	may affect	NLAA	may affect	NLAA	NLAA	NLAA

Description of the Proposed Action

The Federal Action

The proposed action encompasses EPA's six registrations of the uses (as described by product labels) of all pesticides containing 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil. Although NMFS uses the term "action" in this document to refer to EPA's actions collectively, NMFS has analyzed the impacts of the registration of each active ingredient independently. The purpose of the proposed action is to provide tools for pest control that do not cause unreasonable adverse effects to the environment throughout the U.S. and its affiliated territories. Pursuant to FIFRA, before a pesticide product may be sold or distributed in the U.S. it must be exempted or registered with a label identifying approved uses by EPA's OPP. Once registered, a pesticide may not legally be used unless the use is consistent with directions on its approved label (http:www.epa.gov/pesticides/regulating/registering/index.htm). EPA authorization of pesticide uses are categorized as FIFRA sections 3 (new product registrations), 4 (reregistrations and special review), 18 (emergency use), or 24(c) Special Local Needs (SLN).

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

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

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

Pesticide Labels. For this consultation, EPA's proposed action encompasses all approved product labels containing the a.i.s 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil; their degradates, metabolites, and formulations, including other ingredients within the formulations; adjuvants; and tank mixtures. These activities comprise the stressors of the action (Figure 1). The six BEs indicate that the subject a.i.s are labeled for a variety of uses including applications to residential areas, industrial areas, pastures, forested areas, and crop lands (EPA, 2003a, 2003b, 2003c, 2004a, 2004b, 2004f).

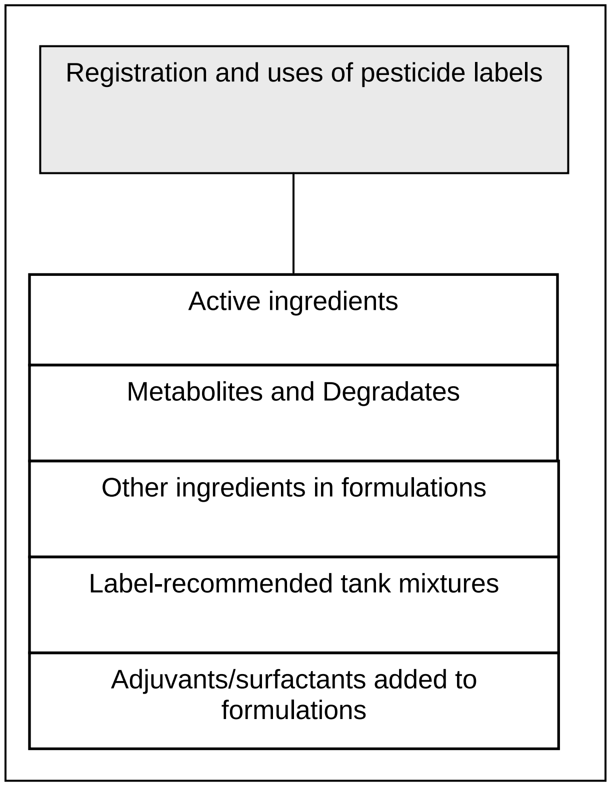


Figure 1. Stressors of the Action

Active and Other Ingredients. 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil are the a.i.s that kill or otherwise affect targeted organisms (listed on the label). However, pesticide products that contain these a.i.s also contain inert ingredients. Inert ingredients are ingredients which EPA defines as not "pesticidally" active. EPA also refers to inert ingredients as "other ingredients". The specific identification of the compounds that make up the inert fraction of a pesticide is not required on the label. However, this does not necessarily imply that inert ingredients are non-toxic, non-flammable, or otherwise non-reactive. EPA authorizes the use of chemical adjuvants to make pesticide products more efficacious. An adjuvant aides the operation or improves the effectiveness of a pesticide. Examples include wetting agents, spreaders, emulsifiers, dispersing agents, solvents, solubilizers, stickers, and surfactants. A surfactant is a substance that reduces surface tension of a system, allowing oil-based and water-based substances to mix more readily. A common group of non-ionic surfactants is the alkylphenol polyethoxylates (APEs), which may be used in pesticides or pesticide tank mixes, and also are used in many common household products. Nonylphenol (NP), one of the APEs, has been linked to endocrine-disrupting effects in aquatic animals.

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

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

Pesticide Registration. The Pesticide Registration Improvement Act (PRIA) of 2003 became effective on March 23, 2004. The PRIA directed EPA to complete REDs for pesticides with food uses/tolerances by August 3, 2006, and to complete REDs for all remaining non-food pesticides by October 3, 2008. The goal of the reregistration program is to mitigate risks associated with the use of older pesticides while preserving their benefits. Pesticides that meet today’s scientific and regulatory standards may be declared “eligible” for reregistration. The eligibility for continued registration may be contingent on label modifications to mitigate risk and can include phase-out and cancellation of uses and pesticide products. The terms of EPA’s regulatory decisions are summarized in RED documents (EPA, 1995, 1998b, 1999b, 2003d, 2004e, 2005).

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

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

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

Registration Information of Pesticide a.i.s under Consultation. As discussed above, the proposed action encompasses EPA's registration of the uses (as described by product labels) of all pesticides containing 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil. EPA provided copies of all active product labels for triclopyr BEE, diuron, linuron, captan, and chlorothalonil; and a master label summarizing all label restrictions for 2,4-D. The following descriptions represent information acquired from review of these labels as well as information conveyed in the EPA BEs, REDs, and other documents.

2,4-D

The herbicide 2,4-D has been used in the United States for more than 60 years (EPA, 2005). It is most commonly used as a post-emergent herbicide for broadleaf control and is available in several chemical forms that are each formulated in multiple end-use products (Table 2) (EPA, 2009a). The isopropyl ester form of 2,4-D is used in citrus crops as a growth regulator to reduce preharvest fruit drop and increase fruit size. 2,4-D is a synthetic auxin that disrupts normal plant growth by mimicking endogenous auxins that act as regulator hormones. Plant injuries include impacts to growth and reproduction. Symptoms may appear almost immediately in plants, but death may not occur for several weeks. Currently, Dow AgroSciences, Nufarm, Ltd., and the Agro-Gor Corporation have registrations with EPA for manufacturing use products containing 2,4-D. These products are formulated into a large number of end-use pesticides which are registered by dozens of companies and applied for a variety of uses (National Pesticide Information Retrieval System http://ppis.ceris.purdue.edu/htbin/cnamlist.com). In total, there are over 600 end-use products that are registered for use on over 300 distinct use sites (e.g. agricultural, residential, aquatic, etc., EPA RED DOCUMENT, 2005). Additionally, there are nine SLN registrations in California, Idaho, Oregon, and Washington (EPA Reg. No. CA040027, CA070017, CA970033, OR050016, OR-940036, WA010009, WA010038, WA070007, and WA9400032). There are no emergency use registrations (section 18) for 2,4-D in California, Idaho, Oregon, or Washington.

Table 2. Chemical forms of 2,4-D products that are currently registered for use in the action area.

EPA PC Code	CAS Number	Chemical Name
030001	94-75-7	2,4D acid
030004	2702-72-9	2,4D sodium salt
030016	5742-19-8	2,4D diethanolamine (DEA) salt
030019	2008-39-1	2,4D dimethylamine (DMA) salt
030025	5742-17-6	2,4D Isoproylamine (IPA) salt
030035	32341-80-3	2,4D triisopropanolamine (TIPA) salt
030053	1929-73-3	2,4D butoxyethyl ester (BEE)
030063	1928-43-4	2,4D 2 ethylhexyl ester (EHE)
030066	94-11-1	2,4D isopropyl ester (IPE)

Usage Information.

EPA estimated 36 million pounds of 2,4-D are applied annually in the United States for agricultural uses, with "heavy use" of 2,4-D in the Pacific Northwest and California (EPA, 2004a). Usage of 2,4-D in California has remained relatively stable, with over 400,000 lbs applied during each year from 1998 – 2008 (CDPR, 2009). Washington State Department of Agriculture estimates total annual usage of 2,4-D for nine of registered agricultural use sites to be 183,630 – 469,843 lbs, based on recent crop patterns and use surveys (WSDA, 2010a). Recent usage information for Oregon and Idaho is not available.

Agricultural Uses. Cereal grains, field and pop corn, sweet corn, sorghum, soybeans, sugarcane, rice, pome fruits (e.g. apples, pears), stone fruits (e.g. cherries, peaches, plums, apricots), nut orchards, pistachios, filberts, pastures, rangeland, fallow land and crop stubble, grass grown for seed or sod, irrigation ditch banks, potatoes, asparagus, hops, strawberries, blueberries, grapes, cranberries, citrus, clover , cottonwood and poplar trees grown for pulp, abandoned orchards, and forestry (site preparation, conifer release, roadsides, etc.).

Non-agricultural Uses. Grasslands not in agricultural production, ornamental turf, tree and brush control, non-cropland such as fencerows, hedgerows, roadsides, ditches, rights-of-way, utility power lines, railroads, airports, industrial sites, and other non-crop areas, and aquatic uses to control floating/emergent aquatic weeds and submerged aquatic weeds (e.g. ponds, lakes, reservoirs, marshes, bayous, ditches, canals, slow moving rivers and streams).

Registered Formulation Types. 2,4-D products are formulated as emulsifiable concentrates, wettable powders, granules, soluble concentrate solids, soluble concentrate liquids, and water dispersible granules (dry flowables). 2,4-D products frequently contain 2 – 4 a.i.s. Other registered herbicides ingredients in currently registered 2,4-D products include atrazine, aminopyralid, bromoxynil, carfentrazone-ethyl, clopyralid, dicamba, fluroxypyr, fenoxaprop-p-ethyl, glyphosate, MCPA, MCPP, MSMA, picloram, pyraflufenethyl, quinclorac, sulfentrazone, and triclopyr.

Methods and Rates of Application.

Methods. 2,4-D can be applied using a variety of methods and equipment. It may be applied as a spot treatment or broadcast application using aircraft (fixed wing or helicopter), ground boom sprayers, granule spreaders, hand held nozzle sprayers, wick applicators, and stump injectors.

Application Rates. Application rates are limited to 1 – 2 lbs of 2,4-D /A on the majority of agricultural use sites (Table 3). Sites with the greatest application rates include forestry and several non-crop use sites that allow a maximum single application of 4 lbs 2,4-D /A. Additionally, up to 10.8 lbs of 2,4-D per acre-foot (4 parts per million) can be applied to aquatic habitats for control of submergent weeds. Multiple applications are permitted on several use sites. Typically, either the maximum number of applications and/or maximum seasonal rate is specified. However, several of the SLN registrations do not specify limitations on either the number of applications or seasonal/annual use rates (CA-070017, CA-970033, OR-940036, and WA-0700070).

Table 3. Summary of all authorized use sites and application restrictions for active 2,4-D.

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
Cereal grains	1.3	2/crop cycle	1.75/crop cycle	NS	Aerial, ground spray	5481-145
Field and Pop Corn	1.5	NS	3.00/season	NS	Aerial, ground spray, granule spreader	5481-145
Sweet Corn	1.0 preplant or preemergence, 0.5 post emergence	1/crop cycle	1.50/season	21	Aerial, ground spray, granule spreader	5481-145
Sorghum	1.0 for amines, acids, and salts; 0.5 for esters	1/crop cycle	1.00 for amines, acids, and salts; 0.50 for esters /season	NA	Aerial, ground spray, granule spreader	34704-120
Soybeans	1.0	NS	1.00/ season	NS	Preplant aerial or ground spay	5481-145
Sugarcane	2.0 amines, acids, and salts	1/crop cycle	4.00/ season	NS	Aerial, ground spray	5481-145
Rice	1.0 preplant, 1.5 post emergence amines, acids, and salts	1/crop cycle	1.50/season	NS	Aerial, ground spray	228-260
Pome fruits (e.g. apples, pears)	2.0 amines, acids, and salts	2	4.00/season	75	Post emergence Ground spray	5481-145
Stone fruits (e.g. cherries, peaches, plums, apricots)	2.0 amines, acids, and salts	2	4.00/season	75	Post emergence Ground spray	5481-145
Nut orchards, Pistachios	2.0 amines, acids, and salts	2	4.00/season	30	Post emergence Ground spray	228-260
Filberts	NS[3] amines, acids, and salts	4	NS	30	Post emergence Ground spray	34704-120
Pastures, rangeland, grasslands not in agricultural production	2.0	NS	4.00/season	30	Post emergence Aerial, ground spray	5481-145
Ornamental and residential turf	1.5	2	3.00/season	NS	Ground spray	42750-19, 961-394
Grass grown for seed or sod	2.0	NS	4.00/season	NS	Ground spray	5481-145

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
Follow land and crop stubble	2.0	NS	4.00/season	30	Aerial, ground spray	5481-145
Forestry: roadsides, site prep., conifer release	4.0	NS	4.00 for broadcast, NS for other	NS	Aerial, ground spray, tree injection	5481-145
Tree and brush control	4.0	NS	4.00 for broadcast, NS for other	NS	Aerial, ground spray, tree injection	2217-703
Non-cropland such as fencerows, hedgerows, roadsides, ditches, rights-of-way, utility power lines, railroads, airports, industrial sites, and other non-crop areas	4.0	NS	4.00/season	30	Aerial, ground spray	5481-145
Irrigation ditch banks	2.0	2	4.00/season	30	Aerial spray, boat spray- Allow no more than 2 feet overspray onto water.	5481-145
Floating/emergent aquatic weeds	4.0	2	NS	21	NS	5481-145
Submerged aquatic weeds (e.g. ponds, lakes, reservoirs, marshes, bayous, ditches, canals, slow moving rivers and streams	10.8 lbs / Acre foot (4 parts per million)	2	NS	21	NS	5481-145
Potatoes	0.1	2	0.14/season	10-14	Post-emergent aerial or ground spray	228-139
Asparagus	2.0 amines, acids, and salts	2	4.00/season	30	NS	34704-120
Hops	0.5 amines, acids, and salts	3	1.50/season	30	Apply to ground between rows	34704-120
Strawberries (not in CA)	1.50 amines, acids,	1	1.50/season	NA	Aerial, ground spray	34704-120

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
	and salts					
Low bush blueberries	NS[4] amines, acids, and salts	1	NS	NA	Wipe and spot applications on weeds	Master label
High bush blueberries	1.4 amines, acids, and salts	1/crop cycle	2.80/season	NS	Directed ground spray	Master label
Grapes (CA only)	1.4 amines, acids, and salts	1/crop cycle	NS	NA	Directed ground spray	228-260
Cranberries	4.0 granular ester 1.2 amines, acids, and salts	1 dormant season, 2 growing season	4.00 lbs in dormant season, 2.40 lbs in growing season	NS	Ground: granule spreader, spot spray, wipe	228-61
Citrus	0.1 Isopropyl ester to increase fruit size, NS[5] to prevent pre-harvest drop	1/crop stage	NS	NS	Aerial, ground spray	5481-145
24 (c) CA: Mandarin growth regulator	NS[2]	1	NS	NA	Air blast, other ground spray	5481-145; CA040027
24 (c) CA: Ladino Clover (seed crop)	1.0	NS	NS	NS	Aerial, ground spray	34704-120; CA-070017
24 (c) CA: Citrus floor	1.6	NS, as needed	NS	NS	Directed ground spray- apply to the point of runoff	228-260; CA-970033
24 (c) OR: Blueberries	1.4	2	2.8	NS- Once in spring and once in fall	Ground spray	34704-803; OR-050016
24 (c) OR: Cottonwood and poplar trees for pulp	1.4	NS	NS	NS	Ground spray or wick application	228-145; OR-940036
24 (c) WA: Blueberries	1.4	2	2.8	NS- Once in spring and once after harvest	Ground spray	34704-803; WA-010009
24 (c) WA: Abandoned orchards	2.0 lb. triclopyr BEE /A + 4 lb 2,4-D	1	2 lb. triclopyr BEE /A + 4 lb 2,4-D	NA	Bark spray or hack and squirt trees	62719-260; WA-010038
24 (c) WA: Eurasian milfoil	As needed to achieve 1 – 4 mg/L	NS	NS	21 days for “broadcast” applications;	Subsurface drip application in	62719-3; WA-070007

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
	maintained for 48 hrs			NS for "spot treatments"	slowly moving surface waters	

1. NS = not specified
2. No upper limit placed on maximum application rate. Used as growth regulator not herbicide. Minimum 500 gal per acre of spray material per ace = 0.10 lbs a.i./A
3. Use rate per acre NS. Maximum rate of 1.00 lb a.i./100 gallons of spray solution. Wet leaves and stems of suckers April through August.
4. Use rate per acre NS. Maximum rate of 1.0 lb a.i./10 gallons of spray solution.
5. Use rate per acre NS. Maximum rate of 200 ppm a.i for spray solution.

Metabolites and Degradates.

Several degradates of parent 2,4-D have been identified in environmental fate studies including 1,2,4-benzenetriol; 2,4-dichlorophenol (2,4-DCP); 2,4 dichloroanisol (2,4-DCA); 4-chlorophenol; 2-chlorophenol; 4-chlorophenoxyacetic acid; and chlorohydroquinone (EPA, 2009a).

Triclopyr butoxyethyl ester (triclopyr BEE)

Triclopyr is a systemic broad-spectrum herbicide that is in the pyridinyloxyacetic acid family. It acts as a plant growth regulator and is used to control broadleaf weeds and woody plants. Triclopyr acid is formulated as a manufacturing product, and is used to formulate triclopyr BEE and triclopyr triethylamine salt (TEA) pesticides (EPA, 2009c). The current consultation is for approved uses of the Triclopyr BEE, and does not evaluate triclopyr TEA. Triclopyr TEA was first registered in 1979 for use on non-crop areas and forestry. Triclopyr BEE was registered in 1980 for use on the same sites. Both chemical forms of triclopyr were registered for use on turf in 1984. Triclopyr BEE is registered for use on rangeland and permanent grass pastures. Triclopyr TEA is registered for use on rice and BEE is not. Currently, Albaugh, Inc., Dow AgroSciences, Makhteshim Agan of North America, Inc., and NuFarm Americas, Inc. have registrations with EPA for manufacturing use products containing triclopyr. There are 31 active labels for end use products containing triclopyr BEE registered by 10 companies. Additionally, there is one SLN registration for control of unwanted trees in abandoned orchards (WA-010038). There are no section 18 registrations for use of triclopyr BEE products in California, Idaho, Oregon, or Washington.

Usage Information.

EPA estimates approximately 200,000 lbs of triclopyr BEE are applied within the action area each year (EPA, 2004f). California use reports indicate more than 70,000 lbs of triclopyr are applied annually in the state. However, the reports do not distinguish between use of triclopyr BEE and triclopyr TEA products (CDPR, 2009). Recent usage information for Washington, Oregon and Idaho is not available.

Agricultural Uses. Uses include range and pasture treatments, ornamental turf (sod farms), non-crop agricultural areas such as abandoned orchards, around farm buildings, fence rows, roads, and non-irrigations ditch banks.

Non-agricultural Uses. Non-agricultural uses of triclopyr BEE include rights-of-way, forest management (site preparation and conifer release), and applications to golf course and residential turf, and industrial areas.

Registered Formulation Types. Triclopyr BEE enduse products are typically formulated as emulsifiable concentrates or ready to use liquids that are spray applied. There is also a granular fertilizer product that contains Triclopyr BEE (EPA Reg. No. 961-394). Triclopyr BEE is frequently formulated with other a.i.s. Several formulations contain 2,4-D. (*e.g.*, EPA Reg. No. 228-565, 961-394, and 34704-928). It is also formulated with fluroxypyr (EPA Reg. No. 62719-477). Two formulations include MCPA and dicamba (EPA Reg. No. 228-395, 228-317). One formulation partners triclopyr BEE with three other herbicides (sulfentrazone, 2-4,D, and dicamba; EPA Reg. No. 2217-920).

Methods and Rates of Application.

Methods. Triclopyr BEE is typically spray applied by ground application or aerial methods. A granular formulation is applied with ground spreader. Labels frequently authorize tank mixes with other herbicides (EPA Reg. No. 62719-527), liquid fertilizers (EPA Reg. No. 228-317), drift control agents (EPA Reg. No. 74779-8), and/or surfactants (EPA Reg. No. 66222-153).

Application Rates. Active labels allow a maximum single and seasonal application rate of up to 8 lbs triclopyr BEE/A to forests and several non-crop areas (Table 4). The number of applications allowed is 1, 8, or is not specified. Use sites without specifications for the number of applications limit the total amount of product that can be applied either annually or seasonally.

Table 4. Summary of all authorized use sites and application restrictions for active triclopyr BEE products.

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number(s)
Range/pasture	2.0	1	2	NA	Aerial or ground spray	228-521; 228-552
Forests	6.0	1	8	NA	Aerial or ground spray	228-517
Turf (ornamental, commercial, golf course, residential)	1.0 (2.0 for spot treatments)	8	8	4 weeks	Aerial or ground spray	17545-8; 961-394; 62719-566; 66222-153
Non-Crop areas (*e.g.* fence rows, non-irrigation ditch banks, rights-of way, around farm buildings, industrial areas)	8.0	NS	8	NS	Aerial or ground spray	66222-153
Seasonably dry wetlands, flood plains, deltas, marshes, swamps, bogs and transitional areas between uplands and lowlands	8.0	NS	8	NS	Aerial or ground spray	66222-153
24 (c) WA: Abandoned orchards	2.0 lb. Triclopyr BEE;	1	2 lb. Triclopyr BEE; 4 lb. 2,4-D	NA	Ground spray	62719-260 WA010038

	4.0 lb. 2,4-D					
NS = not specified						

Metabolites and Degradates.

Triclopyr BEE and triclopyr TEA both rapidly degrade to for triclopyr acid. In aquatic environments, photodegradation products of the acid/anion include 5-chloro-3,6-dihydroxy-2-pyridinoloxyacetic acid (TCP) and oxamic acid. In soils, TCP and 3,5,6-trichloro-2-methoxypyridine (TMP) are formed through biotic metabolism (EPA, 2009c).

Diuron

Diuron (N-(3,4-dichlorophenyl)-N,N-dimethylurea) is a systemic substituted phenylurea herbicide. Diuron acts by inhibiting the Hill reaction in photosynthesis which limits the production of high energy compounds such as ATP used for various metabolic processes. Diuron is primarily absorbed through plant roots. It is transported upward through the xylem, and exerts its action at the seedling stage when the newly emerged plant starts to photosynthesize. It is effective primarily on annual broadleaved weeds, annual grasses, or newly emerged perennial plants. Established perennial plants are less susceptible, which is the basis for its use in fruit and nut crops (EPA, 2003c). Twenty three companies currently hold active registrations for end-use or technical products that contain diuron (http://ppis.ceris.purdue.edu/htbin/cnamlist.com). There are currently 73 active labels for products containing diuron including 66 end use products and 7 technical/manufacturing use products. There are 11 SLN registrations in California, Idaho, Oregon, or Washington. There are no emergency use registrations for diuron in California, Idaho, Oregon, or Washington.

Usage Information.

EPA estimated approximately 8 million pounds of diuron are applied annually for domestic uses based on pesticide surveys for the years of 1990 through 1999 (EPA, 2003c). Slightly over half was used in non-agricultural areas; about 25% of diuron was used on railroads; other non-agricultural sites of high usage (5-9% of total diuron) were pipelines and industrial facilities, roads, and sanitation/utilities. Among agricultural uses, the highest amounts of diuron were used on oranges (15%), cotton (10%), seed crops (9%), grapefruit (3%), and alfalfa (3%) (EPA, 2003c). Recent use of diuron in California has declined over the preceding decade from approximately 1.5 million pounds applied in 1998 to 730, 000 lbs in 2008 (CDPR, 2009). In 2008 nonagricultural uses continued to account for the largest amount of diuron applied in California. More than 283,000 lbs of diuron were applied for maintenance of right of ways. The largest agricultural use sites in California included oranges (approximately 144,000 lbs) and alfalfa (approximately 121,000 lbs). Based on recent crop patterns and surveys of typical use, Washington State Department of Agriculture estimates the total annual usage of diuron on asparagus, blueberries, iris bulbs, and cane berries is approximately 7,000 – 10,000 lbs, (WSDA, 2010d). Usage estimates for other crops and the nonagricultural high usage sites is not available. Recent use information for Oregon and Idaho is not available.

Agricultural Uses. Diuron is used on a variety of fruit and nut crops, grains, cotton, corn, sorghum, mint, asparagus, sugarcane, seed crops, coffee, hay, cut flowers, and for fallow, idle cropland. It may be used in irrigation and drainage systems when water is not present.

Non-agricultural Uses. Diuron is used on impervious surfaces such as paved areas. It is also used on industrial and rights-of-way areas where total vegetation control is desired; often it is combined with other herbicides for total vegetation control. Such broad-spectrum weed control includes along fence lines, rights-of-way (pipelines, powerlines, railway lines, roads), footpaths, in timber yards and storage areas, around commercial, industrial and farm buildings, electrical substations, and petroleum storage tanks. It has some use as an algacide in ornamental ponds, fountains, and aquaria. Additionally diuron may be used for general weed control in non-crop and non-timber (e.g. rights of way, uncultivated agricultural areas, fence rows, and industrial sites, intermittently flooded areas such as marshes, swamps, and bogs after water has receded). It may be used as a mildewicide in paints used on buildings and structures.

Registered Formulation Types. Diuron is available in wettable powder, dry flowable, liquid suspension, and soluble concentrate formulations. Many of these products contain additional active ingredients. Most are herbicides, but chlorothalonil, a fungicide, is used in the paint preservative formulations. Herbicides formulated with diuron include paraquat, thiadiazuron, bromacil, imazapyr, monosodium methanearsonate, tebuthiuron, sodium chlorate, sodium metaborate, sulfometuron-methyl, and copper sulfate (EPA, 2003c).

Methods and Rates of Application.

Methods. Diuron is a systemic herbicide registered for pre- and post emergent control using ground and aerial equipment. Diuron is typically applied as a pre-emergent herbicide to the soil, and needs to be watered in to be effective. It may persist in the soil throughout much of the season, thus providing continuing control of weeds. It can also be effective as a post-emergent herbicide, especially if applied during high humidity and warm temperatures, and with a surfactant added to enhance penetration into the weeds. In formulations with other herbicides, the other active ingredient(s) typically provides knockdown of established weeds, while the diuron inhibits additional weeds from becoming established (EPA, 2003c). Diuron is often applied in combination with other herbicides such as bromacil, hexazinone, paraquat, thiadiazuron, imazapyr, monosodium, sodium chlorate, sodium metaborate, and copper sulfate (EPA, 2009b).

Application Rates. Active labels allow a maximum single application rate of 12 lbs diuron/A on uncultivated agricultural areas, industrial sites, and intermittently flooded areas such as marshes, swamps, and irrigation ditches when water is not present. The maximum annual application rate at these sites is 24 lbs diuron/A. The maximum application rate in crops is 4 lbs diuron/A for single applications and 8 lbs diuron/A annually (Table 5).

Table 5. Summary of all authorized use sites and application restrictions for active diuron products.

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
Alfalfa (established)	2.4	1	2.4	NA	Aerial or Ground Spray; Chemigation	352-678 352-692
Alfalfa (seed) alfalfa grown for seed	1.2	1	1.2	NA	Aerial or Ground Spray; Chemigation	352-666
Apple	3.2	2	3.2	NS	Ground Spray	352-692
Artichoke in California	3.2	1	3.2	NA	Ground Spray	352-692
Asparagus	3.2	2[2]	4.8	N	Ground Spray	352-692
Barley (Western OR, Western WA)	1.6	1	1.6	NA	Aerial or Ground Spray at planting	352-678
Birdsfoot Trefoil (Western OR)	1.6	1	1.6	NA	Ground Spray	352-692
Blueberry, Caneberry, Gooseberry (Western OR, Western WA)	2.4	2	3.2	one application in Fall and Spring	Ground Spray	352-678
Blackberry, Boysenberry, Dewberry, Loganberry, Raspberry (CA)	2.4	2	3.2	one application in Fall and Spring	Ground Spray	352-692
Citrus (CA)	3.2	2	6.4	60	Ground Spray	352-692
Corn	0.8	1	0.8	NA	Ground Spray	352-692
Cotton (Preplant CA)	1.6	1	2.2	NA	Aerial or Ground Spray	352-692
Cotton (Post emergence CA)	0.6	2	2.2	NS	Ground Spray	352-692
Filberts (except CA)	2.2	2	3.2	150	Ground Spray	352-692
Grape	3.2	2	6.4	90	Ground Spray	352-692
Grass Seed Crops (OR and WA)	2.4	1	2.4	NA	Aerial or Ground Spray	352-692
Oats (spring) (drill planted in OR, WA, ID)	1.2	1	1.2	NA	Ground Spray	352-692
Oats (winter) (drill planted in OR, WA, ID)	1.6	1	1.6	NA	Ground Spray	352-692
Olives (CA)	1.6	2	3.2	NS	Ground Spray	352-692
Papaya	4.0	1	4.0	NA	Ground Spray	352-692
Peas (Austrian field) (Western OR)	1.6	1	1.6	NA	Aerial or Ground Spray	352-692
Peach	2.2 (3.0 in CA)	1	2.2 (3 in CA)	NA	Ground Spray	352-692
Pear	3.2	2	3.2	NS	Ground Spray	352-692
Pecan	3.2	1	3.2	NA	Ground Spray	352-692
Peppermint/Spearmint	2.4	NS	NS	NS	Ground Spray	352-692

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
Red Clover (Western OR)	1.6	1	1.6	NA	Ground Spray	352-692
Sorghum (Southwestern States)	0.4	2	0.4	NS	Ground Spray	352-692
Tree Plantings (CA, OR, WA)	2.4	NS	NS	NS	Ground Spray	352-692
Walnut (English) (CA,OR, WA)	2.2 (3.0 in CA)	2	3.2 (3 in CA)	once in Fall, once in Spring	Ground Spray	352-692
Wheat (winter) (ID, OR, WA, East of the Cascade Range)	1.2	1	1.2	NA	Aerial or Ground Spray	352-692
Wheat (winter) (OR, WA, West of the Cascade Range)	1.6	1	1.6	NA	Aerial or Ground Spray	352-692
General Weed Control in non-crop and non-timber (e.g. rights of way, uncultivated agricultural areas, fence rows, industrial sites, intermittently flooded areas such as marshes, swamps, and bogs after water has receded)	12.0	2	24	90	Ground Spray or dry application of granules with ground equipment	228-654; 352-692
Irrigation and Drainage Ditches (when dry)	12.0	NS	NS	NS	Ground Spray or dry application of granules with ground equipment	352-692
24 (c) CA: citrus in Fresno and Tulare Counties	3.2	2	3.2	once in Fall, once in Spring	Microsprinkler irrigation	352-678; CA-050005
24 (c) CA: Lilly bulbs in Del Norte County	4.0	NS	4	NS	Ground spray	352-678; CA-870038
24 (c) OR: Triticale	1.2 east of Cascades; 1.6 west of Cascades	1	1.2 east of Cascades; 1.6 west of Cascades	NA	Ground spray	352-678; OR-010029
24 (c) OR: Triticale	1.2 east of Cascades; 1.6 west of Cascades	1	1.2 east of Cascades; 1.6 west of Cascades	NA	Ground spray	352-692; OR-010030
24 (c) OR: Triticale	1.2 east of Cascades; 1.6 west of Cascades	1	1.2 east of Cascades; 1.6 west of Cascades	NA	Ground spray	352-678; OR-070032

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
24 (c) OR: Field grown Easter Lilies in Curry County	4.0	2	6	once in Fall, once in Spring	Ground spray	352-678; OR-080020
24 (c) OR: Grasses grown for seed	2.4	1	2.4	NA	Aerial or Ground Spray	9779-329; OR-940025
24 (c) OR: Grasses grown for seed	2.4	1	2.4	NA	Aerial or Ground Spray	9779-318; OR-920023
24 (c) WA: Ryegrass grown for seed	1.6	1	1.6	NA	Aerial or Ground Spray	19713-36; WA-000034
24 (c) WA: Ryegrass grown for seed	1.6	1	1.6	NA	Aerial or Ground Spray	19713-274; WA-000033
1. NS = not specified 2. In Washington, apply a single application only						

Metabolites and Degradates.

Diuron degrades in the environment to four major (>10% of the applied parent) and four minor (<10% of the applied parent) metabolites and degradates. The major metabolites are: carbon dioxide (CO2), N-(3,4-dichlorophenyl)-N-methylurea (DCPMU), N'-(3-chlorophenyl)-N,N-dimethylurea (MCPDMU), and 1,1-dimethyl-3-phenylurea (PDMU). The minor metabolites include: 4-dichlorophenylurea (DCPU); 3,4-dichloroaniline (3,4-DCA), N-(3-chlorophenyl)-N-methylurea (CPMU), and 3,3',4,4'-tetrachlorobenzene (TCAB) (EPA, 2009b). Additionally, diuron products contains two impurities from the manufacturing process, TCAB and 3,3',4,4'-tetrachloroazoxybenzene (TCAOB), both ‘dioxin-like’ substances. TCAB levels between 0.15 and 28 ppm have been found in diuron samples tested. TCAOB is present at lower levels (EPA, 2009b).

Linuron

Linuron is a substituted urea herbicide registered for use on several agricultural crops and some non-agricultural sites. It was first registered in 1966 and is currently used for preplant, preemergence, postemergence, or post-transplant weed control. Linuron is a systemic herbicide that targets grasses and broadleaf weeds by inhibiting the photosystem II reaction center (EPA, 2008). Three companies currently hold active registrations for 11 products containing linuron including four technical products (EPA Reg. No. 19713-158, 19713-386, 61842-22 / 352-679, and 61842-24 / 352-726) and 7 end –use products (EPA Reg. No. 19713-97, 19713-251, 61842-20 / 352-660, 61842-21 / 352-677, 61842-23 / 352-686, 61842-24 / 352-726, 66330-218 / 51036-78). There is one SLN registration in California, and none in Idaho, Oregon, and Washington (EPA Reg. No. CA-020006). There are no emergency use registrations for linuron in California, Idaho, Oregon, or Washington.

Usage Information.

Typical use pattern suggest approximately 400,000 lbs of linuron are applied each year to agricultural use sites in the United States (http://water.usgs.gov/nawqa/pnsp/usage/maps/show_map.php?year=02&map=m1993). Application to sorghum, cotton, potatoes, and carrots account for about 80% of the domestic agricultural uses. In California linuron use has generally declined over the last decade from approximately 82,000 lbs in 1998 to 59,000 lbs in 2008 (CDPR, 2010). Washington State Department of Agriculture estimates total maximum usage of linuron on asparagus, carrots, and wheat at 3,367, 4,952, and 12,066 lbs, respectively (WSDA, 2010e). Use estimates for other crops, and non-crop areas were not available.

Agricultural Uses. Linuron use sites in the action area include asparagus, bulbs (CA), carrots, celery, corn, kenaf, marigolds grown for seed (CA), parsley grown for seed (OR, WA), parsnips, sorghum, soybeans, wheat, post-harvest crop stubble and fallow lands.

Non-agricultural Uses. Non-crop areas such as roadsides and fence rows (EPA Reg. No. 19717-97).

Registered Formulation Types. End use products containing linuron are formulated in wettable powders, flowable concentrates, water dispersible granules, and liquid suspensions. One linuron formulation also contains diuron (EPA Reg. No. 352-660). Otherwise, all active labels of linuron contain a single active ingredient.

Methods and Rates of Application.

Methods. Linuron is applied through chemigation, ground boom, or other ground application methods. Aerial applications are not permitted. Several labels provide recommendations for tank mixtures with surfactants and herbicides. For example, one label suggests possible tank mixtures with alachlor, atrazine, Prowl 3.3 EC, Lexone, gramoxone , glyphosate, metribuzin, and 2,4-D (e.g. EPA Reg. No. 19713-97). Some labels contain soil type restrictions to manage the risk of surface and ground water contamination (EPA Reg. No. 19713-97, 352-660, 352-677).

Application Rates. Active labels allow for a maximum single application rate of up to 4lbs of linuron/A and a maximum annual application rate of up to 6 lbs a.i./A. Most field crops allow 1-2 lbs a.i./A for a single application and ≤ 2 lbs a.i./A annually (Table 6).

Table 6. Summary of all authorized use sites and application restrictions for active linuron products.

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
Asparagus	4.0	NS[1]	4 / season Or 6 /year	NS	Ground Chemigation	19717-97 19713-251
Bulbs (CA only)	1.0	NS	NS	NS[2]	Ground Chemigation	19717-97
Celery	1.5 1.0 (in CA)	1	NS	do not apply within 67 days	Ground	19717 -97
Corn	1.5	NS[3]	NS	NS	Chemigation	19717 -97
Parsnips	1.5	1	1.5	NA	Ground	19717 -97
Sorghum	1.0	2[4]	NS	NS	Chemigation	19717 -97
Soybeans[5]	2.0	NS	NS	NS	Ground	19717 -97

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
Wheat, Drill planted winter (ID,OR,WA)	0.3 to 1.8[6]	2 (Spring and Fall)	0.5 – 3.5	At least 4 months	Ground Chemigation: Preemergent broadcast spray	19717 -97
Non-crop weed control: Roadsides, fence rows, , etc.	3.0	NS	NS	NS	Ground Chemigation	19717-97
Carrots	1.0	NS	2 / year	NS	Ground Chemigation	19710-251
Cotton (not in CA)	0.6[7]	Max 3	1.5 / season	21 day minimum	Ground	352-660
Kenef	1.0	1	1	na	Ground Chemigation	352-677
Post Harvest, Crop stubble, fallow ground, stale seedbed	2.0	1 (fallow season)	2	NA	Ground	352-677
24 (c) OR, WA: Parsley grown for seed	1.0	2 in first growing season and 2 in second growing season	NS	21 day minimum	NS	Supplemental label registration numbers not provided 352-686
24 (c) CA: Marigolds grown for seed	1.0	1	1	NA	Ground	CA-020006 (1812-320)

1. NS = not specified
2. Pre-emergence, only during growing season
3. Single application is specified, but not clear if that is a yearly or seasonal limit
4. One pre- and one post-emergent applications permitted
5. The soybean use directions include multiple types of applications and recommended mixtures. These are further broken down by soil texture and % organic material.
6. West of Cascades = 1.75, East of Cascades (with 10-20 in rainfall) = .25 (varies w/ rainfall)
7. This formulation includes an equal amount diuron as an additional active ingredient

Metabolites and Degradates.

In the soil, linuron degrades to 3,4-dichlorobenxenamine (DCA), n-(3,4-dichlorophenyl)-N-methylurea (DCPMU), N-(3,4-dichlorophynyl)-N” methoyurea (DML), AND (3,4-dichlorophenyl)urea (DCPU). In anaerobic aqueous environments, major degradates include desmethoxy linuron and desmethoxy monolinuron (EPA, 2004b).

Captan

Captan was first registered in 1951 to control fungal disease in fruit crops. It is currently registered as a non-systemic fungicide in orchards, vineyard, turf, ornamentals, and a large variety of food crops. The mode of action of captan is inhibition of normal cell division on a broad spectrum of microorganisms and fungi. Captan inhibits the process of oxidative phosphorylation in fish, invertebrates, and other nontarget aquatic and terrestrial organisms (EPA, 2007c). There are 43 active labels for end use products containing captan that are held by nine registrants. There are five SLN registrations in California, Idaho, Oregon, and Washington (EPA Reg. No.CA020017, CA100006, CA980023, OR030029, OR070024).

Usage Information

EPA estimates more than 5 million lbs of captan are applied annually for domestic uses (EPA, 2004e). Recent data from California indicate agriculture use of captan has declined from over 1.5 million lbs to approximately 350,000 lbs during the last decade (CDPR, 2008). Washington State Department of Agriculture estimates total annual usage of captan in apples, blueberries, cane berries, and strawberries at 10,903 – 104,754 lbs, based on recent crop patterns and surveys of typical use (WSDA, 2010b). Recent usage for other crops and recent information for Oregon and Idaho is not available.

Agricultural Uses. Grasses/turf (seed crops, ornamentals, sod farms, grapes, honeydew, kale, lentils, lespedeza, lettuce, milo, mustard seed, nectarines, oats, onions, okra, peaches, prunes, peanuts, peas, peppers, potatoes, roses, radish, raspberries, rye, rutabaga, strawberries, Swiss chard, soybeans, spinach, squash, safflower, sunflower, sesame, greenhouse, sorghum, sugar beets, tomatillo, tomatoes, turnips, wheat, lily bulbs.

Non-agricultural Uses. Active labels allow captan use on turf (golf course, lawn seed beds) and ornamentals.

Registered Formulation Types. Captan is formulated into more than forty end-use products including liquid, dust, and granular formulations. Ten formulations include mixtures of one or more other active ingredients. Captan is mixed with the fungicides PCNB (*e.g.*, EPA Reg. No. 264-949), thiophonate-methyl (*e.g.*, 264-998), trifloxystrobin (*e.g.*, 264-999), carboxin (*e.g.*, 400-555), metalaxyl (*e.g.*, 400-561), sulfur (*e.g.*, 4-355), and fenhexamid (*e.g.*, 66330-48), and the insecticides imidacloprid (*e.g.*, 400-568), malathion (*e.g.*, 4-59), and carbaryl (*e.g.*, 4-122).

Methods. Captan is a contact fungicide applied as a seed treatment, a root dip, an in-furrow application, and by various ground and aerial foliar applications. Several active labels suggest captan can or should be applied with other fungicides and/or insecticides (*e.g.*, EPA Reg. No. 4-459, 19713-235, 19713-268, 19713-362, 19713-385, 19713-405, 62575-6, 66222-1, 66222-24, 66222-66, 66330-209).

Application Rates. Active labels allow for a maximum single application rate of up to 4.5 lbs captan/A and an annual application rate of up to 35 lbs captan/ acre (Table 7). Many products are applied as seed treatments and consequently only applied once per year. However, up to 8 foliar applications/year are allowed in several crops.

Table 7. Summary of all authorized use sites and application restrictions for active captan products.

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
Alfalfa, Clover, Lespedeza, Trefoil	0.2	1	12	NS	Seed dip	19713-161
Almonds	4.5	1	20	7	Aerial or Boom Spray	66222-58 66222-66 66222-24
Apples	4	8	26.3	14	Aerial or Boom Spray	66330-27 66330-29 66330-54
Apricots	2.5	NS	12.5	NS	Aerial or Boom Spray	66330-209
Artichoke	NS	NS	NS	NS	NS	400-568

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
Azaleas	2.0	1	NS	NS	Root dip	19713-235 19713-258 19713-268 19713-362
Barley	0.04	1	12	NS	Seed dip	264-931 66330-238
Beans (dry and snap)	0.03	1	NS	NS	Seed dip	264-931 66330-238
Beans	0.1	1	NS	NS	Seed dip	264-931 66330-238 400-567
Beets	0.01	1	12	NS	Seed dip	264-928
Begonias	2.0	1	NS	NS	Root dip	19713-235 19713-258 19713-268 19713-362
Blackberries	2.0	8	10	14	Aerial or Boom Spray	19713-258 19713-268
Blueberries	2.5	NS	35	10	Aerial or Boom Spray	19713-268 19713-258
Blue Grass	0.2	1	NS	NS	Seed dip	19713-161
Brassica (Cole)	0.03	1	NS	NS	Seed dip	400-568
Cabbage	NS	1	NS	NS	Seed dip	19713-258 66330-27
Canola	.005	1	NS	NS	Seed dip	400-568 400-567
Cauliflower	NS	1	NS	NS	Seed dip	19713-258
Camellias	2	1	NS	NS	Root dip	19713-235 19713-258

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
						19713-268 19713-362
Cantaloupe, Cucumber	0.003	1	12	NS	Seed dip	264-931 66330-238
Carnations	1	NS	NS	10	Root dip	19713-235 19713-258 19713-268 19713-362
Carrots	NS	1	NS	NS	Seed dip	19713-258
Cherries	2	8	14	14	Aerial or Boom Spray	19713-258 19713-268
Chrysanthemums	2	1	NS	NS	Root dip	19713-235 19713-258 19713-268 19713-362
Cilantro	NS	1	NS	NS	Seed dip	400-568 400-567
Clover	N/A	1	NS	NS	Seed dip	19713-258
Cole Crops (Broccoli, Brussels Sprouts, Cabbage, Cauliflower)	0.05	1	12	NS	Seed dip	264-931 66330-238
Conifers	NS	NS	NS	NS	Seed dip	19713-258
Cotton - Acid Delinted	0.01	1	NS	NS	Seed dip	264-931
Cotton - Fuzzy	0.01	1	NS	NS	Seed dip	264-931
Cotton - Machine Delinted	0.01	1	NS	NS	Seed dip	264-931
Collards	N/A	1	NS	NS	Seed dip	19713-258
Corn - Field	0.02	1	NS	NS	Seed dip	264-931
Corn - Sweet	0.04	1	NS	NS	Seed dip	264-931

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
Cowpeas	0.06	1	NS	NS	Seed dip	66330-238
Crucifers (mustard, radish, rape, turnips)	0.007	1	12	NS	Seed dip	264-931 66330-238
Cucurbits	NS	1	NS	NS	Seed dip	400-568
Dewberries	2	8	10	14	Aerial or Boom Spray	19713-235 19713-258 19713-268
Dichondra	0.01	1	NS	NS	Root dip	66222-66
Eggplant	N/A	N/A	N/A	N/A	N/A	19713-258 400-568 400-567
Flax	0.06	1	NS	NS	Seed dip	264-931 66330-238
Gladiolus	0.8	1	NS	NS	Root dip	19713-235 19713-258 19713-268 19713-362
Gladiola Bulbs	0.04	1	NS	NS	Seed dip	19713-258 19713-268 66330-238
Ginseng	2	8	15.6	10	Aerial or Boom Spray	19713-235
Grasses (seed)	0.1	1	NS	NS	Seep dip	264-931
Grasses (Ornamental)/ Turf (Golf Course)	1	2	4.3	8.6	Aerial or Boom Spray	19713-235 19713-258 19713-268
Grasses (Lawn Seedbeds)/Turf (Sod Farms)	1	2	4.3	8.6	Aerial or Boom Spray	19713-235 19713-258 19713-

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
						268
Grapes	2	8	12	14	Aerial or Boom Spray	19713-258 19713-268
Honeydew	NS	1	NS	NS	Seed dip	19713-258
Kale	NS	1	NS	NS	Seed dip	19713-258
Lentils	0.1	1	12	NS	Seed dip	19713-258 34704-935 66330-238
Lespedeza	0.4	1	NS	NS	Seed dip	19713-258 66330-238
Lettuce	NS	1	NS	NS	Seed dip	19713-258 400-568
Milo	0.3	1	NS	NS	Seed dip	19713-258 66330-238
Mustard Seed	0.9	1	NS	NS	Seed dip	400-568
Nectarines	2	NS	24	14	Aerial or Boom Spray	19713-258 19713-268
Oats	0.05	1	NS	NS	Seed dip	264-931 66330-238
Onions	NS	1	NS	NS	Seed dip	66330-238
Onions (pelleting)	0.04	1	NS	NS	Seed dip	19713-161
Okra	NS	1	NS	NS	Seed dip	19713-258
Paint Additive	NS	NS	NS	NS	Additive	66330-31
Peaches	4.0	NS	32	14	Aerial or Boom Spray	19713-258 19713-268

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
Plums/Fresh Prunes (Western US)	2.0	NS	27	14	Aerial or Boom Spray	19713-258 19713-268
Peanuts	0.1	1	1	NS	Seed dip	66330-238
Peas	0.01	1	12	NS	Seed dip	66330-238
Peppers (California wonder)	0.01	1	NS	NS	Seed dip	19713-161
Peppers	0.7	1	12	NS	Seed dip	66330-238
Potatoes	1.0	1	NS	NS	Dusted at cutting for seed	400-568 2935-536 19713-258
Roses	1.0	NS	NS	14	Aerial or Boom Spray	19713-235 19713-258 19713-268 19713-362
Radish	0.07	NS	NS	NS	Seed dip	66330-238
Raspberries	2	8	10	14	Aerial or Boom Spray	19713-258 19713-268
Rye, Triticale	0.3	1	12	NS	Seed dip	264-931 66330-238
Rutabaga	NS	1	NS	NS	Seed dip	19713-258
Strawberries	3.0	8	24	7	Aerial or Boom Spray	19713-258 19713-268
Swiss Chard	0.01	1	12	NS	Seed dip	66330-238
Soybeans	0.2	1	12	NS	Seed dip	66330-238
Spinach	0.2	1	12	NS	Seed dip	66330-238
Squash, Watermelon, Pumpkin,	0.01	1	12	NS	Seed dip	66330-238

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
Muskmelon						
Safflower	NS	1	NS	NS	Seed dip	19713-258
Sunflower	0.1	1	12	NS	Seed dip	66330-238
Sesame	NS	1	NS	NS	Seed dip	19713-258
Soil/Greenhouse Bench	0.01	1	NS	NS	Root dip/soil condition/ Spray	66330-234 66330-239
Sorghum (Milo)	0.01	1	12	NS	Seed dip	66330-238
Sugar Beets – Western US	0.01	1	12	NS	Seed dip	66330-238
Tomatillo	0.3	1	NS	NS	Seed dip	400-568
Tomatoes	0.3	1	NS	NS	Seed dip	400-568 19713-258
Turnips	0.01	1	NS	NS	Seed dip	66330-238
Wheat	0.03	1	12	NS	Seed dip	264-931 66330-238
24 (c) CA: Strawberry	NS	1	NA	NA	Plant dip	CA-020017; 51036-166
24 (c) CA: Lily bulbs	4	1	4	NA	Soak bulbs in solution then apply solution in-furrow	CA-100006; 1973-156
24 (c) CA: Easter lily bulbs	NS	1	NS	NA	Soak bulbs in solution	CA-980028; 264-931
24 (c) OR: Easter lily	NS	1	NS	NA	Soak bulbs in solution	OR-030029; 19713-156
24 (c) OR: Grass Seeds for Export Only	0.8 grams carboxin and 0.76 grams captan /kg seed	NA	NA	NA	Seed treatment for export, not for use immediately prior to planting	OR-070024; 400-554
1. NS = not specified						

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number

Metabolites and Degradates

Captan degrades in soil or water include tetrahydrophtalate (THPI), trichloromethylthio (TCMT), tetrahydrophthalimic acid (THPAm), cyclohex-4-ene-2-cyano-1-carboxylic acid (THCY), inorganic sulfur and chlorine, and thiophosgene (EPA, 2003a).

Chlorothalonil

Chlorothalonil is a broad spectrum pesticide, used primarily as a fungicide and registered for use on a variety of crop and noncrop sites (*e.g.* nursery, home and garden, golf course). It has also been registered for use as a wood protectant, antimold and antimildew agent, bactericide, mirobiocide, algaecide, insecticide, and acaricide (EPA, 1999b). Chlorothalonil's exact mechanism of toxicity for vertebrate species is unknown, although in fungus it is reported to interfere with cellular respiration by binding glutathione (EPA, 2007b). More than 40 companies hold active registrations for pesticides containing chlorothalonil, and there are more than 100 enduse products containing chlorothalonil are currently registered with EPA (http://ppis.ceris.purdue.edu/htbin/cnamlst2.com). There are eleven SLN registrations in California, Idaho, Oregon, and Washington (EPA Reg. No. CA-030010, CA-960027, OR-000023, OR-030008, OR-990038, OR-990037, OR-990039, OR-990040, WA-000003, WA-020012, WA 000014). There are no emergency use registrations (section 18) for chlorothalonil in California, Idaho, Oregon, or Washington.

Usage Information.

EPA reported an average domestic use of approximately 15 million lbs of chlorothalonil per year for the period 1999-2000 (EPA, 2003b). Recent data show use of chlorothalonil has declined in California over the last decade from approximately 1.8 million lbs in 1998 to 558 thousand lbs in 2008 (CDPR, 2009). In recent years, the greatest use of chlorothalonil in California has been on tomatoes, almonds, and landscape maintenance.

Washington State Department of Agriculture estimates total annual usage of chlorothalonil on potatoes exceeds 96,000 lbs and approximately 20, 000 – 28,000 lbs of chlorothalonil are used on onions. Annual use of chlorothalonil on cranberry (1,000 – 12,000 lbs), Christmas trees (1,000-18,000 lbs), and iris and tulip bulbs (600-1,200 lbs) account for a smaller proportion of current use of the chemical in Washington (WSDA, 2010c). Estimated use on other sites in Washington and recent usage information for Oregon and Idaho is not available.

Agricultural Uses. Chlorothalonil is approved for use on a variety of vegetables, field, orchard, turf, and ornamental crops.

Non-agricultural Uses. Non-agricultural uses of chlorothalonil include golf courses, conifers, lawns around commercial and industrial buildings, and collegiate and professional athletic fields. They also include landscape areas around residential, institutional, public, commercial and industrial buildings, parks, recreational areas and athletic fields. It is also used as a wood protectant, antimold, and antimildew agent. Although labels allow use on forest stands of conifers, Syngenta indicates that in practice it is not used for general forestry management. Further, Syngenta is working with all existing chlorothalonil registrants to get all existing chlorothalonil labels amended to clarify that conifer use includes nursery beds, Christmas tree and bough production plantations, tree seed orchards, and landscaping, but not applications to forests (Syngenta, 2011).

Registered Formulation Types. End use products containing chlorothalonil are available in a variety of liquid applied formulations including emulsifiable concentrates, wettable powders, and water dispersable granules. Chlorothalonil end-use pesticides frequently contain other active ingredients such as propiconazole (*e.g.*, EPA Reg. No. 100-1347), azoxystrobin (100-1315), mandipropamid (100-1279), mefenoxam (100-1221), and fludioxonil (100-1231).

Methods and Rates of Application.

Methods. Chlorothalonil may be spray-applied by aerial and ground application methods, including chemigation. Tank dip applications are also authorized for bulbs and several plants. Some labels specify that the product must not be applied within 150 feet for aerial and air-blast applications, or 25 feet for ground applications of marine/estuarine water bodies (EPA Reg. No. 50534-201). Several labels indicate that chlorothalonil products can be tank mixed with many commonly used insecticides, fungicides, and spray adjuvants (EPA Reg. No. 100-800).

Application Rates. Active labels allow for a maximum single application rate of up to 11.3 lbs of chlorothalonil/A on golf courses and some lawn and turf use sites (Table 8). Most agricultural applications are restricted to single application rates of < 5 lbs chlorothalonil/A. However, the maximum seasons application rates are quite high for some field crops (celery 18 lbs a.i./A), nut and fruit crops (pistachio 22.5 lbs a.i./A), ornamentals (36.4 lbs a.i./A), and golf course applications (27 and 73 lbs a.i./A for fairways and greens, respectively).

Table 8. Summary of all authorized use sites and application restrictions for active chlorothalonil products.

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
Brassica, head and stem	1.5	NS	8.8 per growing season	7-10	Air, ground or chemigation	60063-7
Chinese (napa) cabbage	1.5	NS	8.8 per growing season	7-10	Air, ground or chemigation	60063-7
Curbits: Cucumber, Cantaloupe, Muskemelon, Honeydew melon, Watermelon, Squash, Pumpkin, Zucchini Additional Crops: Chayote, Chinese waxgourd, Gourds, Momordica spp.	2.5	NS	15.75 per growing season	7-10	Air, ground or chemigation	60063-7

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
Fruiting Vegetables (does not include tomatos): Eggplant, Groundcherry, Okra, Pepino, Peppers	1.2	NS	9.0 per growing season	7-10	Air, ground or chemigation	50534-201; 50534-188
Ginseng	1.5	NS	12.0 per growing season	7-10	Air, ground or chemigation	60063-7
Horseradish	2.3	NS	18.0 per growing season	7-10	Air, ground or chemigation	60063-7
Lupine, Lentil	1.1	NS	6.0 per growing season	7-10	Air, ground or chemigation	60063-7
Persimmon	0.9	NS	4.7 per growing season	14	Air, ground or chemigation	60063-7
Rhubarb	2.3	NS	13.5 per growing season	7-10	Air, ground or chemigation	60063-7
Yam	1.1	NS	11.25 per growing season	10-14	Air, ground or chemigation	60063-7
Turf / sod farms	11.3	NS	13.0	7	Air, ground or chemigation	50534-202
Turf / lawns around commercial/industrial buildings, collegiate and professional athletic fields, ornamental turfgrass (lawns at homes, apartments, etc., excluded)	11.3	1-NS	26.0	7	NS - Spray application	50534-209
Golf courses tees, greens and fairways	11.3	1-2 depending on rate	73.0 greens; 52.0 tees; 26.0 fairways[2]	Variable depending on concentration rates	Air, ground or chemigation	50534-202; 50534-209
Ornamentals	2.1 lbs./100 gallons water[6]; 1.5 lb./ac	NS	36.4 per growing season	7-14	Air, ground or chemigation	50534-202; 66222-149
Flowering bulbs (caladium, crocus, daffodil, iris, lily, tulip)	4.1[3]	NS	36.4[3]	Dip once prior to planting bulbs	Dip tank, then spray apply material to field with ground application equipment.	50534-202

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
Conifers Christmas tree plantations; forestry applications[7].	4.1	For as long as conditions are favorable for disease.	16.5	7-21 depending on size of trees.	Air, ground spray	66222-149; 50534-201
Asparagus	3.0	NS	9.0	14-28	Ground	66222-149; 50534-201
Beans, Dry: Including but not limited to: Navy, Pinto, Kidney, Lima, Broad, Pink, Jack, Cow pea, Chick pea (Garbanzo), Black-eyed pea, Southern Pea	1.5	NS	6.0	7-10	Air, ground spray or chemigation	66222-149; 50534-201
Beans, Snap	2.3	NS	9.0	7	Air, ground spray	66222-149; 50534-201
Blueberry	3.0	NS	9.0	10-14	Air, ground spray	66222-149; 50534-201
Cabbage, Broccoli, Cauliflower, Brussels sprouts, Chinese Mustard Cabbage	1.5	NS	12.0	7-10	Air, ground spray	66222-149; 50534-188
Carrot	1.5	NS	15.0	7-10	Air, ground spray or chemigation	66222-149; 50534-201
Celery	2.3	NS	18.0	7	Air, ground spray or chemigation	66222-149; 50534-188
Corn	1.5	NS	9.0	7	Air, ground spray	66222-149; 50534-201
Cranberry	5.0[4]	NS	15.0	10-14	Air, ground spray or chemigation	66222-149; 50534-201
Grasses grown for seed	1.5	NS	4.5	14	Air, ground spray or chemigation	66222-149; 50534-201
Mango	2.6	NS	24.0	7-14	Air, ground spray	66222-149; 50534-201
Mint (Oregon)	1.0	NS	3.0	7-10	Air, ground spray	66222-149
Onion (dry bulb), and Garlic	2.3	NS	15.0	7-10	Air, ground spray, or chemigation	50534-201
Onion (green bunching) Leek, Shallot, Onion, Garlic grown for seed	2.3	NS	6.75	7-10	Air, ground spray or chemigation	66222-149; 50534-201; 50534-188
Papaya	2.3	NS	6.75	14	Ground spray	66222-149; 50534-201; 50534-188
Parsnip	1.5	NS	6.0	7-10	Air, ground spray or chemigation	66222-149
Peanut	1.1	NS	9.0	14	Recommended	66222-149;

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
					to alternate chemigation with air or ground spray	50534-201
Potato	1.1	NS	11.25	5-10	Air, ground spray or chemigation	66222-149; 50534-201
Soybean	1.8	NS	4.5	10-14	Air, ground spray or chemigation	66222-149; 50534-201
Tomato	2.2	NS	15.1	7-14	Air, ground spray or chemigation	66222-149; 50534-201
Strawberry (non-bearing nurseries)	1.2	NS	15.0	10-14	Air, ground spray or chemigation	66222-149
Almonds	3.0	NS	18.75	NS	Air, ground spray	66222-149; 50534-201
Filberts	3.0	NS	9.0	14-28	Air, ground spray	66222-149; 50534-201
Fruit Trees: Apricot, Cherry (sweet and tart), Nectarine, Peach Plum, Prune	3.1	NS	15.5	10-14	Air, ground spray	66222-149; 50534-201
Pistachio	4.5	NS	22.5	28	Air, ground spray	66222-149; 50534-201; 50534-188
Passion Fruit	1.5	NS	7.5	14	Ground spray	50534-201
Wood protectant and antimildew and antimildew agent	16.3[5]	NS	NS	NS	Brush, spray, or dip application	577-544
24 (c) CA: Strawberry Transplants	1.1 / 100 gallons water	One dip per season	15	NS	Tank dip only	CA-960027; 50534-188
24 (c) CA: Strawberry (non-bearing nurseries)	1.1 / 100 gallons water	NS	15	10-14	Ground spray or chemigation	CA-960027; 50534-188
24 (c) WA: treatment for "bulb rot"	4.1 / 100 gallons water	NS	36.4	NS	Tank dip only	WA-000003: 50534-202
24 (c) OR: Sugar beets (seed production only)	1.3	NS	NS	NS	Air, ground spray or chemigation	OR-990040; re:50534-188
24 (c) OR: Mint	1.1	3	3.3	7-10	Air, ground spray or chemigation	OR-990038; 50534-188-10182
24 (c) OR: Mint	1.0	3	3.0	7-10	Air, ground spray or chemigation	OR-990037;

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
						50534-201
24 (c) OR: Ornamental Bulbs	4.1 / 100 gallons water	NS	NS	NS	Tank dip only	OR-000023; 50534-202-100
24 (c) WA: Chickpeas	1.5	5	6.0 per growing season	7-10	Air, ground or chemigation	WA-020012; 60063-7
24 (c) WA: Conifer seedlings forest tree nursery and green house management.	4.2	For as long as conditions are favorable for disease.	16.5 per growing season	7-14	Ground spray	WA 000014; 50534-202
24 (c) OR: Chickpeas	1.5	5	6.0 per growing season	7-10	Air, ground spray or chemigation	OR-030008; 66222-149
24 (c) OR: Sugar beets (seed production only)	1.3	NS	NS	NS	Air, ground spray or chemigation	OR-990039; 0534-201
24 (c) CA: Gladiolus corms	2.0 / 100 gallons water	One dip before storage, one before planting	NS	NS	Tank dip only	CA-030010; 50534-209

1. NS = not specified
2. Label discrepancy allows for much higher annual use rate than could be achieved considering single application maximum and limits on number of applications
3. Bulbs dipped in solution at rate of up to 5.0 lbs. product / 100 gallons of water. This ratio is equivalent to 4.125 lbs. a.i. / acre as calculated assuming 100 gallons of solution applied/A.
4. Label specifies not to apply to bogs when flooded or allow release of irrigation water from bogs for at least 3 days following application.
5. Wood stain product containing 0.7% chlorothalonil and 0.3% bis(tributyltin) oxide. Apply at rates of up to 1 gallon / 150 ft^2. Assumed net weight of 8 lbs/gallon as net weight not provided.
6. Syngenta indicated they will remove pachysandra from their labels so that the maximum single application rates for ornamentals will be reduced to 1.16 lb a.i./A.
7. Syngenta indicated they will clarify labels to indicate that the only conifer uses will include: conifer nursery beds, Christmas tree and bough production plantations, tree seed orchards and landscaping.

Metabolites and Degradates.

EPA has identified 4-hydroxy-2,5,6-trichloro-1,3-dicyanobenzene (SDS-3701) as a degradate of concern for terrestrial organisms due to its elevated toxicity and persistence relative to chlorothalonil (EPA, 2007b). It is ubiquitous in the environment and consistently found at 10 – 40% of the applied parent. Four other degradates/metabolites (SDS-19221, SDS-46851, SDS-47523/SDS-47524, and SDS-47525) have been identified as products in aerobic soil or anaerobic aquatic conditions (EPA, 1999b).

Action Area

The action area is defined as all areas to be affected directly or indirectly by the federal action and not merely the immediate area involved in the action (50 CFR §402.02). Given EPA's nationwide authorization of these pesticides, the action area would encompass the entire U.S. and its territories. 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 waters of California, Oregon, and Washington. The action area also includes all freshwater surface waters in Idaho (Figure 2).

2,4-D, triclopyr BEE, diuron, linuron, captan and chlorothalonil are the fourth 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 threatenedin 2005. The Puget Sound steelhead and the Oregon Coast coho salmon were listed as threatened in 2007 and 2008, respectively.

EPA's consultation with NMFS remains incomplete until it analyzes the effects of its authorization of pesticide product labels with these six 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.

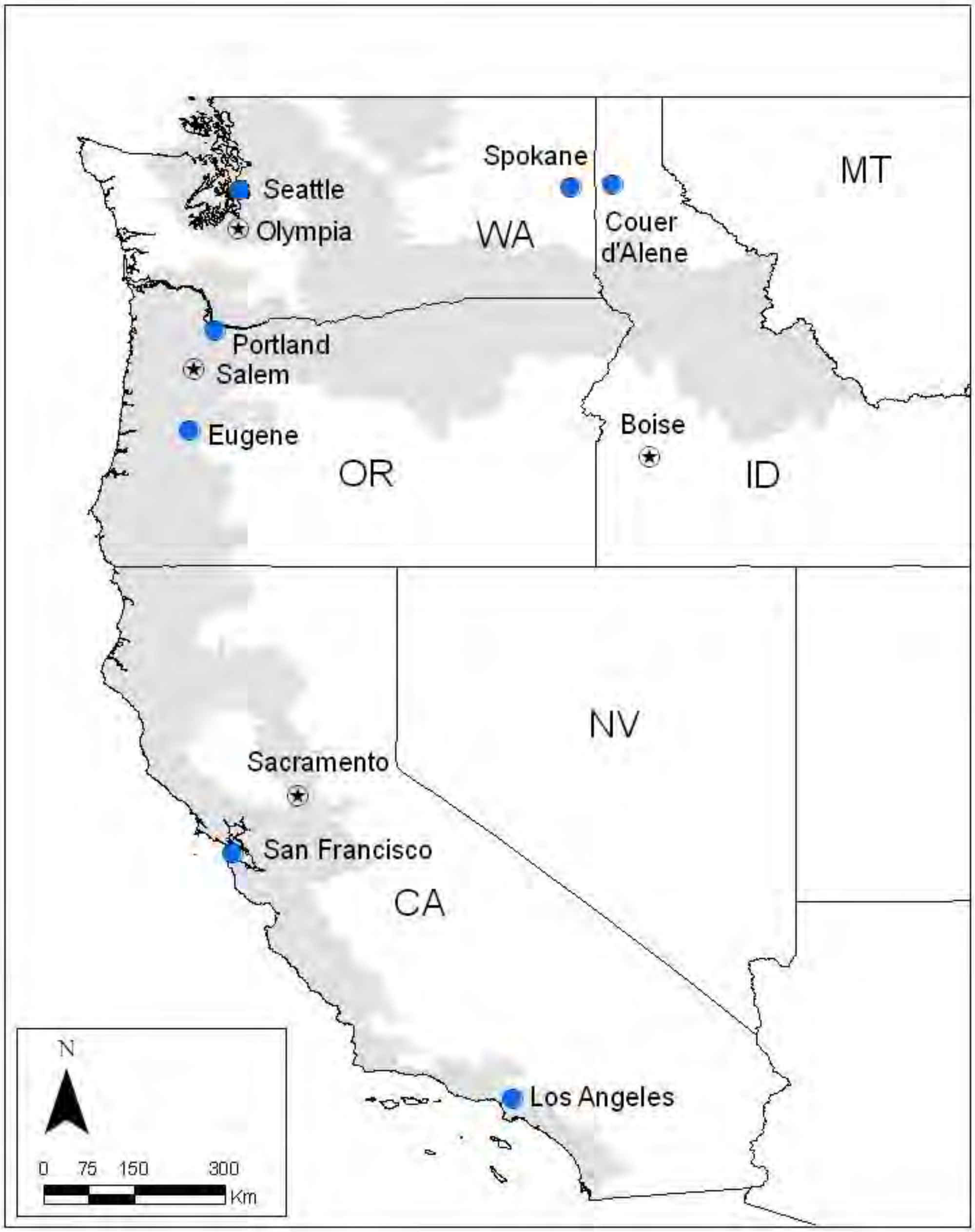


Figure 2. Map showing extent of inland action area with the range of all ESU and DPS boundaries for ESA listed salmonids highlighted in gray.

Approach to this Assessment

Overview of NMFS' Assessment Framework

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

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

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

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

In the final steps of our analyses, we establish the risks posed to listed species and to designated critical habitat. This part of the analysis is found within the *Integration and Synthesis section* where spatial analyses are used to determine overall risk to listed resources.

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

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

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

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

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

The critical habitat analysis focuses on reductions in the quality, quantity, or availability of 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 and water quality are the primary assessment endpoints addressed when evaluating the effects of the action on designated critical habitat. Information evaluated for effects to prey include prey survival, prey growth, prey drift, prey reproduction, abundance of prey, health of invertebrate aquatic communities, and recovery of aquatic communities following pesticide exposure. Information evaluated for degradation of water quality include anticipated exposure concentrations leading to toxic responses within aquatic organisms (including salmonids and their prey) as well as instances of water bodies not meeting local, state, or federal water quality standards and criteria.

Evidence Available for the Consultation

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

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

Application of Approach in this Consultation

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

Problem Formulation

Problem foumulation includes conceptual models based on our initial evaluation of the relationships between stressors of the action (pesticides and other identified chemical stressors) and potential receptors (listed species and habitat). Unlike OPP's pesticide ERAs[1], which begin with the use, fate, and toxicity properties of the a.i.s, and evaluate risk based on broad categories of taxa, 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. In brief, we employ a species-centric approach, rather than a chemical-centric approach. Assessment endpoints and measures may vary by life stage and are presented in Table 9. Some of the relevant measures are not ones commonly considered in the field of toxicology, especially in a regulatory context. They may, however, be commonly used in the disciplines of fisheries management, conservation biology, or ecological assessment.

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

Table 9. Salmonid life stage and habitat assessment endpoints and measures.

Salmonid Life Stage	Assessment Endpoint	Assessment Measure
	Individual fitness	**Measures of changes in individual fitness**
Egg* * If egg appears permeable to pesticides, may vary by pesticide type, K_{ow}, or formulation	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 (smolt)	Na/K ATPase activity, sea water challenge tests
Returning adult	Development	Length, weight, malformations
	Survival	LC_{50}, (dose-response slope). Percent dead at a given concentration

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

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

Species Risk Hypotheses

1. Exposure to the stressors of the action is sufficient to:
    a. kill salmonids from direct, acute exposure;
    b. reduce salmonid survival through impacts to growth or development;
    c. reduce salmonid growth through impacts to salmonid prey;
    d. reduce survival, migration, and reproduction through impacts to olfactory-mediated behaviors; and
2. Exposure to the herbicides 2, 4-D, triclopyr BEE, diuron, and linuron is sufficient to:

a. reduce aquatic primary producers thereby affecting salmonid prey communities and salmonids and natural cover;

b. reduce riparian vegetation to such an extent that stream temperatures are elevated, erosion increases, and reductions in natural coverage results through reduced inputs of woody debris and other organic matter.

3. Exposure to mixtures of diuron and linuron can act in combination to increase adverse effects to salmonids and salmonid habitat.
4. Exposure to active ingredient degradates, adjuvants, tank mixtures, and other active and other ingredients in pesticide products containing 2, 4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil cause adverse effects to salmonids and their habitat.
5. Exposure to other pesticides present in the action area can act in combination with 2, 4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil to increase effects to salmonids and their habitat.
6. Exposure to elevated temperatures can enhance the toxicity of the stressors of the action.

Critical Habitat

When designated critical habitat for the species is identified, primary constituent elements (PCEs) of that habitat are also identified Table 10. To determine potential effects to designated critical habitat, NMFS evaluates the effects of the action by first looking at whether PCEs of critical habitat are affected by the stressors of the action. Effects to PCEs include changes to the functional condition of salmonid habitat caused by the action in the action area. Properly functioning salmonid PCEs are important to the conservation of the ESU/DPS. The stressors of the action for this Opinion are chemicals introduced into the environment by application of pesticide products containing the six a.i.s. Key PCEs potentially affected are freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, estuarine areas, and nearshore marine areas where exposure to those stressors is anticipated.

Table 10. Essential physical and biological features named as PCEs in all salmonid critical habitat designations.

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

[a] Natural cover includes shade, large wood, 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.

Based on the PCEs and life stage potentially affected Table 10, 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.

Critical Habitat Risk Hypotheses

1. Exposure to the stressors of the action is sufficient to degrade water quality, and substrate in freshwater spawning sites;
2. Exposure to the stressors of the action is sufficient to degrade water quality, reduce prey availability (forage), and/or reduce natural cover in rearing sites;
3. Exposure to the stressors of the action is sufficient to degrade water quality, prey availability, and/or reduce natural cover in freshwater migration corridors;

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

Evaluating Exposure and Response

As part of the problem formulation phase, we consider the toxic mode and mechanism of action of chemical stressors, particularly for the pesticide a.i.s to provide insight into potential physiological 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). We consider authorized pesticide use sites, and group them into landuse categories to determine spatial overlap between the use and the species or its designated critical habitat. We consider fate properties of the pesticides and evaluate how that affects exposure. Conceptual diagrams are shown in Figure 3 and Figure 4.

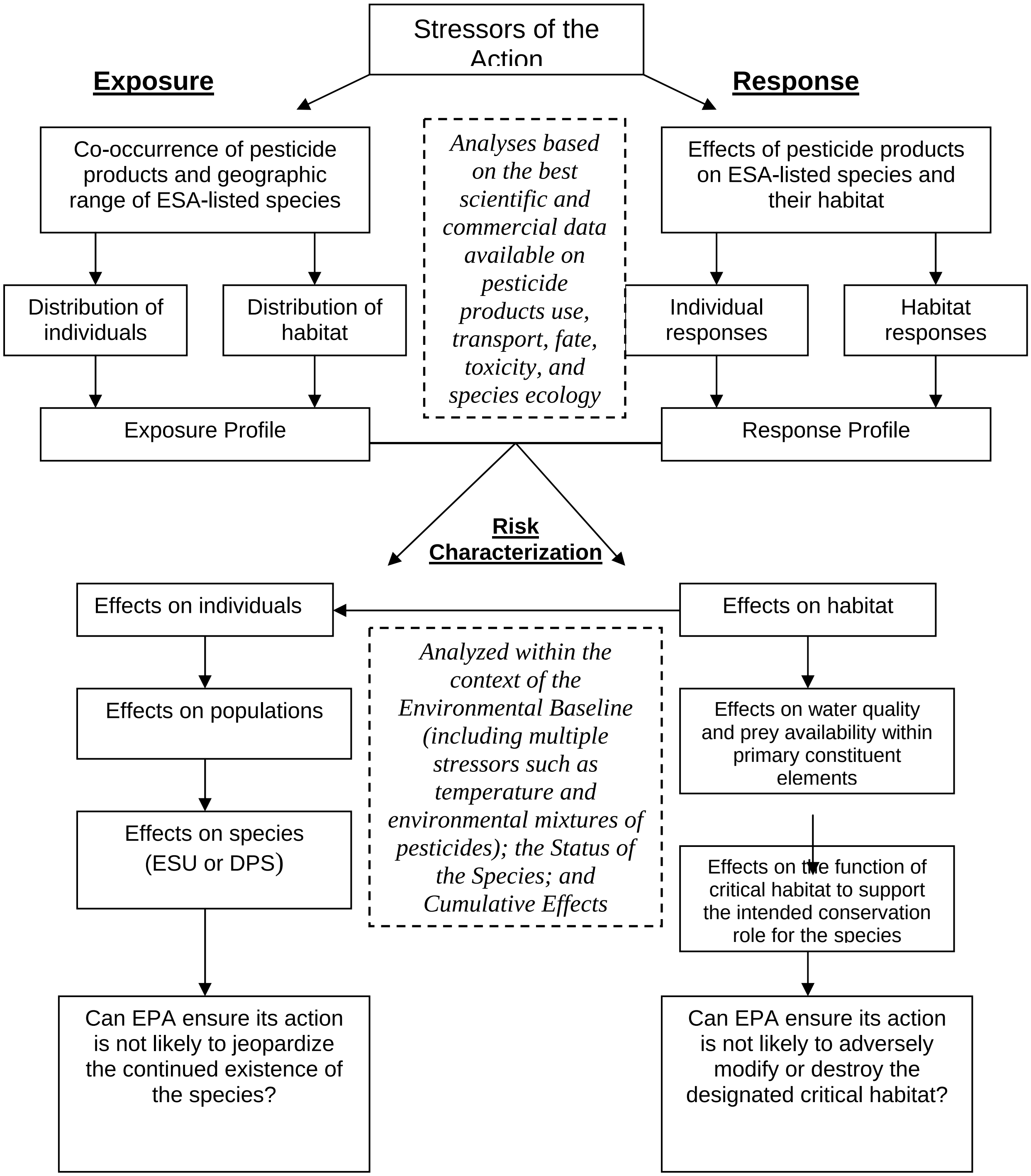


Figure 3. Conceptual framework for assessing risks of EPA's action to ESA listed resources.

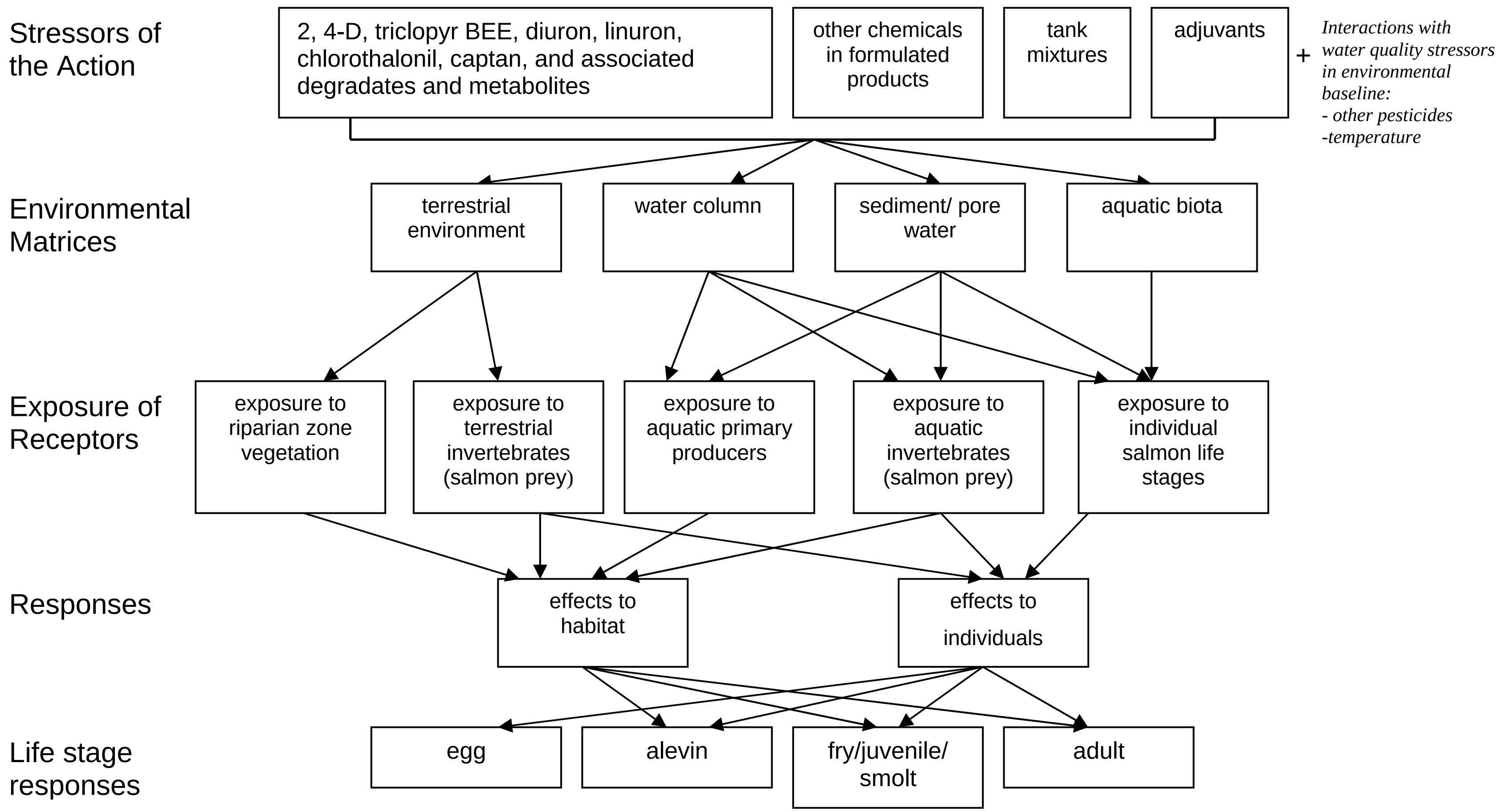


Figure 4. Exposure pathways for stressors of the action, and general response of Pacific salmonids and habitat.

Analysis Plan

Status of the Species

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

Environmental Baseline

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

Effects of the Proposed Action to Threatened and Endangered Pacific Salmonids

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

Integration and Synthesis

We begin Integration and Synthesis with with a summary of risk associated with each of the a.i.s. In separate sections for listed species and critical habitat, we combine these risk conclusions regarding the effects of the proposed action with information in the *Status of* *the Species* and *Environmental Baseline* to determine potential effects on populations and species.

Conclusion

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

Other Considerations

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

In recent pesticide risk assessments, probabilistic techniques have been used to evaluate the probability of exceeding a “toxic” threshold for aquatic organisms by combining pesticide monitoring data with species sensitivity distributions (Geisy et al., 1999; Giddings, 2009). There is utility in information generated by probabilistic approaches if supported by robust data. NMFS considered the use of probabilistic risk assessment techniques for addressing risk at population and species (ESU and DPS) scales for the stressors of the action. However, we encountered significant limitations in available data that suggested the information was not sufficient to define exposure and/or response probabilities necessary to determine the probability of risk. Probabilistic techniques were not used in the Opinion due to issues with data collection, paucity of data, non-normal distributions of data, and quality assurance and quality control. For example, it was not deemed appropriate to pair the salmonid prey responses with exposure probabilities based on monitoring results given the limitations of that data set discussed in the *Effects of the Proposed Action*. To evaluate population consequences associated with potential lethality from pesticide exposure in salmon, NMFS selected the lowest reported salmonid LC_{50} from the available information to ensure risk was not underestimated. When we consider the data limitations coupled with the inherent complexity of EPA's proposed action in California, Idaho, Oregon, and Washington, we find that probabilistic assessments at population and species scales introduce an unquantifiable amount of uncertainty that undermines confidence in derived risk estimates. These same studies do not factor the status of the 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.

Status of Listed Resources

The purpose of this section is to characterize the condition of the 28 salmonid species[2] 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 2,4-D, triclopyr BEE, diuron, linuron, captan and chlorothalonil - containing products (Table 11). More detailed information on the status of these species and critical habitat are found in a number of published documents including recent recovery plans, status reviews, stock assessment reports, and technical memorandums. Many are available on the Internet at http://www.nmfs.noaa.go/pr/species/.

Table 11. 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
Steelhead (Puget Sound)	*Oncorhynchus mykiss*	Threatened
Steelhead (Lower Columbia River*)		Threatened
Steelhead (Upper Willamette River*)		Threatened

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

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

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

Species Status

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

Before assessing population viability, we first identify the historic and current populations that constitute a species. How NMFS defines a population and its function are found in McElhany *et al. (2000)* and in Bjorkstedt *et al.(2005),* NMFS' Pacific salmon Technical Recovery Teams (TRTs) have identified historic populations within ESUs/DPSs. These historical populations have been categorized based on their distribution and demographic role (*i.e.*, functionally independent, potentially independent, or dependent). Functionally independent (independent) populations were sufficiently large to be viable in isolation, (*i.e.*, a negligible extinction risk). Potentially independent populations were potentially viable in isolation, but were likely influenced by immigrants from adjacent populations. Dependent populations were unlikely to persist over a 100-year time period in isolation. However, immigration from other nearby populations reduced the extinction risk for dependent populations. The historical conditions of the populations for each ESU/DPS serve as a point of reference for evaluating the current viability of populations[3] 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).

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

A population's tendency to increase in abundance and its variation in annual population growth defines a viable population (McElhany, et al., 2000; Morris & 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 & Doak, 2002).

Thus, in our status reviews of each listed salmonid species, we provide information on population abundance and annual growth rate of extant populations. We use the median annual population growth rate (denoted as lambda, λ) from available time series of abundance for independent populations (T. P. Good, Waples, & Adams, 2005). Several publications provide a detailed description of the calculation of lambda (T. P. Good, et al., 2005; McClure, Holmes, Sanderson, & Jordan, 2003). The lambda values for salmonid populations presented in these papers are summarized in *Appendix 2*.

Conservation Role of Critical Habitat for the Species

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

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

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

NMFS has identified six common PCEs for 7 California listed Chinook salmon 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.

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

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

Chinook Salmon

Description of the Species

Chinook salmon are the largest of the Pacific salmon and historically ranged from the Ventura River in California to Point Hope, Alaska in North America, and in northeastern Asia from Hokkaido, Japan to the Anadyr River in Russia (M.C. 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 (Groot & Margolis, 1991; M.C. Healey, 1991). One race, the “stream-type,” resides in fresh water for a year or more following emergence from gravel nests. Juveniles migrate to sea as yearlings. Stream-type Chinook salmon normally returns in late winter and early spring (spring-run) as immature adults and reside in deep pools during summer before spawning in fall. The other race, the “ocean-type,” migrate to the ocean within their first year (sub-yearlings) and usually return as full mature adults in fall (fall-run). Fall-run adults spawn soon after river entry.

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

Successful incubation depends on several factors including dissolved oxygen (DO) levels, temperature, substrate size, amount of fine sediment, and water velocity. Chinook salmon egg incubation time is highly correlated with water temperature (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 (R.A. 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 (T. C. Bjornn & 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 (J. R. Brett, 1952; C. H. 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, Gray, & Fairley, 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 (J. A. Miller & Simenstad, 1997; Rondorf, et al., 1990; Sommer, Nobriga, Harrell, Batham, & Kimmerer, 2001; Tabor, Gearns, McCoy III, & Camacho, 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 (J. A. Miller & 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 (D. W. Chapman & Bjornn, 1969; Everest & Chapman, 1972; S. W. Johnson, Thedinga, & Koski, 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 & Beauchamp, 2006; Tiffan, Clark, Garland, & Rondorf, 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 & 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 (M. C. Healey, 1982).

Chinook salmon fry may also move into non-natal tributaries (*i.e.*, streams other than those where they incubated) to rear (Limm & Marchetti, 2009; Teel, Baker, Kuligowski, Friesen, & Shields, 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 & 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 & 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, Opperman, & Moyle, 2008; Limm & 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 & Hilborn, 2003). Ocean-type Chinook salmon migrate downstream as fry immediately after emerging from spawning beds (M.C. Healey, 1991). These smaller fry and sub-yearlings extensively use shallow water habitat and sloughs within the estuary to rear to the smolt stage (K. L. Fresh, Casillas, Johnson, & Bottom, 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 (K. L. 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 (M.C. Healey, 1991). The larger stream-type immature fish disappear from the surface waters of the Strait of Georgia in early summer. In contrast, during their first ocean year, ocean-type fish are abundant in the sheltered surface waters and estuaries of the Strait of Georgia and the Puget Sound from July through November and some continue to be present throughout winter. Estuaries provide the only shelter along the open coasts of Washington, Oregon, and California; in these areas, ocean-type fry remain longer in their native estuaries. After ocean entry, immature Chinook salmon may move into large estuaries and bays as they migrate along the coast. Chinook salmon remain at sea for one to six years (more commonly two to four years), with the exception of a small proportion of yearling males (called jack salmon) which mature in fresh water or return after two or three months in salt water.

Status and Trends

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

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

Puget Sound Chinook Salmon

The Puget Sound ESU (Figure 5) 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 12). 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 12. Puget Sound Chinook salmon - preliminary population structure, abundances, and hatchery contributions (Good et al 2005).

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

Life History

Puget Sound Chinook salmon populations exhibit both early-returning (August) and late-returning (mid-September and October) Chinook salmon spawners (M.C. 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, Beamer, & McClure, 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, Hayman, & Smith, 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, Higgins, Cordell, & Stamatiou, 2004).

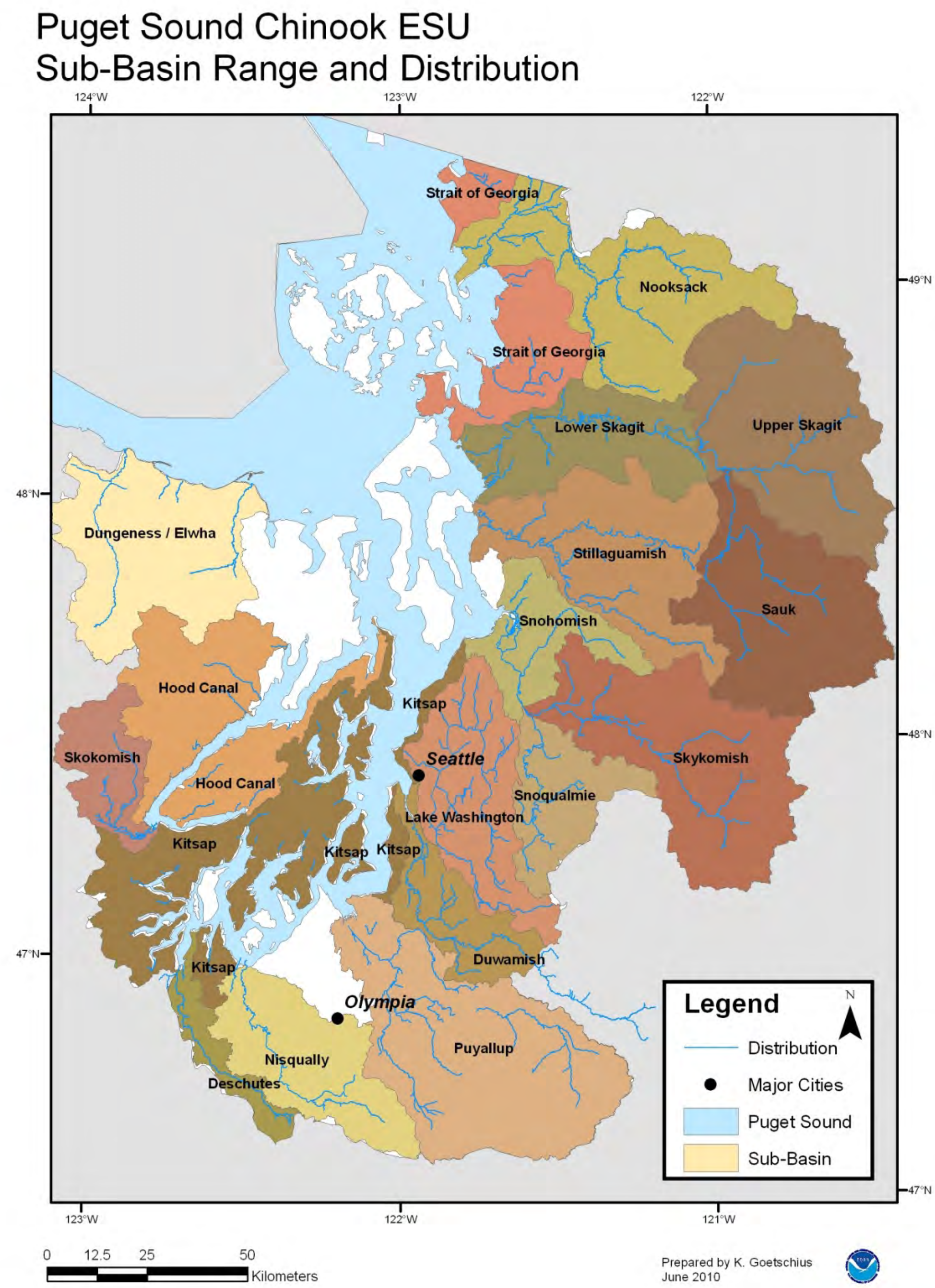


Figure 5. Puget Sound Chinook salmon distribution.

Status and Trends

NMFS listed Puget Sound Chinook salmon as threatened in 1999 (64 FR 14308) and reaffirmed its status as threatened on June 28, 2005 (70 FR 37160). Historically, the ESU included 31 rivers or river systems that supported historic independent populations. Of the historic populations, only 22 are extant (Mary H. Ruckelshaus et al., 2006) (Table 12). A disproportionate loss of an early-run life history represents a significant loss of the evolutionary legacy of the ESU (Mary H. 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 (T. P. 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 (T. P. Good, et al., 2005). Populations with the greatest long-term population growth rates are the North Fork Nooksack and White rivers.

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52630). It includes 1,683 km of stream channels, 41 square km of lakes, and 3,512 km of nearshore marine habitat. Of 61 watersheds (5th field Hydrological Units or HUC 5) reviewed in NMFS' assessment of critical habitat for the Puget Sound ESU, 9 watersheds were rated as having a medium conservation value, 12 were rated as low, and the remaining watersheds (40), where the bulk of federal lands overlap with this ESU, were rated as having a high conservation value for Puget Sound Chinook salmon (Figure 6). The 19 nearshore marine areas were all given a high conservation value rating. (Table 13).

Table 13. 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.

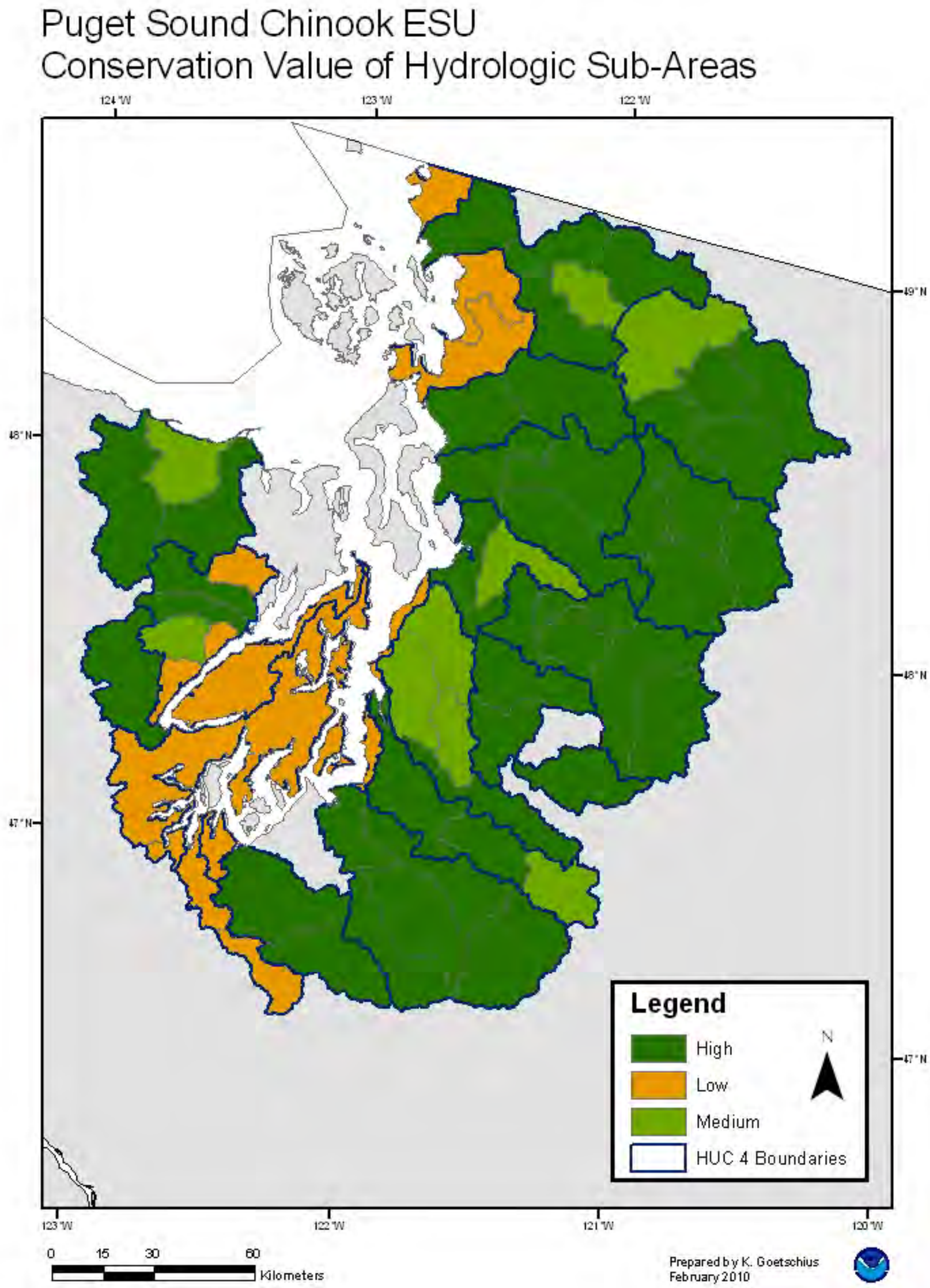


Figure 6. Puget Sound Chinook salmon Conservation Values per Sub-watershed.

Lower Columbia River Chinook Salmon

The Lower Columbia River (LCR) Chinook salmon ESU (Figure 7) 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.

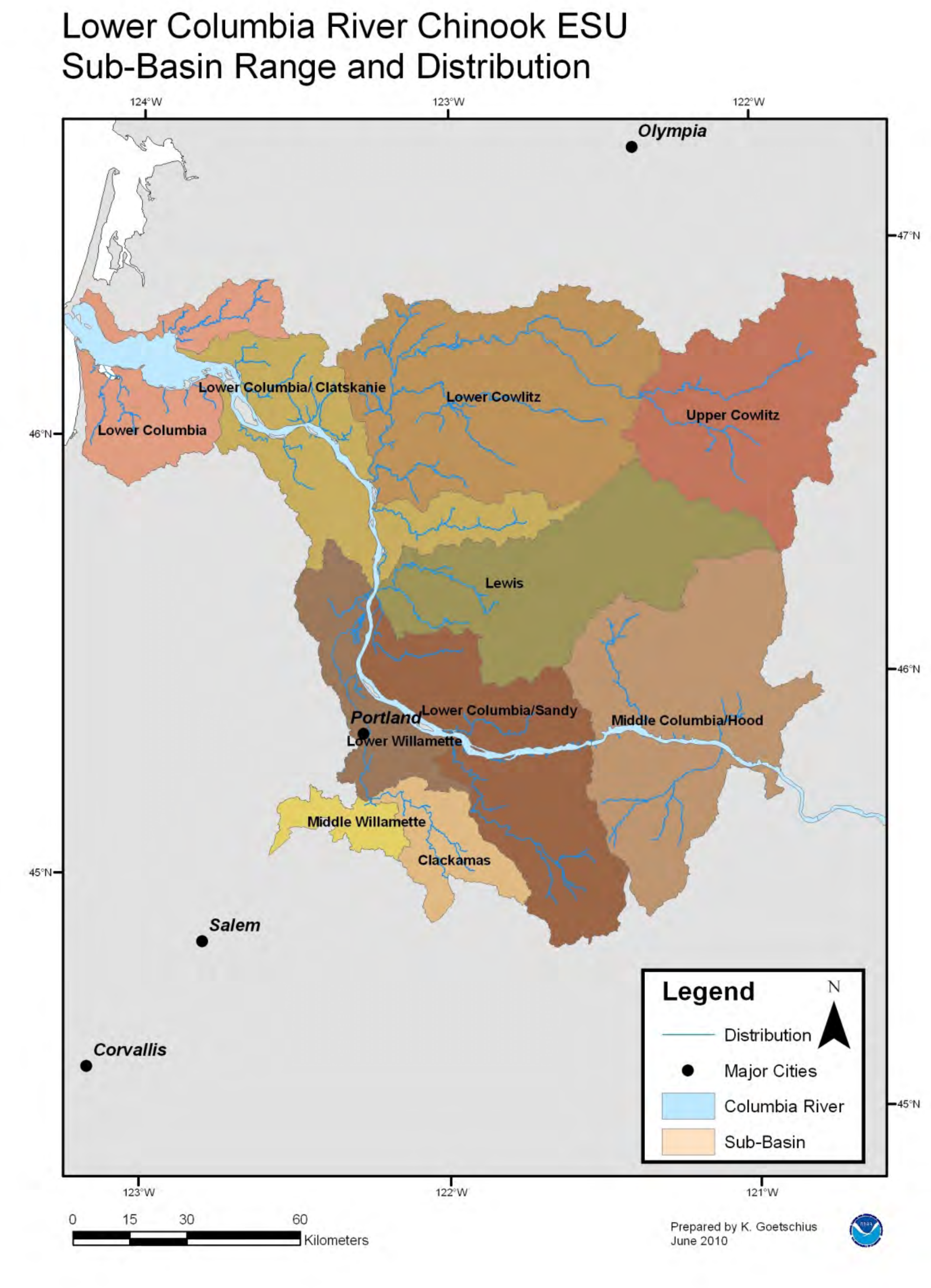


Figure 7. Lower Columbia River Chinook salmon distribution.

Life History

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

Offspring of fall-run spawning may migrate as fry to the ocean soon after yolk absorption (*i.e.*, ocean-type), at 30–45 mm in length (M.C. 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 (J. 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 (M.C. Healey, 1991; J. Myers, et al., 2006). Adults return to tributaries in the lower Columbia River predominately as three- and four-year-olds for fall-run fish and four- and five-year-olds for spring-run fish.

Status and Trends

NMFS originally listed LCR Chinook salmon as threatened on March 24, 1999 (64 FR 14308), and reaffirmed their threatened status on June 28, 2005 (70 FR 37160). Thirty-one independent Chinook salmon populations – 22 fall- and late fall-runs and 9 spring-runs – are estimated to have existed historically in the Lower Columbia River (J. 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 (T. P. Good, et al., 2005). Table 14 identifies populations within the LCR Chinook salmon ESU, their abundances, and hatchery input.

Table 14. Lower Columbia River Chinook salmon - population structure, abundances, and hatchery contributions (T. P. Good, et al., 2005; J. 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

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

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

Historical records of Chinook salmon abundance are sparse. However, cannery records suggest a peak run of 4.6 million fish [43 million lbs see (Lichatowich, 1999) in 1883]. Historically, the number of spring-run Chinook salmon returning to the Lower Columbia River may have almost equaled that of fall-run Chinook salmon (J. 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; T. P. 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, Chilcote, Myers, & Beamesderfer, 2007). The Sandy River is the only stream system supporting a natural production of spring-run Chinook salmon of any amount. However, the population is at risk from low abundance and negative to low population growth rates (McElhany, et al., 2007).

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

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

Critical Habitat

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

As shown in Figure 8, 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 15). 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 15. 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.

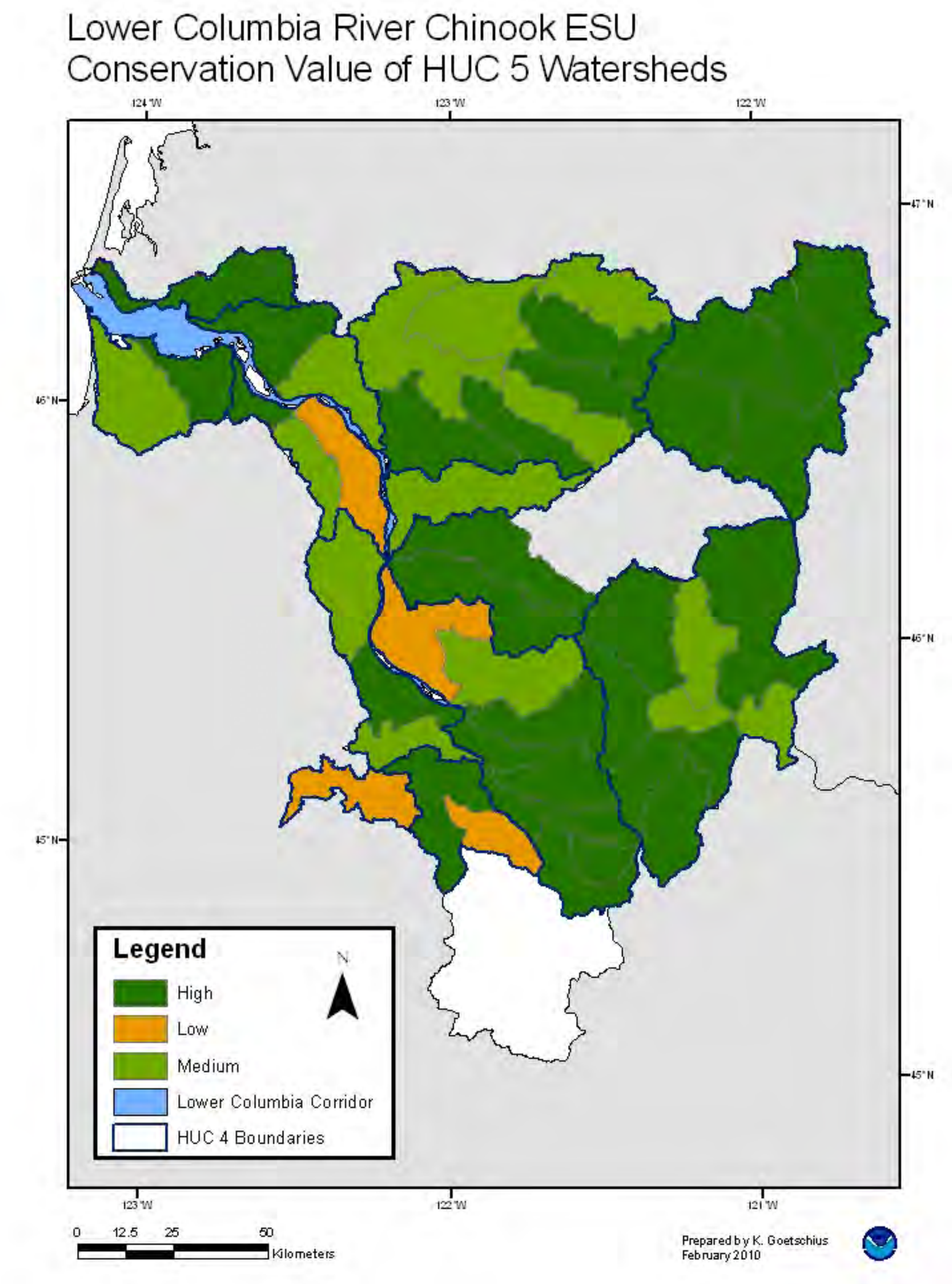


Figure 8. Lower Columbia River Chinook salmon Conservation Values per Sub-Area.

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 9).

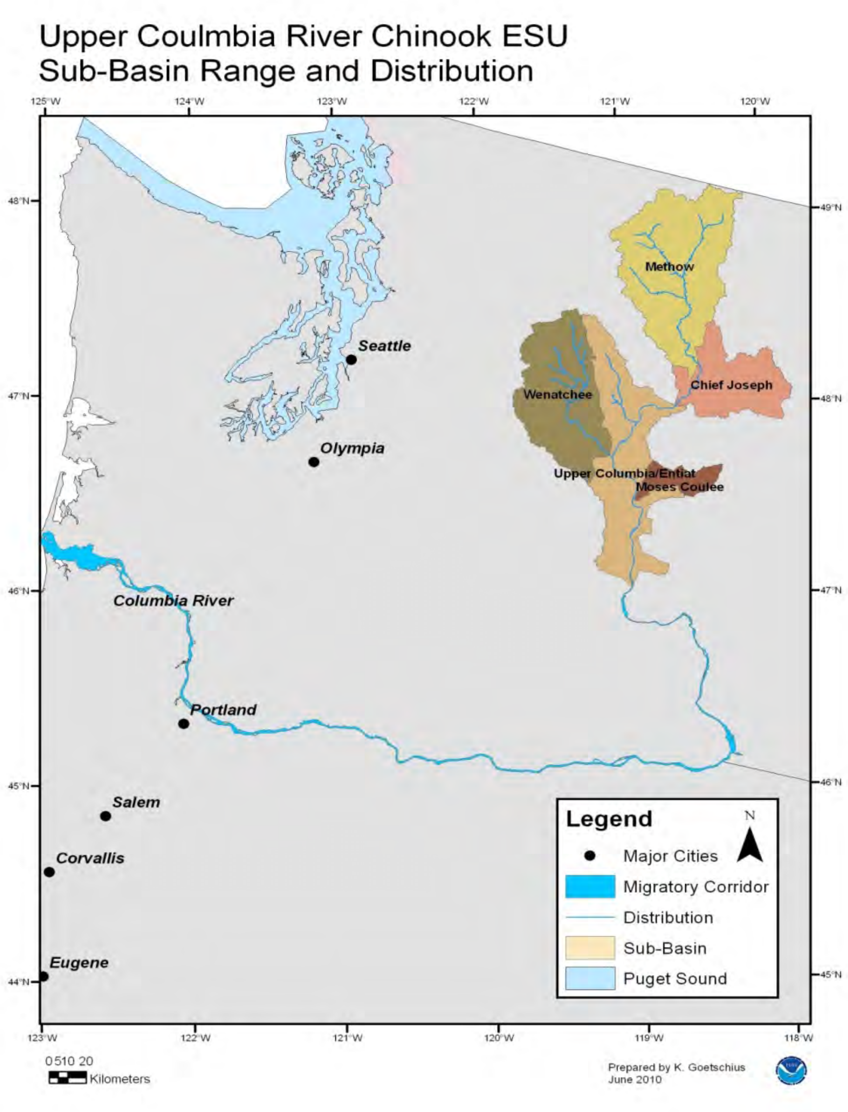


Figure 9. Upper Columbia River Chinook salmon distribution.

Several hatchery populations are also listed (70 FR 37160). These artificially propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within this ESU.

Life History

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

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

Table 16. 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-	47%

		1,046)	
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 (T. P. 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, 2008a, 2008b, 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) (T. Fisher & Hinrichsen, 2006). The long-term trend for abundance and lambda for individual populations indicate a decline for all three populations (T. P. Good, et al., 2005). Short-term lambda values indicate an increasing trend for the Methow population, but not for the Wenatchee and Entiat populations (ICTRT, 2008a, 2008b, 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 (T. P. Good, et al., 2005). However, this increase did not manifest itself in subsequent years. Thus, recent available data on population viability suggest that the ESU continues to be at high risk from small population size; all three UCR Spring-run Chinook salmon populations are affected by low abundances and failing recruitment. Should population growth rates continue at the 1980-2004 levels, UCR Spring-run Chinook salmon populations have a high probability of decline within 50 years. The genetic integrity of all populations has been compromised by periods of low effective population size and low proportion of natural-origin fish.

Critical Habitat

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

The UCR Spring-run Chinook salmon ESU has 31 watersheds within its range. Five watersheds received a medium rating and 26 received a high rating of conservation value to the ESU (Table 17). The Columbia River rearing/migration corridor downstream of the spawning range was rated as having a high conservation value (Figure 10).

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 17. 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.

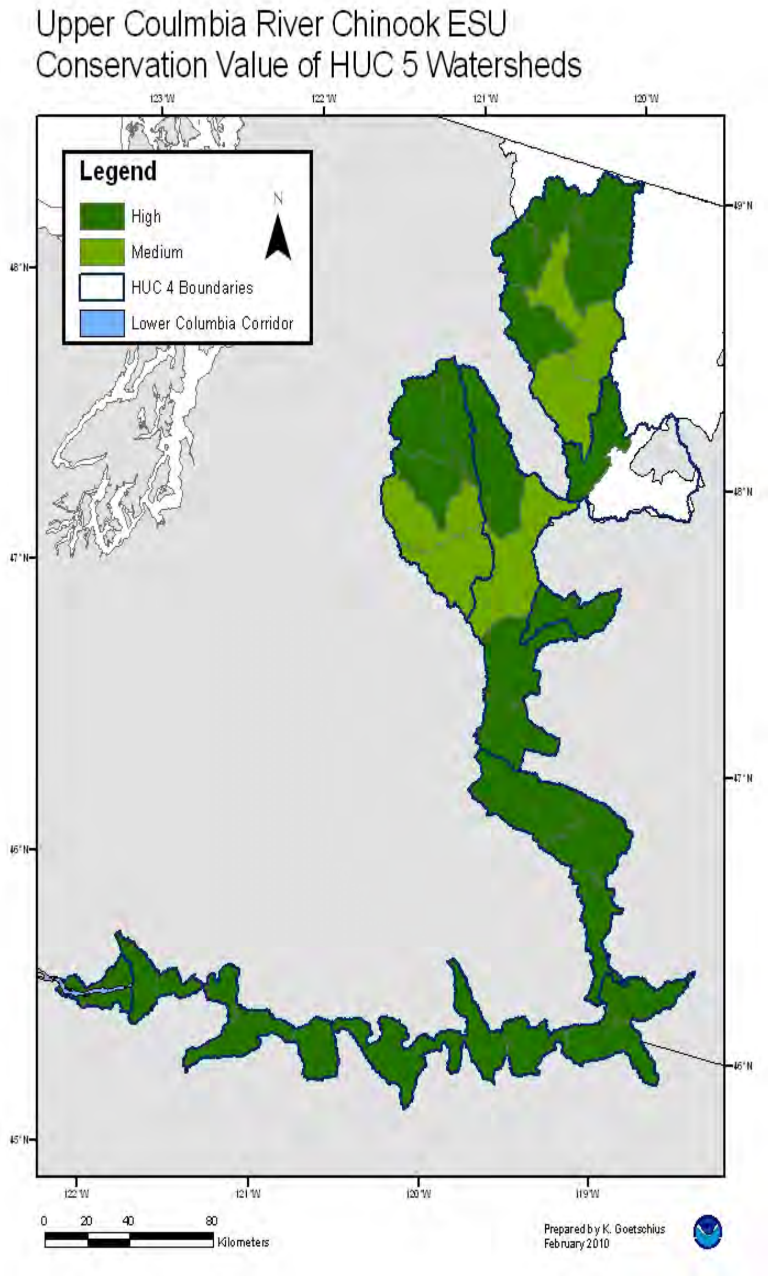


Figure 10. Upper Columbia River Spring-run Chinook salmon Conservation Values per Sub-Area

Snake River Fall-run Chinook Salmon

The Snake River (SR) Fall-run Chinook salmon ESU (Figure 11) 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.

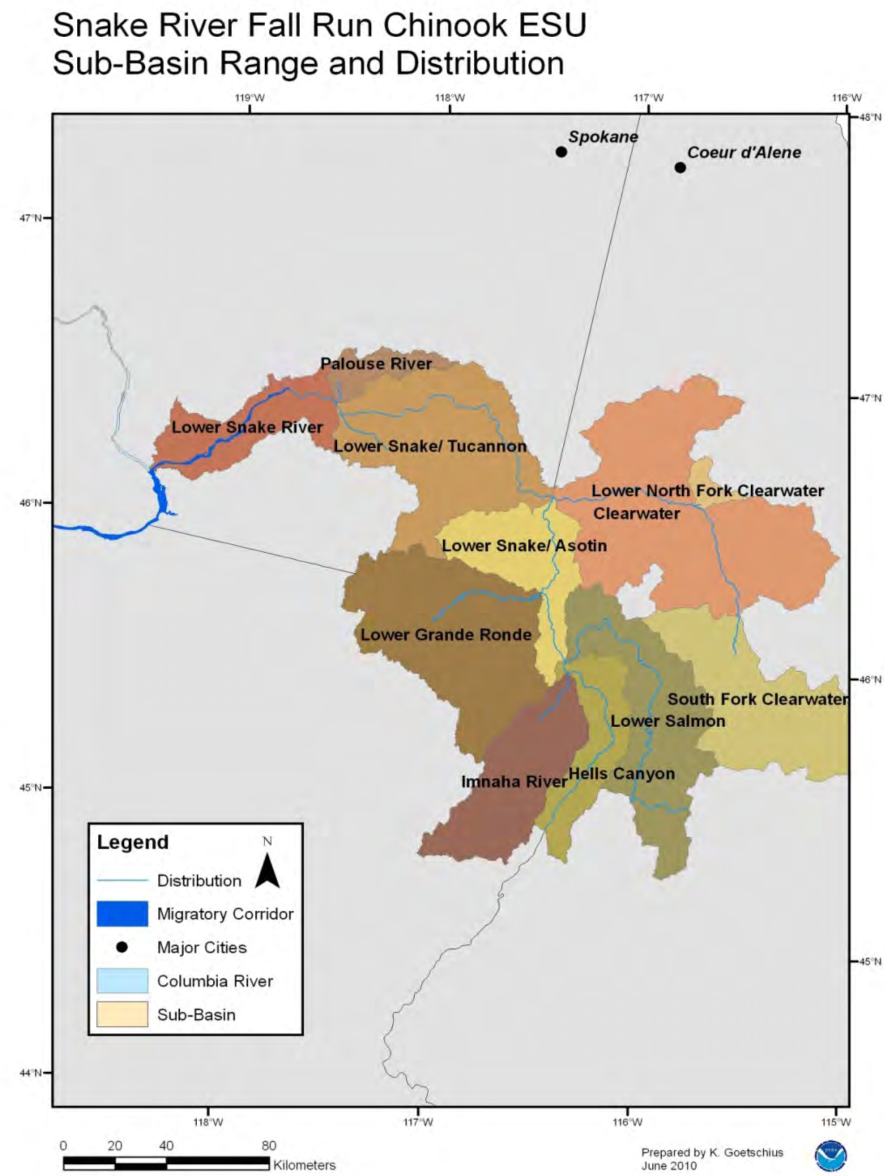


Figure 11. Snake River Fall-run Chinook salmon distribution.

Life History

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

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

Status and Trends

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

Estimated annual returns for the period 1938 to 1949 were at 72,000 fish. By the 1950s, numbers had declined to an annual average of 29,000 fish (T. C. Bjornn & 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.

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.

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 12). 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.

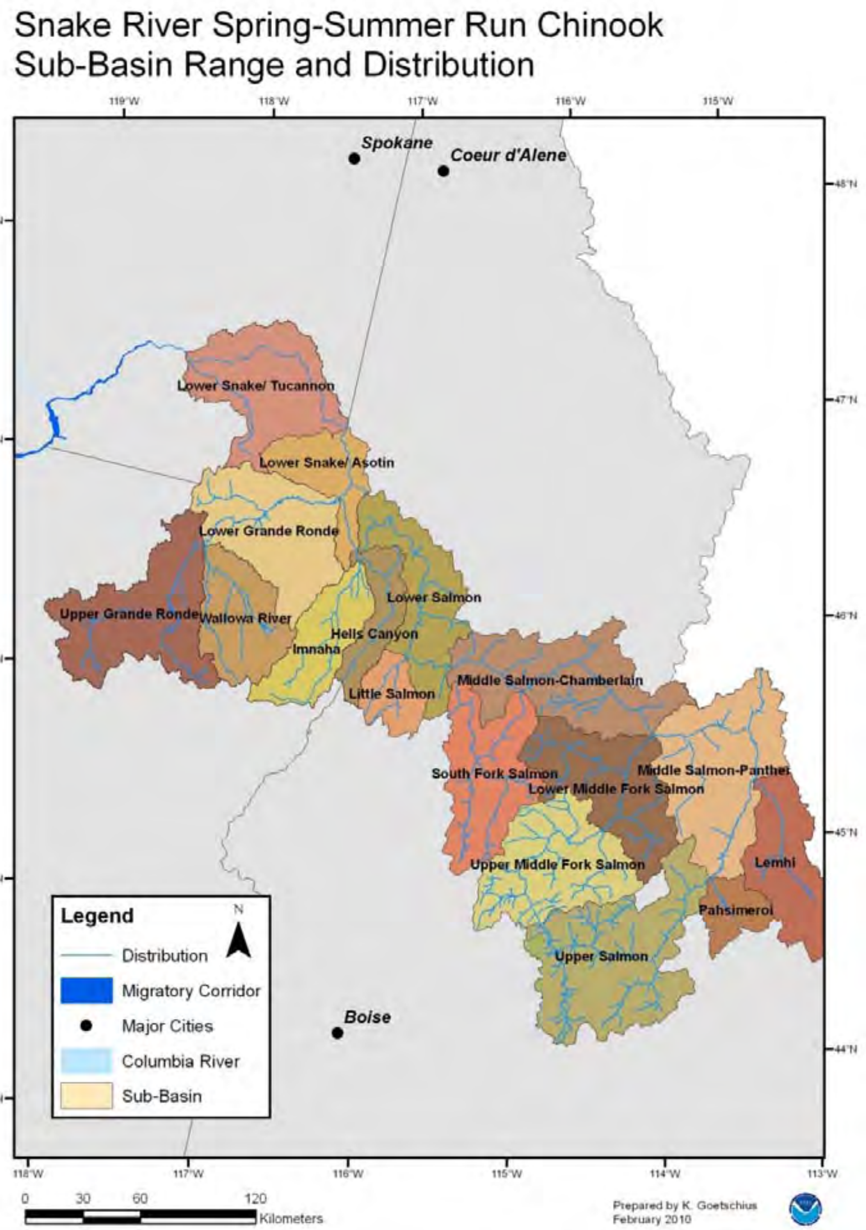


Figure 12. **Snake River Spring/Summer-run Chinook salmon distribution.**

Life History

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

Status and Trends

NMFS originally listed SR Spring/Summer-run Chinook salmon as threatened on April 22, 1992 (57 FR 14653), and reaffirmed their threatened status on June 28, 2005 (70 FR 37160). The ICBTRT has identified 31 historic populations (Table 18). 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 18 identifies populations within the Snake River Spring/Summer-run Chinook salmon ESU, their abundances, and hatchery input.

Table 18. Snake River Spring/Summer-run Chinook salmon populations, abundances, and hatchery contributions (T. P. 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 & 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 (T. P. 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 (T. P. 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 (T. P. Good, et al., 2005).

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 (D. J. Chapman & 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.

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 13). 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.

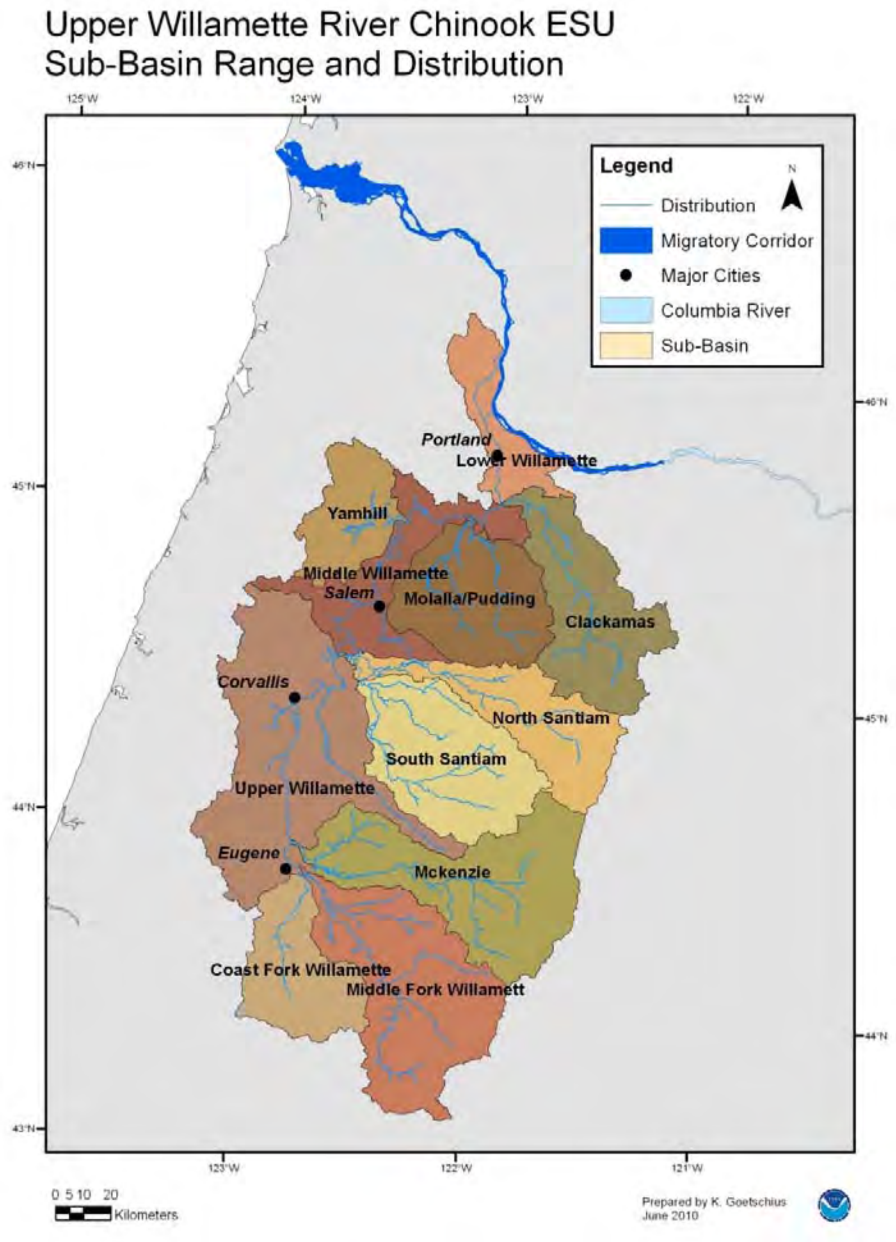


Figure 13. Upper Willamette River Chinook salmon distribution.

Life History

UWR Chinook salmon exhibit an earlier time of entry into the Columbia River than other spring-run Chinook salmon ESUs (J. M. 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 (J. Myers, et al., 2006). Juvenile migration varies with three distinct juvenile emigration "runs": fry migration in late winter and early spring; sub-yearling (0 yr +) migration in fall to early winter; and yearlings (1 yr +) migrating in late winter to spring. Sub-yearlings and yearlings rear in the mainstem Willamette River where they also use floodplain wetlands in the lower Willamette River during the winter-spring floodplain inundation period.

Status and Trends

NMFS originally listed UWR Chinook salmon as threatened on March 24, 1999 (64 FR 14308), and reaffirmed their threatened status on June 28, 2005 (70 FR 37160). Historically, this ESU included sizable numbers of spawning salmon in the Santiam River, the middle fork of the Willamette River, and the McKenzie River, as well as smaller numbers in the Molalla River, Calapooia River, and Albiqua Creek. Table 19 identifies populations within the UWR Chinook salmon ESU, their abundances, and hatchery input.

The W/LCRTRT identified seven historical independent populations (J. Myers, et al., 2006) (Table 19). 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 19. Upper Willamette River Chinook salmon independent populations core (C) and genetic legacy (G) populations, and hatchery contributions (T. P. 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 (J. Myers, et al., 2006).

Critical Habitat

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

NMFS assessed the conservation value of 59 watersheds within the range of the UWR Chinook salmon (Table 20). 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 20. 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

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 14. Seven artificial propagation programs are part of this ESU. These artificially propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within this ESU.

Life History

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

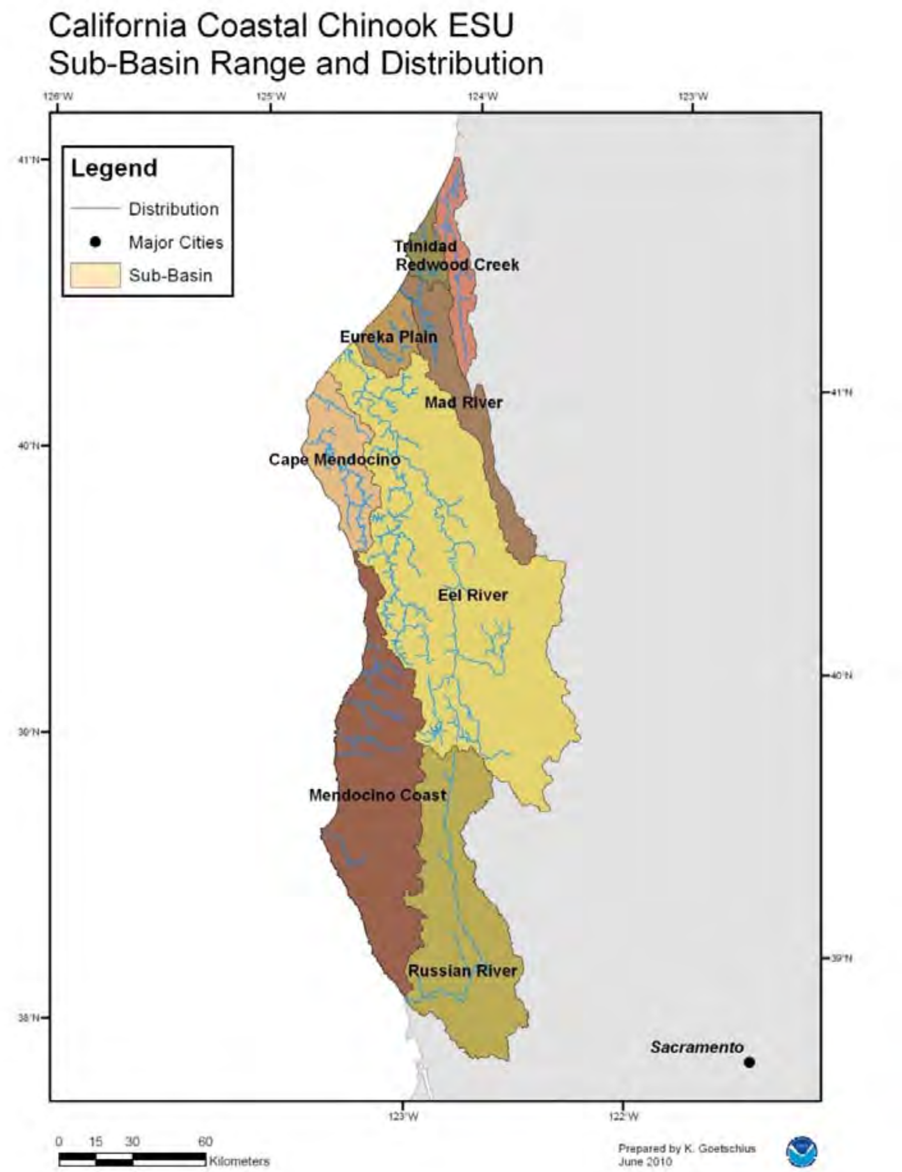


Figure 14. California Coastal Chinook salmon distribution.

Table 21. California Coastal Chinook salmon fall-run populations-preliminary population structure, abundances, and hatchery contributions (T. P. 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	

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; Brian C. Spence, et al., 2008). The number of extant populations is uncertain.

Historical estimates of escapement suggest abundance was roughly 73,000 in the early 1960s, with the majority of fish spawning in the Eel River, and about 21,000 in the 1980s (T. P. Good, et al., 2005). Table 21 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; T. P. 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 (Brian C. Spence, et al., 2008). The Eel River interior fall-run populations are severely depressed (Brian C. Spence, et al., 2008). Two functionally independent populations are believed to have existed along the southern coastal portion of the ESU; of these two, only the Russian River currently has a run of any significance (Bjorkstedt, et al., 2005). This is also the only population with abundance time series. The 2000 to 2007 median observed (at Mirabel Dam) Russian River Chinook salmon run size is 2,991 with a maximum of 6,103 (2003) and a minimum of 1,125 (2008) adults (Cook, 2008; Sonoma County Water Agency (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.

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 22). 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 15).

Table 22. 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.

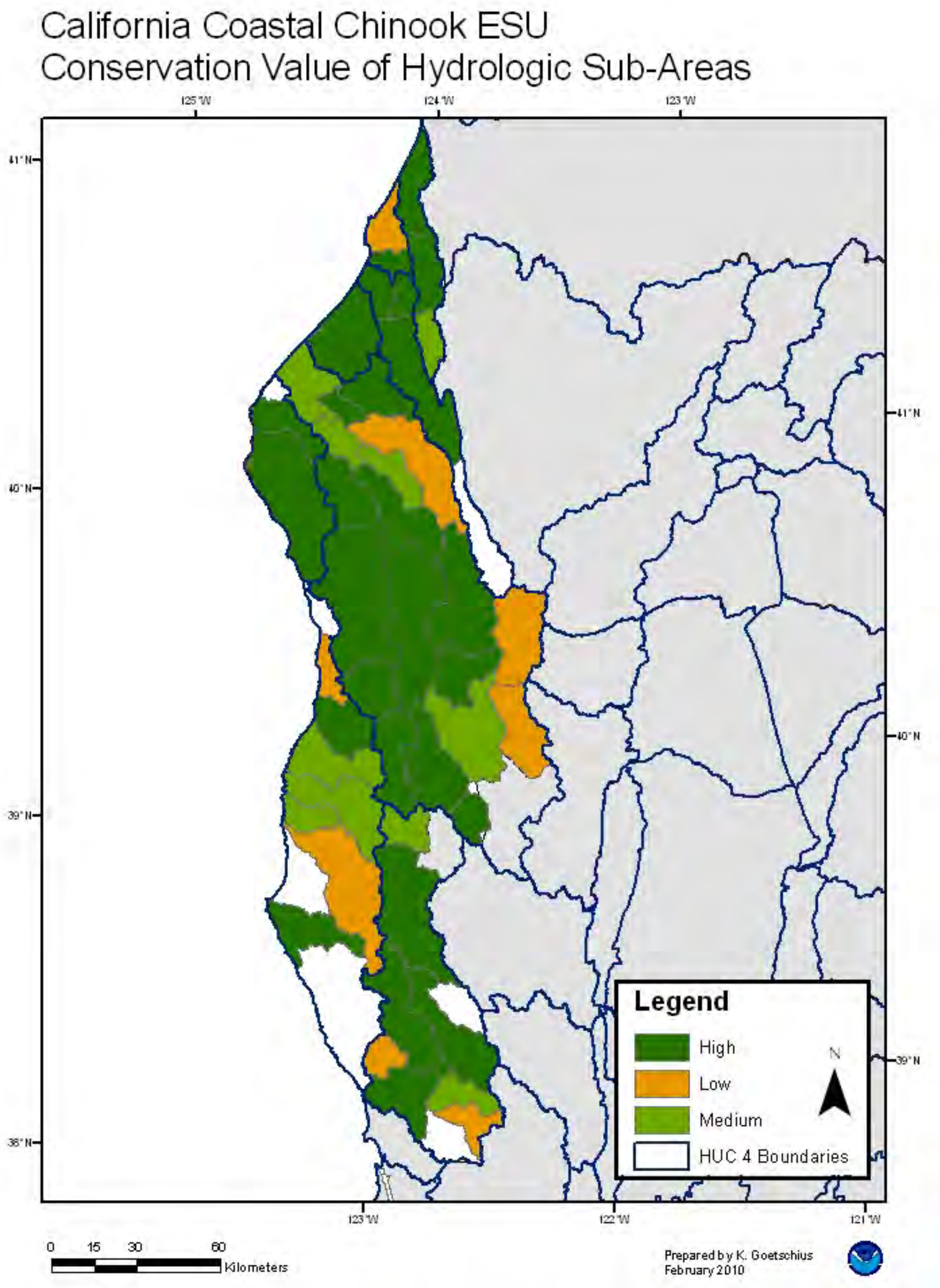


Figure 15. California Coastal Chinook salmon Conservation Values per Sub-Area.

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 16). 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 23 identifies populations within the CV Spring-run Chinook salmon ESU, their abundances, and hatchery input.

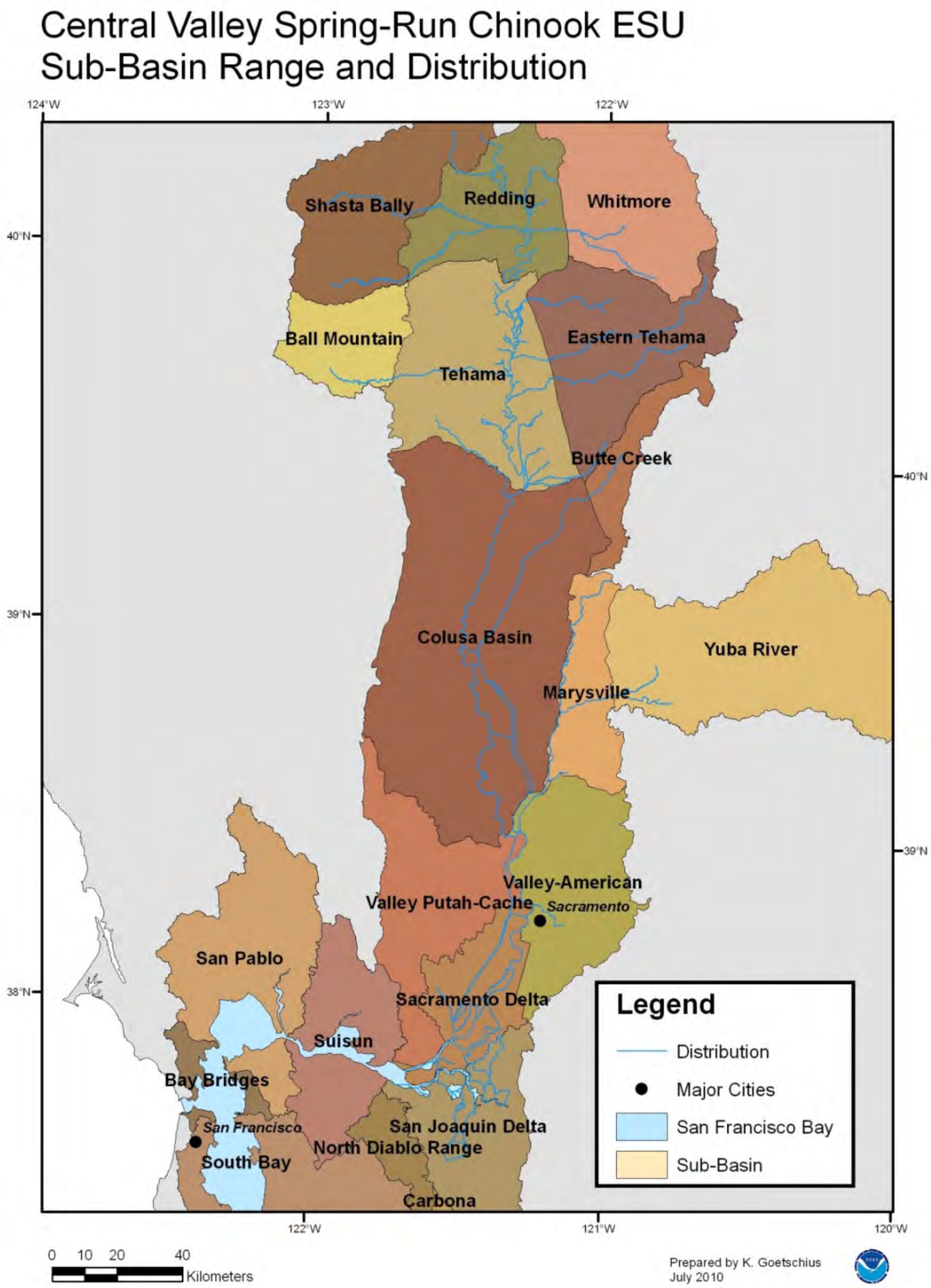


Figure 16. Central Valley Spring-run Chinook salmon distribution.

Life History

CV Spring-run Chinook salmon enter the Sacramento River from March to September and spawn from late August through early October, with a peak in September. Chinook salmon require cool fresh water while they mature over the summer. Adult upstream migration may be blocked by temperatures above 21ºC (McCullough, 1999). Fry emerge from the gravel November to March. Juvenile spring-run emigration in the Sacramento River is highly variable and they may migrate either as soon as they emerge from the gravel or as yearlings. The majority of spring-run fry emerging in the tributaries migrate downstream from December through February during high flows. Juvenile CV Spring-run Chinook salmon have been observed rearing in the lower reaches of non-natal tributaries and intermittent streams in the Sacramento Valley during the winter months. Peak fry/sub-yearling movements are observed farther downstream in lower Sacramento River (Knights Landing) and the Delta during March and April. Up to 25% of juveniles may remain in the tributaries to rear and outmigrate as yearlings the next fall, normally starting in December.

Status and Trends

NMFS originally listed CV Spring-run Chinook salmon as threatened on September 16, 1999 (64 FR 50393), and reaffirmed their threatened status on June 28, 2005 (70 FR 37160). Historically, spring-run Chinook salmon were predominant throughout the Sacramento and San Joaquin River drainages. All runs within the San Joaquin River basin are now extirpated. Naturally spawning populations of CV Spring-run Chinook salmon currently are restricted to accessible reaches of the upper mainstem Sacramento River and its tributaries Butte, Deer, and Mill Creeks. Limited spawning occurs in the basins of smaller tributaries (CDFG, 1998).

Table 23. Central Valley Spring-run Chinook salmon--preliminary population structure, historic and most recent natural production, spawner abundance, and hatchery contributions (T. P. Good, et al., 2005; USFWS & Reclamation, 2007).

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

1. Includes catches
2. *i.e.*, escapement

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

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

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

Critical Habitat

NMFS designated critical habitat for this species on September 2, 2005 (70 FR 52488). The critical habitat boundary includes the Sacramento River and several tributaries from the Big Chico tributary with Sacramento River upstream to Shasta Dam (Table 24).

There are 38 occupied HSA watersheds within the freshwater and estuarine range of this ESU. As shown in Figure 17, 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 24. 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.

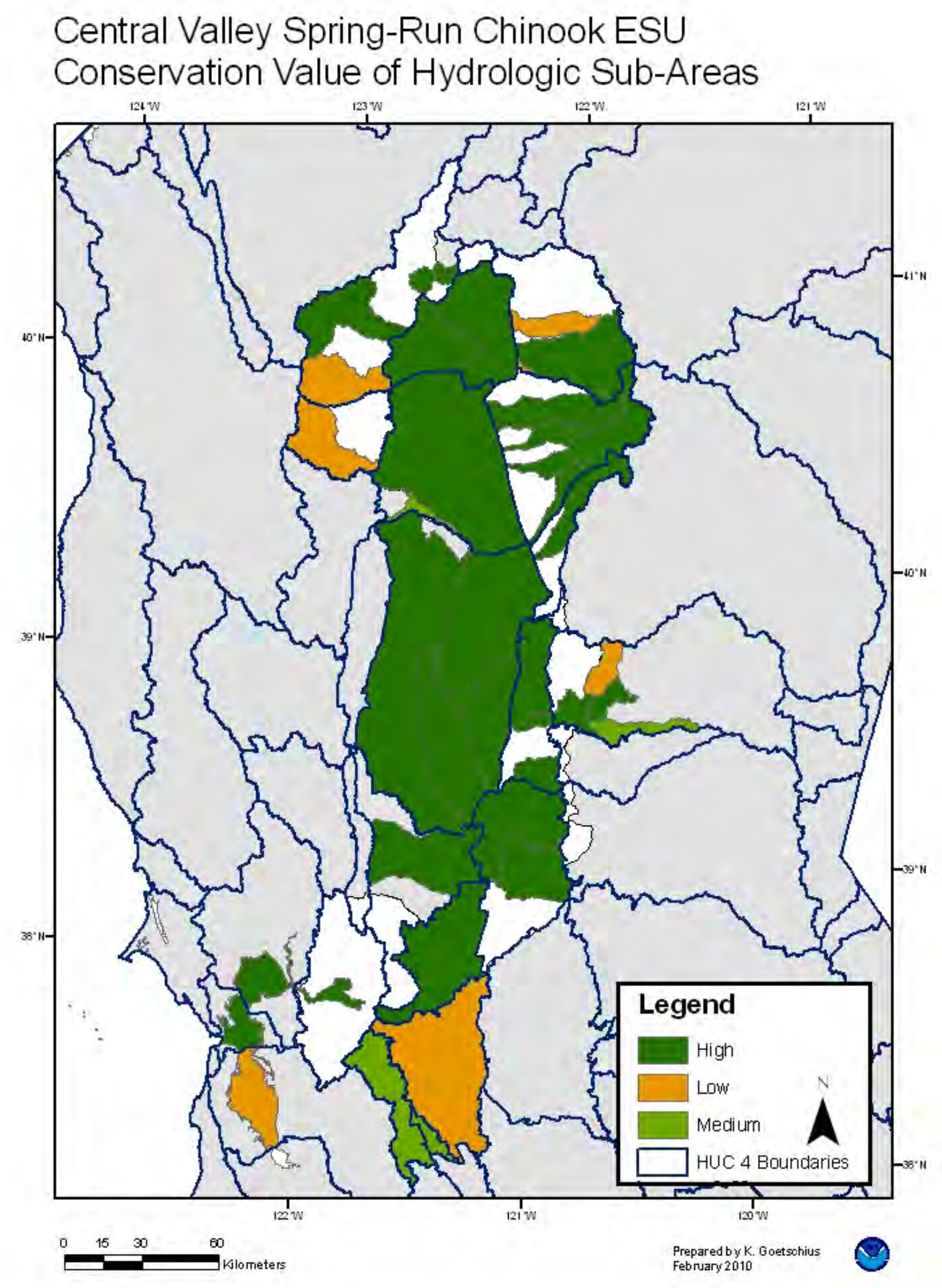


Figure 17. Central Valley Spring-run Chinook salmon Conservation Values per Sub-Area.

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 18). The ESU now consists of a single spawning population.

Life History

The winter-run Chinook salmon have characteristics of both stream- and ocean-type races (M.C. 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 (F. W. 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 (F. W. Fisher, 1994; J. M. Myers, et al., 1998). The winter-run race matures between two and six years of age with the majority returning as three-year olds.

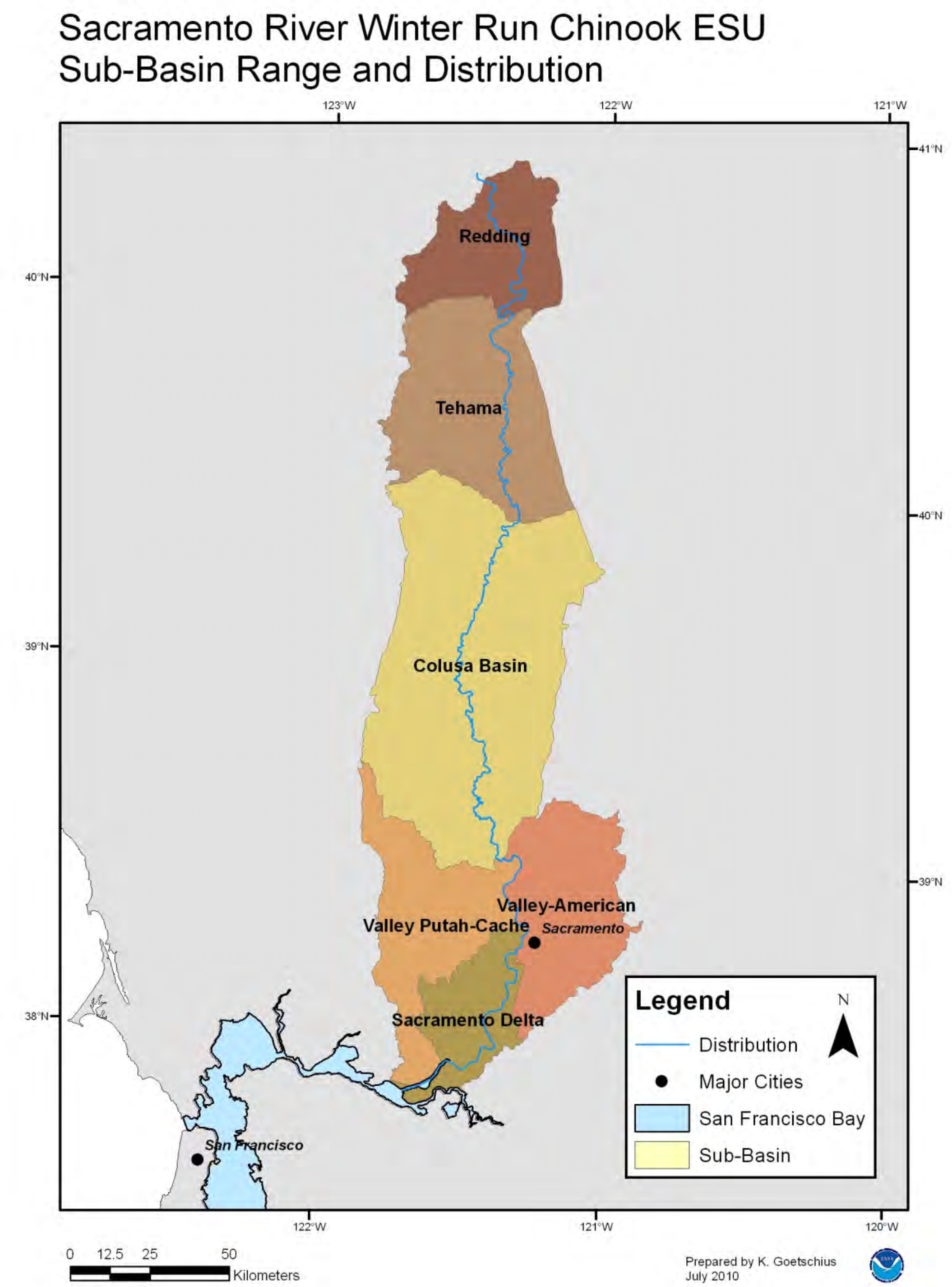


Figure 18. Sacramento River Winter-run Chinook salmon distribution.

Status and Trends

NMFS listed Sacramento River Winter-run Chinook salmon as endangered on January 4, 1994 (59 FR 440), and reaffirmed their endangered status on June 28, 2005 (70 FR 37160). The winter-run Chinook salmon spawned and reared in the upper Sacramento River and its tributaries (Slater, 1963; Yoshiyama, Gerstung, Fisher, & Moyle, 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 (F. W. 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 & 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.

Critical Habitat

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

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

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

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

Chum Salmon

Description of the Species

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

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

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

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

Status and Trends

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

Hood Canal Summer-run Chum Salmon

The Hood Canal (HC) Summer-run chum salmon ESU (Figure 19) 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 25 identifies populations within the HC Summer-run chum salmon ESU, their abundances, and hatchery input.

Table 25. Hood Canal Summer-run Chum salmon populations, abundances, and hatchery contributions (T. P. 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.

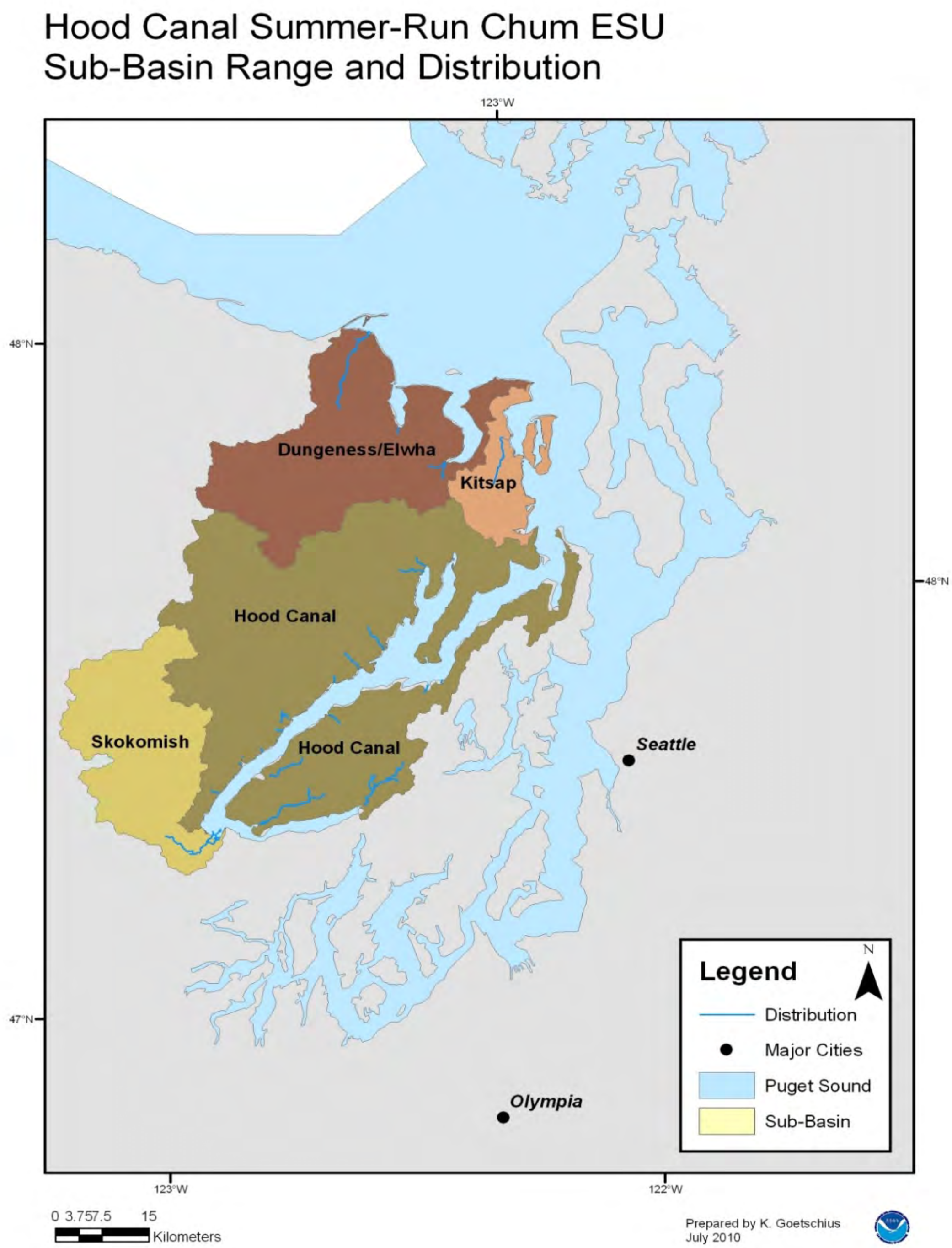


Figure 19. Hood Canal Summer-run Chum salmon distribution.

Life History

Run-timing data from as early as 1913 indicated temporal separation between summer- and fall-run chum salmon in Hood Canal (O. W. 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 (O. W. Johnson, et al., 1997). The juveniles rear for an average of 23 days in the subestuary deltas which support a diverse array of habitats (tidal channels, mudflats, marshes, and eelgrass meadows). These habitats provide essential rearing and transition environments for this ESU and juveniles rear in these habitats before entering the ocean. Fry in Hood Canal have not been observed to display daily tidal migrations (Bax, 1983). Fry movement is associated with prey availability. Juveniles feed primarily on plankton and epibenthic organisms, while subadults feed on similar items as well as larger prey (including fishes and squid).

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

Status and Trends

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

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

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

Canal populations have long-term trends above replacement; long-term lambda values range from 0.85 to 1.39 (T. P. Good, et al., 2005). Long-term trends in productivity are above replacement only for the Quilcene and Union River stocks.

Critical Habitat

Critical habitat for this species was designated on September 2, 2005 (70 FR 52630). Of 11 watersheds reviewed in NMFS' assessment of critical habitat for the Hood Canal Summer-run chum salmon ESU (Figure 20), nine watersheds were rated as having a high conservation value while three were rated as having a medium value for conservation (Table 26). 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 26. Hood Canal Summer-run chum salmon watersheds with conservation values.

HUC 4 Subbasin	HUC 5 Watershed conservation Value (CV)					
	High CV	PCE(s) [1]	Medium CV	PCE(s) [1]	Low CV	PCE(s) [1]
Skokomish	0		1	(1, 3)	0	
Hood Canal	6	(1, 3)	1	(1)[2]	0	
Kitsap	1	(1)	0		0	
Dungeness/Elwha	2	(1)	1	(3, 1)	0	
Total	9		3		0	

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

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

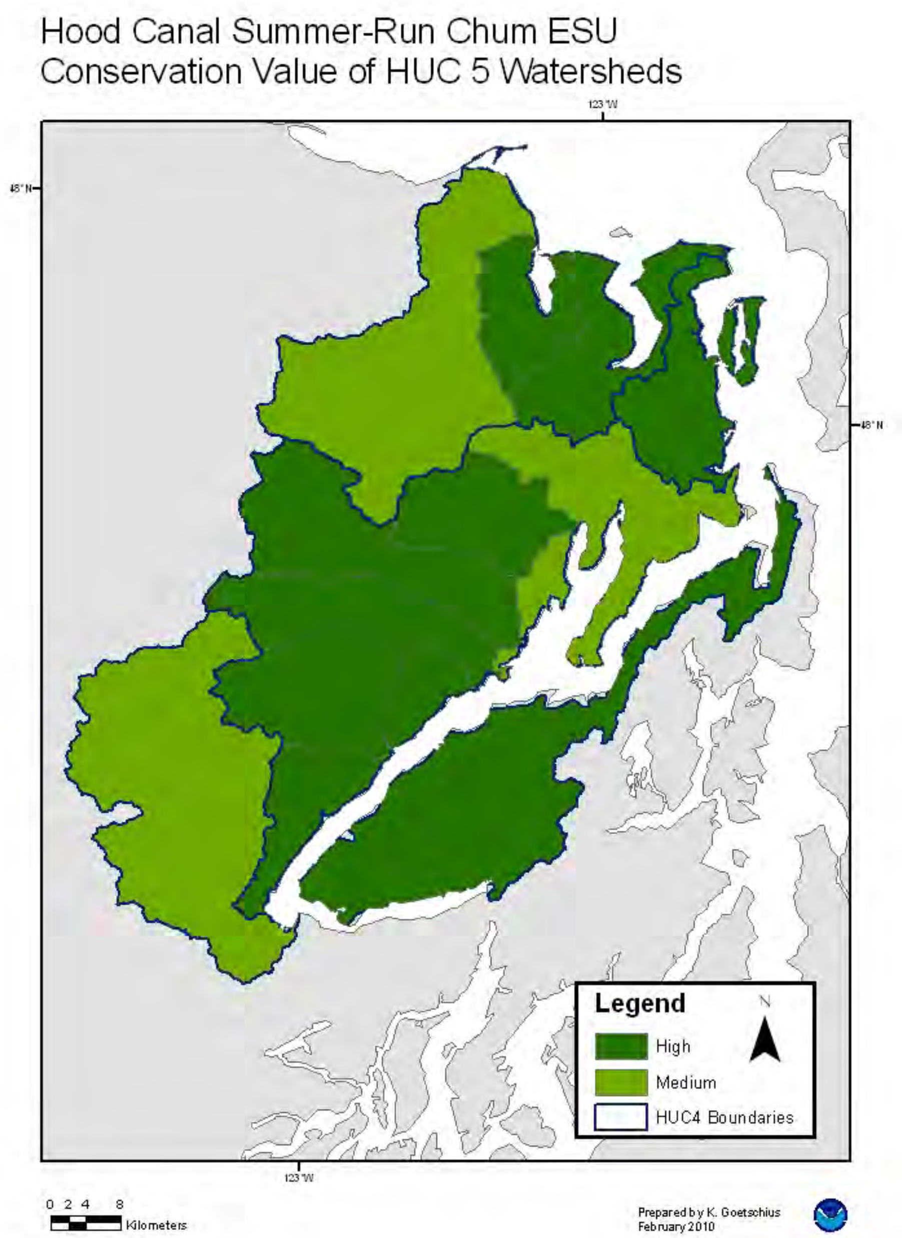


Figure 20. Hood Canal Summer-run Conservation Values per Sub-area.

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 21). This ESU also includes three artificial hatchery programs. These artificially propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within this ESU.

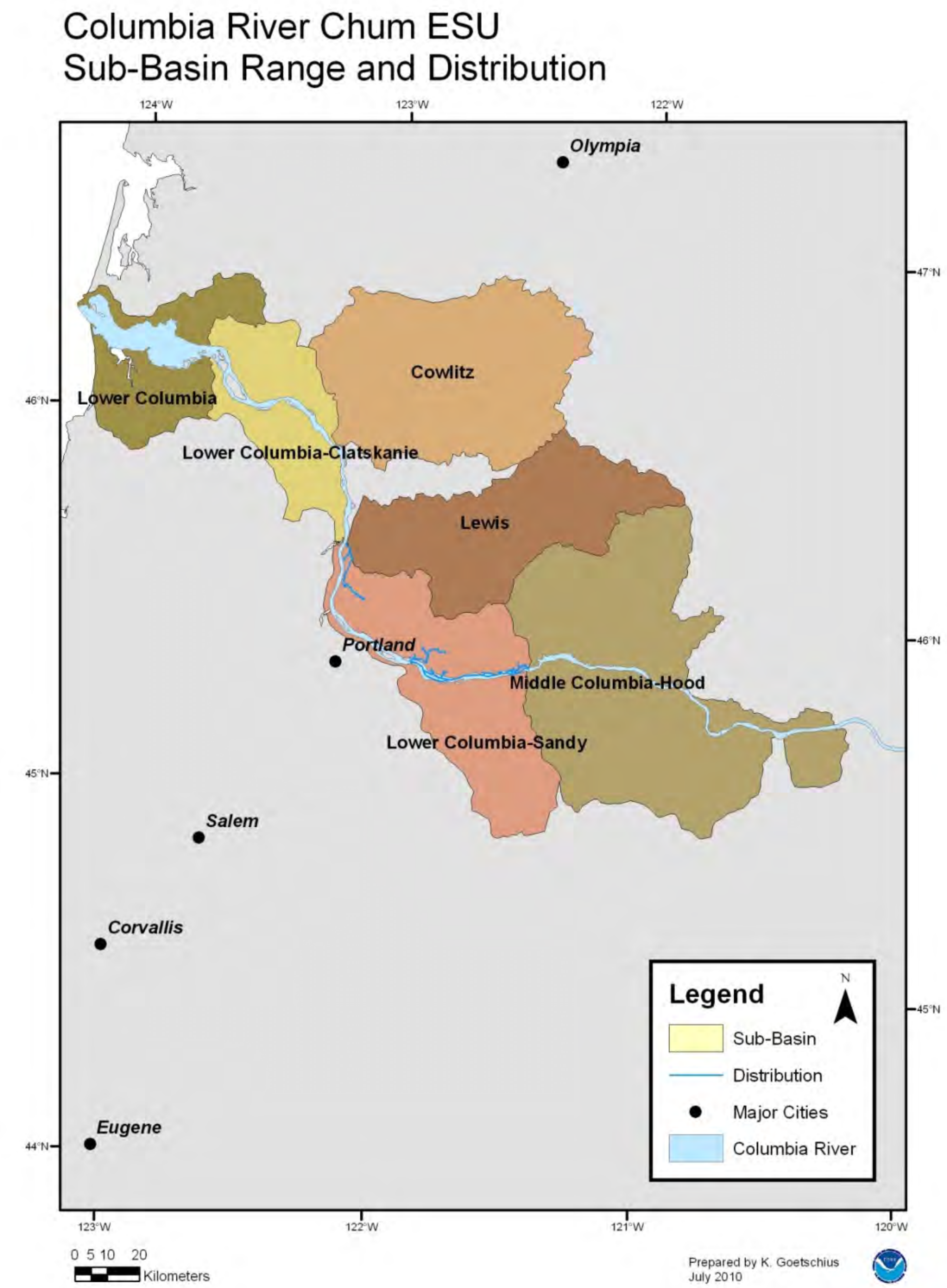


Figure 21. **Columbia River Chum salmon distribution.**

Table 27. Populations within the Columbia River chum salmon ESU, their abundances, and hatchery input (T. P. Good, et al., 2005).

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

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

Life History

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

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

Status and Trends

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

Historically, the ESU was composed of 17 populations in Oregon and Washington between the mouth of the Columbia River and the Cascade crest (J. Myers, et al., 2006) (Table 27). Only two populations with any significant spawning remain today, both on the Washington side (T. P. Good, et al., 2005). They are the Grays River and the Lower Gorge (which include Hardy and Hamilton Creeks) populations (T. P. 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 (T. P. 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 (T. P. 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 (T. P. Good, et al., 2005). A significant increase in spawner abundance occurred in 2001 and 2002 to around 10,000 adults (T. P. 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 (T. P. 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 (T. P. 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 (T. P. Good, et al., 2005). The 10-year trend (up to 2001) was negative for the Grays River population and just over 1.0 for the Lower Gorge. Long- and short-term productivity trends for populations are at or below replacement.

Critical Habitat

Critical habitat was originally designated for the CR chum salmon on February 16, 2000 (65 FR 7764) and was re-designated on September 2, 2005 (70 FR 52630). Sixteen of the 19 subbasins reviewed in NMFS' assessment of critical habitat for the CR chum salmon ESU were rated as having a high conservation value (Table 28). The remaining three subbasins were given a medium conservation value (Figure 22). Washington's federal lands were rated as having high conservation value to the species.

Table 28. CR chum salmon watersheds with conservation values.

HUC 4 Subbasin	HUC 5 Watershed conservation Value (CV)					
	High CV	PCE(s) [1]	Medium CV	PCE(s) [1]	Low CV	PCE(s) [1]
Middle Columbia/Hood	3	(3)	0		0	
Lower Columbia/Sandy	3	(3, 1)	0		0	
Lewis	2	(3)	0		0	
Lower Columbia/Clatskanie	3	(3, 2, 1)	0		0	
Cowlitz	3	(3)	3	(3)	0	
Lower Columbia	2	(3, 2, 1)	0		0	
Lower Columbia Corridor	all	(3, 1)	0		0	
Total	16		3		0	

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

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

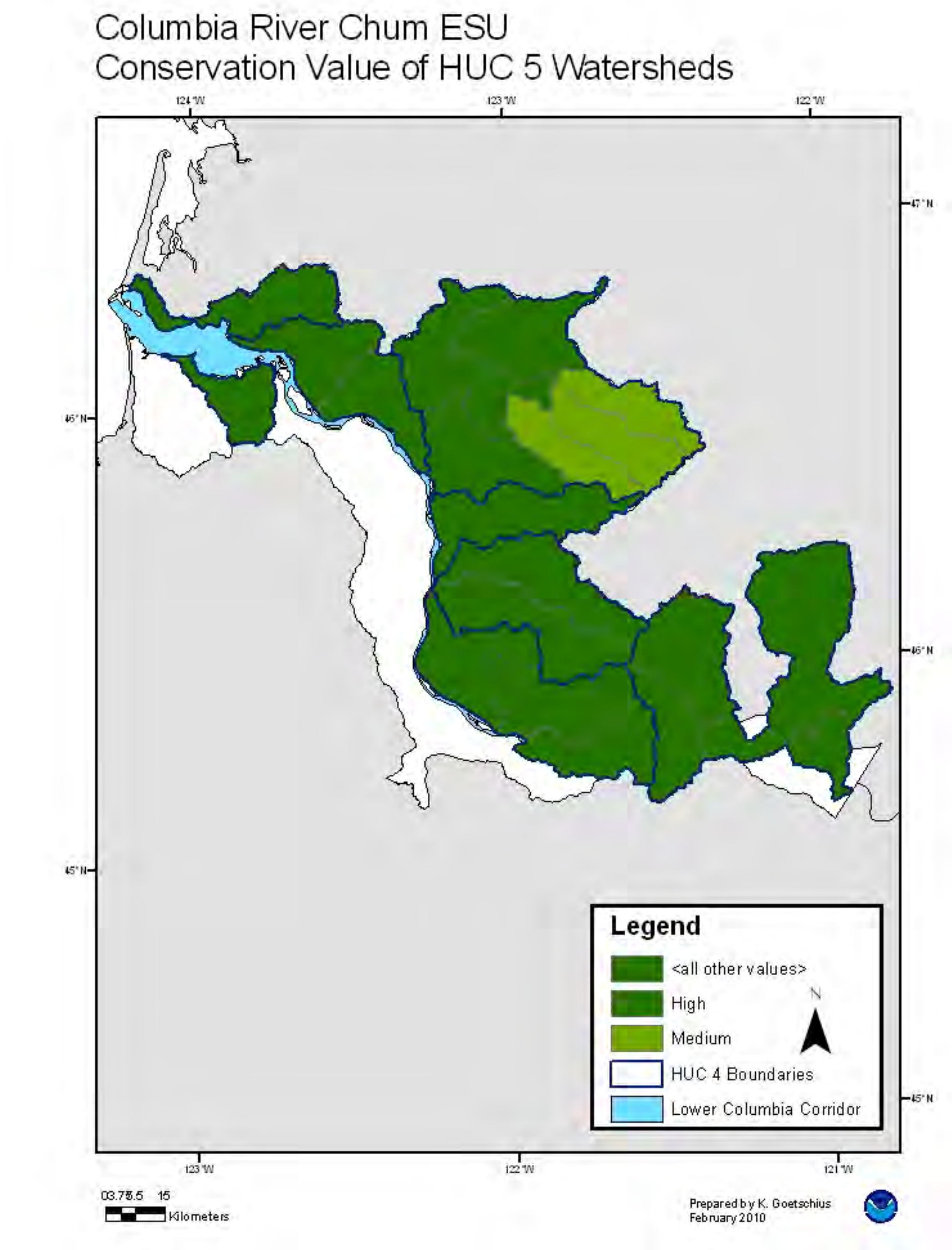


Figure 22. Columbia River Chum salmon Conservation Values per Sub-area.

Coho Salmon

Description of the Species

Coho salmon occur naturally in most major river basins around the North Pacific Ocean from central California to northern Japan (Laufle, Pauley, & Shepard, 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, Pess, & Beechie, 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, Reimers, & Hall, 1972). The likelihood of juvenile coho salmon occupying habitat that exceed 16.3°C maximum weekly average temperature declines significantly (Welsh, Hodgson, Roche, & Harvey, 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, Bjornn, & Meehan, 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, Caldwell, & DeMond, 2002; Fausch, 1984, 1993; J. Rosenfeld, Porter, & Parkinson, 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 & Healey, 1999; J. S. Rosenfeld, Leiter, Lindner, & Rothman, 2005). High prey availability also reduces territory size and may increase a stream's rearing capacity (Dill & Fraser, 1984; Dill, Ydenberg, & Fraser, 1981; Mason, 1976). Reduction in food availability reduces growth by subdominants and less for dominant fish (J. S. 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 (Robert E. Bilby & Bisson, 1987; R. E. Bilby & Bisson, 2001; Fausch & Northcote, 1992; Tschaplinski & Hartman, 1983). High densities of juvenile coho salmon also occur in log jams (G. T. Brown, 1985; Tschaplinski & 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 (G. T. Brown, 1985; Bustard & Narver, 1975a; Thomas E. Nickelson, Rodgers, Johnson, & Solazzi, 1992; N. P. Peterson, 1982; Tschaplinski & 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 (G. T. Brown, 1985; Olegario, 2006; Quinn & Peterson, 1996; Swales, Caron, Irvine, & Levings, 1988).

Availability of suitable overwintering habitat has been suggested to determine smolt production in streams (Bustard & Narver, 1975b; Thomas E. Nickelson, et al., 1992). Adult return or smolt production is related to the area of wetlands, lakes, and ponds within watersheds (Timothy J. Beechie, Beamer, & Wasserman, 1994; Pess et al., 2002; Sharma & 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 & Hilborn, 2003; Thorpe, 1994). Estuarine residence times can average one to three days (B. A. Miller & 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 (B. A. Miller & 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.

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).

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 23). This ESU also includes 25 artificial propagation programs (70FR 37160, June 28, 2005).

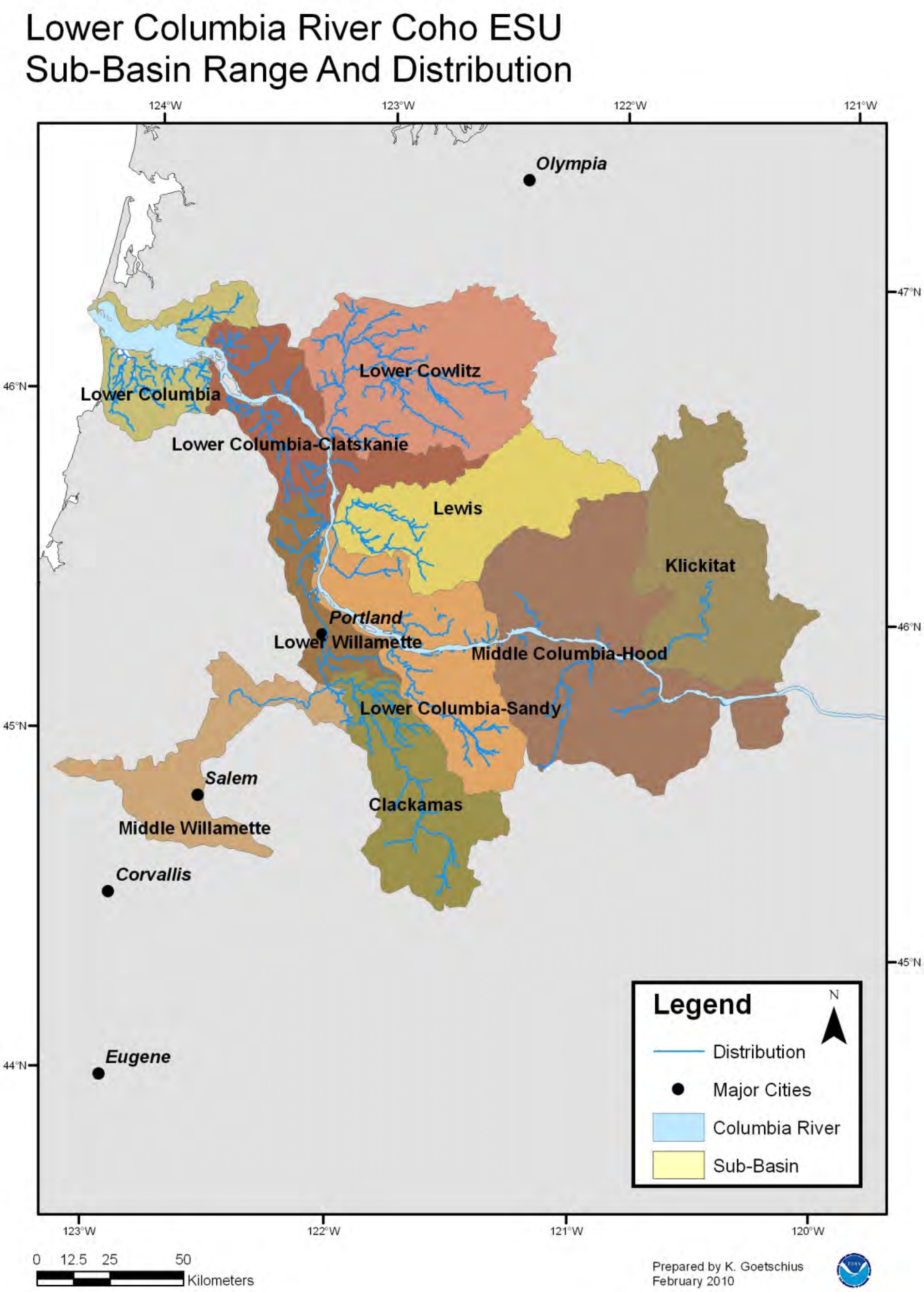


Figure 23. LCR coho salmon distribution.

Life History

The majority of the LCR coho salmon are of hatchery origin. Hatchery runs are currently managed for two distinct runs: early returning (Type S) and late returning (Type N) (O. W. Johnson, Flagg, Maynard, Milner, & Waknitz, 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 (T. P. Good, et al., 2005). The naturally produced coho salmon return to spawn between December and March (O. W. 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 (O. W. Johnson, et al., 1991).

Status and Trends

NMFS listed the LCR coho salmon as threatened on June 28, 2005 (70 FR 37160). The LCR coho salmon ESU historically consisted of 25 independent populations. The vast majority (over 90%) of these are either extirpated or nearly so (Table 29). 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 29. Lower Columbia River coho salmon populations, estimated natural spawner abundances, and hatchery contributions (T. P. 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 (O. W. 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 (T. P. 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 (T. P. Good, et al., 2005). The long-term trend (1977 to 2002) for this same population shows that the population has been decreasing (trend=0.54); there is a 43% probability that the median population growth rate (lambda) was less than one.

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

Critical Habitat

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

Oregon Coast Coho Salmon

The Oregon Coast (OC) coho salmon ESU includes all naturally spawned populations of coho salmon in Oregon coastal streams south of the Columbia River and north of Cape Blanco (63 FR 42587, August 10, 1998; Figure 24). One hatchery stock, the Cow Creek (ODFW stock # 37) hatchery coho, is included in the ESU. This artificially propagated population is no more divergent relative to the local natural populations than would be expected between closely related populations within this ESU.

Life History

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

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

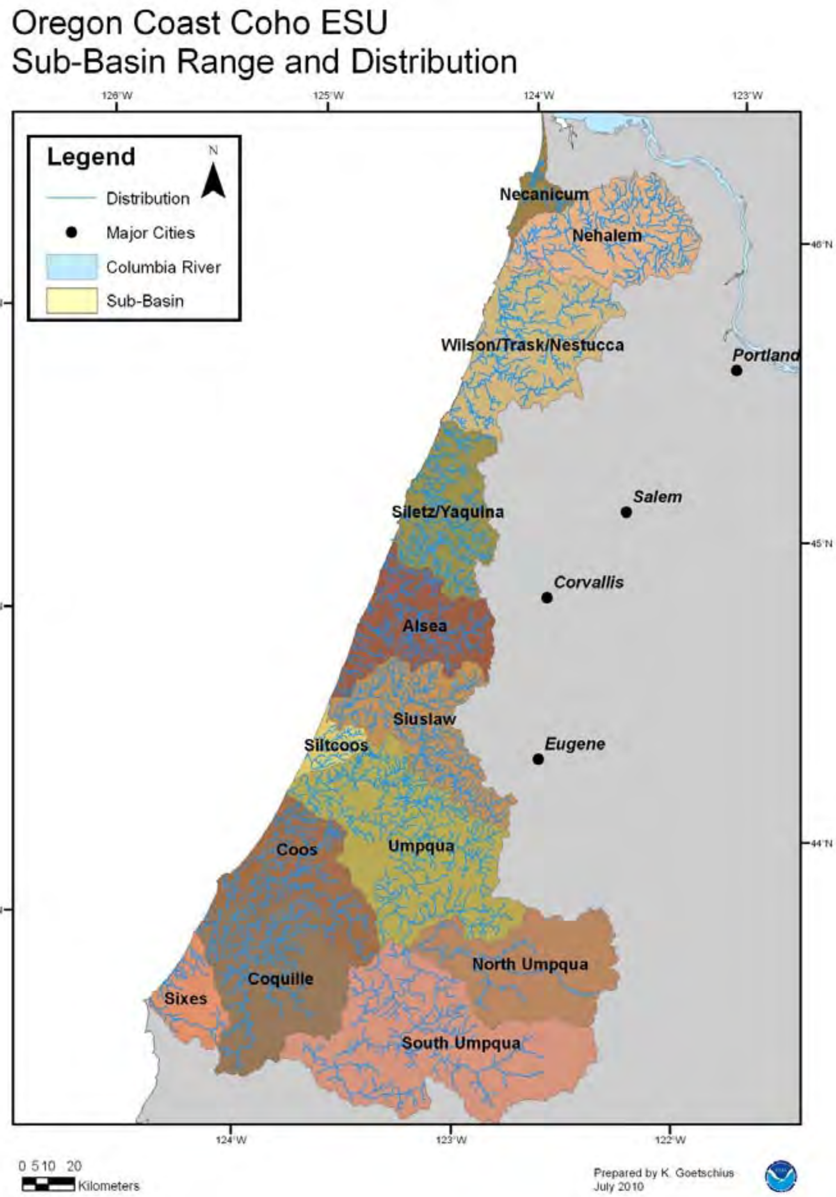


Figure 24. Oregon Coast Coho salmon distribution.

Status and Trends

NMFS listed the OC coho salmon as a threatened species on February 11, 2008 (73 FR 7816). Lawson *et al.* (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 30. Oregon Coast Coho salmon potential historic and estimated recent spawner abundances, and hatchery contributions (T. P. Good, et al., 2005; Lawson, et al., 2007).

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

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

period 2000 through 2008 with a range from a low of 66,169 to a high of 260,000 naturally produced spawners. Table 30 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 (T. P. 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 (T. P. 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 (T. P. 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 (T. P. 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).

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 31, and Figure 25).

Table 31. 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.

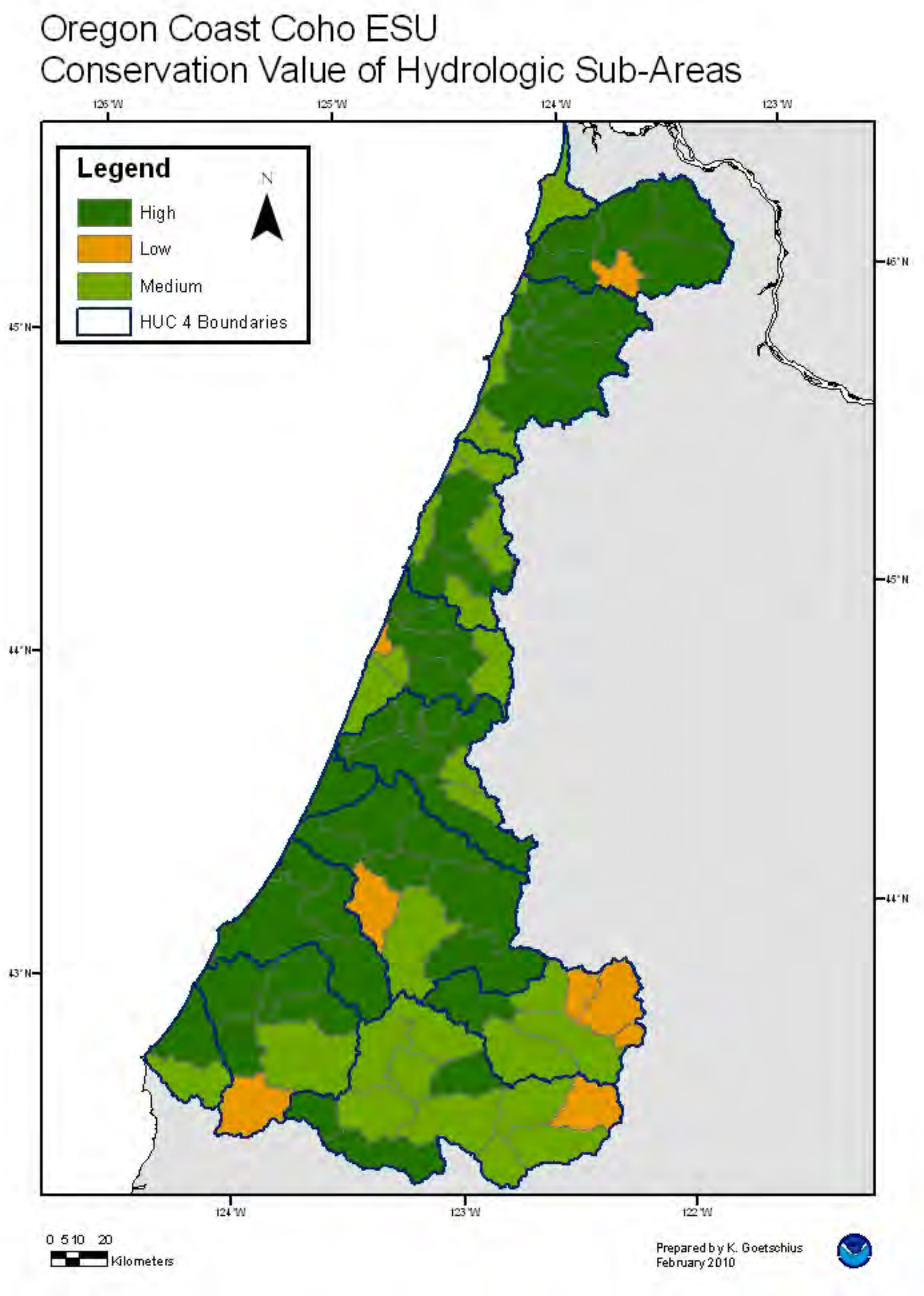


Figure 25. Oregon Coast Coho salmon Conservation Values per Sub-area.

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 26). This ESU also includes three artificial propagation programs. These artificially propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within this ESU.

Life History

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

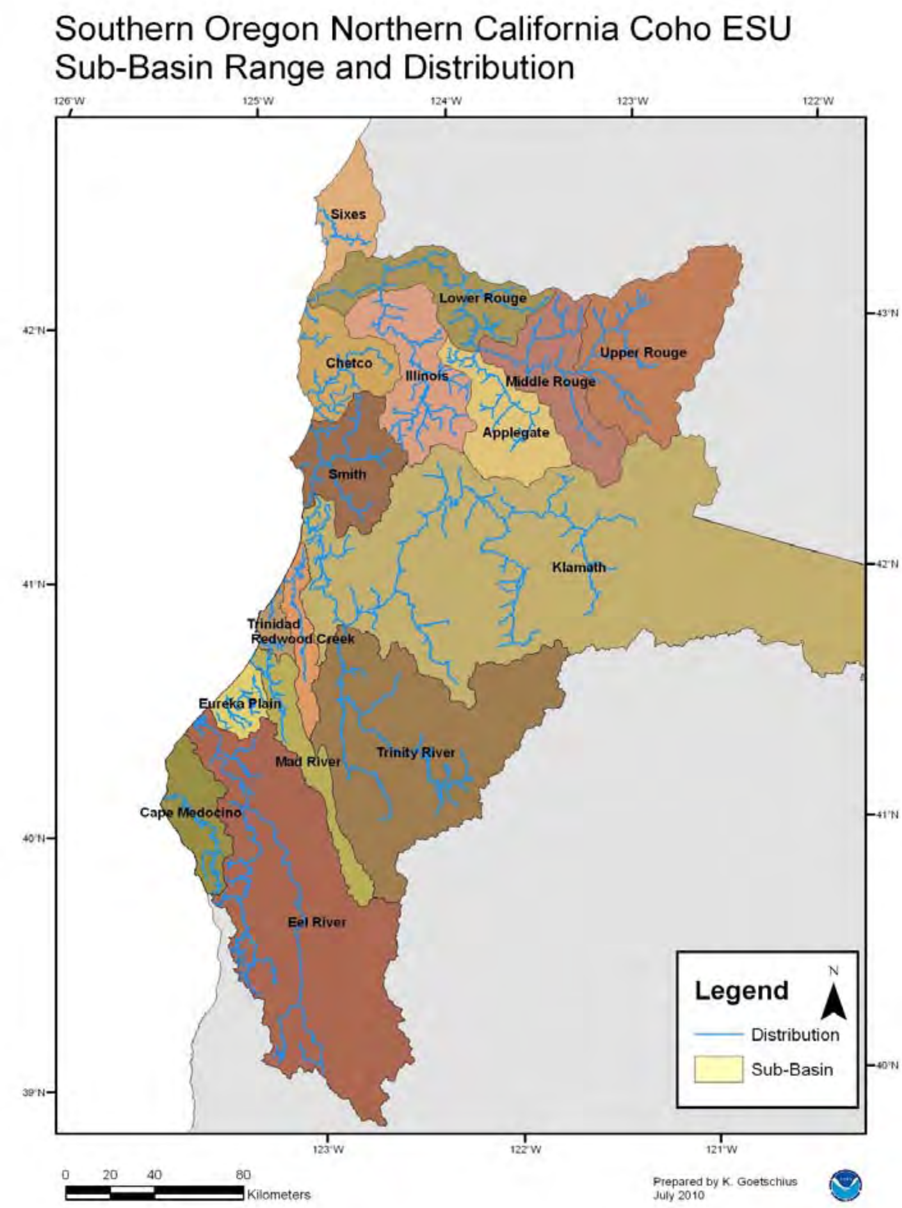


Figure 26. SONCC coho salmon distribution.

Status and Trends

NMFS listed SONCC coho salmon as threatened on May 7, 1997 (62 FR 24588), and reaffirmed their threatened status on June 28, 2005 (70 FR 37160). The ESU consists of three major basins: the Rough (OR), Klamath (OR/CA), and the Eel (CA) Rivers. Three historically independent interior populations have been identified for the Rough River basin, eight for the Klamath River basin, and six in the Eel River basin (Williams et al., 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.* (L. R. 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 (T. P. 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; T. P. Good, et al., 2005; Jepsen & Leader, 2008).

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

Critical Habitat

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

Critical habitat designated for the SONCC coho salmon is generally of good quality in northern coastal streams. Spawning PCE has been degraded throughout the ESU by logging activities that has increased fines in spawning gravel. Rearing PCE has been considerably degraded in many inland watersheds 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.

Central California Coast Coho Salmon

The Central California Coast (CCC) coho salmon ESU includes all naturally spawned populations of coho salmon from Punta Gorda in northern California south to and including the San Lorenzo River in central California, as well as populations in tributaries to San Francisco Bay, excluding the Sacramento-San Joaquin River system (Figure 27) The ESU also includes four artificial propagation programs. These artificially propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within this ESU.

Life History

In general, coho salmon within California exhibit a three-year life cycle. However, two-year old males commonly occur in some streams. Both run and spawn timing of coho salmon in this region are late (both peaking in January) relative to northern populations, with little time spent in fresh water between river entry and spawning. Spawning runs coincide with the brief peaks of river flow during the fall and winter. Most CCC coho salmon juveniles undergo smoltification and start their seaward migration one year after emergence from the redd. Juveniles spending two winters in fresh water have, however, been observed in at least one coastal stream within the range of the ESU (Bjorkstedt, et al., 2005). Smolt outmigration generally peaks in April and May (Shapovalov & Taft, 1954; Weitkamp et al., 1995).

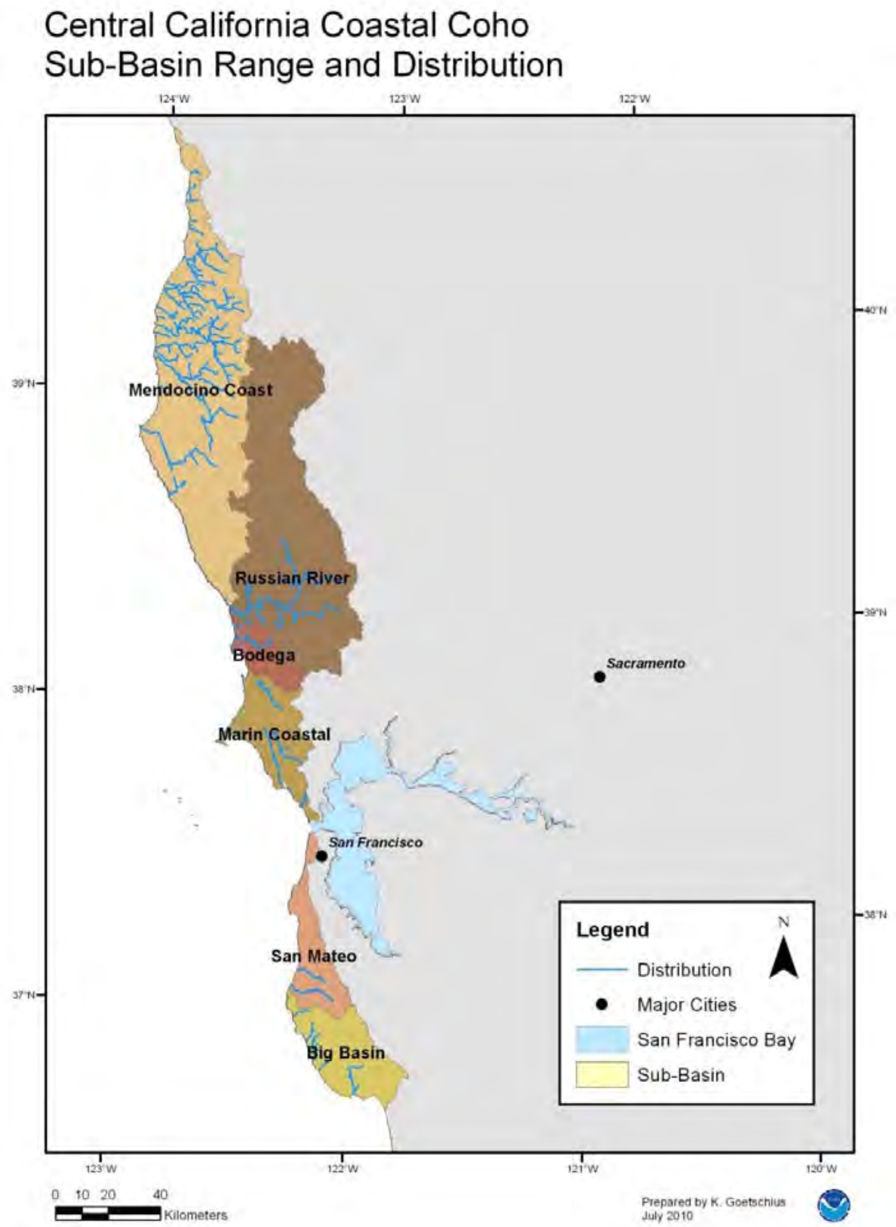


Figure 27. CCC Coho salmon distribution.

Status and Trends

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

Limited information exists on abundance of coho salmon within the CCC coho salmon ESU. About 200,000 to 500,000 coho salmon were produced statewide in the 1940s (T. P. 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) (T. P. 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 (T. P. Good, et al., 2005; Brian C. Spence, et al., 2008). There are no long-term time series of spawner abundance for individual river systems. Returns increased in 2001 in streams within the northern portion of the ESU (T. P. Good, et al., 2005). However, recent CCC coho salmon returns (2006/07 and 2007/08) have been discouragingly low (McFarlane, Hayes, & Wells, 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 32. Central California Coast Coho salmon populations, abundances, and releases of hatchery raised smolt (Bjorkstedt, et al., 2005; T. P. Good, 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 32). 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 (L. R. 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 (T. P. 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.

Critical Habitat

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

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

Sockeye Salmon

Description of the Species

Sockeye salmon occur in the North Pacific and Arctic oceans and associated freshwater systems. This species ranges south as far as the Klamath River in California and northern Hokkaido in Japan, to as far north as Bathurst Inlet in the Canadian Arctic and the

Anadyr River in Siberia. We discuss the distribution, life history diversity, status, and critical habitat of the two endangered and threatened sockeye species separately.

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

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

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

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

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

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

Status and Trends

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

Ozette Lake Sockeye Salmon

Distribution

This ESU includes sockeye salmon that migrate into and rear in the Ozette Lake near the northwest tip of the Olympic Peninsula in Olympic National Park, Washington (Figure 28). 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.

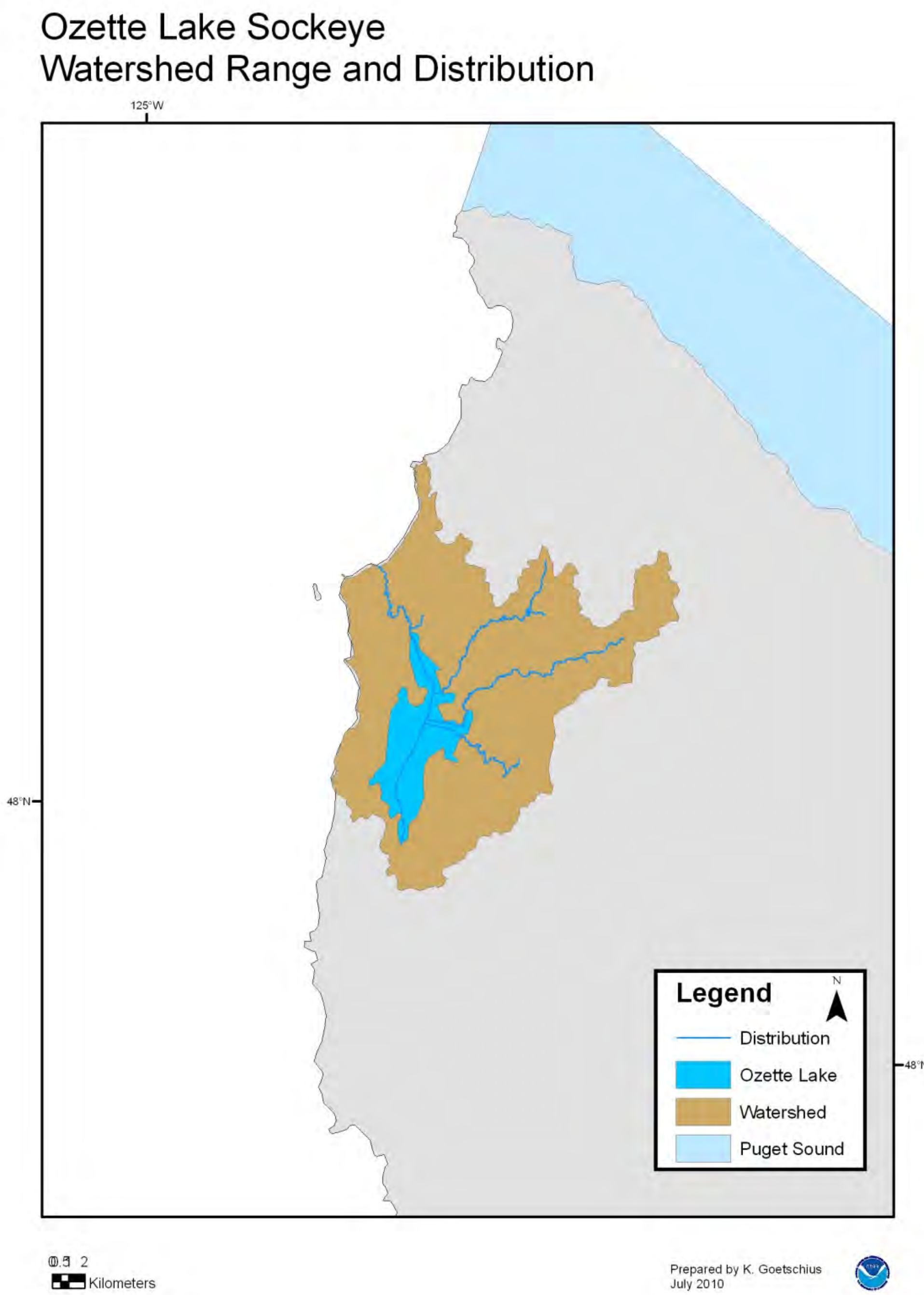


Figure 28. Ozette Lake Sockeye salmon distribution.

Life History

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

Egg incubation occurs from October through May. Emergence and dispersal in the lake occurs from late-February through May. Fry disperse to the limnetic zone in Ozette Lake, where the fish rear. Tributary fry also migrate to the lake soon after emergence. In their second spring after one year of rearing, Ozette Lake sockeye salmon emigrate seaward as age 1+ smolts. The lake is highly productive and water fleas dominate the diet. Sockeye salmon smolts produced in Ozette Lake are documented as the third largest, averaging 4 ½ to 5 inches in length, among west coast sockeye populations examined for average smolt size. The majority of Ozette Lake sockeye salmon return to spawn after two years in the ocean (NMFS, 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).

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 (T. P. 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, Jones, Delarm, & Waples, 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.

Critical Habitat

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

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

Snake River Sockeye Salmon

The Snake River (SR) sockeye salmon ESU includes all anadromous and residual sockeye from the Snake River basin, Idaho, as well as artificially propagated sockeye salmon from the Redfish Lake Captive Broodstock Program (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 29).

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, Myers, & Nielsen, 1994). The residents also spawn at the same time and in the same location as anadromous sockeye salmon.

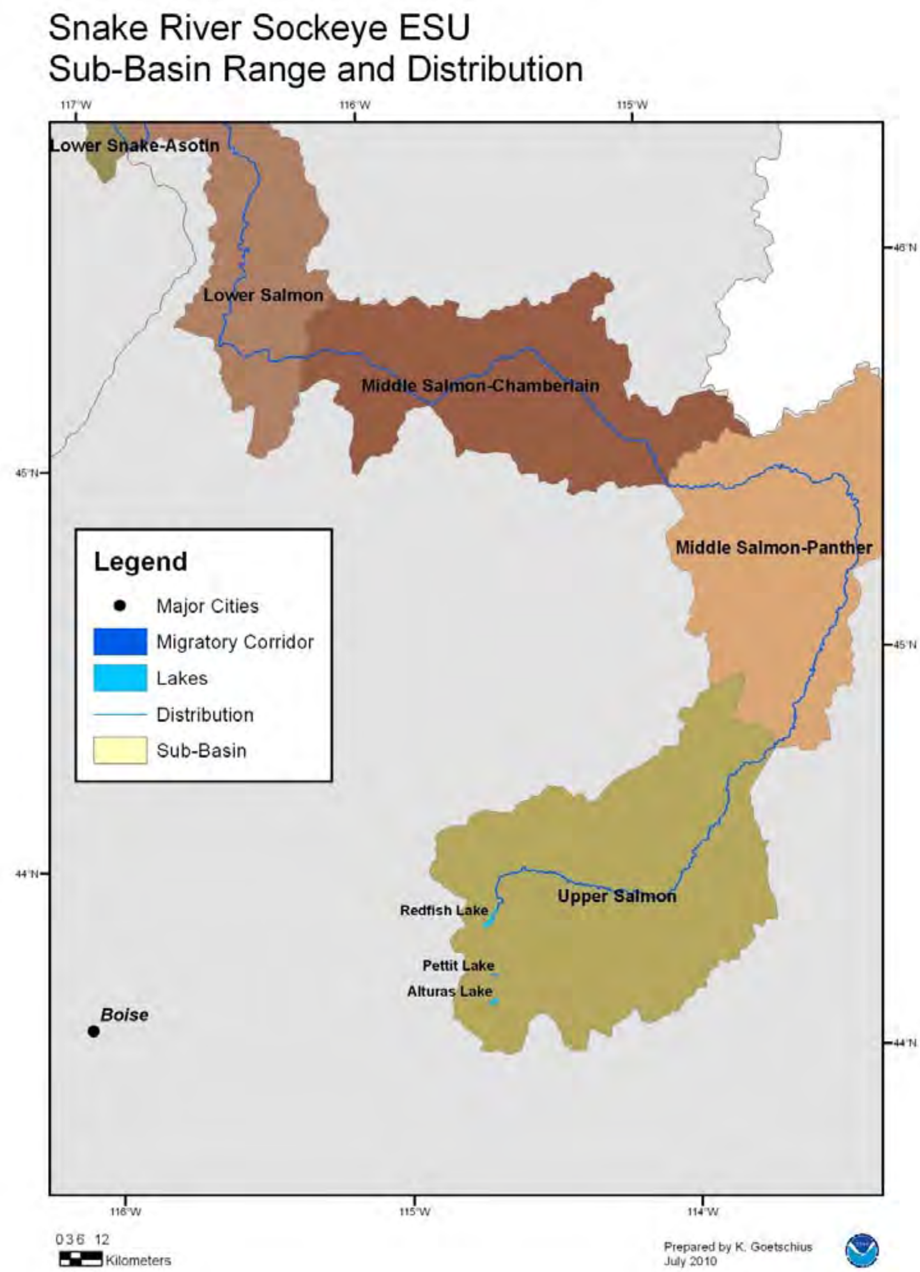


Figure 29. SR Sockeye Salmon distribution.

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.

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 (D. Chapman & 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 (D. Chapman & 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 (T. Bjornn, Craddock, & Corley, 1968). In 1985, 1986, and 1987, 11, 29, and 16 sockeye, respectively, were counted at the Redfish Lake weir (T. P. 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, Kline, Taki, & Flagg, 2004).

Critical Habitat

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

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

The quality and quantity of rearing and juvenile migration PCEs have been reduced 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.

Steelhead

Description of the Species

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

Steelhead have a protracted run time relative to Pacific salmon and do not tend to travel in large schools. Nevertheless, steelhead can be divided into two basic run-types: the stream-maturing type, or summer steelhead, and the ocean-maturing type, or winter steelhead. The summer steelhead enters fresh water in a sexually immature condition between May and October (Busby et al., 1996; T.E. Nickelson et al., 1992). 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; T.E. Nickelson, et al., 1992). Summer steelhead most commonly occur in streams where snowmelt contributes substantially to the annual hydrograph. The winter steelhead enters fresh water between November and April with well-developed gonads and spawns shortly after river entry (Busby, et al., 1996; T.E. Nickelson, et al., 1992). Variations in migration timing exist between populations. Some adults enter coastal streams in the spring, just before spawning (Meehan & 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 (T.E. Nickelson, et al., 1992). 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 (T.E. Nickelson, et al., 1992). As they grow, steelhead juveniles commonly occupy faster flowing water such as riffles. Older and larger juveniles are more risk averse; they stay in deeper water and keep close to cover (Peter A. Bisson, Nielsen, Palmson, & Grove, 1982; Peter A. Bisson, Sullivan, & Nielsen, 1988). Some older juveniles move downstream to rear in larger tributaries and mainstem rivers (T.E. Nickelson, et al., 1992).

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 (Peter A. 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 (Peter A. Bisson, et al., 1988; Kalleberg, 1958). Older steelhead commonly uses deeper pools (Peter A. Bisson, et al., 1982; Peter A. Bisson, et al., 1988).

Juvenile steelhead are opportunistic and feed on a wide variety of aquatic and terrestrial insects (D. W. Chapman & 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 & 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 & Dell, 1986; T.E. Nickelson, et al., 1992). Steelhead typically reside in marine waters for two or three years prior to returning to their natal stream to spawn as four or five-year olds.

Status and Trends

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

Puget Sound Steelhead DPS

This DPS includes all naturally spawned anadromous winter-run and summer-run steelhead in streams in the river basins of the Strait of Juan de Fuca, Puget Sound, and Hood Canal, Washington, bounded to the west by the Elwha River (inclusive) and to the north by the Nooksack River and Dakota Creek (inclusive), as well as the Green River natural and Hamma Hamma winter-run steelhead hatchery stocks (Figure 30). The remaining hatchery programs are not considered part of the DPS because they are more than moderately diverged from the local native populations.

Life History

The Puget Sound steelhead DPS contains both winter-run and summer-run steelhead. Adult winter-run steelhead generally return to Puget Sound tributaries from December to April (NMFS, 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.

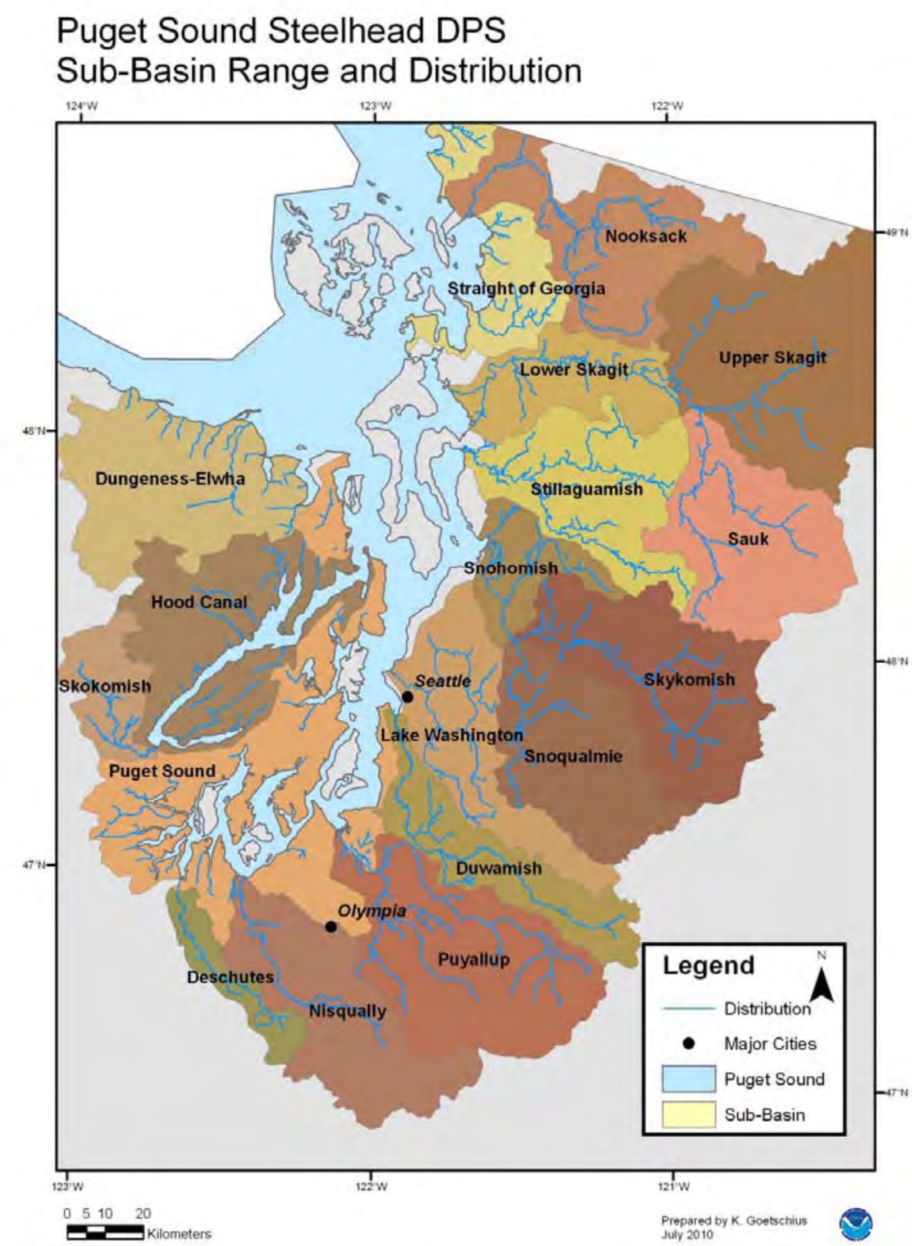


Figure 30. Puget Sound steelhead distribution.

Status and Trends

NMFS listed Puget Sound steelhead as threatened on May 11, 2007 (72 FR 26722). Fifty-three populations of steelhead have been identified in this DPS, of which 37 are winter-run. Summer-run populations are distributed throughout the DPS but are concentrated in northern Puget Sound and Hood Canal; only the Elwha River and Canyon Creek support summer-run steelhead in the rest of the DPS. The Elwha River run, however, is descended from introduced Skamania Hatchery summer-run steelhead. Historical summer-run steelhead in the Green River and Elwha River were likely extirpated in the early 1900s. Table 33 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

Table 33. 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)

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.

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, Leider, Hulett, Baker, & Johnson, 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, Marshall, & Phelps, 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.

Critical Habitat

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

Lower Columbia River Steelhead

The LCR steelhead DPS includes all naturally spawned steelhead populations below natural and manmade impassable barriers in streams and tributaries to the Columbia River between the Cowlitz and Wind Rivers, Washington (inclusive), and the Willamette and Hood Rivers, Oregon (inclusive) (Figure 31). Two hatchery populations are included in this species, the Cowlitz Trout Hatchery winter-run population and the Clackamas River population but neither was listed as threatened.

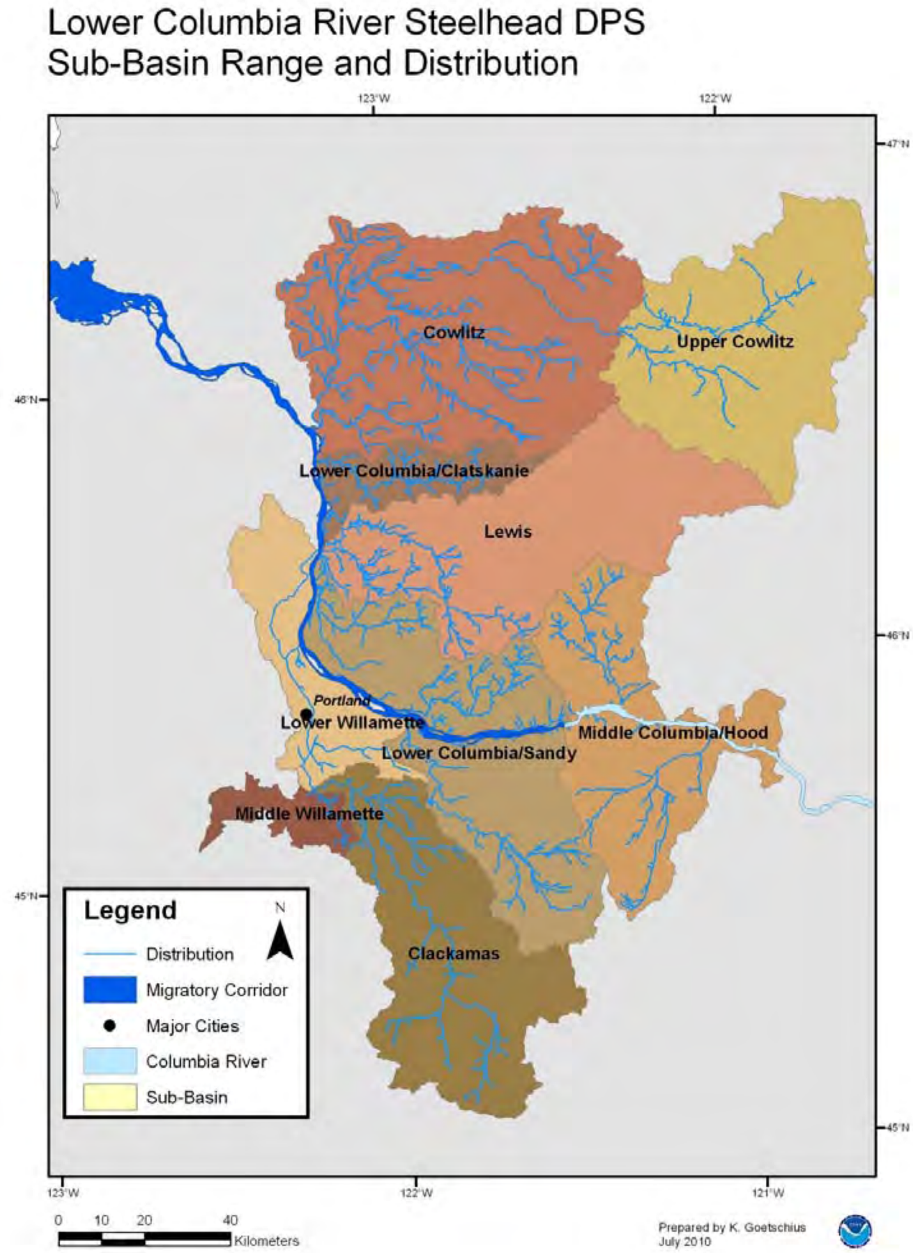


Figure 31. Lower Columbia River steelhead distribution.

Life History

The LCR steelhead DPS includes both summer- and winter-run stocks (Table 34). Summer-run steelhead return sexually immature to the Columbia River from May to

November, and spend several months in fresh water prior to spawning. Winter-run steelhead enter fresh water from November to April, are close to sexual maturation during freshwater entry, and spawn shortly after arrival in their natal streams. Where both races spawn in the same stream, summer-run steelhead tend to spawn at higher elevations than the winter-run.

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

Status and Trends

NMFS listed LCR steelhead as threatened on March 19, 1998 (63 FR 13347), and reaffirmed their threatened status on January 5, 2006 (71 FR 834). The LCR steelhead had 17 historically independent winter steelhead populations and 6 independent summer steelhead populations (McElhany et al., 2003; J. 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

Table 34. LCR Steelhead salmon populations, historic abundances (T. P. Good, et al., 2005), 1998 – 2002 and 2004 to 2005 geometric mean abundance (T. P. Good, et al., 2005)(Salmon Scape Query 2009), and hatchery contributions (T. P. Good, et al., 2005; McElhany, et al., 2003).

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%

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 (T. P. Good, et al., 2005; McElhany, et al., 2007). The Gorge summer-run steelhead is at the highest risk over the long-term as the Hood River population is at high risk of being lost (McElhany, et al., 2007).

Critical habitat

Critical habitat was designated for the LCR steelhead on September 2, 2005 (70 FR 52488). Of 41 subbasins listed as critical habitat for the LCR steelhead, 28 subbasins were rated as having a high conservation value. Eleven subbasins were rated as having a medium value and two were rated as having a low value to the conservation of the DPS (Table 35).

Table 35. 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 32).

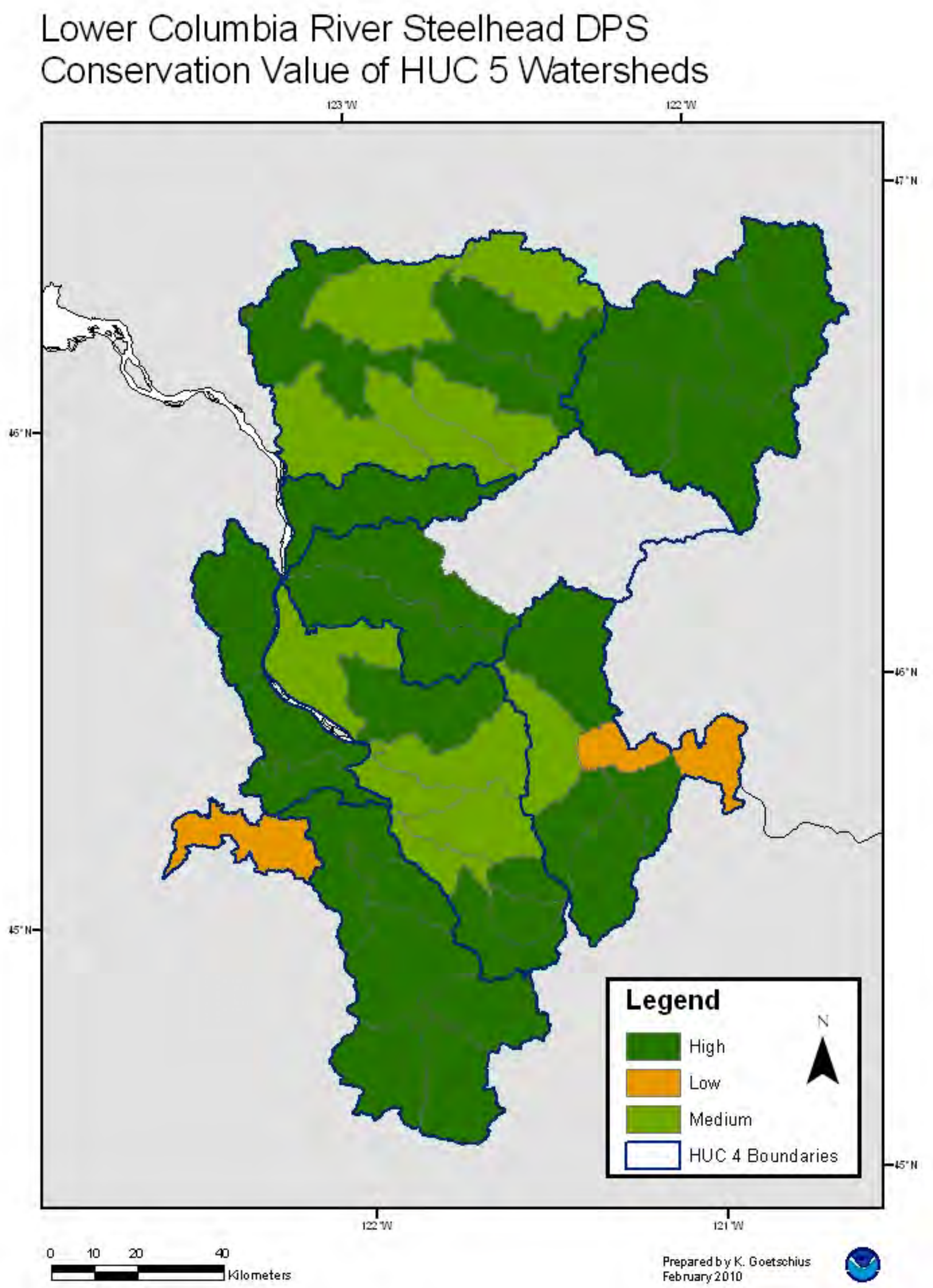


Figure 32. Lower Columbia River Steelhead Conservation Values per Sub-area.

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 33). No artificially propagated populations that reside within the historical geographic range of this DPS are included in this listing. Hatchery summer-run steelhead occur in the Willamette Basin but are an out-of-basin population that is not included in this DPS.

Life History

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

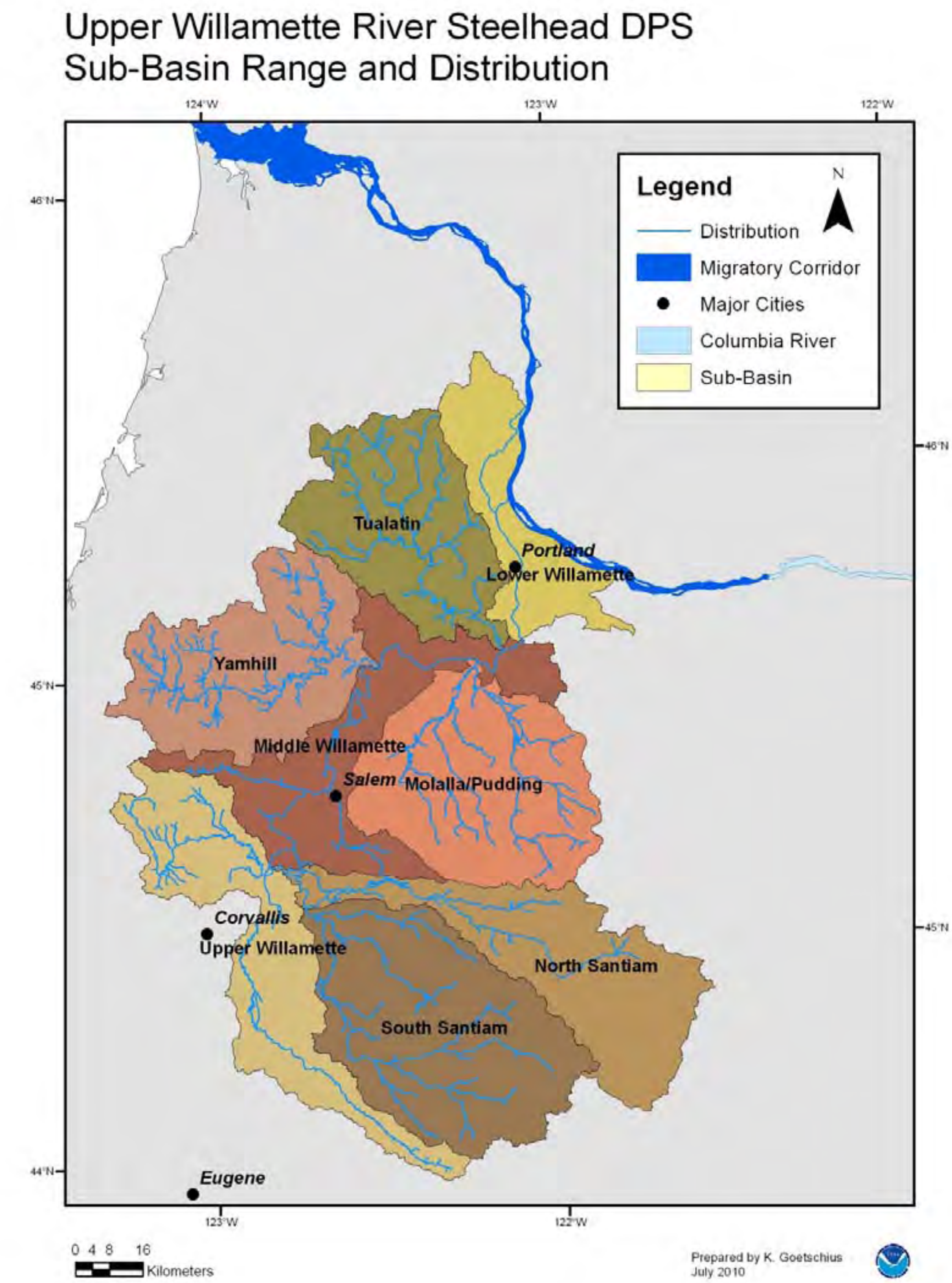


Figure 33. UWR Steelhead distribution.

Status and Trends

NMFS originally listed UWR steelhead as threatened on March 25, 1999 (64 FR 14517), and reaffirmed their threatened status on January 5, 2006 (71 FR 834). Four basins on the east side of the Willamette River historically supported independent populations for the UWR steelhead, all of which remain extant. Data reported in McElhaney et al. (2007) indicate that currently the two largest populations within the DPS are the Santiam River populations. 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 36. Upper Willamette River steelhead salmon populations, core (C) and genetic legacy (G) populations, abundances, and hatchery contributions (T. P. Good, et al., 2005; McElhany, et al., 2003).

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 (T. P. 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 36). Nevertheless, remains of the early run still exist and the release of hatchery fish has been discontinued.

Critical Habitat

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

Table 37. 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 37).

The current condition of critical habitat designated for the UWR steelhead is degraded (Figure 34), 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.

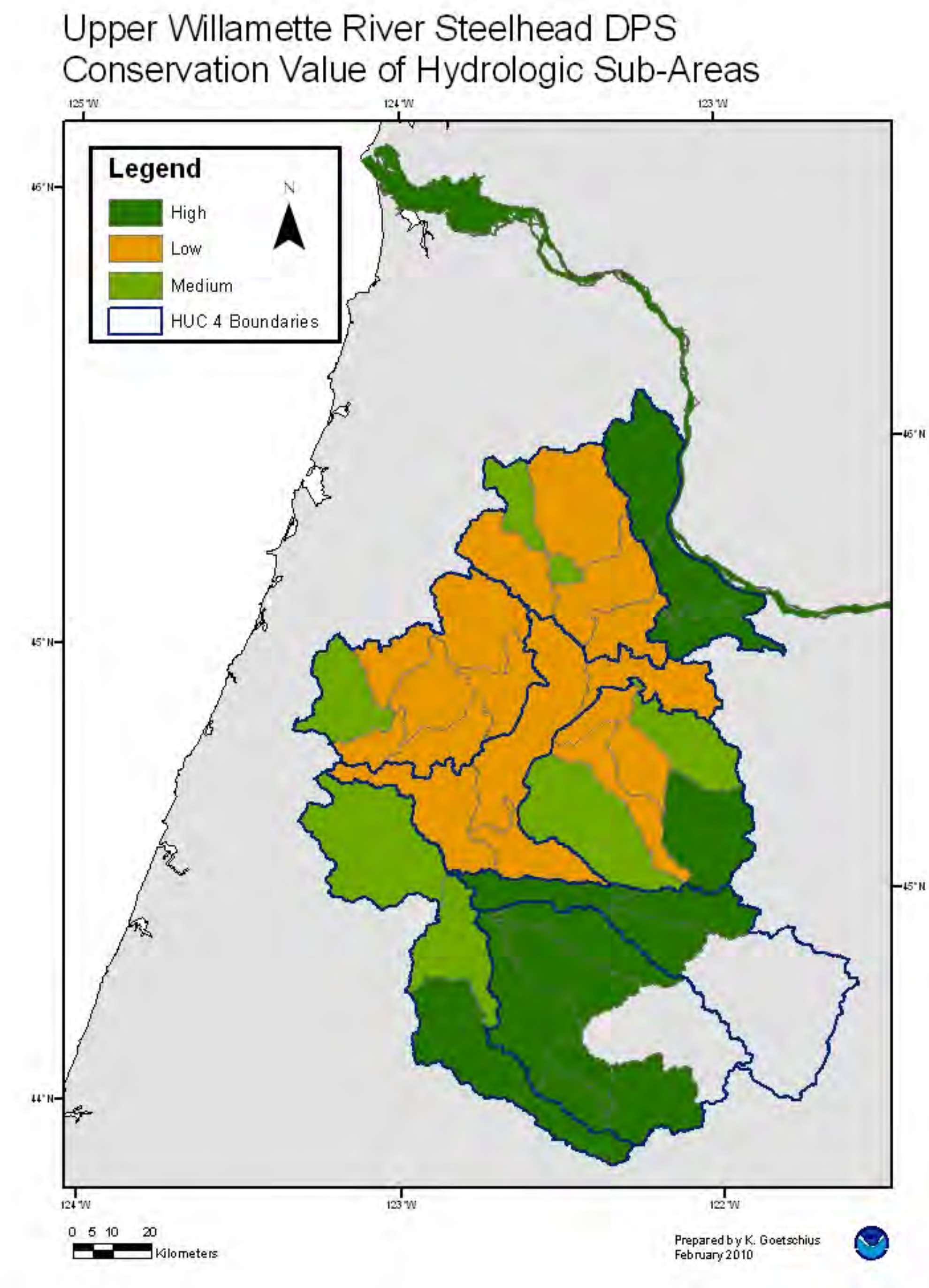


Figure 34. Upper Willamette River Steelhead Conservation Values per Sub-Area.

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 35). 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).

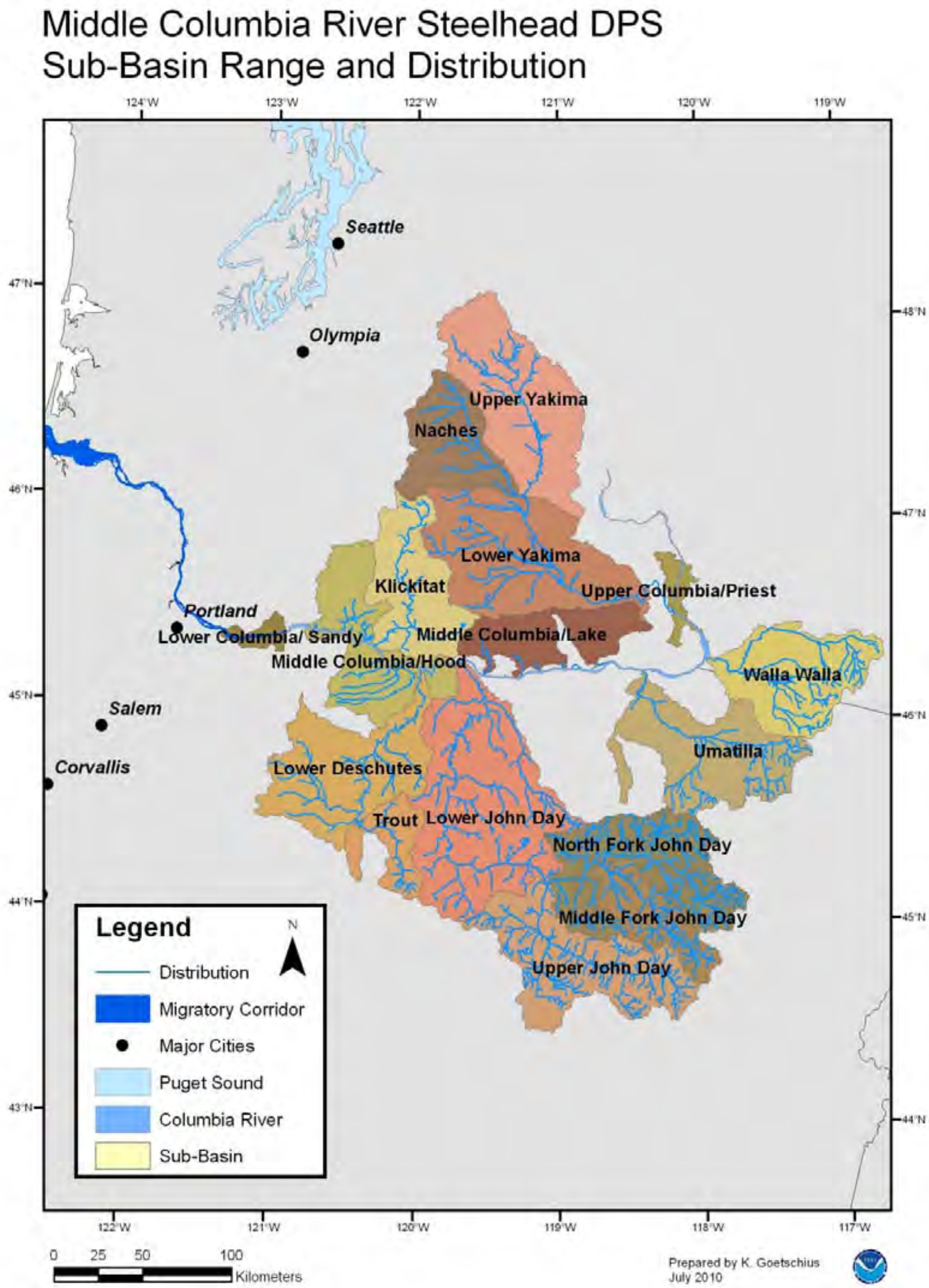


Figure 35. MCR Steelhead distribution.

Life History

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

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

Status and Trends

NMFS listed MCR steelhead as threatened on March 25, 1999 (64 FR 14517), and reaffirmed their threatened status on January 5, 2006 (71 FR 834). The 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.

Table 38. Middle Columbia River steelhead independent populations, abundances, and hatchery contributions (T. P. Good, et al., 2005; ICTRT, 2003).

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

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

Historic run estimates for the Yakima River imply that annual species abundance may have exceeded 300,000 returning adults (Busby, et al., 1996). The five-year average (geometric mean) return of natural MCR steelhead for 1997 to 2001 was up from previous years' basin estimates. Returns to the Yakima River, the Deschutes River, and sections of the John Day River system were substantially higher compared to 1992 to 1997 (T. P. Good, et al., 2005). The five-year average for these basins is 298 and 1,492 fish, respectively (T. P. Good, et al., 2005).

Good *et al.* (2005) calculated that the median estimate of long-term trend over 12 indicator data sets was –2.1% per year (–6.9 to 2.9), with 11 of the 12 being negative. Long-term annual population growth rates (λ) were also negative (T. P. 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 (T. P. 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 (T. P. Good, et al., 2005). The Cascade populations are at risk by the only population with large runs being dominated by out-of-basin strays (T. P. 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 (T. P. 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 (T. P. Good, et al., 2005).

Table 38 summarizes MCR steelhead independent populations, abundances and hatchery contributions(T. P. Good, et al., 2005; ICTRT, 2003). 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 (T. P. Good, et al., 2005).

Critical Habitat

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

The CHART assessment for this DPS addressed 15 (HUC4) subbasins containing 106 occupied watersheds (HUC5), as well as the Columbia River rearing/migration corridor (NMFS, 2005a). Of all the watersheds, 73 were rated as having a high conservation value, 24 as medium value, and 9 as low value (Table 39). The lower Columbia River rearing/migration corridor downstream of the spawning range is also considered to have a high conservation value.

Table 39. 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 36). 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.

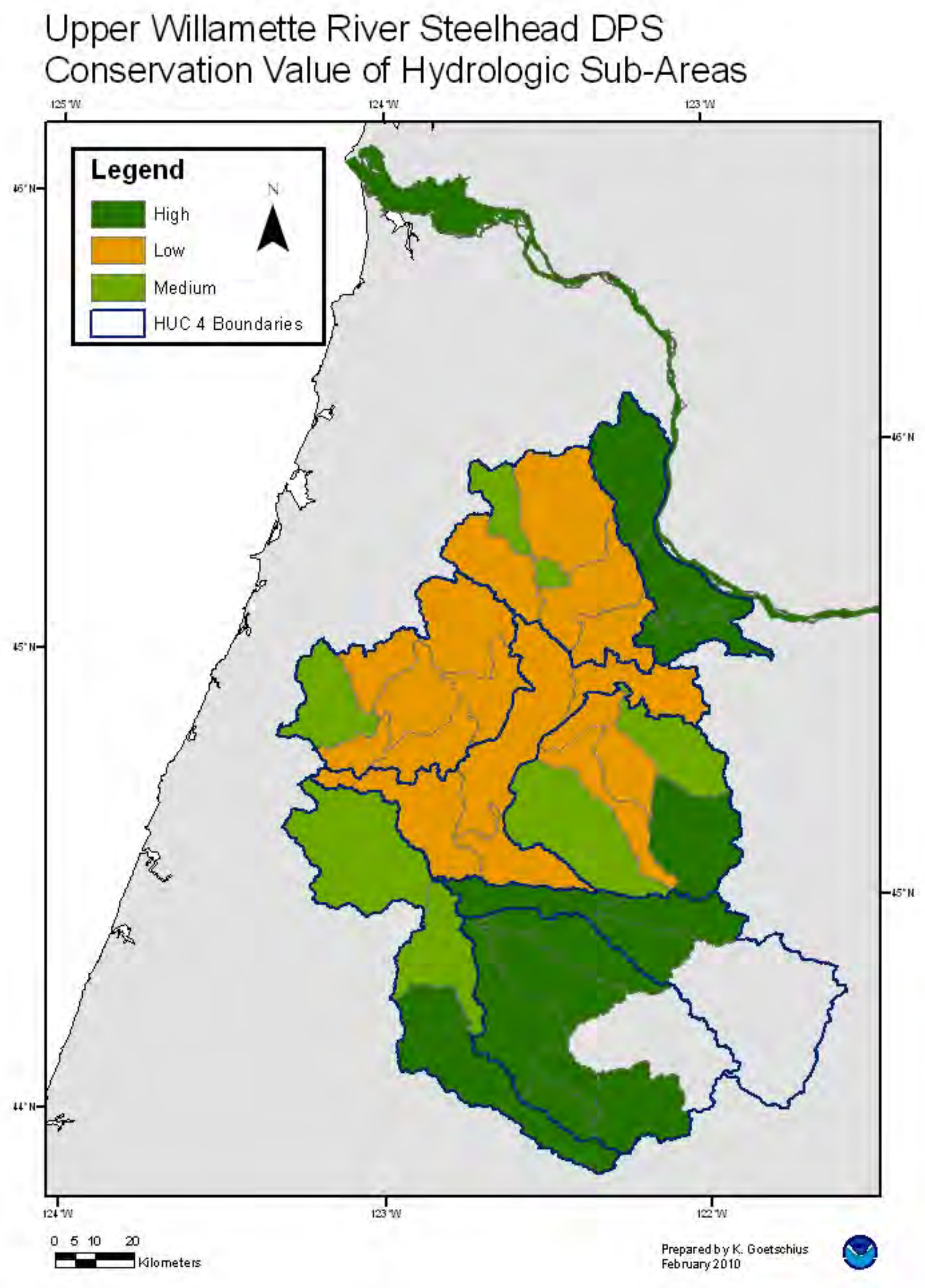


Figure 36. Upper Willamette River Steelhead Conservation Values per Sub-area.

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 37). The UCR steelhead DPS also includes six artificial propagation programs: the Wenatchee River, Wells Hatchery (in the Methow and Okanogan Rivers), Winthrop NFH, Omak Creek, and the Ringold steelhead hatchery programs. These artificially propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within this DPS.

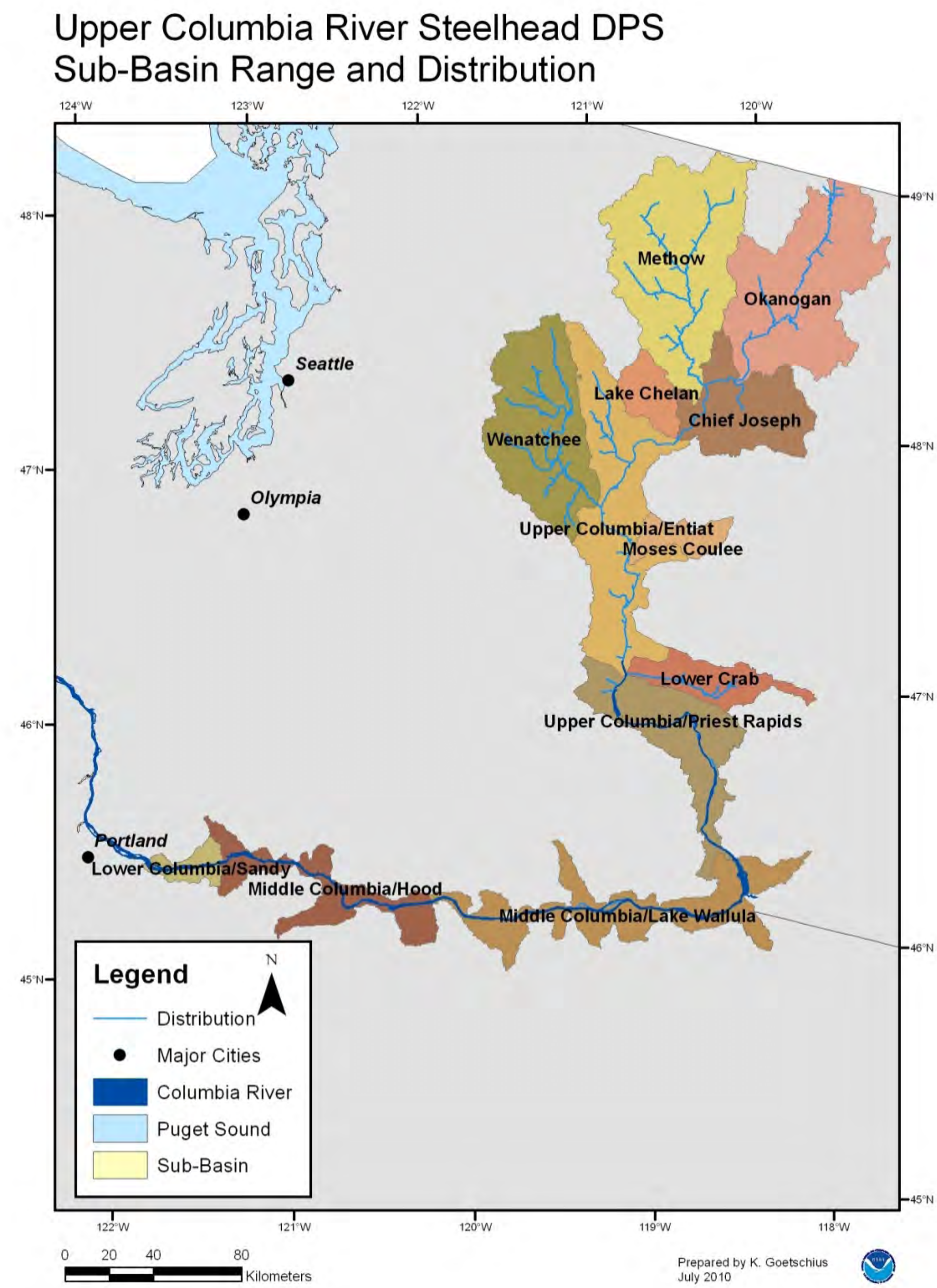


Figure 37. UCR Steelhead distribution.

Life History

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

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 (T. P. 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 40. Upper Columbia River Steelhead salmon populations, abundances, and hatchery contributions (T. P. 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 40), 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 (D. Chapman et al., 1994).

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52630). The CHART assessment for this ESU addressed 10 (HUC4) subbasins containing 41 occupied watersheds (HUC5), as well as the Columbia River rearing/migration corridor. Thirty-one of the watersheds were rated as having a high conservation value, seven as medium value, and three as low value (Table 41). The lower Columbia River rearing/migration corridor downstream of the spawning range is of high conservation value.

Table 41. UCR Steelhead watersheds with conservation values.

HUC 4 Subbasin	HUC 5 Watershed conservation Value (CV)					
	High CV	PCE(s) [1]	Medium CV	PCE(s) [1]	Low CV	PCE(s) [1]
Chief Joseph	1	(3, 2)	0		2	(2)
Okanogan	2	(3, 1)	3	(3)	0	
Similkameen	1	(3)	0		0	
Methow	7	(1, 3)	0		0	
Lake Chelan	0		1	(1, 3)	0	
Upper Columbia/Entiat	3	(3, 1)	1	(3)	0	
Wenatchee	4	(1, 2, 3)	1	(3, 1)	0	
Moses Coulee	0		0		1	(2)
Lower Crab	0		1	(3)	0	
Upper Columbia/Priest Rapids	3	(3)	0		0	
Middle Columbia/Lake Wallula	5	(3)	0		0	
Middle Columbia/Hood	4	(3)	0		0	
Lower Columbia/Sandy	1	(3)	0		0	
Lower Columbia Corridor	all	(3)	0		0	
Total	31		7		3	

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

The current condition of critical habitat designated for the UCR steelhead is moderately degraded. Habitat quality in tributary streams varies from excellent in wilderness and roadless areas to poor in areas subject to heavy agricultural and urban development (Figure 38). 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.

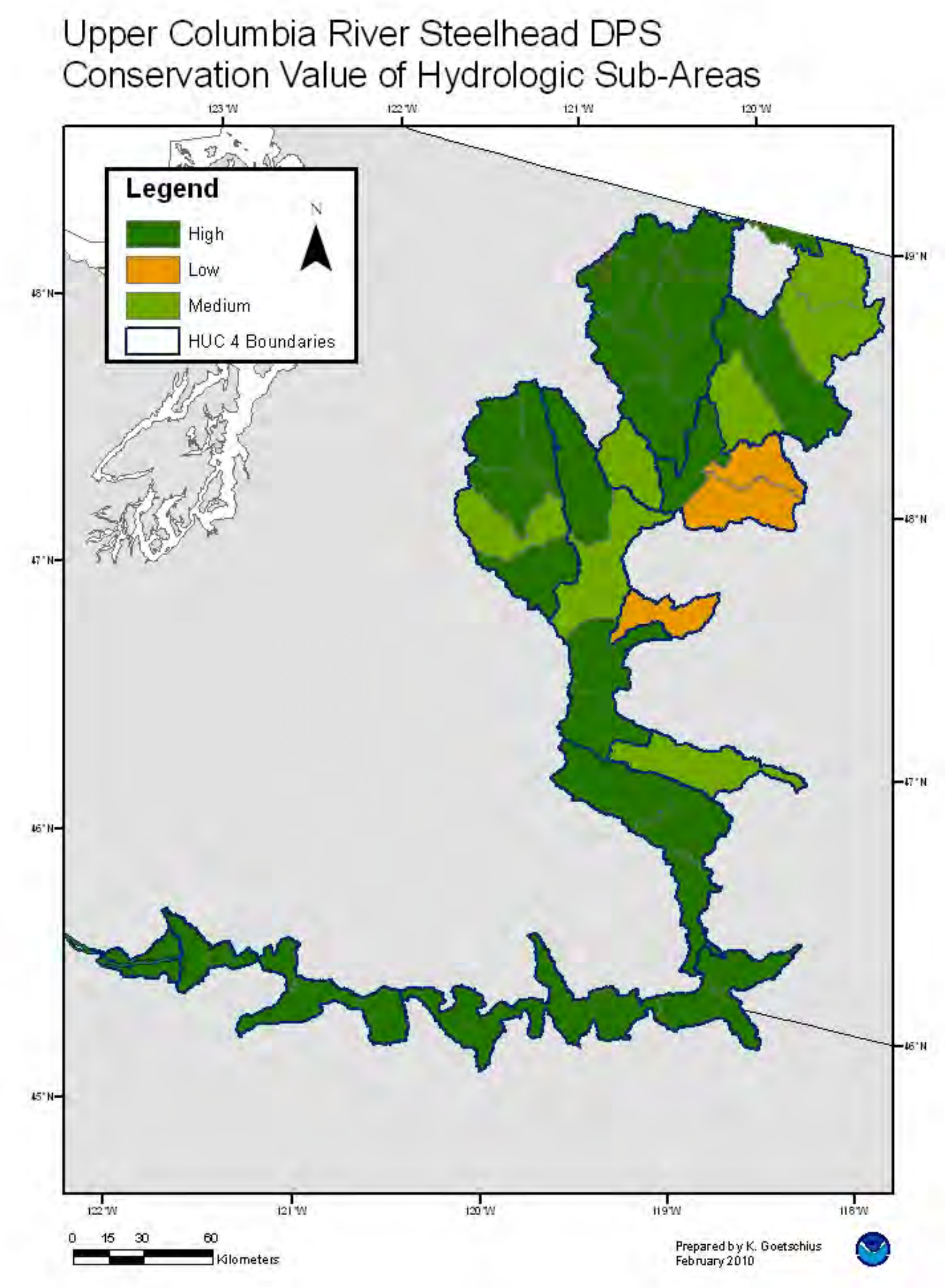


Figure 38. Upper Columbia River Steelhead Conservation Values per Sub-area.

Snake River Steelhead

The Snake River (SR) basin steelhead DPS includes all naturally spawned steelhead populations below natural and man-made impassable barriers in streams in the Columbia River Basin upstream from the Yakima River, Washington, to the U.S. - Canada border (Figure 39). Six artificial propagation programs are also included in the DPS: the Tucannon River, Dworshak National Fish Hatchery, Lolo Creek, North Fork Clearwater, East Fork Salmon River, and the Little Sheep Creek/Imnaha river hatchery programs. These artificially propagated populations are no more divergent relative to the local natural populations than what would be expected between closely related natural populations within the DPS.

Life History

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

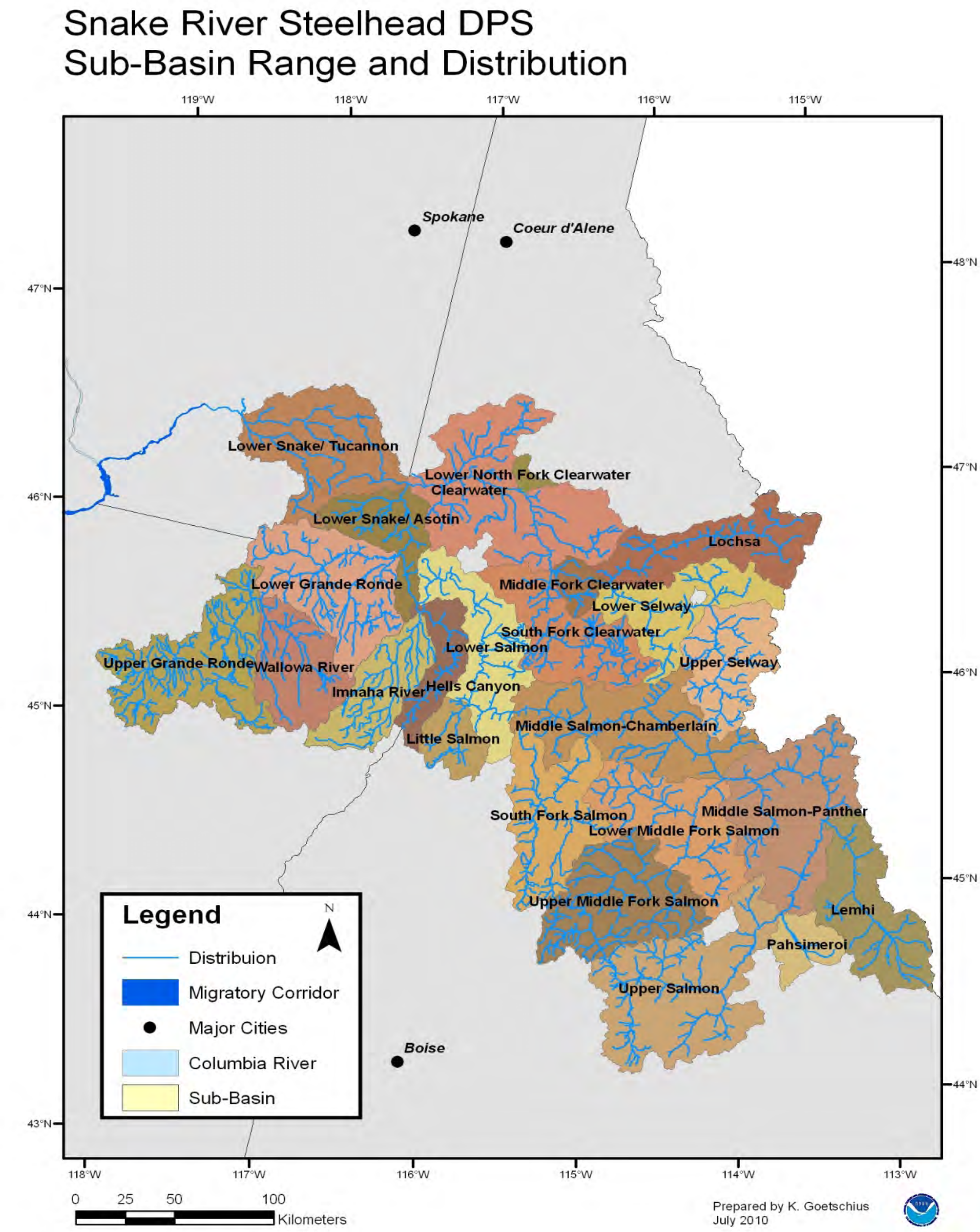


Figure 39. **SR Basin Steelhead distribution.**

Status and Trends

NMFS listed SR basin steelhead as threatened on August 18, 1997 (62 FR 43937), and reaffirmed their threatened status on January 5, 2006 (71 FR 834). The 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 (T. P. Good, et al., 2005). The SR basin steelhead B- run populations remain particularly depressed.

Table 42. SR Basin Steelhead salmon populations, abundances, and hatchery contributions (T. P. 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 42). 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 (T. P. Good, et al., 2005) is summarized below.

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

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

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52630). Figure 40 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 43). 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.

Table 43. 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.

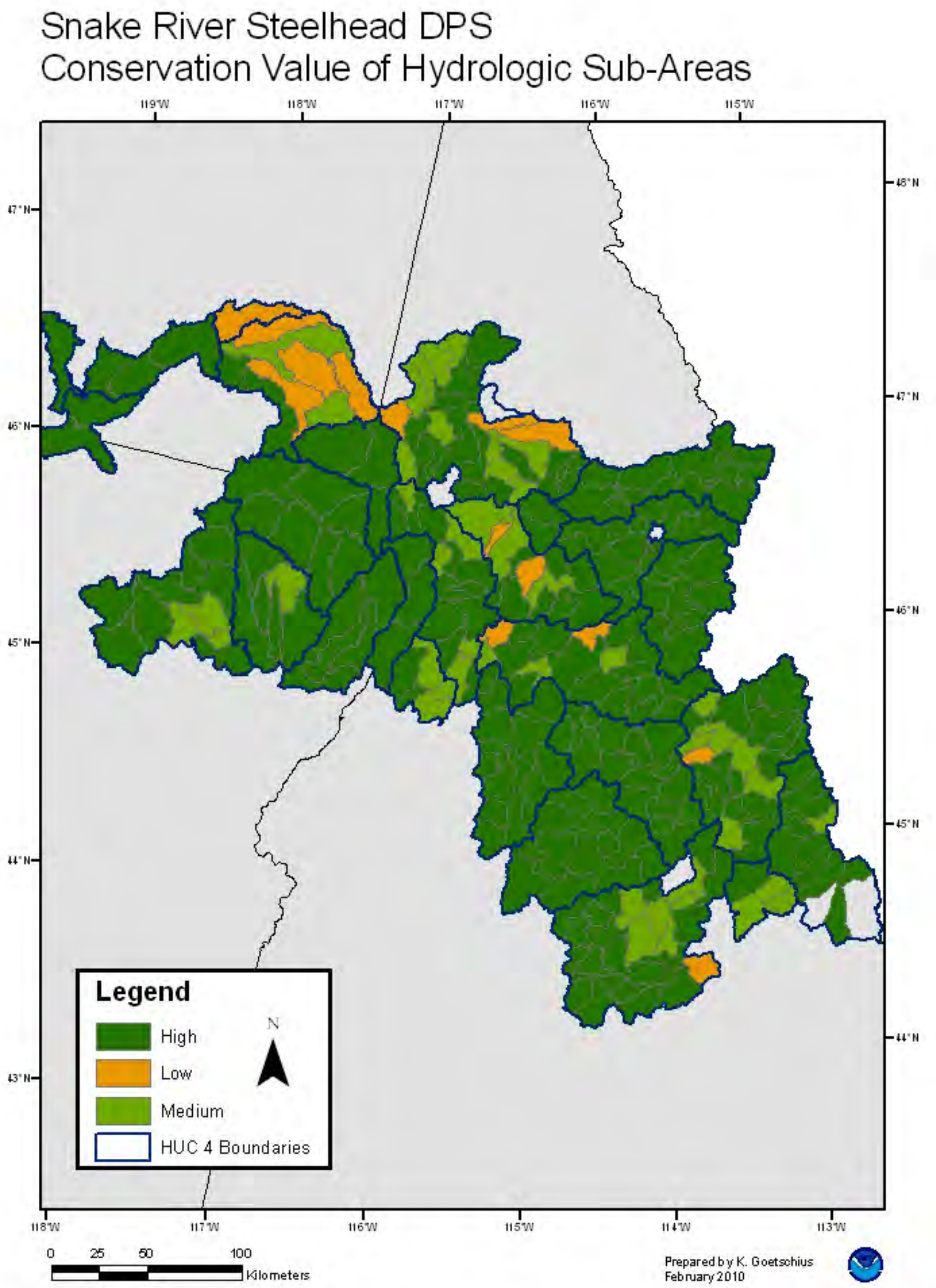


Figure 40. Snake River Steelhead Conservation Values per Sub-area.

basin. These factors have substantially reduced the rearing PCEs contribution to the conservation value necessary for species recovery. Several dams affect adult migration PCE by obstructing the migration corridor.

Northern California Steelhead

The NC steelhead DPS includes all naturally spawned steelhead populations below natural and manmade impassable barriers in California coastal river basins from Redwood Creek southward to, but not including, the Russian River, as well as two artificial propagation programs: the Yeager Creek Hatchery, and North Fork Gualala River Hatchery (Gualala River Steelhead Project) steelhead hatchery programs (Figure 41).

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 & 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).

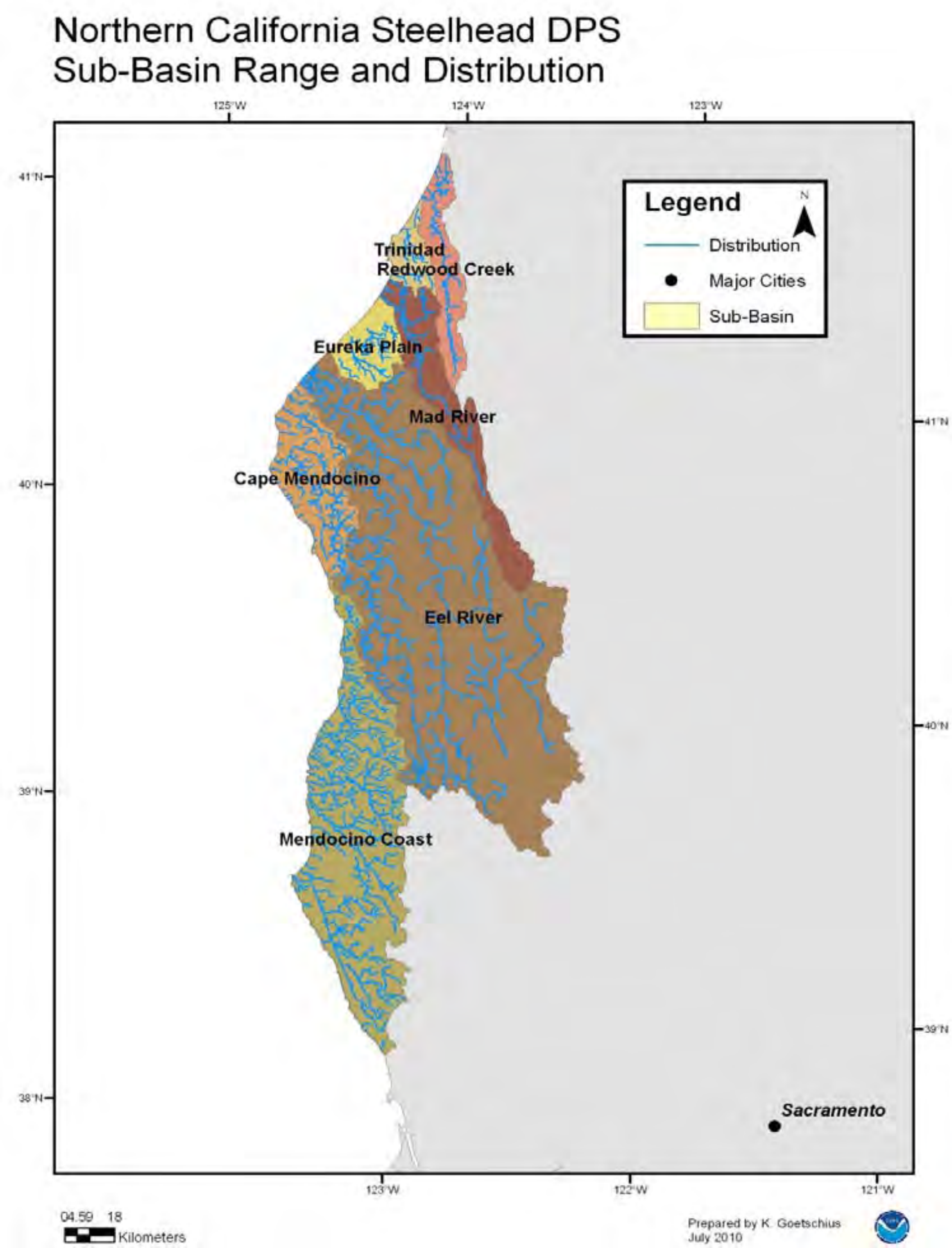


Figure 41. Northern California Steelhead distribution.

Status and Trends

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

Long-term data sets are limited for the NC steelhead. Before 1960, estimates of abundance specific to this DPS were available from dam counts in the upper Eel River (Cape Horn Dam–annual avg. no. adults was 4,400 in the 1930s), the South Fork Eel River (Benbow Dam–annual avg. no. adults was 19,000 in the 1940s), and the Mad River (Sweasey Dam– annual avg. no. adults was 3,800 in the 1940s). Estimates of steelhead spawning populations for many rivers in this DPS totaled 198,000 by the mid-1960s (Table 44).

Table 44. NC Steelhead salmon historic functionally independent populations and their abundances and hatchery contributions (T. P. Good, et al., 2005).

Population	Historical Abundance	Recent Spawner Abundance	Hatchery Abundance Contributions
Mad River (S)	6,000	162-384	2%
MF Eel River (S)	Unknown	384-1,246	0%
NF Eel River (S)	Uknown	Extirpated	N/A
Mattole River (S)	Unknown	9-30*	Unknown
Redwood Creek (S)	Unknown	6*	Unknown
Van Duzen (W)	10,000	Unknown	Unknown
Mad River (W)	6,000	Unknown	Unknown
SF Eel River (W)	34,000	2743-20,657	Unknown
Mattole River (W)	12,000	Unknown	Unknown
Redwood Creek (W)	10,000	Unknown	Unknown
Humboldt Bay (W)	3,000	Unknown	Unknown
Freshwater Creek (W)		25-32	
Ten Mile River (W)	9,000	Unknown	Unknown
Noyo River (W)	8,000	186-364*	Unknown
Big River (W)	12,000	Unknown	Unknown
Navarro River (W)	16,000	Unknown	Unknown
Garcia River (W)	4,000	Unknown	Unknown
Gualala River (W)	16,000	Unknown	Unknown
Total	198,000	Unknown	

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

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

More recent time series data are from snorkel counts conducted on adult summer-run steelhead in the Middle Fork Eel River. Good *et al.* (2005) estimated lambda at 0.98 with a 95% confidence interval of 0.93 and 1.04. The result is an overall downward trend in both the long- and short- term. Juvenile data were also recently examined. Both upward and downward trends were apparent (T. P. 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 (Brian C. Spence, et al., 2008). Hatchery influence has likely been limited.

Critical Habitat

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

There are 50 occupied CALWATER Hydologic Subareas (HSA) watersheds within the freshwater and estuarine range of this ESU. Nine watersheds received a low rating, 14 received a medium rating, and 27 received a high rating of conservation value to the ESU (NMFS, 2005a) (Table 45, and Figure 42). Two estuarine habitat areas used for rearing and migration (Humboldt Bay and the Eel River Estuary) also received a high conservation value rating.

Table 45. 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.

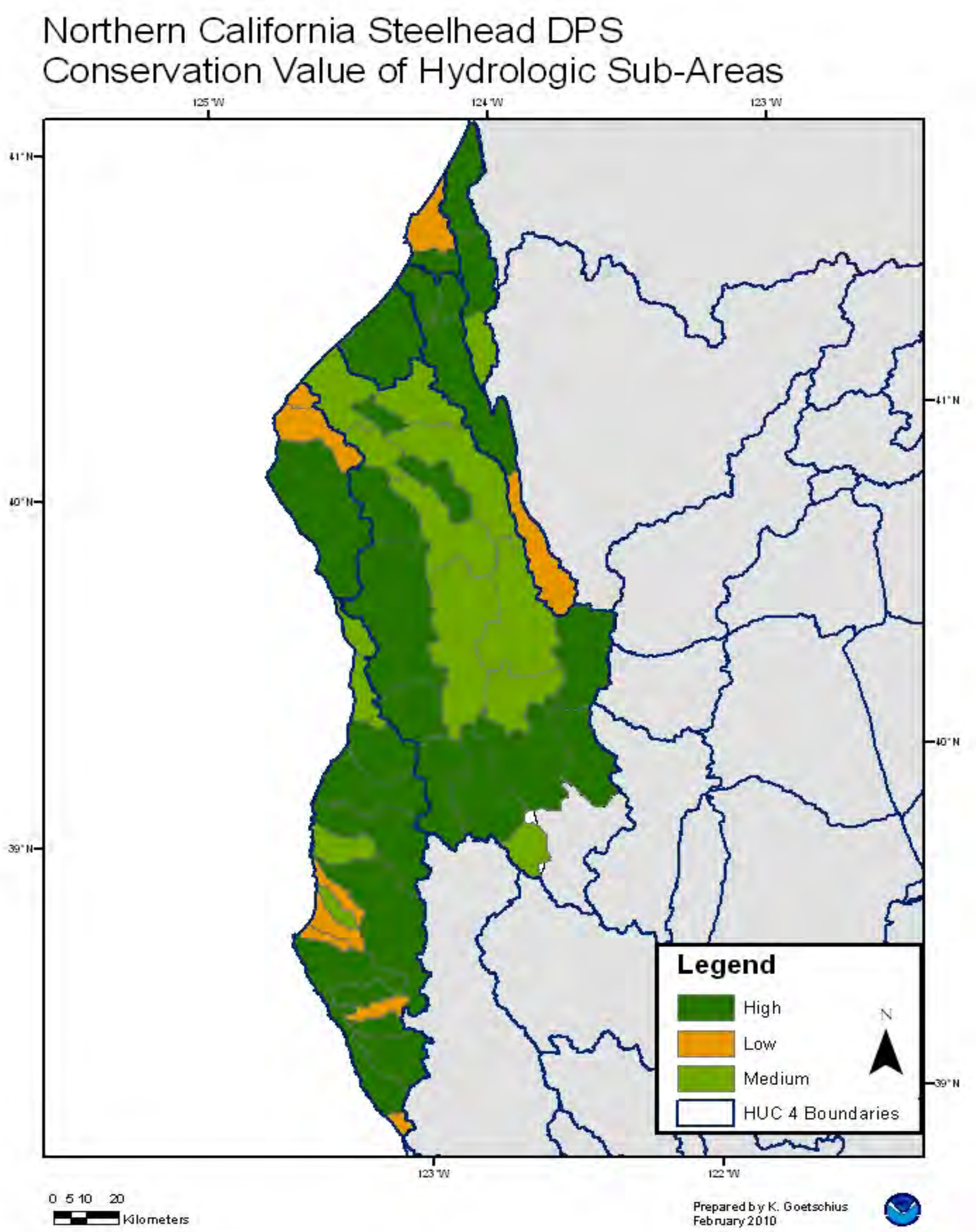


Figure 42. Northern California Steelhead Conservation Values per Sub-area.

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 43).

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) (Hayes, Bond, Hanson, & MacFarlane, 2004; Shapovalov & Taft, 1954). 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, Williams, & Fish, 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).

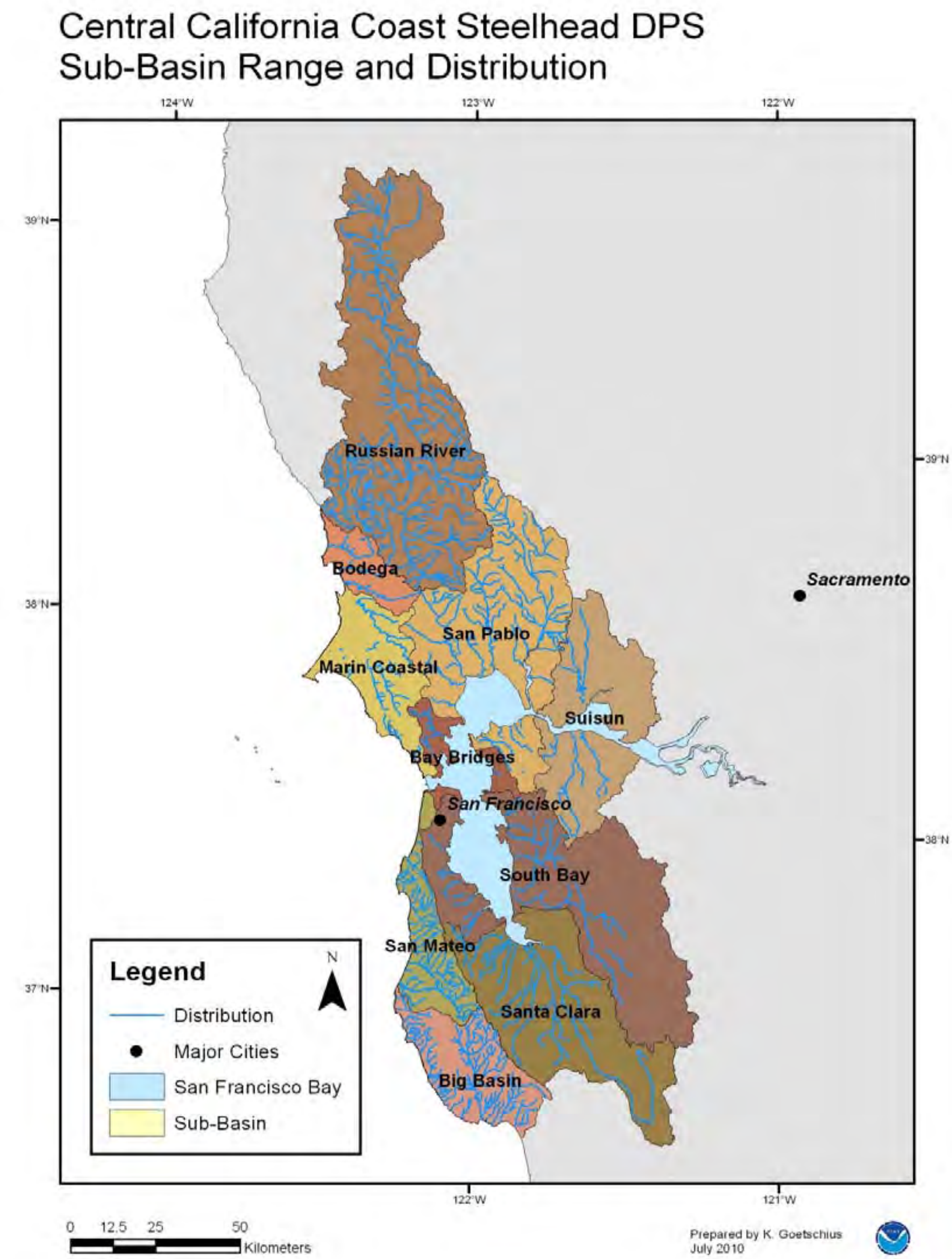


Figure 43. CCC steelhead distribution.

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 46), both have been estimated at less than 15% of their abundances just 30 years earlier (T. P. Good, et al., 2005). Steelhead access to significant portions of the upper Russian River has also been blocked (Busby, et al., 1996; NMFS, 2008a).

Table 46. CCC Steelhead populations, historic population type, abundances, and hatchery contributions (T. P. 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 (Robert A. Leidy, Becker, & Harvey, 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 (T. P. 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; T. P. 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 (T. P. 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.

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 44, 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 47). 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 47. 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.

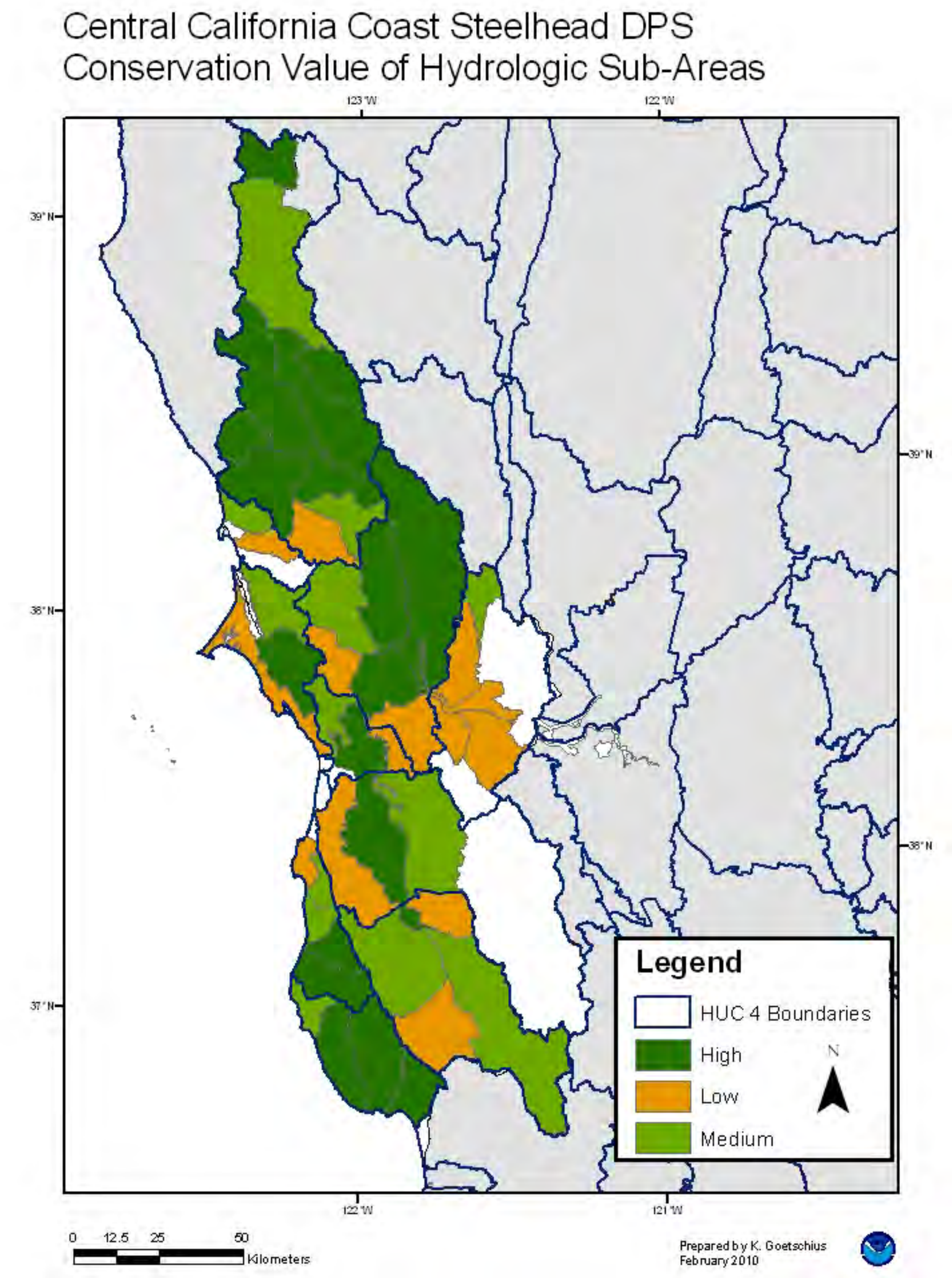


Figure 44. Central California Coast Steelhead Conservation Values per Sub-area.

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 45).

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 though March, in small streams and tributaries where cool, well oxygenated water is available year-round (Hallock, Van Woert, & Shapovalov, 1961; D. McEwan & 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 (D. McEwan & 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).

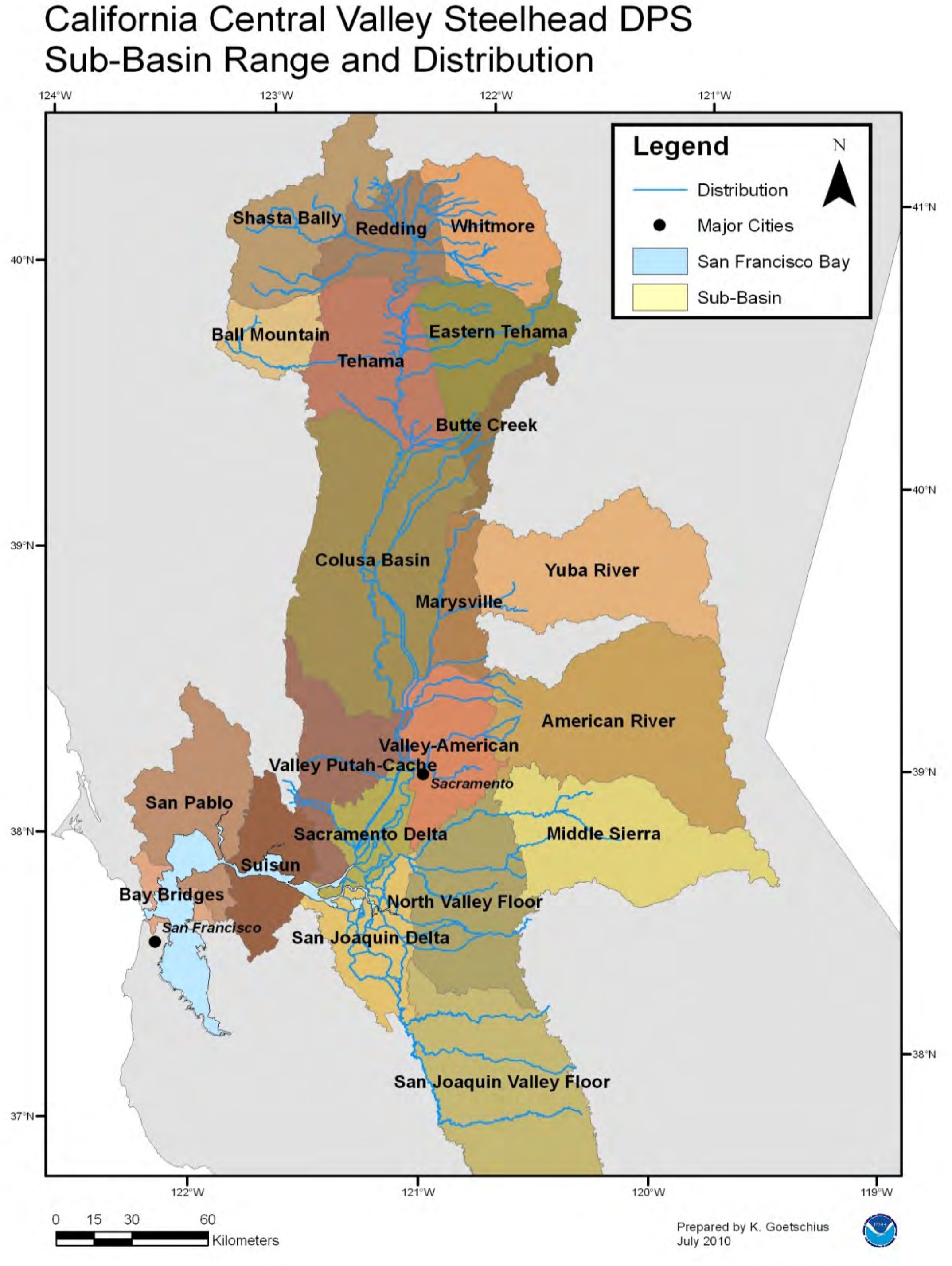


Figure 45. CCV steelhead distribution.

Status and Trends

NMFS originally listed CCV steelhead as threatened on March 19, 1998, and reaffirmed their threatened status on January 5, 2006 (71 FR 834). The CCV steelhead DPS may have consisted of 81 historical and independent populations (Lindley 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 (T. P. Good, et al., 2005). Steelhead have also been observed in Clear Creek and Stanislaus River (Demko & Cramer, 2000; T. P. Good, et al., 2005). 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 (T. P. Good, et al., 2005). In 2004, a total of 12 steelhead smolts were collected in monitoring trawls at the Mossdale station in the lower San Joaquin River (CDFG unpublished data).

Historic CCV steelhead run size may have approached one to two million adults annually (D. R. McEwan, 2001). By the early 1960s, the steelhead run size had declined to about 40,000 adults (D. R. 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 (D. McEwan & Jackson, 1996; D. R. 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 (T. P. 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.

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 46). 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 48. 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 48). 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.

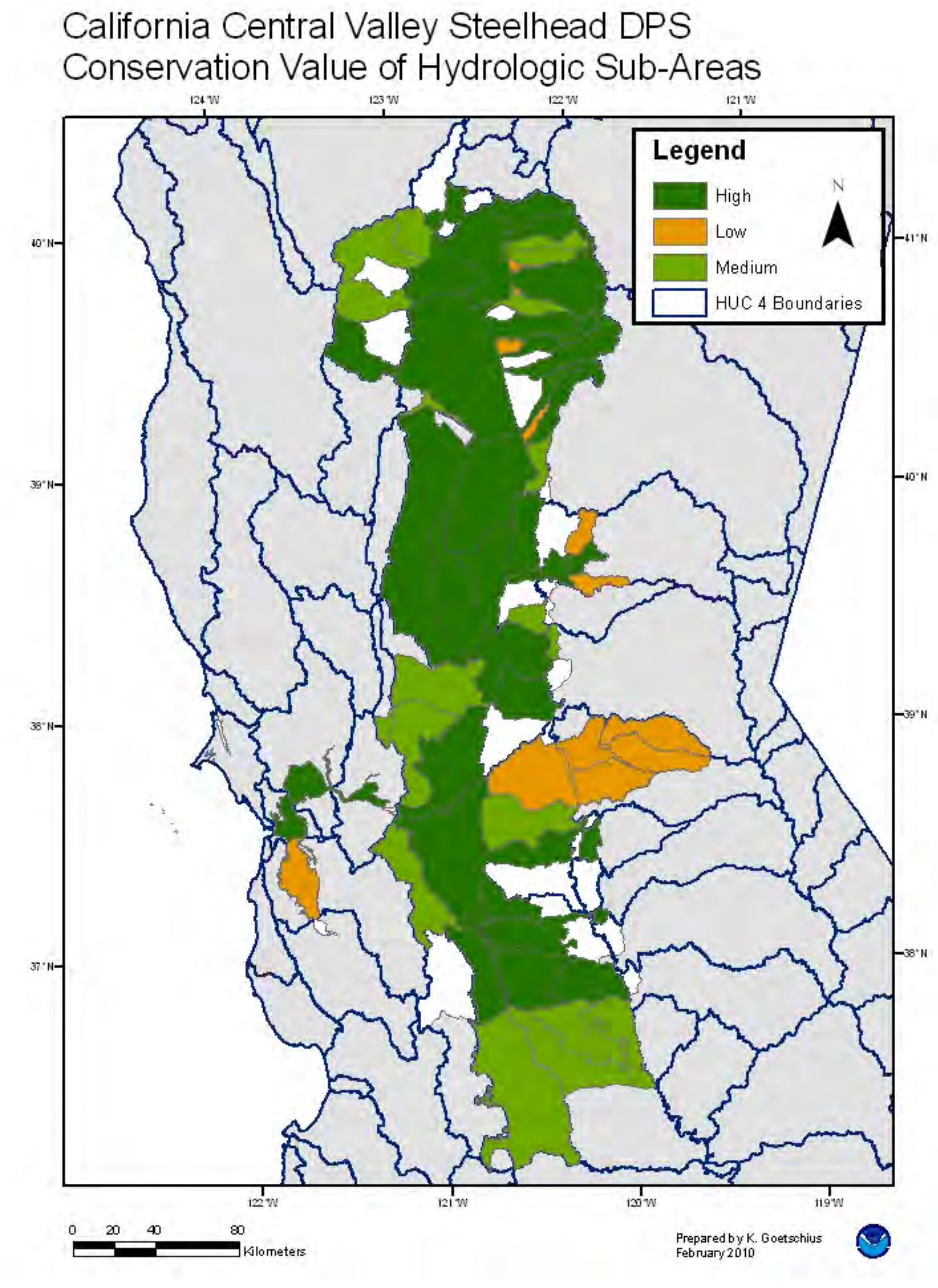


Figure 46. California Central Valley Steelhead Conservation Value per Sub-area.

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 48).

Life History

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

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52488). There are 29 occupied HSA watersheds within the freshwater and estuarine range of this ESU. Figure 47 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 49)(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 .

Table 49. Number of South-Central California Coast steelhead CALWATER HSA watersheds with conservation values.

HUC 4 Subbasin	HUC 5 Watershed conservation Value (CV)					
	High CV	PCE(s) [1]	Medium CV	PCE(s) [1]	Low CV	PCE(s) [1]
Pajaro River	2	(2, 3, 1)	3	(2, 3, 1)	0	
Carmel River	1	(1, 2, 3)	0		0	
Santa Lucia	1	(1, 2, 3)	0		0	
Salinas	2	(2, 3, 1)	1	(1, 2)	4	(2, 3, <1)
Estero Bay	6	(2, 1, 3)	7	(1, 2, 3)	2	(1, 2, 3)
Total	12		11		6	

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

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

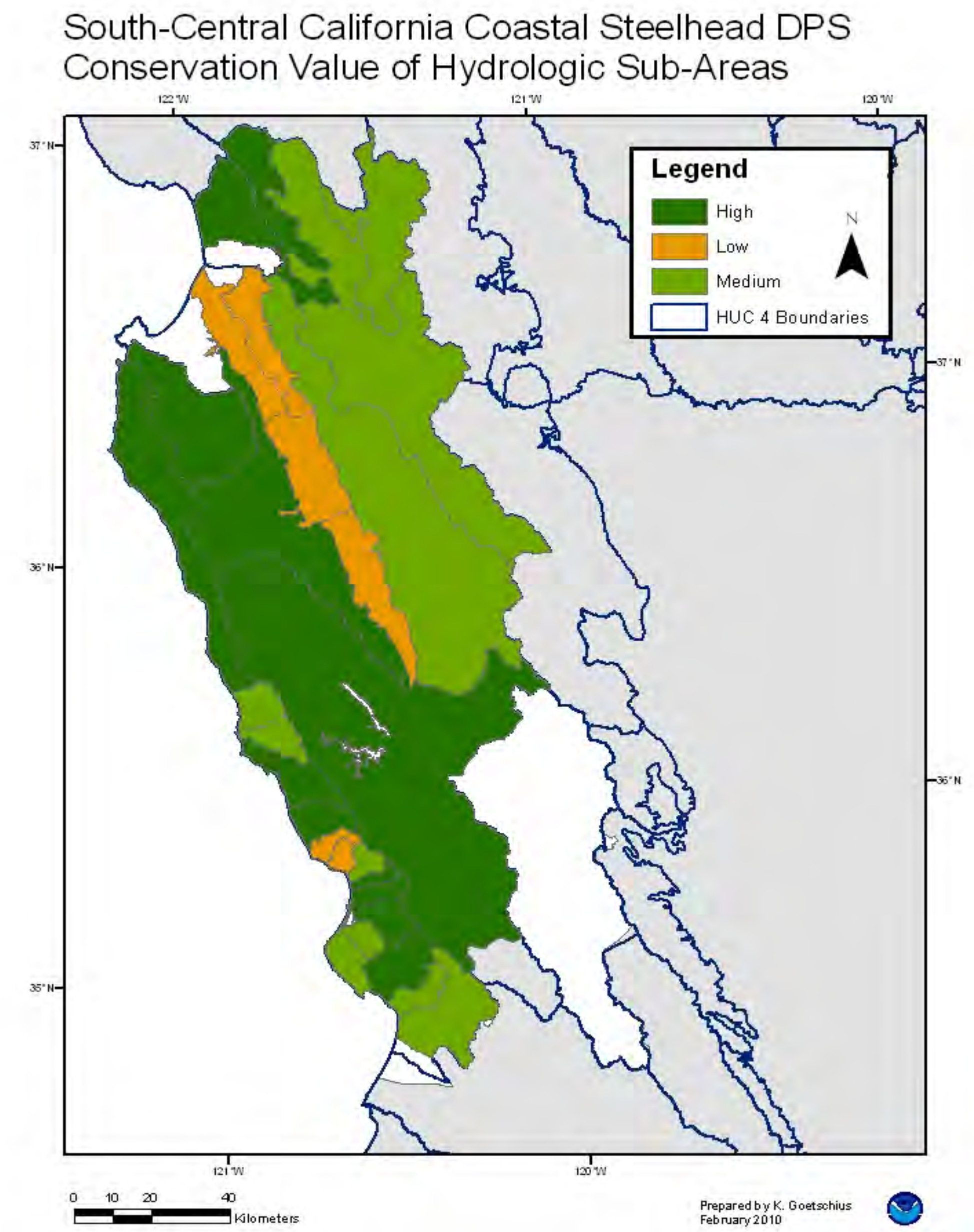


Figure 47. South-Central California Coast Steelhead Conservation Values per Sub-area.

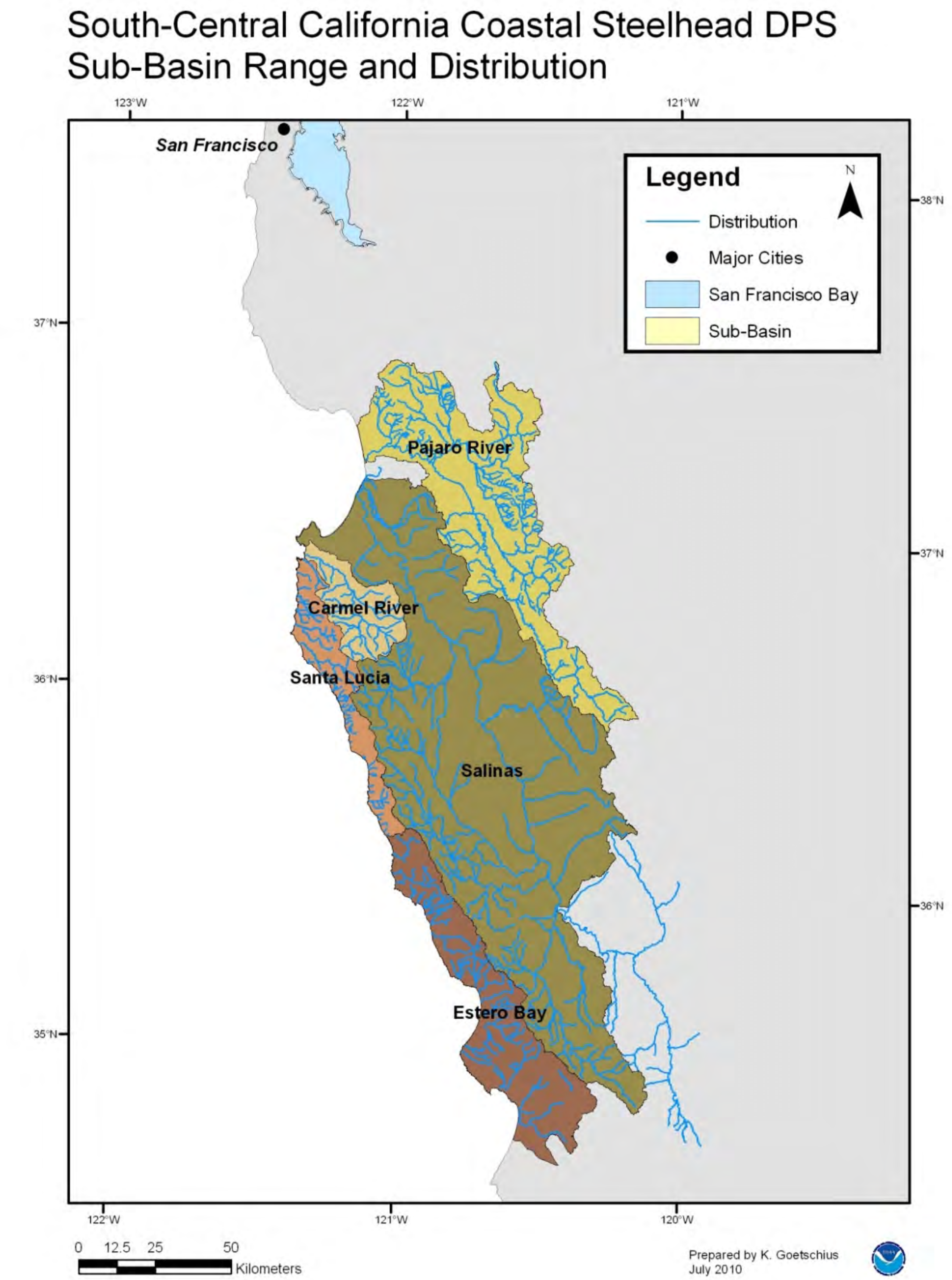


Figure 48. S-CCC steelhead distribution.

Southern California Steelhead

The Southern California (SC) steelhead DPS includes all naturally spawned steelhead populations below natural and manmade impassable barriers in streams from the Santa Maria River, San Luis Obispo County, California, (inclusive) to the U.S. - Mexico Border (Figure 49). Artificially propagated steelhead that reside within the historical geographic range of this DPS are not included in the listing.

Life History

There is limited life history information for SC steelhead. In general, migration and life history patterns of SC steelhead populations are dependent on rainfall and stream flow (Moore, 1980). 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).

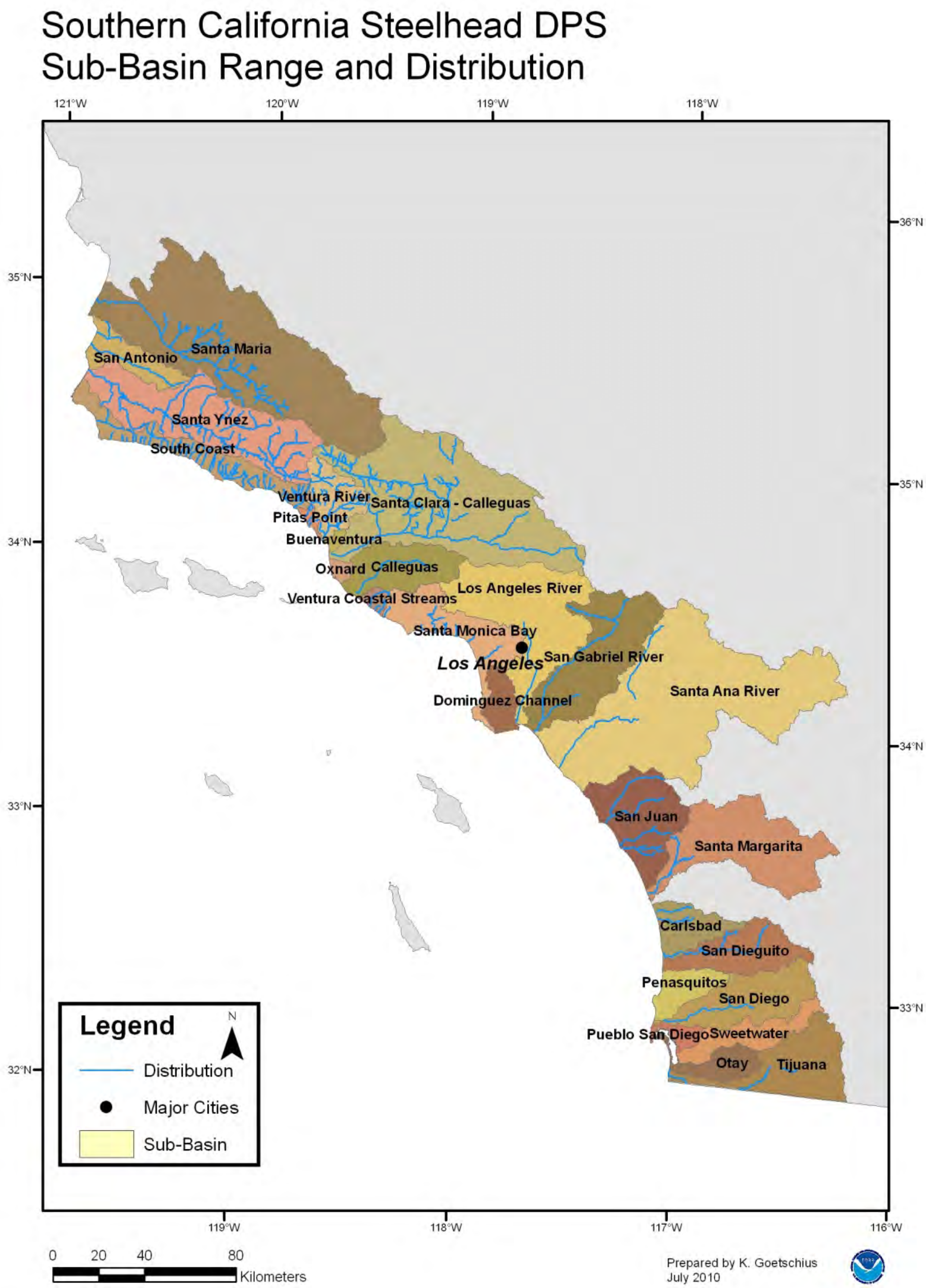


Figure 49. Southern California steelhead distribution.

Status and Trends

NMFS listed the SC steelhead as endangered on August 18, 1997 (62 FR 43937), and reaffirmed their endangered status on January 5, 2006 (71 FR 834). Historic population structure and evaluation of potential stratification of the DPS have not been conducted for this DPS (Table 50).

Table 50. Southern California Steelhead salmon populations, abundances, and hatchery contributions (T. P. 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 50 identifies populations within the SC Steelhead salmon ESU, their abundances, and hatchery input.

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52630). There are 29 HSA watersheds within the freshwater and estuarine range of this ESU designated as critical habitat (Table 51). Figure 50 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 51. 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.

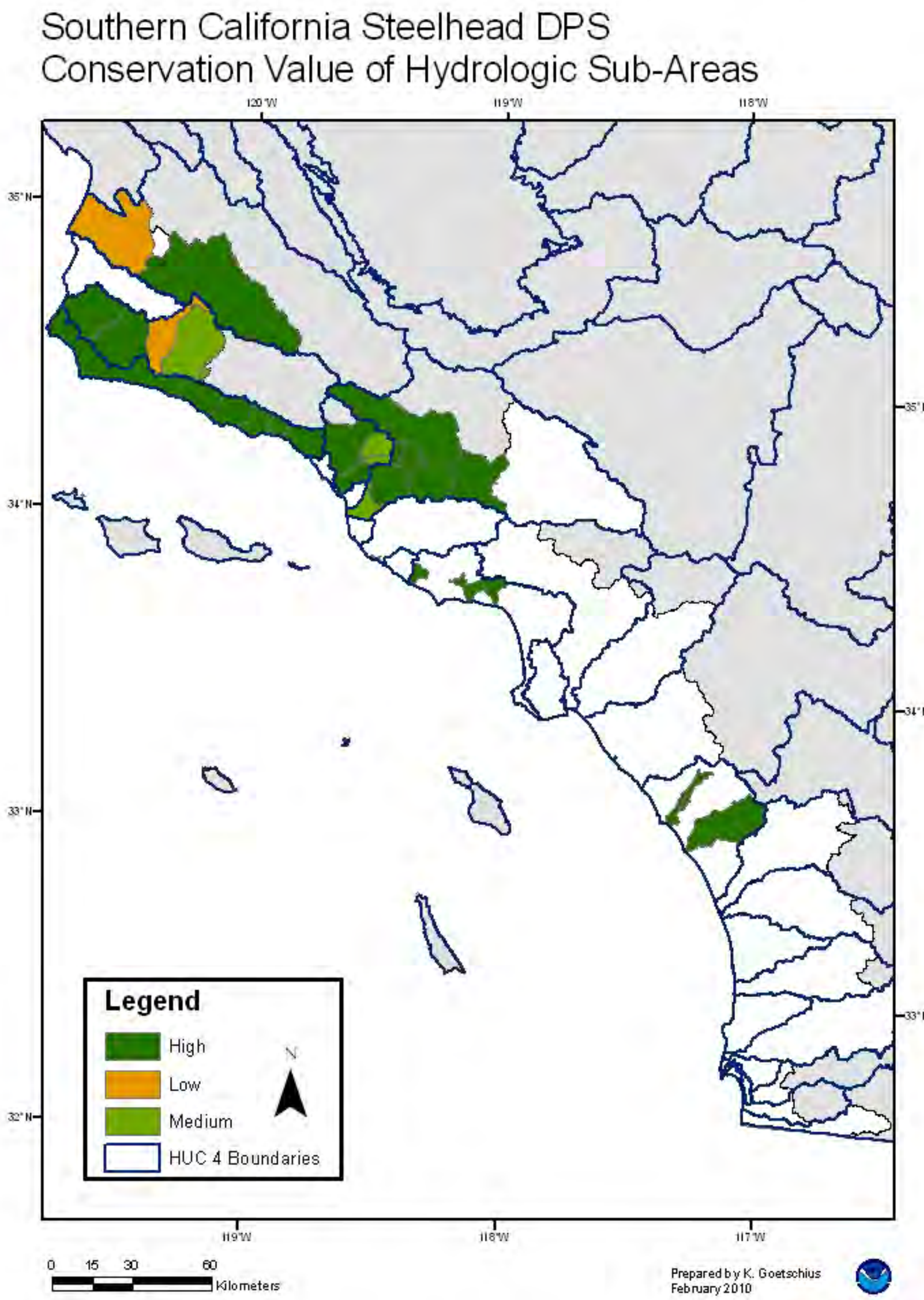


Figure 50. Southern California Steelhead Conservation Values per Sub-area.

Environmental Baseline

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

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

The proposed action under consultation is 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.

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 (Bradford, et al., 1997; Brodeur et al., 2004). 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; D. E. Marshall & 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.

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 (Foott, Harmon, & Stone, 2003; Kier Associates, 1991; Walker & Foott, 1993). However, disease outbreaks may occur when water quality is diminished and fish are stressed from crowding and diminished flows (Guillen, 2003; B.C. Spence, Lomnicky, Hughs, & Novitzki, 1996). Young coho salmon or other salmonid species may become stressed and lose their resistance in higher temperatures (B.C. 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 (CDFG, 2003; Guillen, 2003).

Predation

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

Marine Mammal Predation

Marine mammals are known to attack and eat salmonids. Harbor seals (*Phoca vitulina*), California sea lions (*Zalophus californianus*), and killer whales (*Orcinus orca*) prey on juvenile or adult salmon. Killer whales have a strong preference for Chinook salmon (up to 78% of identified prey) during late spring to fall (Ford & Ellis, 2006; B. Hanson, Baird, & Schorr, 2005; Hard, et al., 1992). 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).

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, Ricciardi, Riva, & Provini, 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 (Collis, 2007; D.D. Roby et al., 2006).

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 (Daniel D. Roby et al., 2008). There are few quantitative data on avian predation rates on Snake River sockeye salmon. Based on the above, avian predators are assumed to have a minimal effect on the long-term survival of Pacific salmon (NMFS, 2008d).

Fish Predation

Pikeminnows (*Ptychocheilus oregonensis*) are significant predators of yearling juvenile migrants (Friesen & Ward, 1999). Chinook salmon were 29% of the prey of northern pikeminnows in lower Columbia reservoirs, 49% in the lower Snake River, and 64% downstream of Bonneville Dam. Sockeye smolts comprise a very small fraction of the overall number of migrating smolts (Ferguson, 2006) 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 & 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 (R.J. Beamish & Neville, 1995; R .J. Beamish, Thomson, & Farlane, 1992; Pearcy, 1992).

Wildland Fire

Wildland fires that are allowed to burn naturally in riparian or upland areas may benefit or harm aquatic species, depending on the degree of departure from natural fire regimes. Although most fires are small in size, large size fires increase the chances of adverse effects on aquatic species. Large fires that burn near the shores of streams and rivers can have biologically significant short-term effects. They include increased water temperatures, ash, nutrients, pH, sediment, toxic chemicals, and large woody debris (Buchwalter, Sandahl, Jenkins, & Curtis, 2004; Rinne, 2004). Nevertheless, fire is also one of the dominant habitat-forming processes in mountain streams (P.A. 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 (Greswell, 1999; Rinne, 2004). 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 & 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, Jenkins, & Curtis, 2003; Buchwalter, et al., 2004; Minshall, Royer, & Robinson, 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).

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 (R. J. Beamish & Bouillon, 1993; R. J. Beamish, Sweeting, & Neville, 2009; Finney, Gregory-Eaves, Douglas, & Smol, 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 (R. J. 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 (R. J. Beamish, et al., 2009). The abundance and distribution of salmon and zooplankton also relate to shifts in North Pacific atmosphere and ocean climate (Francis & 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 & Mantua., 2000; Mantua, Hare, Zhang, Wallace, & Francis, 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 & 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 (R. J. Beamish, et al., 2000).

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

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

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

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 activites 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: and (6) removal or impairment of riparian vegetation – resulting in increased water temperatures and streambank erosion.

Prior to discussion of each geographic region, three major issues are highlighted: pesticide contamination, elevated water temperature, and loss of habitat/habitat connectivity. These three factors are the most relevant to the current analysis. We provide information on pesticide detections in the aquatic environment and highlight their background levels from past and ongoing anthropogenic activities. This information is pertinent to EPA's proposed registration of 2,4-D, triclopyr, diuron, linuron, captan, and chlorothalonil in the U.S. and its territories. Some of these 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.

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 (R.J. Gilliom et al., 2006). We chose this approach because the NAWQA studies present the same level of analysis for each area. Further, given the lack of uniform reporting standards, we are unable to present a comprehensive basin-specific analysis of detections from other sources.

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

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

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 (R.J. Gilliom, et al., 2006).

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 urbanareas. Two of the herbicides used primarily in urban areas are 2,4-D and diuron. Both herbicides were detected roughly 12% of the time in agricultural streams and between 20% and 25% of the time in urban streams. 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, 2009e).

Another dimension of pesticides and their degradates in the aquatic environment is their simultaneous occurrence as mixtures (R.J. Gilliom, et al., 2006). 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 (R.J. Gilliom, et al., 2006). 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 (R.J. Gilliom, et al., 2006). 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%) (R.J. Gilliom, et al., 2006). Finally, organochlorine pesticides that were discontinued 15 to 30 years ago still exceeded benchmarks for aquatic life and fish-eating wildlife in bed sediment or fish tissue samples from many streams.

National Pollutant Discharge Elimination System

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

Baseline Water Temperature - Clean Water Act

Elevated temperature is considered a pollutant in most states with approved Water Quality Standards under the federal Clean Water Act (CWA) of 1972. Under the authority of the CWA, states periodically prepare a list of all surface waters in the state for which beneficial uses - such as drinking, recreation, aquatic habitat, and industrial use

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

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

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 (McCullough, 1999; B.C. Spence, et al., 1996). 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 (S. V. Gregory & Bisson, 1997).

Sublethal temperatures (above 24ºC) could be detrimental to salmon by increasing susceptibility to disease (Colgrove & Wood, 1966) or elevating metabolic demand (J.R. 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 52. 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)

(7-DADMax) exceeds the temperature threshold (Table 52).

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

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

Table 53. Number of kilometers of river, stream and estuaries included in state 303(d) lists due to temperature that are located within each salmonid ESU/DPS. Data was taken from the most recent GIS layers available from state water quality assessments reports.*

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

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

Baseline Habitat Condition

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 (P. A. Bisson & 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 (R. E. Bilby, Fransen, Walter, & Scarlett, 2001). While not necessary for pool formation, LWD is associated with around 80% of pools in northern California, Washington, and the Idaho pan-handle (R. E. Bilby & Bisson, 2001).

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

Floodplains are relatively flat areas adjacent to larger streams and rivers. They allow for the lateral movement of the main channel and provide storage for floodwaters during periods of high flow. Water stored in the floodplain is later released during periods of low flow. This process ensures adequate flows for salmonids during the summer months, and reduces the possibility of high-energy flood events destroying salmonid redds (C. J. 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, Miller, Dunne, Reeves, & Agee, 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 (C. J. 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 (C. J. 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.

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, 2009d) 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. Most recently, NMFS published conclusions regarding the registration of 12 different a.i.s (NMFS, 2010). 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.

Additionally, some of the a.i.s discussed in this Opinion have been approved for use in Federal weed control programs. The US Forest Service and BLM both have invasive plant control programs that have gone through the ESA section 7 consultation process (NMFS, 2007b, 2009c). These programs include the use of formulations of 2,4-D, triclopyr, and/or diuron within the range of at least one salmon population. Each opinion concluded that the weed control program would not jeopardize the existence of listed salmonids or destroy adversely modify critical habitat because of the limited scope of each project and the management practices included in the action. Each project covered use of the herbicides in a few subbasins, often limiting the number of acres that can be treated. Treatment methods are specified as well, and are frequently limited to spot spraying, wicking, dipping, painting, and injecting. Generally, only 2,4-D products labeled for aquatic use were permitted to be used within 15 feet of open water. In one case, triclopyr was approved for use within this range, but on only a select number of species and was limited to an area of 0.1 acre per occurrence and 5 acres total per year

(NMFS, 2009b). Most proposals included a minimum distance from water and/or intermittent streams for each product or mixture of products. Among the BMPs included in the action were monitoring weather, using only specified adjuvants, diluting the formulation, marking riparian buffers before application begins, and buffers. As these practices are mandatory, NMFS had to consider them as part of the federal action. Further, in the Incidental Take Statements, NMFS specified compliance monitoring and other types of oversight to be sure these BMPs were being followed. In general, the extent of incidental take for these opinions is defined by the amount of treated land within a given distance of salmon bearing streams. These Opinions recognized the large degree of uncertainty inherent in the effects analysis due to a paucity of information regarding sublethal effects and toxicity of mixtures. As such, the conclusions depended on limited use of the compounds and BMPs to keep products out of the water. In some cases, subsequent more targeted consultations were required. If adverse effects were to occur, NMFS Biologists concluded that they would be limited to a specific area to avoid impact to the ESU/DPS as a whole.

Geographic Regions

For a more fine scale analysis, we divided the action area into geographic regions: the Southwest Coast Region (California and the southern parts of the State of Oregon) and the Pacific Northwest Region (Idaho, Oregon, and Washington). The Pacific Northwest Region was further subdivided according to ecoregions or other natural features important to NMFS trust resources. Use of these geographic regions is consistent with previous NMFS consultations conducted at the national level (NMFS, 2007a). We summarize the principal anthropogenic factors occurring in the environment that influence the current status of listed species within each region. Table 54 provides a breakdown of these regions and includes the USGS subregions and accounting units for each region. It also provides a list of ESUs/DPSs found in each accounting unit, as indicated by Federal Register listing notices.

Table 54. USGS Subregions and accounting units within the Northwest and Southwest Regions, along with ESUs/DPSs present within the area (Seaber, Kapinos, & Knapp, 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
	Willamette River Basin	—	OR	170900	Upper Willamette Chinook; Upper Willamette Steelhead; Lower Columbia Chinook; Lower Columbia Steelhead; Lower Columbia Coho
Pacific Northwest: Coastal Drainages	Oregon-Washington Coastal Basin	Washington Coastal	WA	171001	Ozette Lake Sockeye
		Northern Oregon Coastal	OR	171002	Oregon Coast Coho

Region	USGS Subregion	Accounting Unit	State	HUC no.	ESU/DPS
		Southern Oregon Coastal	OR	171003	Oregon Coast Coho; Southern Oregon and Northern California Coast Coho
Pacific Northwest: Puget Sound	Puget Sound	—	WA	171100	Puget Sound Chinook; Hood Canal Summer - Run Chum; Puget Sound Steelhead
Southwest Coast	Klamath-Northern California Coastal	Northern California Coastal	CA	180101	Southern Oregon and Northern California Coast Coho; California Coastal Chinook; Northern California Steelhead; Central California Coast Steelhead; Central California Coast Coho
		Klamath River Basin	CA, OR	180102	Southern Oregon and Northern California Coast Coho
	Sacramento River Basin	Lower Sacramento River Basin	CA	180201	Central Valley Spring-run Chinook; California Central Valley Steelhead; Sacramento River Winter-run Chinook
	San Joaquin River Basin	—	CA	180400	California Central Valley Steelhead
	San Francisco Bay	—	CA	180500	Central California Coast Steelhead; Southern Oregon and Northern California Coast Coho; Central California Coast Coho; Sacramento River Winter-run Chinook
	Central California Coastal	—	CA	180600	Central California Coast Steelhead; Southern Oregon and Northern California Coast Coho; South-Central California Coast Steelhead; Southern California Steelhead; Central California Coast Coho; Sacramento River Winter-run Chinook
	Southern California Coastal	Ventura-San Gabriel Coastal	CA	180701	Southern California Steelhead
		Laguna-San Diego Coastal	CA	180703	Southern California Steelhead

Southwest Coast Region

The basins in this section occur in the States of California and the southern parts of Oregon. Ten of the 28 species addressed in the Opinion occur in the Southwest Coast Region. They are the California Coastal Chinook (CC) salmon, Central Valley (CV) Spring-run Chinook salmon, Sacramento River winter-run Chinook salmon, Southern Oregon/Northern California Coast (SONCC) coho salmon, Central California Coast (CCC) coho salmon, Northern California (NC) steelhead, Central California Coast (CCC) steelhead, California Central Valley (CCV) steelhead, South-Central California Coast (S-CCC) steelhead, and Southern California (SC) steelhead (Table 54). Table 55 and Table 56 show land area in km^2 for each ESU/DPS located in the Southwest Coast Region.

Table 55. Area of land use categories within the range Chinook and Coho Salmon ESUs in km² where bolded numbers are totals for each category. Land cover image data were taken from Multi-Resolution Land Characteristics (MRLC) Consortium, a consortium of nine federal agencies (USGS, EPA, USFS, NOAA, NASA, BLM, NPS, NRCS, and USFWS) (National Land Cover Data 2001). Land cover class definitions are available at: http://www.mrlc.gov/nlcd_definitions.php

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

Table 56. Area of Land Use Categories within the Range of Steelhead Trout DPSs (km^2). Land cover image data were taken from Multi-Resolution Land Characteristics (MRLC) Consortium, a consortium of nine federal agencies (USGS, EPA, USFS, NOAA, NASA, BLM, NPS, NRCS, and USFWS) (National Land Cover Data 2001). Land cover class definitions are available at: http://www.mrlc.gov/nlcd_definitions.php

Land Cover		Steelhead				
sub category	code	Northern CA	Central CA Coast	CA Central Valley	South-Central CA coast	Southern CA
Water		**106**	**1,406**	**409**	**127**	**159**
Open Water	11	106	1,406	409	127	159
Perennial Snow/Ice	12	0	0	0	0	0
Developed Land		**757**	**3,677**	**3,252**	**1,759**	**7,327**
Open Space	21	610	1,224	1,431	1,019	1,952
Low Intensity	22	50	876	693	247	1,787
Medium Intensity	23	32	1,223	744	168	2,726
High Intensity	24	3	327	181	23	767
Barren Land	31	63	26	202	303	95
Undeveloped Land		**16,117**	**11,041**	**19,216**	**14,959**	**13,057**
Deciduous Forest	41	763	179	751	1	1
Evergreen Forest	42	9,790	2,506	3,990	1,721	984
Mixed Forest	43	1,159	2,086	598	1,925	1,025
Shrub/Scrub	52	2,878	2,253	3,745	4,952	8,375
Herbaceous	71	1,478	3,588	9,435	6,194	2,539
Woody Wetlands	90	32	36	248	93	83
Emergent Wetlands	95	17	392	450	73	50
Agriculture		**193**	**522**	**10,724**	**1,500**	**1,059**
Hay/Pasture	81	179	36	1,671	203	179
Cultivated Crops	82	14	486	9,054	1,297	880
TOTAL (inc. open water)		17,173	16,645	33,601	18,345	21,602
TOTAL (w/o open water)		17,067	15,240	33,193	18,218	21,446

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 57).

Table 57. Select rivers in the southwest coast region (Carter & 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

Figure 51displays major landuse chatagories 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 58). 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, Izbicki, & Paybins, 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 58. Land uses and population density in several southwest coast watersheds (Carter & 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

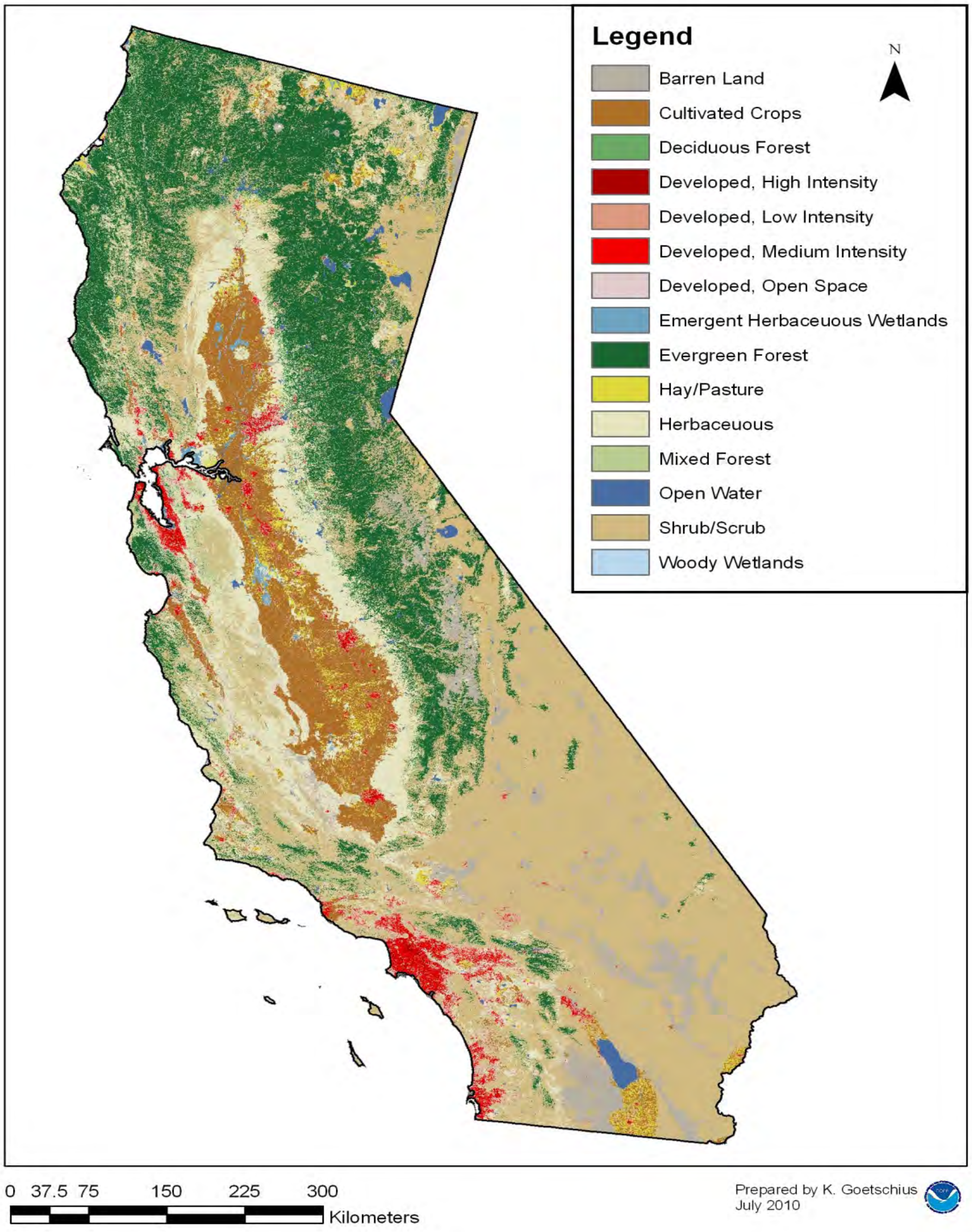


Figure 51. Landuse in Southwest Region.

As a watershed becomes urbanized, population increases and changes occur in stream habitat, water chemistry, and the biota (plants and animals) that live there. The most obvious effect of urbanization is the loss of natural vegetation which results in an increase in impervious cover and dramatic changes to the natural hydrology of urban streams. Urbanization generally results in land clearing, soil compaction, modification and/or loss of riparian buffers, and modifications to natural drainage features (Richter, 2002). 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 six 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 of the Proposed Action* chapter.

Pesticide Reduction Programs in the Southwest Coast Region

When using these six 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. 2,4-D, triclopyr BEE, diuron, and captan all have formulations that are registered as restricted materials in California. This designation means that use of these products is regulated and monitored via licensing and reporting requirements. The degree to which this designation provides additional environmental protections is unclear. Surface water protections exist in draft form for pesticides "that have been determined to have a high potential to contaminate surface water", including diuron and linuron. While it has undergone public comment and revision, this legislation has not been formally proposed yet.

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, Kalman, & Klassen, 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 diazinon surface water contamination in the Sacramento River Basin. The organization has developed best-practice literature for pesticide use in both urban and agricultural settings (www.curesworks.org). CURES also works with California's Watershed Coalitions to standardize their Watershed Evaluation Reports and to keep the Coalitions informed. The organization has worked with local organizations, such as the California Dried Plum Board and the Almond Board of California, to address concerns about diazinon, pyrethroids, and sulfur. The CURES site discusses alternatives to organophosphate dormant spray applications. It lists pyrethroids and carbaryl as alternatives, but cautions that these compounds may impact non-target organisms. The CURES literature does not specifically address the a.i.s discussed in this Opinion.

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

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

The CDPR publishes voluntary interim measures for mitigating the potential impacts of pesticide usage to listed species. These measures are available online as county bulletins (http://www.cdpr.ca.gov/docs/endspec/colist.htm). Measures that apply to 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil use in salmonid habitat are:

- Do not use in currently occupied habitat. (captan and chlorothalonil)
- 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 (2,4 D, triclopyr, diuron and linuron)

- Provide a 20 ft minimum strip of vegetation (on which pesticides should not be applied) along rivers, creeks, streams, wetlands, vernal pools and stock ponds, or on the downhill side of fields where runoff could occur. Prepare land around fields to contain runoff by proper leveling, etc. Contain as much water "on-site" as possible. The planting of legumes or other cover crops for several rows adjacent to off-target water sites is recommended. Mix pesticides in areas not prone to runoff such as concrete mixing/loading pads, disked soil in flat terrain or graveled mix pads, or use a suitable method to contain spills and/or rinsate. Properly empty and triple-rinse pesticide containers at time of use. (captan and chlorothalonil)

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

- 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. (all six compounds)

- Do not apply within 30 yards upslope of habitat unless a suitable method is used to contain or divert runoff waters. (diuron and linuron)

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 & 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

Currently, California has over 500 water bodies on its 303(d) list (Wu, 2000). The 2006 list includes 779 stream segments, rivers, lakes, and estuaries and 12 pollutant categories (CEPA, 2007). Pollutants represented on the list include pesticides, metals, sediments, nutrients or low dissolved oxygen, temperature, bacteria and pathogens, and trash or debris. There are 2,237 water body/pollutant listings; a water body is listed separately for each pollutant detected (CEPA, 2007). The 2006 303(d) list identifies water bodies listed due to elevated temperature (Table 59). See species ESU/DPS maps for NPDES permits and 303(d) waters co-located within listed salmonid ESUs/DPSs in California.

Table 59. California's 2006 Section 303(d) List of Water Quality Limited Segments: segments listed for exceeding temperature limits (CEPA, 2007).

Pollutant	Estuary Acres Affected	River / Stream Miles Affected	# Water Bodies
Temperature	-	16,907.2	41

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). 2,4-D, triclopyr, and linuron were tested for but not detected. Of the 85 VOCs routinely analyzed for, 49 were detected. VOCs included methyl *tert*-butyl ether (MTBE), chloroform, and trichloroethylene (TCE). Organochlorine compounds were also detected in bed sediment and fish tissue. Organochlorine concentrations were also higher at urban sites than at undeveloped sites in the Santa Ana Basin. Organochlorine compounds include DDT and its breakdown product diphenyl dicloroethylene (DDE), and chlordane. Other contaminants detected at high levels included trace elements such as lead, zinc, and arsenic. According to Belitz *et al.* (2004), 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, Kratzer, Brown, Gronberg, & Burow, 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. Diuron was detected in roughly 54% of samples, while 2,4-D was found in 12 % . The other two compounds were found much less frequently at 1% for triclopyr and <1% for linuron.

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 four herbicides assessed in this Opinion. Land use differences between sites are reflected in pesticide detections. Diuron was detected in 66.7 % of agricultural samples, but 85.7% of urban samples (Domagalski, 2000). Similarly, 2,4-D and triclopyr both had a detection rate of 19% in agricultural samples, but had higher raters, 28.6% amd 32.1% repectively, in urban samples. Linuron detections were lower and more stable, at 4.8% for agricultural and 3.6% for urban samples. Some pesticides were detected at concentrations higher than criteria for the protection of aquatic life in the smaller streams, but were diluted to safer levels in the mainstem river. Intensive agricultural activities also impact water chemistry. In the Salinas River and in areas with intense agriculture use, water hardness, alkalinity, nutrients, and conductivity are also high.

Other Land Uses in the Southwest Coast Region

Habitat Modification

The Central Valley area, including San Francisco Bay and the Sacramento and San Joaquin River Basins, has been drastically changed by development. Salmonid habitat has been reduced to 300 miles from historic estimates of 6,000 miles (CDFG, 1993). 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 & 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, Squire, Mason, & Flegal, 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 (see species distribution maps). In all, there are about 1,400 dams within the State of California, more than 5,000 miles of levees, and more than 140 aqueducts (Mount, 1995). 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 (Figure 52). 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 & Resh, 2005).

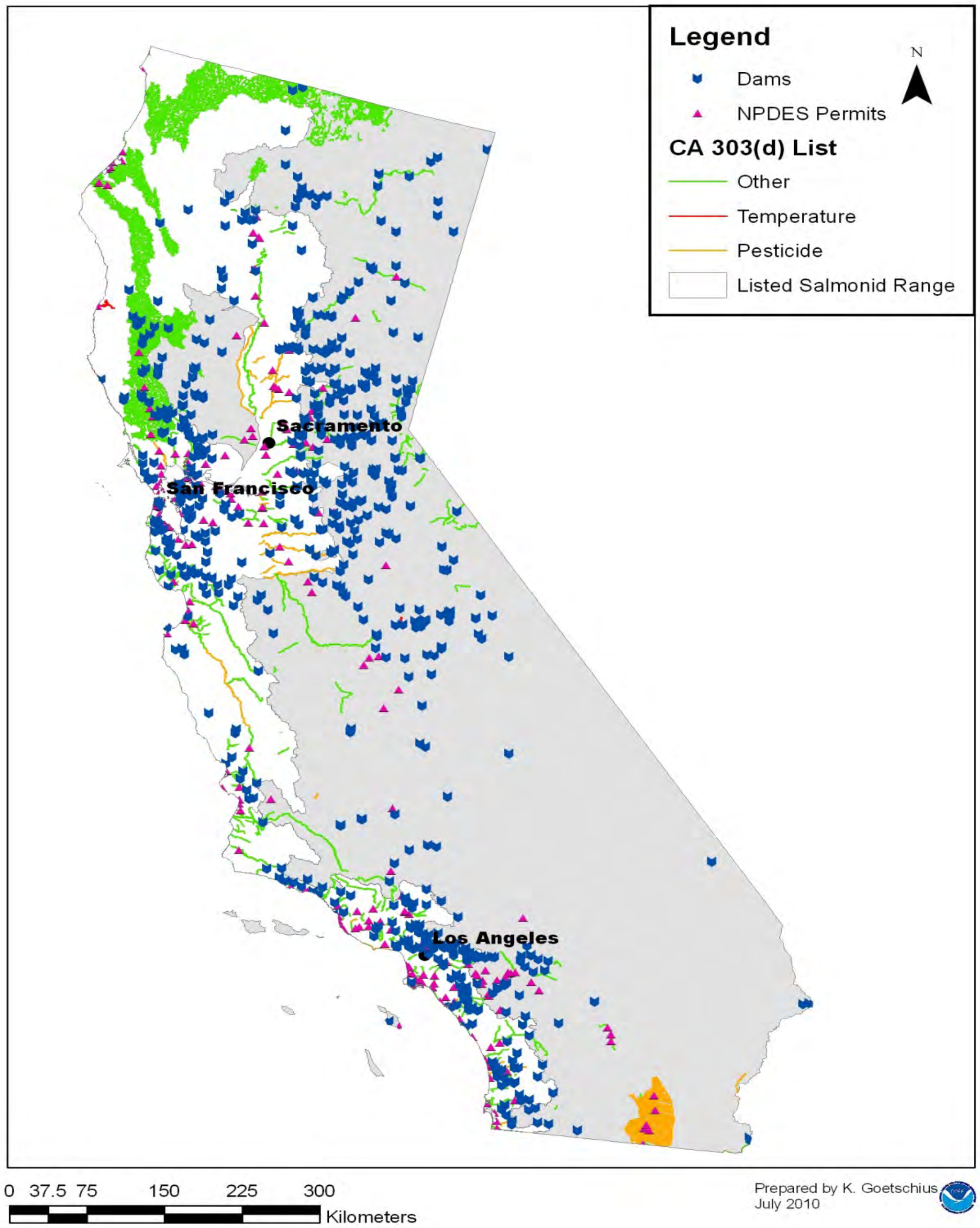


Figure 52 Southest Coast 303(d) waters, dams, and NPDEs permit sites.

Artificial Propagation

Anadromous fish hatcheries have existed in California since establishment of the McCloud River hatchery in 1872. There are nine state hatcheries: the Iron Gate (Klamath River), Mad River, Trinity (Trinity River), Feather (Feather River), Warm Springs (Russian River), Nimbus (American River), Mokelumne (Mokelumne River), and Merced (Merced River). The California Department of Fish and Game (CDFG) also manages artificial production programs on the Noyo and Eel rivers. The Coleman National Fish Hatchery, located on Battle Creek in the upper Sacramento River, is a federal hatchery operated by the USFWS. The USFWS also operates an artificial propagation program for Sacramento River winter run Chinook salmon.

Of these, the Feather River, Nimbus, Mokelumne, and Merced River facilities comprise the Central Valley Hatcheries. Over the last ten years, the Central Valley Hatcheries have released over 30 million young salmon. State and the federal (Coleman) hatcheries work together to meet overall goals. State hatcheries are expected to release 18.6 million smolts in 2008 and Coleman is aiming for more than 12 million. There has been no significant change in hatchery practices over the year that would adversely affect the current year class of fish. A new program marking 25% of the 32 million Sacramento River Fall-run Chinook smolts may provide data on hatchery fish contributions to the fisheries in the near future.

Commercial and Recreational Fishing

The region is home to many commercial fisheries. The largest in terms of total California landings in 2006 were northern anchovy, Pacific sardine, Chinook salmon, sablefish, Dover sole, Pacific whiting, squid, red sea urchin, and Dungeness crab (CDFG, 2007). 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-rn Chinook salmon returns. California State sport and commercial ocean salmon seasons were closed by the CFGC through August 28, 2009. There was a 10-day ocean sport fishery in the Klamath Management Zone (Horse Mountain to the California-Oregon border) from August 29 through September 7, 2009. A limited in-river salmon season was considered by the CFGC at its May meeting. The CFGC decided to leave open the Sacramento River between the Highway 113 bridge near Knight's Landing and just below the Lower Red Bluff (Sycamore) Boat Ramp from November 16 through December 31,

2009. The Klamath-Trinity River Basin had a salmon sport fishing season for Klamath River fall Chinook salmon that began August 15, 2009.

Non-native Species

Plants and animals that are introduced into habitats where they do not naturally occur are called non-native species. They are also known as non-indigenous, exotic, introduced, or invasive species, and have been known to affect ecosystems. Non-native species are introduced through infested stock for aquaculture and fishery enhancement, through ballast water discharge and from the pet and recreational fishing industries (http://biology.usgs.gov/s+t/noframe/x191.htm.). The Aquatic Nuisance Species Task Force suggests that it is inevitable that cultured species will eventually escape confinement and enter U.S. waterways. Non-native species were cited as a contributing cause in the extinction of 27 species and 13 subspecies of North American fishes over the past 100 years (R. R. Miller, Williams, & Williams, 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, Ode, Sowby, & Ashe, 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.

Pacific Northwest Region

This region encompasses Idaho, Oregon, and Washington and includes parts of Nevada, Montana, Wyoming, and British Columbia. In this section we discuss three major areas that support salmonid populations within the action area. They include the Columbia River Basin and its tributaries, the Puget Sound Region, and the coastal drainages north of the Columbia River (Figure 53).

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 54). Table 60, Table 61, and Table 62 show the types and areas of land use within each salmonid ESU/DPS.

Table 60. Area of land use categories within Chinook Salmon ESUs in km² where bolded numbers are totals of each category. Land cover image data were taken from Multi-Resolution Land Characteristics (MRLC) Consortium, a consortium of nine federal agencies (USGS, EPA, USFS, NOAA, NASA, BLM, NPS, NRCS, and USFWS) (NLCD, 2001). Land cover class definitions are available at: http://www.mrlc.gov/nlcd_definitions.php

Landcover Type		**Chinook Salmon**					
sub category	code	Puget Sound	Lower Columbia River	Upper Columbia River Spring Run	Snake River Fall Run	Snake River Spring/ Summer Run	Upper Willamette River
Water		**6,485**	**653**	**203**	**236**	**293**	**130**
Open Water	11	6,172	641	188	233	253	124
Perennial Snow/Ice	12	313	12	16	3	40	7
Developed Land		**5,271**	**1,861**	**847**	**543**	**974**	**2,008**
Open Space	21	1,601	649	203	401	328	632
Low Intensity	22	1,694	517	218	79	113	722
Medium Intensity	23	668	290	55	20	30	322
High Intensity	24	266	118	11	2	2	112
Barren Land	31	1,042	287	360	41	500	220
Undeveloped Land		**22,481**	**10,692**	**16,155**	**31,231**	**52,573**	**14,159**
Deciduous Forest	41	999	551	21	30	10	248
Evergreen Forest	42	14,443	6,497	8,138	18,447	27,701	9,531
Mixed Forest	43	2,526	927	7	16	4	1,130
Shrub/Scrub	52	2,415	1,598	6,100	6,315	13,618	1,940
Herbaceous	71	957	520	1,737	6,358	11,053	801
Woody Wetlands	90	648	377	92	35	96	431
Emergent Wetlands	95	492	223	59	30	92	78
Agriculture		**1,447**	**825**	**964**	**5,557**	**4,316**	**5,972**
Hay/Pasture	81	1,188	547	327	59	456	3,617
Cultivated Crops	82	258	278	636	5,497	3,860	2,355
TOTAL (inc. open water)		35,683	14,031	18,168	37,566	58,157	22,269
TOTAL (w/o open water)		29,511	13,390	17,981	37,331	57,904	22,146

Table 61. Area of land use categories within chum and coho ESUs in km². Land cover image data were taken from Multi-Resolution Land Characteristics (MRLC) Consortium, a consortium of nine federal agencies (USGS, EPA, USFS, NOAA, NASA, BLM, NPS, NRCS, and USFWS) (NLCD, 2001). Land cover class definitions are available at: http://www.mrlc.gov/nlcd_definitions.php

Landcover Type		**Chum Salmon**		**Coho Salmon**		**Sockeye Salmon**	
sub category	code	Hood Canal Summer Run	Columbia River	Lower Columbia River	Oregon Coast	Ozette Lake	Snake River
Water		**755**	**656**	**687**	**200**	**30**	**36**
Open Water	11	704	655	675	200	30	19
Perennial Snow/Ice	12	51	1	12	0	0	18
Developed Land		**403**	**1,684**	**1,990**	**1,807**	**3**	**15**
Open Space	21	134	605	708	1,107	1	3
Low Intensity	22	77	463	563	163	0	2
Medium Intensity	23	20	258	305	49	0	0
High Intensity	24	6	110	124	20	0	0
Barren Land	31	166	247	290	467	2	9
Undeveloped Land		**3,324**	**8,198**	**13,254**	**24,589**	**195**	**1,259**
Deciduous Forest	41	97	548	575	418	3	0
Evergreen Forest	42	2,477	4,294	8,487	14,943	158	755
Mixed Forest	43	200	892	999	4,126	3	0
Shrub/Scrub	52	299	1,353	1,982	3,134	14	185
Herbaceous	71	61	363	386	263	8	269
Woody Wetlands	90	56	222	225	226	8	16
Emergent Wetlands	95	133	526	600	1,478	1	34
Agriculture		**66**	**746**	**1,028**	**925**	**0**	**13**
Hay/Pasture	81	64	533	680	860	0	12
Cultivated Crops	82	2	213	348	64	0	1
TOTAL (inc. open water)		4,548	11,284	16,959	27,520	228	1,323
TOTAL (w/o open water)		3,843	10,628	16,284	27,320	199	1,304

Table 62. Area of land use categories within sockeye ESUs and steelhead DPSs in km² where bolded numbers are totals for each category. Land cover image data were taken from Multi-Resolution Land Characteristics (MRLC) Consortium, a consortium of nine federal agencies (USGS, EPA, USFS, NOAA, NASA, BLM, NPS, NRCS, and USFWS) (NLCD, 2001). Land cover class definitions are available at: http://www.mrlc.gov/nlcd_definitions.php

Landcover		Steelhead					
sub category	Code	Puget Sound	Lower Columbia River	Upper Willamette River	Middle Columbia River	Upper Columbia River	Snake River
Water		**6,485**	**262**	**62**	**588**	**375**	**327**
Open Water	11	6,172	250	62	575	359	285
Perennial Snow/Ice	12	313	12	0	13	16	42
Developed Land		**5,271**	**1,601**	**1,278**	**2,304**	**1,092**	**1,205**
Open Space	21	1,601	518	382	1,276	343	515
Low Intensity	22	1,694	506	513	627	294	144
Medium Intensity	23	668	287	231	192	80	40
High Intensity	24	266	116	75	25	13	3
Barren Land	31	1,042	174	77	183	361	504
Undeveloped Land		**22,481**	**10,339**	**6,942**	**53,790**	**19,621**	**67,839**
Deciduous Forest	41	999	382	171	54	25	35
Evergreen Forest	42	14,443	7,023	4,133	18,347	8,223	39,556
Mixed Forest	43	2,526	611	791	41	7	17
Shrub/Scrub	52	2,415	1,589	994	32,089	9,351	15,644
Herbaceous	71	957	398	519	2,752	1,823	12,361
Woody Wetlands	90	648	244	292	217	109	116
Emergent Wetlands	95	492	93	43	291	81	111
Agriculture		**1,447**	**927**	**4,373**	**12,771**	**3,684**	**6,690**
Hay/Pasture	81	1,188	605	2,529	863	448	463
Cultivated Crops	82	258	322	1,844	11,908	3,236	6,227
TOTAL (inc. open water)		35,683	13,128	12,655	69,453	24,771	76,061
TOTAL (w/o open water)		29,511	12,878	12,593	68,878	24,411	75,777

Pesticide Reduction Programs in the Pacific Northwest Region

When using any of the six 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. All three states have some limitations on the use of pesticides as a part of their Administrative codes. Most are regulations are focused on chemical use in forestry applications. Table 63 summarizes the existing legislation in the North West.

Table 63. Summary of State-level limitations on pesticide use in Idaho, Oregon, and Washington. All materials are available on organization web-sites.

State	Limitation	Source	Code
Oregon	Forestry Aerial Application: No fungicides applied by aircraft within 300 feet of significant wetlands, type F streams (with salmonids or other game fish), large lakes, areas of other lakes used by fish, or within 60 feet of other perennial streams	Dept. of Forestry	629-620-0400
Oregon	Forestry Aerial: No chemicals applied by aircraft within 60 feet of significant wetlands, type F streams , large lakes, and areas of other lakes used by fish	Dept. of Forestry	629-620-0400
Oregon	Forestry Ground: No chemicals applied within 10 feet of significant wetlands, type F and D streams , large lakes, areas of other lakes used by fish, and areas of standing open water larger than one quarteracre at the time of application	Dept. of Forestry	629-620-0400
Oregon	Forestry: Requires additional state permit for use of isopropyl ester of 2,4-D or any other ester of equal or higher volatility with regard to plant damage	State Forest Laws	634.372

State	Limitation	Source	Code
Washington	Rights of Way: 2,4-D amine formulations and Triclopyr Ester cannot be used within 60 feet of water; Diuron cannot be used in western Washington, and not within 60 feet of water in eastern Washington	Washington State Dept. of Transportation	NA
Washington	Forestry Ground Application: application with power equipment is prohibited within the core and inner zone, channel migration zone of Type S and F Waters; Operators shall maintain a 25 foot no application buffer strip around Type A or B Wetlands and on all sides of all other surface waters.	Washington Administrative Code Forest Practices Board	222-38-020
Washington	Forestry Hand Application: No pesticides may be applied within the core zone, channel migration zone of Type S and F Waters ; Pesticides must be applied to specific targets, such as vegetation, trees, stumps, etc.	Washington Administrative Code Forest Practices Board	222-38-020
Washington	Forestry Aerial Application: mandatory buffers depending on nozzle type, application height, weather conditions, and width of stream; Ranges from 10 to 325 feet	Washington Administrative Code Forest Practices Board	222-38-021 222-38-022
Idaho	Forestry Aerial Application: must leave a buffer of untreated land (minimum off 100 ft) on each side of all Class I streams, flowing Class II streams and other areas of open water.	Idaho Administrative Code, Dept of Lands, Forest Practices Act	20.02.01
Idaho	Forestry Ground Application: must leave a buffer of untreated land (minimum off 100 ft) on each side of all Class I streams, flowing Class II streams and other areas of open water.	Idaho Administrative Code, Dept of Lands, Forest Practices Act	20.02.01
Idaho	Forestry Hand Application: apply only to specific targets	Idaho Administrative Code, Dept of Lands, Forest Practices Act	20.02.01

State	Limitation	Source	Code
Idaho	Home and garden: use of high volatile liquid ester formulations of 2,4-D prohibited - both homeowner use and professional applications to home/garden locations	Idaho Administrative Code, Dept of Agriculture, Pesticide & Chemigation Use & Application Rules	02.03.03 500.01
Idaho	Home and garden: Low volatile liquid ester formulations of 2,4-D; 2,4-DP; MCPA and MCPB shall not be applied around any home or garden between May 1 and October 1 of any year or at any time when air temperature exceeds eighty (80) degrees Fahrenheit.	Idaho Administrative Code, Dept of Agriculture, Pesticide & Chemigation Use & Application Rules	02.03.03 500.02
Idaho	Agriculture Aerial Application: No aircraft pilot shall apply high volatile ester formulations of 2,4-D in Latah, Nez Perce, and Clearwater Counties, or within five miles of a susceptible crop or hazard area in any other county in Idaho	Idaho Administrative Code, Dept of Agriculture, Pesticide & Chemigation Use & Application Rules	02.03.03 550.01
Idaho	Agriculture Aerial Application: No aircraft pilot shall apply low volatile ester formulations of 2,4-D in Latah, Nez Perce, and Clearwater Counties unless ambient air temperatures are not above or expected to exceed eighty-five degrees Fahrenheit within twenty-four hours of the expected application time, or within one miles of a susceptible crop or hazard area in any other county in Idaho	Idaho Administrative Code, Dept of Agriculture, Pesticide & Chemigation Use & Application Rules	02.03.03 550.02

The Idaho State Department of Agriculture has published a BMP guide for pesticide use. The BMPs include eight “core” voluntary measures that will prevent pesticides from leaching into soil and groundwater. These measures include applying pest-specific controls, being aware of the depth to ground water, and developing an Irrigation Water Management Plan.

Oregon has PURS legislation that requires all agricultural uses of registered pesticides be reported. In this case “agricultural” use includes applications to parks, golf courses, and most livestock uses. Oregon requires reporting if application is part of a business, for a government agency, or in a public place. However, the Governor of Oregon has suspended the PURS program until January 2013 due to budget shortages.

Oregon has also implemented a voluntary program. The Pesticide Stewardship Partnerships (PSP) program began in 1999 through the Oregon Department of Environmental Quality. The PSP’s goal is to involve growers and other stakeholders in water quality management at a local level. Effectiveness monitoring is used to provide feedback on the success of mitigation measures. As of 2006, there were six pilot PSPs planned or in place. Early results from the first PSPs in the Columbia Gorge Hood River and in Mill Creek demonstrate reductions in chlorpyrifos and diazinon levels and detection frequencies. DEQ’s pilot programs suggest that PSPs can help reduce contamination of surface waters.

Oregon is in the process of developing a Pesticide Management Plan for Water Quality Protection, as required under FIFRA. This plan describes how government agencies and stakeholders will collaboratively reduce pesticides in Oregon water supplies. The PSP program is a component of this plan, and will provide information on the effectiveness of mitigation measures.

Washington State has a Surface Water Monitoring Program that looks at pesticide concentrations in some salmonid bearing streams and rivers. The program was initiated in 2003 and now monitors four areas. Three of these were chosen due to high overlap with agriculture: the Skagit-Samish watershed, the Lower Yakima Watershed, and the Wenatchee and Entiat watersheds. The final area, in the Cedar-Sammamish watershed, is an urban location, intended to look at runoff in a non-agriculture setting. It was chosen due to detection of pesticides coincident with pre-spawning mortality in coho salmon. The Surface Water Monitoring program is relatively new and will continue to add watersheds and testing for additional pesticides over time.

Washington State also has a voluntary program that assists growers in addressing water rights issues within a watershed. Several watersheds have elected to participate, forming Comprehensive Irrigation District Management Plans (CIDMPs). The CIDMP is a collaborative process between government and landowners and growers; the parties determine how they will ensure growers get the necessary volume of water while also guarding water quality. This structure allows for greater flexibility in implementing mitigation measures to comply with both the CWA and the ESA.

The Columbia Gorge Fruit Growers Association is a non-profit organization dedicated to the needs of growers in the mid-Columbia area. The association brings together over 440 growers and 20 shippers of fruit from Oregon and Washington. It has issued a BMP handbook for OPs, including information on alternative methods of pest control. The mid-Columbia area is of particular concern, as many orchards are in close proximity to streams.

Stewardship Partners is a non-profit organization in Washington State that works to build partnerships between landowners, government, and non-profit organizations. In large part, its work focuses on helping landowners to restore fish and wildlife habitat while maintaining the economic viability of their farmland. Projects include restoring riparian areas, reestablishing floodplain connectivity, and removing blocks to fish passage. Another current project is to promote rain gardens as a method of reducing surface water runoff from developed areas. Rain gardens mimic natural hydrology, allowing water to collect and infiltrate the soil.

Stewardship Partners also collaborates with the Oregon-based Salmon-Safe certification program. Salmon-Safe is an independent eco-label recognizing organizations who have adopted conservation practices that help restore native salmon habitat in Pacific Northwest rivers and streams. These practices protect water quality, fish and wildlife habitat, and overall watershed health. While the program began with a focus on agriculture, it has since expanded to include industrial and urban sites as well. The certification process includes pesticide restrictions. Salmon-Safe has produced a list of "high risk" pesticides which, if used, would prevent a site from becoming certified. If a grower wants an exception, they must provide written documentation that demonstrates a clear need for use of the pesticide, that no safer alternatives exist, and that the method of application (such as timing, location, and amount used) represents a negligible risk to water quality and fish habitat. Bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, naled, and phosmet are all on the high risk list. Over 250 farms and businesses currently have the Salmon-Safe certification.

In addition to pesticide usage for agriculture, this land use further affects available salmonid aquatic habitat. The amount and extent of water withdrawals or diversions for agriculture impact streams and their inhabitants via reduced water flow/velocity and dissolved oxygen levels. These impacts are described below.

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 64 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 64. Select tributaries of the Columbia River (Carter & 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, Hauer, Gregory, & Synder, 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 65 for a summary of land uses and population densities in several subbasins within the Columbia River watershed [data from (Stanford, et al., 2005)].

Table 65. 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.

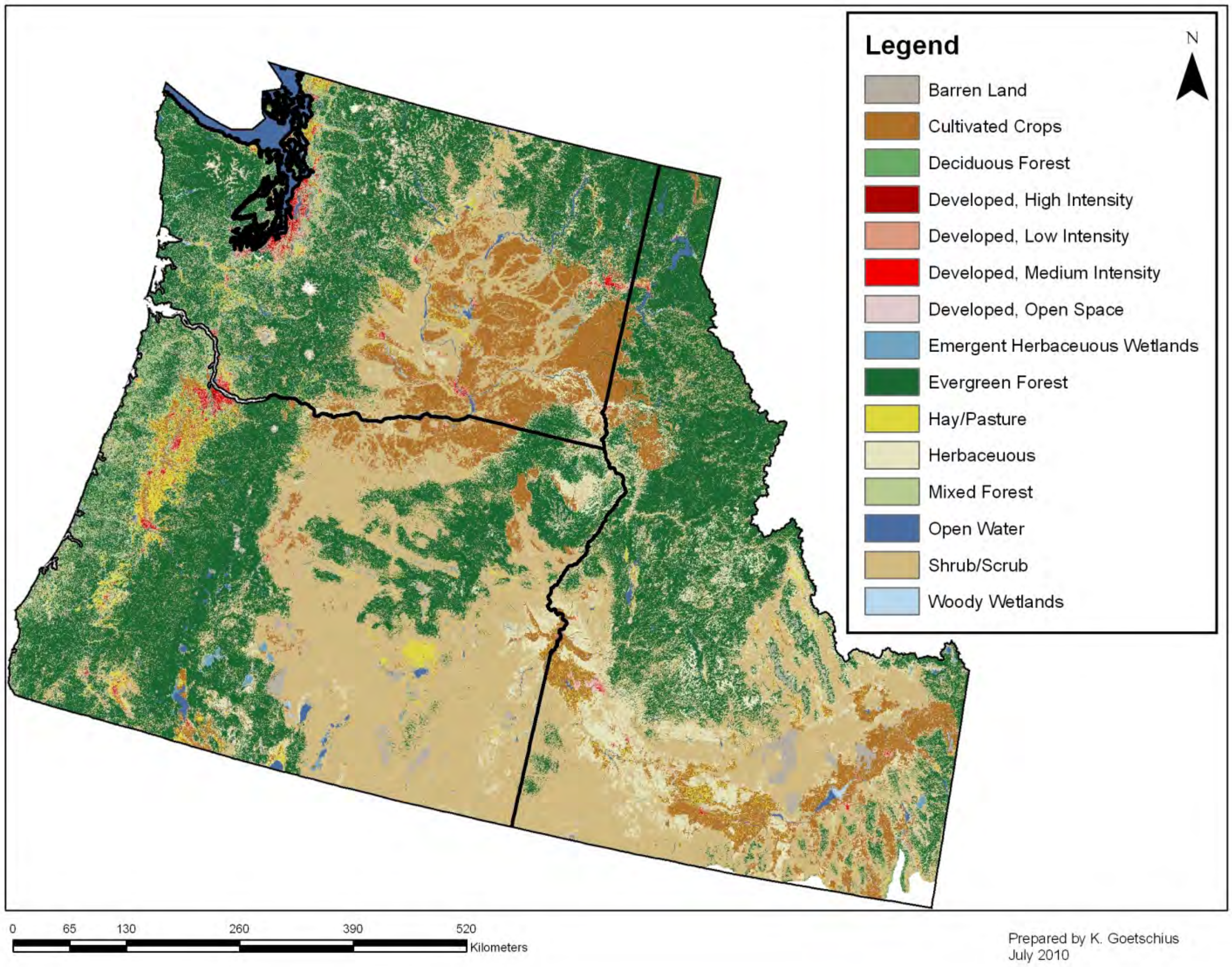


Figure 53. Pacific Northwest Landuse.

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 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 (T. F. Cuffney, M. R. Meador, S. D. Porter, & M. E. Gurtz, 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 (including 2,4-D and diuron), 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. Diuron was not detected in samples from the Yakama River, but was found in 23% of tributary samples. The exception to this trend was 2,4-D, which was found in 67% of Yakima River samples, but only 59% of tributary samples.

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). 2,4-D was detected in over 80% of samples, and diuron was detected in over 30% of samples. Additionally, roughly 30% of samples contained both a.i.s. Linuron was screened for but not detected. The most frequently detected pesticides in the Yakima River Basin are total DDTs, dichloro-diphenyl-dichloroethane (DDD), and dieldrin (Fuhrer, et al., 2004; A. Johnson & Newman, 1983; Joy, 2002; Joy & Madrone, 2002). 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 (J. Ebbert & 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. Linuron was the only chemical monitored for in this study that was assessed in this Opinion; it was only detected once.

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). All four herbicides addressed in this Opinion were detected in samples from this region, though at different frequencies (Williamson, et al., 1998). 2,4-D was detected in 27% of samples, diuron in 20%, linuron in 3%, and triclopyr <1%.

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). Diuron was found in 53% of samples with a maximum concentration of 14 µg/L. The other herbicides assessed in this Opinion were detected less frequently: 2,4-D in 12%, triclopyr in 9%, and linuron in 1% of samples. 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 abut 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. All four herbicides addressed in this Opinion were detected in samples throughout the study area (Table 66).

Table 66. Summaraized detection information from (Carpenter, et al., 2008). Note that percentages aren't comparable because results were pooled from multiple sources.

Pesticide	Percent Detection	Tributary Detections	Clackamas River Detections
2,4-D	35	28	4
Triclopyr	22	20	1
Diuron	44	22	15
Linuron	2	1	0

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. 2,4-D was present in 14% of samples, while diuron was found in 1%. Triclopyr and linuron were screened for but not detected (Clark, et al., 1998).

Hood River Basin

The Hood River Basin ranks fourth in the state of Oregon in total agricultural pesticide usage (J. Jenkins, Jepson, Bolte, & Vache, 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 67).

Table 67. Summaraized 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	Organo-phosphate	22,294
Metam-Sodium	Carbamate	17,114
Phosmet	Organo-phosphate	15,919
Chlorpyrifos	Organo-phosphate	14,833

The Hood River basin contains approximately 400 miles of perennial stream channel, of which an estimated 100 miles is accessible to anadromous fish. These channels are important rearing and spawning habitat for salmonids, making pesticide drift a major concern for the area.

Other Land Use in the Pacific Northwest Region

Urban and Industrial Development

The largest urban area in the basin is the greater Portland metropolitan area, located at the mouth of the Willamette River. Portland's population exceeds 500,000 (Hinck, 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 & 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 & Whitney, 1979). Similarly, over one third (2,000 km) of coho salmon habitat is no longer accessible (T. P. 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 (P. D. Anderson, Dugger, & Burke, 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 (P. D. Anderson, et al., 2007; Stanford, et al., 2005). 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 54). 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, Farris, Moore, Smith, & Cooper, 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, BPA, & Reclamation, 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.

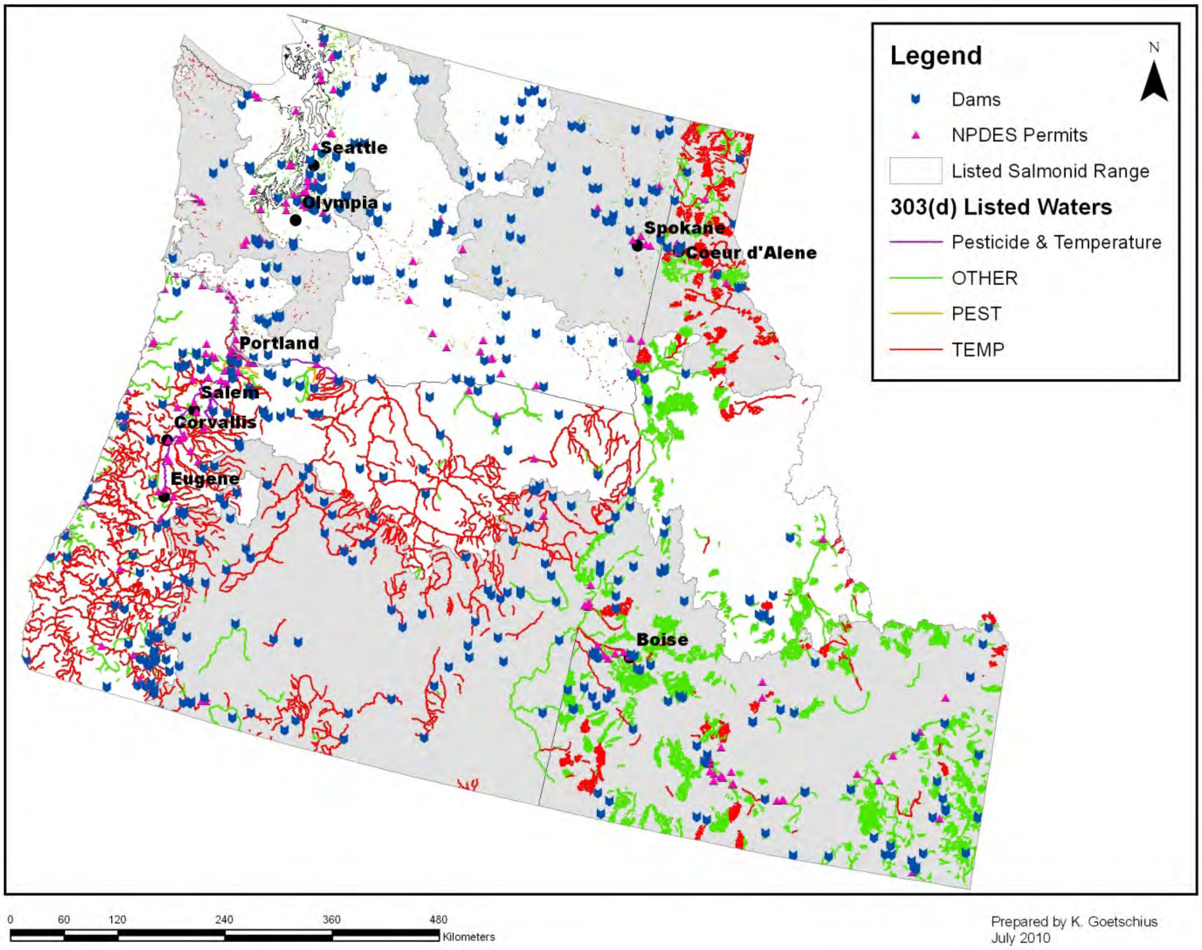


Figure 54. Pacific Northwest 303(d) waters, dams, and NPDES permit sites.

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 (T. J. Beechie, Liermann, Beamer, & Henderson, 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, Waknitz, Maynard, Milner, & Mahnken, 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; Reisenbichler, 1997; Riggs, 1990), disease transfer, increased competition for food, habitat, or mates, increased predation, altered migration, and the displacement of natural fish (K. D. Fresh, 1997; Hard, et al., 1992; Steward & Bjornn, 1990). 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 (ODFW, 2007; Washington Department of Fish and Wildlife (WDFW), 2005). The program is designed to achieve HSRG objectives related to controlling the number of hatchery-origin fish on the spawning grounds and in the hatchery broodstock.

Coho salmon hatchery programs in the lower Columbia have been tasked to compensate for impacts of fisheries. However, hatchery programs in the LCR have not operated specifically to conserve LCR coho salmon. These programs threaten the viability of natural populations. The long-term domestication of hatchery fish has eroded the fitness of these fish in the wild and has reduced the productivity of wild stocks where significant numbers of hatchery fish spawn with wild fish. Large numbers of hatchery fish have also contributed to more intensive mixed stock fisheries. These programs largely overexploited wild populations weakened by habitat degradation. Most LCR coho salmon populations have been heavily influenced by hatchery production over the years.

Commercial, Recreational, and Subsistence Fishing

Despite regulated fishing programs for salmonids, listed salmonids are also caught as bycatch. There are several approaches under the ESA to address tribal and state take of ESA-listed species that may occur as a result of harvest activities. Section 10 of the ESA provides for permits to operate fishery harvest programs. ESA section 4(d) rules provide exemptions from take for resource, harvest, and hatchery management plans. Furthermore, there are several treaties that have reserved the right of fishing to tribes in the North West Region.

Management of salmon fisheries in the Columbia River Basin is a cooperative process involving federal, state, and tribal representatives. The Pacific Fishery Management Council sets annual fisheries in federal waters from three to 200 miles off the coasts of Washington, Oregon, and California. Salmon and steelhead fisheries in the Columbia River and its tributaries are co-managed by the states of Washington, Oregon, Idaho, four treaty tribes, and other tribes that traditionally have fished in those waters. A federal court oversees Columbia River harvest management through the U.S. v. Oregon proceedings. Inland fisheries are those in waters within state boundaries, including those extending out three miles from the coasts. The states of Oregon, Idaho, and Washington issue salmon fishing licenses for theses areas.

Fisheries in the Columbia River basin are managed within the winter/spring, summer, and fall seasons. There are Treaty Indian and non-Treaty fisheries which are managed subject to state and tribal regulation, consistent with provisions of a U.S. v. Oregon 2008 agreement. The winter/spring season extends from January 1 to June 15. Commercial, recreational, and ceremonial subsistence fisheries target primarily upriver spring Chinook stocks and spring Chinook salmon that return to the Willamette and lower Columbia River tributaries. Some steelhead are also caught incidentally in these fisheries. The summer season extends from June 16 to July 31. Commercial, recreational, and ceremonial and subsistence fisheries are managed primarily to provide harvest opportunity directed at unlisted UCR summer Chinook salmon. Summer fisheries are constrained primarily by the available opportunity for UCR summer Chinook salmon, and by specific harvest rate limits for SR sockeye salmon and harvest rate limits on steelhead in non-Treaty fisheries. Fall season fisheries begin on August 1 and end on December 31. Commercial, recreational, and ceremonial and subsistence fisheries target primarily harvestable hatchery and natural origin fall Chinook and coho salmon. Fall season fisheries are constrained by specific ESA related harvest rate limits for listed SR fall Chinook salmon, and SR steelhead.

Treaty Indian fisheries are managed subject to the regulation of the Columbia River Treaty Tribes. They include all mainstem Columbia River fisheries between Bonneville Dam and McNary Dam, and any fishery impacts from tribal fishing that occurs below Bonneville Dam. Tribal fisheries within specified tributaries to the Columbia River are included.

Non-Treaty fisheries are managed under the jurisdiction of the states. These include mainstem Columbia River commercial and recreational salmonid fisheries at the river mouth of Bonneville Damn, designated off channel Select Area fisheries, mainstem recreational fisheries between Bonneville Dam and McNary Dam, recreational fisheries between McNary Dam and Highway 305 Bridge in Pasco, Washington, recreational and Wanapum tribal spring Chinook fisheries from McNary Dam to Priest Rapids Dam, and recreational spring Chinook fisheries in the Snake River upstream to Lower Granite Dam.

Archeological records indicate that indigenous people caught salmon in the Columbia River more than 7,000 years ago. One of the most well known tribal fishing sites within the basin was located near Celilo Falls, an area in the lower river that has been occupied by Dalles Dam since 1957. Salmon fishing increased with better fishing methods and preservation techniques, such as drying and smoking. Salmon harvest substantially increased in the mid-1800s with canning techniques. Harvest techniques also changed over time, from early use of hand-held spears and dip nets, to riverboats using seines and gill nets. Harvest techniques eventually transitioned to large ocean-going vessels with trolling gear and nets and the harvest of Columbia River salmon and steelhead from California to Alaska (T. J. 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 (T. J. 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 (T. J. Beechie, et al., 2005).

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

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 (Kruckeberg, 1991; Wydoski & Whitney, 1979). 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 (M. H. Ruckelshaus & 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, O'Neill, & Scholz, 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 (M. H. Ruckelshaus & 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 (M. H. Ruckelshaus & McClure, 2007). Projected land cover changes indicate that trends are likely to continue over the next several decades with population changes (M. H. Ruckelshaus & 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 68). Other ions generally elevated in urban streams include calcium, sodium, potassium, magnesium, and chloride ions where sodium chloride is used as the principal road deicing salt (Paul & Meyer, 2001). The combined effect of increased concentrations of ions in streams is the elevated conductivity observed in most urban streams.

Table 68. Examples of Water Quality Contaminants in Residential and Urban Areas.

Contaminant groups	Select constituents	Select example(s)	Source and Use Information
Fertilizers	Nutrients	Phosphorus Nitrogen	lawns, golf courses, urban landscaping
Heavy Metals	Pb, Zn, Cr, Cu, Cd, Ni, Hg, Mg	Cu	brake pad dust, highway and parking lot runoff, rooftops
Pesticides including- Insecticides (I) Herbicides (H) Fungicides (F) Wood Treatment chemicals (WT) Legacy Pesticides (LP) Other ingredients in pesticide formulations (OI)	Organophosphates (I) Carbamates (I) Organochlorines (I) Pyrethroids (I) Triazines (H) Chloroacetanilides (H) Chlorophenoxy acids (H) Triazoles (F) Copper containing fungicides (F) Organochlorines (LP) Surfactants/adjuvants (OI)	Chlorpyrifos (I) Diazinon (I) Carbaryl (I) Atrazine (H) Esfenvalerate (I) Creosote (WT) DDT (LP) Copper sulfate (F) Metalaxyl (F) Nonylphenol (OI)	golf courses, right of ways, lawn and plant care products, pilings, bulkheads, fences
Pharmaceuticals and personal care products	Natural and synthetic hormones soaps and detergents	Ethinyl estradiol Nonylphenol	hospitals, dental facilities, residences, municipal and industrial waste water discharges
Polyaromatic hydrocarbons (PAHs)	Tricyclic PAHs	Phenanthrene	fossil fuel combustion, oil and gasoline leaks, highway runoff, creosote-treated wood
Industrial chemicals	PCBs PBDEs Dioxins	Penta-PBDE	utility infrastructure, flame retardants, electronic equipment

Many other metals have been found in elevated concentrations in urban stream sediments including arsenic, iron, boron, cobalt, silver, strontium, rubidium, antimony, scandium, molybdenum, lithium, and tin (Wheeler, Angermeier, & Rosenberger, 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 (B.C. 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, Collier, & Scholz, 2004). PAHs tend to adsorb to organic or inorganic matter in sediments, where they can be trapped in long-term reservoirs (L. Johnson, Collier, & Stein, 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 (Meador, Stein, Reichert, & Varanasi, 1995; Varanasi, Stein, & Nishimoto, 1989; Varanasi et al., 1992).

PAHs and their metabolites in invertebrate prey can be passed on to consuming fish species, PAHs are metabolized extensively in vertebrates, including fishes (L. 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 (L. 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, Chrzastowski, & Helgerson, 1980). Tidal wetlands in Puget Sound amount to roughly 18% of their historical extent (Collins & 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 & 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; M. H. Ruckelshaus & 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 (M. H. Ruckelshaus & 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 69.

Table 69. 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. 2,4-D, triclopyr, diuron, and linuron were all detected in Puget sound samples (Ebbert, 2000). NAWQA results reported that the herbicides atrazine, prometon, simazine and tebuthiuron were the most frequently detected herbicides in surface and ground water (Bortleson & 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 & 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 & 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 & 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 & 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 (M. H. Ruckelshaus & 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 (NMA, 2007; Palmisano, Ellis, & Kaczynski, 1993). 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 (T. P. 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 (A. R. 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 (T. P. 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 (M. H. Ruckelshaus & 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 (M. H. Ruckelshaus & 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 (M. H. Ruckelshaus & McClure, 2007; Wunderlich, Winter, & Meyer, 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 theses areas. Adult salmon returning to Washington migrate through both U.S. and Canadian waters and are harvested by fishermen from both countries. The 1985 Pacific Salmon Treaty helps fulfill conservation goals for all members and is implemented by the eight-member bilateral Pacific Salmon Commission. The Commission does not regulate salmon fisheries, but provides regulatory advice.

Most of the commercial landings in the region are groundfish, Dungeness crab, shrimp, and salmon. Many of the same species are sought by Tribal fisheries and by charter and recreational anglers. Nets and trolling are used in commercial and Tribal fisheries. Recreational anglers typically use hook and line, and may fish from boat, river bank, or docks. Entanglement of marine mammals in fishing gear is not uncommon and can lead to mortality or serious injury.

Harvest impacts on Puget Sound Chinook salmon populations average 75% in the earliest five years of data availability and have dropped to an average of 44% in the most recent five-year period (T. P. 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.

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; Carter & Resh, 2005; Kagan, Hak, Csuti, Kiilsgaard, & Gaines, 1999).

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 (S. 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 & 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 (S. 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, McIntosh, & Bowles, 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 70 summarizes the change in area of tidal wetlands for several Oregon estuaries (J. W. Good, 2000).

Table 70. Change in total area (acres^2) of tidal wetlands in Oregon (tidal marshes and swamps) due to filling and diking between 1870 and 1970 (J. W. 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 (NMA, 2007; Palmisano, et al., 1993). 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 & 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 54). 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 theses areas.

Most commercial landings in the region are groundfish, Dungeness crab, shrimp, and salmon. Many of the same species are sought by Tribal fisheries, as well as by charter, and recreational anglers. Nets and trolling are used in commercial and Tribal fisheries.

Recreational anglers typically use hook and line and may fish from boat, river bank, or docks.

Integration of the Environmental Baseline on Listed Resources

Collectively, the components of the environmental baseline for the action area include sources of natural mortality as well as influences from natural oceanographic and climatic features in the action area. Climatic variability may affect the growth, reproductive success, and survival of listed Pacific salmonids in the action area. Temperature and water level changes may lead to: (1) Reduced summer and fall stream flow, leading to loss of spawning habitat and difficulty reaching spawning beds; (2) increased winter flooding and disturbance of eggs; (3) changes in peak stream flow timing affecting juvenile migration; and (4) rising water temperature may exceed the upper temperature limit for salmonids at 64ºF (18ºC) (JISAO, 2007). 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 2,4-D, triclopyr BEE, diuron, linuron, captan and chlorothalonil in California, Idaho, Oregon, and Washington. These salmonids are and have been exposed to the components of the environmental baseline for decades. The activities discussed above have some level of effect on all 28 ESUs/DPSs in the proposed action area. They have also eroded the quality and quantity of salmonid habitat – including designated critical habitat. We expect the combined consequences of those effects, including impaired water quality, temperature, and reduced prey abundance, may increase the vulnerability and susceptibility of overall fish health to disease, predation, and competition for available suitable habitat and prey items. The continued trend of anthropogenic impairment of water quality and quantity on Pacific salmonids and their habitats may further compound the declining status and trends of listed salmonids, unless measures are implemented to reverse this trend.

Effects of the Proposed Action to Threatened and Endangered Pacific Salmonids

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

Exposure Analysis

In this section, we identify and evaluate potential exposure of salmonids to the stressors of the action (Figure 55). We begin by presenting general life history information of vulnerable life stages of Pacific salmon and steelhead. Next, we present a general discussion of the physical and chemical properties of the six a.i.s 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 the salmon habitat with the stressors of the action by comparing the distribution of sites authorized for pesticide use by product labeling to the distribution of each species and their designated critical habitat. To further characterize exposure where co-occurrence exists, we summarize EPA exposure estimates presented in the six BEs, present additional exposure estimates for shallow floodplain habitats utilized by salmonids, and summarize the available water quality monitoring data. Finally, we conclude with a summary of anticipated ranges of exposure when pesticide use is proximate to salmon habitats and characterize the uncertainty contained in this analysis. Because the ESA section 7 consultation process is intended to insure that the agency action is not likely to jeopardize listed species or destroy or adversely modify critical habitat, NMFS considers a variety of scenarios in addition to those presented in EPA's BEs. These scenarios provide estimates for the range of habitats used by listed salmonids.

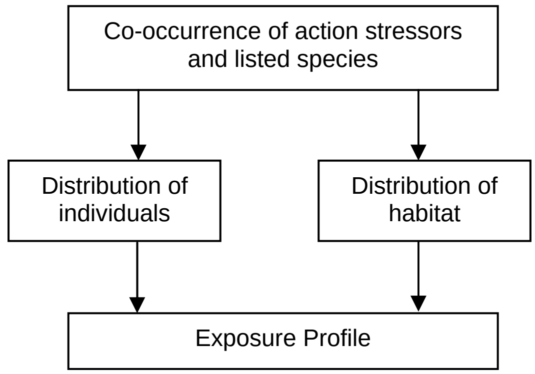


Figure 55. Exposure analysis.

Threatened and Endangered Pacific Salmonids use of Aquatic Habitats

Within the Status Section we discussed salmonid lifecycles, life histories, and the use and significance of aquatic habitats. Listed salmonids occupy a variety of aquatic habitats that range from shallow, low-flow freshwaters to open reaches of the Pacific Ocean. All listed Pacific salmonid species use freshwater, estuarine, and marine habitats at some point during their life. The temporal and spatial use of habitats by salmonids depends on the species and the individuals' life history and life stage. General life history descriptions describing use of aquatic habitats is provided below in Table 71. Additionally information on timing of presence of the ESUs/DPSs in the habitats is presented in *Appendix 6*. Many species migrate hundreds or thousands of miles during their lifetime, increasing the likelihood that they will come in contact with aquatic habitats contaminated with pesticides.

Table 71. General life-histories of Pacific salmonids.

Species (number of listed ESUs or DPSs)	General Life History Descriptions		
	Spawning Migration	Spawning Habitat	Juvenile Rearing and Migration
Chinook (9)	Mature adults (usually 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 to as far as 1,200 miles from the sea. Chinook salmon migrate and spawn in four distinct runs (spring, fall, summer, and winter). Chinook salmon are semelparous[1].	Generally spawn in the middle and upper reaches of main stem rivers and larger tributary streams.	The alevin life stage primarily resides just below the gravel surface until they approach or reach the fry stage. Immediately after leaving the gravel, fry distribute to habitats that provide refuge from fast currents and predators. Juveniles exhibit two general life history types: Ocean-type fish migrate to sea in their first year, usually within six months of hatching. Ocean-type juveniles may rear in the estuary for extended periods. Stream-type fish migrate to the sea in the spring of their second year.
Coho (4)	Mature adults (usually two to four years old) enter the rivers in the fall. The timing varies depending on location and other variables. Coho salmon are semelparous.	Spawn throughout smaller coastal tributaries, usually penetrating to the upper reaches to spawn. Spawning takes place from October to March.	Following emergence, fry move to shallow areas near stream banks. As fry grow they distribute up and downstream and establish territories in small streams, lakes, and off-channel ponds. Here they rear for 12-18 months. In the spring of their second year juveniles rapidly migrate to sea. Initially, they remain in nearshore waters of the estuary close to the natal stream following downstream migration.
Chum (2)	Mature adults (usually three to four years old) enter rivers as early as July, with arrival on the spawning grounds occurring from September to January. Chum salmon are semelparous.	Generally spawn from just above tidewater in the lower reaches of mainstem rivers, tributary stream, or side channels to 100 km upstream.	The alevin life stage primarily resides just below the gravel surface until they approach or reach the fry stage. Immediately after leaving the gravel, swim-up fry migrate downstream to estuarine areas. They reside in estuaries near the shoreline for one or more weeks before migrating for extended distances, usually in a narrow band along the Pacific Ocean's coast.

Species	General Life History Descriptions		
(number of listed ESUs or DPSs)	Spawning Migration	Spawning Habitat	Juvenile Rearing and Migration
Sockeye (2)	Mature adults (usually four to five years old) begin entering rivers from May to October. Sockeye are semelparous.	Spawn along lakeshores where springs occur and in outlet or inlet streams to lakes.	The alevin life stage primarily resides just below the gravel surface until they approach or reach the fry stage. Immediately after leaving the gravel, swim-up fry migrate to nursery lakes or intermediate feeding areas along the banks of rivers. Populations that migrate directly to nursery lakes typically occupy shallow beach areas of the lake's littoral zone; a few cm in depth. As they grow larger they disperse into deeper habitats. Juveniles usually reside in the lakes for one to three years before migrating to off shore habitats in the ocean. Some are residual, and complete their entire lifecycle in freshwater.
Steelhead (11)	Mature adults (typically three to five years old) may enter rivers any month of the year, and spawn in late winter or spring. Migration in the Columbia River system extends up to 900 miles from the ocean in the Snake River. Steelhead are iteroparous[2].	Usually spawn in fine gravel in a riffle above a pool.	The alevin life stage primarily resides just below the gravel surface until they approach or reach the fry stage. Immediately after leaving the gravel, swim-up fry usually inhabit shallow water along banks of stream or aquatic habitats on streams margins. Steelhead rear in a wide variety of freshwater habitats, generally for two to three years, but up to six or seven years is possible. They smolt and migrate to sea in the spring.

1 spawn only once
2 spawn more than once

Freshwater, estuarine, and marine near-shore habitats are areas subject to pesticide loading from runoff and drift given their proximity to pesticide application sites. Small streams and many floodplain habitats are more susceptible to higher pesticide concentrations than other aquatic habitats used by salmon because their physical characteristics provide less dilution and dissipation. Examples of floodplain habitats include alcoves, channel edge sloughs, overflow channels, backwaters, terrace tributaries, off-channel dredge ponds, off-channel ponds, and braids (S. E. Anderson, 1999; T.

Beechie & Bolton, 1999; Swift III, 1979). The transition from yolksac fry to exogenous feeding is a critical life stage for all salmon species and depends upon availability of prey. Diverse, abundant communities of invertebrates (many of which are salmonid prey items), also populate floodplain habitats and, in part, are responsible for juvenile salmonids' reliance on these habitats. Juvenile coho salmon, stream-type Chinook salmon, and steelhead use floodplain habitats for extended durations (several months). Although these habitats typically vary in surface area, volume, and flow, they are frequently shallow, low to no-flow systems protected from a river's or a stream's primary flow. Thus, rearing and migrating juvenile salmonids use these habitats extensively (T. Beechie & Bolton, 1999; T. J. Beechie, et al., 2005; Caffrey, 1996; Henning, Gresswell, & Fleming, 2006; Montgomery, 1999; Morley, Garcia, Bennett, & Roni, 2005; Opperman & Merenlender, 2004; Roni, 2002).

Exposure Pathways to Salmonids Habitats

Aquatic habitats can be contaminated by pesticides applied to terrestrial target sites through several alternative pathways. For example, spray drift or primary drift refers to the off-target deposition of droplets from spray-applied pesticides at the time of application. The likelihood of spray drift to an aquatic habitat is determined by the application method, the proximity to the habitat, and meteorological conditions at the time of application. Some pesticides are applied directly to surface water for control of plants, mosquitoes, and other aquatic pests. Other pathways of surface water contamination are influenced primarily by the environmental fate properties of the chemical. For example, secondary drift or vapor drift is dependent on a chemical's volatility and refers to the redistribution of pesticides from plant and soil surfaces through volatilization and subsequent atmospheric deposition. Runoff and leaching, the horizontal and vertical movement of pesticides with rainwater or irrigation water, are influenced by chemical-specific properties that determine the compound's persistence and mobility in soil and water. Standardized tests are typically used to characterize mobility (*e.g.* solubility, K_d and K_{oc}) and persistence under different environmental conditions (*e.g.* hydrolysis, photolysis, and metabolism half-lives in aerobic and anaerobic environments). Below we present environmental fate properties of the six a.i.s to characterize the relative importance of these exposure pathways in terms of the potential for the active ingredients and their toxic degradates to contaminate salmonid bearing habitats and designated critical habitats.

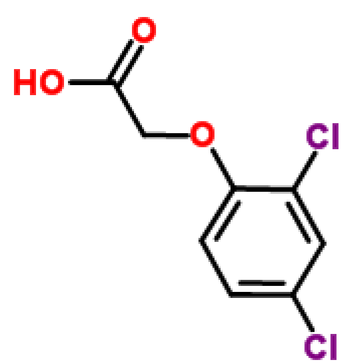
Summary of Chemical Fate of the Six Active Ingredients

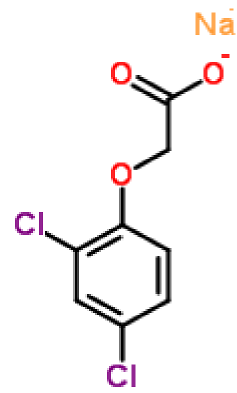
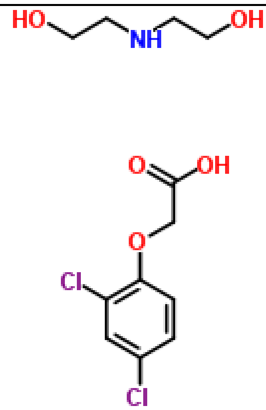
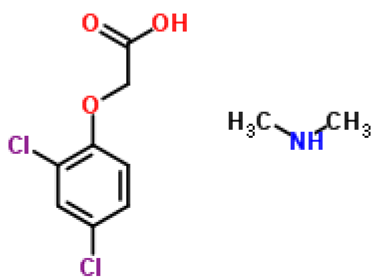
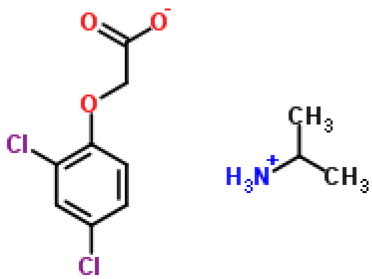
Pesticides can contaminate surface waters via runoff, erosion, leaching, spray drift from application at terrestrial sites or direct application to aquatic habitats, and atmospheric deposition. The six a.i.s are primarily registered for use in terrestrial habitats although some 2,4-D products are registered for use as aquatic herbicides and may be directly applied to a variety of habitats utilized by listed salmonids including ponds, lakes, reservoirs, marshes, ditches, canals, and slow moving rivers and streams. Fish are most likely exposed to the six a.i.s from the water column where the chemicals enter the fish during respiration, (*i.e.*, across the gills), or where fish sensory systems come in direct contact with contaminated water (*i.e.*, olfactory sensory neurons). Other secondary routes may contribute to overall exposure including incidental ingestion of the chemical in sediment or ingestion of the chemical in food items. Below we summarize chemical fate properties of the six a.i.s reported by EPA in the salmon BEs and red-legged frog BEs. Where discrepancies existed between the two documents, we deferred to the more recent document.

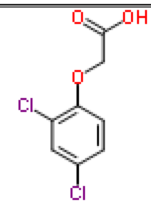
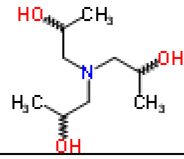
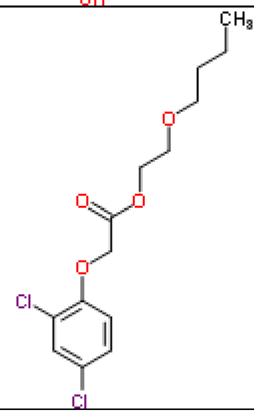
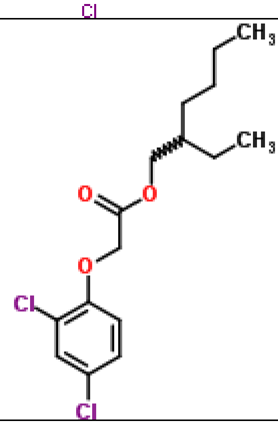
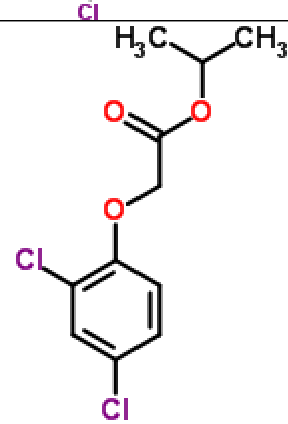
2,4-D

EPA developed an environmental fate bridging strategy to account for the differences among the different chemical forms of 2,4-D (Table 72)(Section 2.4.1 in (EPA, 2009a)). The strategy is based on the rapid degradation, under most environmental conditions, of 2,4-D esters and 2,4-D amine salts to 2,4-D acid. In most aquatic and terrestrial environments 2,4-D amine salts dissociate to form 2,4-D acid instantaneously. The conversion of 2,4-D esters to 2,4-D acid is also rapid (<2.9 d) in agricultural soils and water (EPA, 2009a). Physical and chemical fate parameters of 2,4-D acid are provided in Table 73. 2,4-D has a relatively low potential for volatilization from soil and water suggesting secondary drift is not a likely pathway of high exposure. 2,4-D is frequently detected in rainfall although concentrations are generally < 1 ug/L. The low octanol water partition coefficient also suggest bioaccumulation in fish would be relatively low. The degradation of 2,4-D acid is dependent on oxidative microbial-mediated mineralization and photodegradation in water. In buffered solution, the photodgradation half-life was 12.98 days. In soil, the photodegradation half-life is 68 days. Thirty terrestrial field studies found dissipation half-lives of 2,4-D acid that ranged from 1 – 43 d, and had a median half-life of 6 d. Aquatic field studies suggest that persistence of 2,4-D in the water column is dependent on chemical form and site specific conditions (e.g. pH). 2,4-D half-lives in the water column ranged from 1 – 40 days. 2,4-D acid has low binding affinity in mineral soils and its mobility is characterized as intermediate to very mobile depending on soil type. These properties suggest that drift and runoff are the most likely pathways of deposition of 2,4-D into aquatic habitats (EPA, 2009a).

Table 72. Chemical structures and the molecular weight ratios of various chemical forms of 2,4-D[1].

Chemical Name	Chemical Structure	Molecular Weight Ratio relative to 2,4-D acid
2,4-D acid		1.00

2,4-D sodium salt		1.10
2,4-D diethanolamine salt (DEA)		1.48
2,4-D dimethylamine salt (DMA)		1.20
2,4-D isopropylamine salt (IPA)		1.27

2,4-D triisoproanolamine salt (TIPA)	 	1.87
2,4-D butoxyethyl ester (BEE)		1.45
2,4-D ethylhexyl ester (EHE)		1.51
2,4-D isopropyl ester (IPE)		1.19

1 (EPA, 2009a)

Table 73. Environmental fate characteristics of 2,4-D acid[1].

Parameter	Value
Water solubility	569 mg/L at 20° C
Vapor pressure	1.47×10^{-7} mm Hg at 25° C
Henry's law constant	4.74×10^{-10} atm m^3 mol^{-1}
Octanol/Water partition coefficient	Log $K_{ow} = 2.81$
Hydrolysis ($t_{1/2}$) pH 5, pH 7, & pH 9	stable
Aqueous photolysis ($t_{½}$)	12.98 d
Soil photolysis ($t_{½}$)	68 d
Aerobic soil metabolism ($t_{½}$)	1.44 to 12.4 d
Anaerobic soil metabolism ($t_{½}$)	Not Specified
Aerobic aquatic metabolism ($t_{½}$)	15 d
Anaerobic aquatic metabolism ($t_{½}$)	41 to 333 d
Soil partition coefficient	K_d = 0.17 – 0.36; K_{oc} = 56 – 117 L/kg_{soil}

1 (EPA, 2009a)

Triclopyr BEE

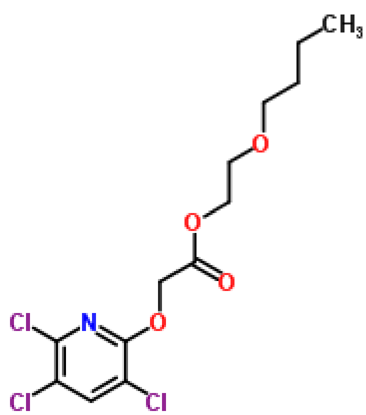


Figure 56. Chemical structure of triclopyr BEE.

Tricolpyr BEE (Figure 56) is non-volatile and has low solubility (6.8 mg/L). It quickly hydrolyzes in the environment to form triclopyr acid/anion and butoxyethyl ester (0.5 d at pH 6.7). Hydrolysis occurs more rapidly at higher pHs. Butoxyethyl ester is rapidly dissipated by microbial degradation. The predominant moiety present in the environment is triclopyr anion when either triclopyr BEE or triclopyr TEA are applied (EPA, 2009c). The environmental fate parameters for triclopyr acid are presented below in Table 74. Triclopyr acid/anion is relatively persistent and mobile in the environment. Tryclopyr acid primarily degrades through photodegradation in water and through microbial processes in the soil. Microbial degradation of triclopyr in soil produces the major metabolite TCP (>10% of applied parent), which is likely to be transported to surface waters because it is both persistent and mobile (EPA, 2009c). Triclopyr BEE products are applied in broadcast applications by ground and aerial application methods to several use sites suggesting transport to surface waters via primary drift is likely. Its persistence and mobility suggest runoff is a likely pathway of exposure to aquatic habitats, and its relatively low volatility suggest secondary drift and long range transport are pathways of less concern.

Table 74. Environmental fate characteristics of triclopyr acid[1].

Parameter	Value
Water solubility	440 mg/L at 25° C
Vapor pressure	1.26 X 10^{-6} mm Hg
Henry's law constant	9.66 X 10^{-7} atm m^3 mol^{-1}
Octanol/Water partition coefficient	Not specified
Hydrolysis ($t_{½}$)	Stable at pH 5,7,9
Aqueous photolysis ($t_{½}$)	0.3 – 1.7 d
Soil photolysis ($t_{½}$)	Not Specified
Aerobic soil metabolism ($t_{½}$)	8 – 18 d
Anaerobic soil metabolism ($t_{½}$)	Stable (1300 d)
Aerobic aquatic metabolism ($t_{½}$)	142 d
Anaerobic aquatic metabolism ($t_{½}$)	Not Specified
Soil partition coefficient	K_d = 0.165-0.975; K_{oc} = 25-134 L/kg_{soil}

1- (EPA, 2009c)

Diuron

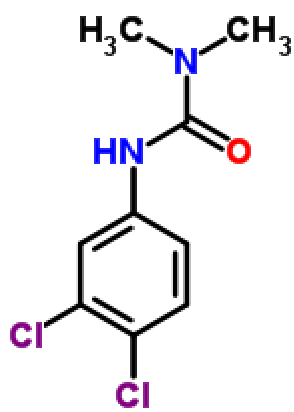


Figure 57. Chemical structure of diuron

Environmental fate studies indicate diuron (Figure 57) is moderately to highly persistent (Table 75). Field residue studies indicate highly variable half-lives of diuron in the soil that range from 1 month to more than 1 year. EPA considers the average field dissipation half-life of diuron to be 115 d. It is primarily degraded through microbial processes and to a lesser degree from photodegradation (EPA, 2009b). In water, microbial breakdown is the primary degradation pathway. Diuron is stable to hydrolysis at the pH range of 5 - 9 and has a photolysis half-life of 43 d. In aquatic metabolism studies, half-lives of 5 d and 33 d were determined for anaerobic and aerobic conditions, respectively. Diuron tends to not sorb well to soil, and its mobility is inversely correlated with soil organic matter. The metabolites are less mobile than the parent. Diuron is prone to surface water runoff and leaching given its persistence and mobility in soils. Studies have found relatively high peak concentrations of diuron in runoff under simulated rainfall conditions (600-1700 μg/L) and in runoff monitoring in fields months after application (200-890 μg/L). Primary spray drift is also a likely pathway of exposure to aquatic organisms given broadcast ground and aerial application methods. Secondary drift is less likely considering its relatively low volatility (EPA, 2009b). Information to characterize the potential for accumulation in aquatic organisms is lacking.

Table 75. Environmental fate characteristics of diuron[1].

Parameter	Value
Water solubility	42 mg/L at 20° C
Vapor pressure	6.9×10^{-8} mm Hg
Henry's law constant	5.10×10^{-10} atm m^3 mol^{-1}
Octanol/Water partition coefficient	Not Specified
Hydrolysis ($t_{½}$) pH 5, pH 7, & pH 9	Stable
Aqueous photolysis ($t_{½}$)	43 d
Soil photolysis ($t_{½}$)	173 d
Aerobic soil metabolism ($t_{½}$)	372 d
Anaerobic soil metabolism ($t_{½}$)	1,000 d
Aerobic aquatic metabolism ($t_{½}$)	33 d
Anaerobic aquatic metabolism ($t_{½}$)	5 d
Soil partition coefficient	K_{oc} = 468-1666; K_d = 14 L/kg_{soil}

1 (EPA, 2009b)

Linuron

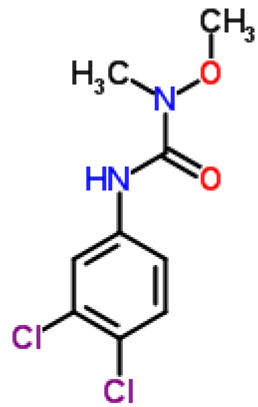


Figure 58. Chemical structure of linuron.

Data are not adequate to fully characterize the terrestrial dissipation of linuron (Figure 58) in the field. However, environmental fate studies suggest half-lives of < 60d in soil (Table 76). Linuron dissipates principally through microbial degradation in soils. In water, linuron is stable to hydrolysis and can be degraded through metabolism and secondarily through photolysis (EPA, 2008). Its half-life in an anaerobic metabolism study was less than 21 d. However, persistence may increase under conditions of low microbial activity. Significant fractions of linuron may exist in the water column given its low to intermediate tendency to partition to sediment (EPA, 2008). Linuron can be transported to surface waters through primary drift from ground spray applications. Secondary drift is less likely given low potential to volatilize based on its physico-chemical properties. Linuron is only slightly mobile in high organic content soils. It has greater mobility in permeable soils and soils with low organic matter. Transport of linuron to surface water is likely from linuron dissolved in surface runoff and suspended sediment. Bioconcentration factors (40-240x) indicate some accumulation in aquatic organism may occur (EPA, 2008).

Table 76. Environmental fate characteristics of linuron[1].

Parameter	Value
Water solubility	810 mg/L (estimate)
Vapor pressure	1.5×10^{-5} mm Hg at 26° C
Henry's law constant	2.6×10^{-6} atm m^3 mol^{-1}
Octanol/Water partition coefficient	Not Specified
Hydrolysis (t½)	stable
Aqueous photolysis (t½)	49 d
Soil photolysis (t½)	Not Specified; 79% remaining at 15 d
Aerobic soil metabolism (t½)	49 d
Anaerobic soil metabolism (t½)	Not Specified
Aerobic aquatic metabolism (t½)	48 d
Anaerobic aquatic metabolism (t½)	21 d
Soil partition coefficient[2]	K_d= 2.7-7.2 L/kg; K_{oc}= 370 L/kg_{soil}

1 (EPA, 2008)
2 (EPA, 1995)

Captan

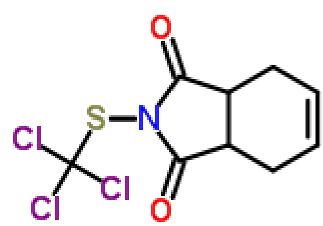


Figure 59. Chemical structure of captan.

The environmental fate characteristics of captan suggest it is quickly degraded by hydrolysis and aerobic metabolism (Table 77). Photodegradation on soil also occurs, but is secondary to hydrolysis and metabolism. Captan (Figure 59) has a relatively short half-life, generally less than 10 d in both soils and water. Terrestrial field dissipation studies found captan dissipated with half lives of 2.5 – 24 d and was relatively immobile to slightly mobile at six sites. Transport of captan to aquatic habitats may occur from primary drift associated with ground and aerial-spray pesticide applications. Secondary drift of captan is less likely given relatively low volatility. Captan is expected to be moderately mobile in the soil based on its soil adsorption properties although field studies found it to be only slightly mobile (Kd = 3-8 L/Kg). These data suggest surface water runoff of dissolved or suspended material is also a likely transport mechanism for captan in aquatic habitats (EPA, 2007c).

Table 77. Environmental fate characteristics of captan[1].

Parameter	Value
Water solubility	3.3 mg/L
Vapor pressure	8×10^{-8} mm Hg at 26° C
Henry's law constant	9.6×10^{-10} atm m^3 mol^{-1}
Octanol/Water partition coefficient	2.79
Hydrolysis pH 5, 7, and 9($t_{½}$)	0.8, 0.25, 0.006 d
Aqueous photolysis ($t_{½}$)	0.42 d
Soil photolysis ($t_{½}$)	Not Specified
Aerobic soil metabolism ($t_{½}$)	<1 d
Anaerobic soil metabolism ($t_{½}$)	1.85 d
Aerobic aquatic metabolism ($t_{½}$)	<1 d
Anaerobic aquatic metabolism ($t_{½}$)	Not Specified
Soil partition coefficient	K_d= 3-8 L/kg_{soil}

1 (EPA, 2007c)

Chlorothalonil

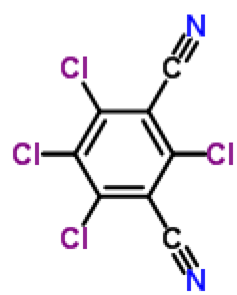


Figure 60. Chemical structure of chlorothalonil.

Chlorothalonil degrades rapidly in clear, shallow water through photolyis (Table 78). Aqueous photolysis is limited to aquatic habitats exposed to direct sunlight. It is also degraded through biotic metabolism with reported aerobic aquatic half-lifes ranging from 0.1 – 16 days (EPA, 2007b; Syngenta, 2011). Chlorothalonil (Figure 60) is stable to hydroloysis at pHs below 9. At pH of 9 and greater chlorothalonil is degraded by hydrolysis with a half-life of 40-60 d. The main route of dissipation for chlorothalonil in the environment is though biotic metabolism (EPA, 2007b). Bioaccumulation factors are moderately high for bivalves (2600) and fish (2700). Chlorothalonil has low mobility in loamy soils and low to moderate mobility in sandy soils. Its moderate persistence suggest the potential of transport to surface waters through runoff. Its major metabolite 4-hydroxy-2,5,6-trichloro-1,3-dicyanobenze (SDS-3701) has a greater potential for runoff given greater persistence and mobility in soil (EPA, 2007b). Transport to aquatic habitats from primary spray drift is likely given ground and aerial broadcast spray applications. Its physical properties suggest chlorothalonil can volatilize from soil and water, and therefore transport from secondary drift may also occur. Air monitoring has detected trace amounts of chlorothalonil up to a mile away for pesticide application sites (EPA, 2007b). However, concentrations were not considered high enough to suggest that long range transport through secondary drift would result in a substantial increase in risk to nontarget species (EPA, 2007b).

Table 78. Environmental fate characteristics of chlorothalonil[1].

Parameter	Value
Water solubility	0.8 mg/L at 20° C
Vapor pressure	5.7×10^{-7} mm Hg
Henry's law constant	2.6×10^{-7} atm m^3 mol^{-1}
Octanol/Water partition coefficient	Log K_{ow} = 3.8
Hydrolysis ($t_{½}$) pH 5, pH 7, & pH 9	Stable, stable, 40-60 d
Aqueous photolysis ($t_{½}$)	0.42 d
Soil photolysis ($t_{½}$)	Not Specified
Aerobic soil metabolism ($t_{½}$)	22-68 d
Anaerobic soil metabolism ($t_{½}$)	Not Specified
Aerobic aquatic metabolism ($t_{½}$)	7-16 d
Anaerobic aquatic metabolism ($t_{½}$)	21-29 d
Soil partition coefficient	K_d = 3-29 L/kg_{soil}

1 (EPA, 2007b)

Degradates of the 6 active ingredients

The molecular structure of a pesticide may be modified by biotic (*e.g.* microbial metabolism) or abiotic processes (*e.g.* photolysis and hydrolysis). The products of these processes typically have different toxicities, environmental fate characteristics, and risks compared to the parent pesticide. Several degradates of parent 2,4-D have been identified in laboratory environmental fate studies suggesting possible exposure to salmonids and their habitats. 1,2,4-benzenetriol is a product of photodegradation. However, it may be less likely to occur in many environments as degradation appears to be dependent primarily on oxidative microbial-mediated mineralization. Metobolic products (Table 79) that may result in exposure in the aquatic environment include 2,4-DCP; 2,4-DCA; 4-chlorophenol; 2-chlorophenol; 4-chlorophenoxyacetic acid; and chlorohydroquinone (EPA, 2009a).

Table 79. Metabolites and degradates of 2,4-D.

Laboratory study	Degradates/metabolites of 2,4-D acid
Direct aqueous photolysis	1,2,4-benzenetriol (37% of applied)
Aerobic soil metabolism	2,4-DCP (3.5% of applied) 2,4-DCA (2.8% of applied)
Anaerobic aquatic metabolism	2,4-DCP, 4-chlorophenol, and 2-chlorophenol
Aerobic aquatic metabolism	2,4-DCP, 4-chlorophenol, 4-chlorophenoxyacetic acid, and chlorohydroquinone

(EPA, 2009a)

Triclopyr BEE may be quickly degraded to triclopyr acid through hydrolysis (Table 80). Conversion to the acid through hydrolysis is pH-dependent with a half-life of 7 hrs at pH 9 and a half-life of 84 days at pH 5 (EPA, 2004f). Laboratory studies indicate triclopyr acid is somewhat persistent and mobile. In aquatic environments, photodegradation products of triclopyr acid include TCP and oxamic acid. In soils, TCP and TMP are formed through biotic metabolism. TCP is both mobile and persistent suggesting likely transport to aquatic habitats (EPA, 2009c).

Table 80. Metabolites and degradates of triclopyr BEE.

Laboratory study	Degradates/metabolites of triclopyr BEE
Hydrolysis	Triclopyr acid
Aqueous photolysis	Oxamic acid (48% of applied)
Aerobic soil metabolism	TCP (26.4% of applied) TMP (7.8%)
Anaerobic aquatic metabolism	TCP (26% of applied)
Aerobic aquatic metabolism	TCP (<5% of applied)

(EPA, 2009c)

Although relatively resistant to abiotic degradation, diuron can be broken down somewhat through hydrolysis and photolysis (Table 81). It is also metabolized in the soil and aquatic environments which may lead to exposure to several metabolic products including DCPMU, MCPDMU, and PDMU. Other degradates which may be deposited or form in the aquatic environment include DCPU, 3,4-DCA, CPMU, and TCAB (EPA, 2009b). TCAB and TCAOB are 'dioxin-like' substances that are also impurities in the diuron manufacturing process. TCAB levels between 0.15 and 28 ppm have been found in diuron samples tested. TCAOB is present at lower levels (EPA, 2009b).

Table 81. Metabolites and degradates of diuron.

Laboratory study	Degradates/metabolites of diuron
Hydrolysis	3,4-DCA (~2% of applied)
Soil photolysis	DCPMU (3.6% of applied) DCMU (≤0.7% of applied) 3,4-DCA (0.370 ppm) TCAP (0.038 ppm)
Aerobic soil metabolism	DCPMU (22.5% of applied) DCPU (3.4% of applied) CO_2 (3.36% of applied)
Anaerobic soil metabolism	DCPMU (10.3% of applied)
Anaerobic aquatic metabolism	MCPDMU (83% of applied) PDMU (13% of applied) MCPMU (23% of applied)
Aerobic aquatic metabolism	MCPDMU (25% of applied) DCPMU (9.2% of applied) CPMU (8% of applied)

(EPA, 2009b)

Linuron and diuron have similar chemical structures and they degrade into several common metabolites (Table 82). In the soil, linuron degrades to 3,4- DCA, DCPMU, DML, and DCPU. In aquatic environments, linuron degrades to desmethoxy linuron, desmethoxy monolinuron, and several minor metabolites (EPA, 2008).

Table 82. Metabolites and degradates of linuron.

Laboratory study	Degradates/metabolites of linuron
Hydrolysis	3,4-DCA (~1% of applied) DCPMU (~1% of applied) DML (~1% of applied) DCPU (~1% of applied)
Direct aqueous photolysis	3-(3-chloro-4-hydroxyphenyl)-1-methoxy-1-methylurea, 3,4-ichlorophenylurea, and 3-(3,4-dichlorophenyl)-1-methylurea
Soil photolysis	Norlinuron (<8.4% of applied) Desmethyl linuron (<8.4% of applied) 3,4-DCA (<8.4% of applied)
Aerobic soil metabolism	Desmethoxy linuron (3% of applied) Desmethyl linuron (2.1% of applied) Norlinuron (1.9% of applied)
Anaerobic aquatic metabolism	Desmethoxy linuron (46.7% of applied) Desmethoxy monolinuron (78% of applied) Desmethyl linuron (<5% of applied) Norlinuron (<5% of applied) 3,4-DCA (<5% of applied)
Aerobic aquatic metabolism	Desmethoxy linuron (<10% of applied) Desmethyl linuron (<10% of applied) Norlinuron (<10% of applied) 3,4-DCA (<10% of applied)

(EPA, 2008)

Captan degrades rapidly in soil and water. The major routes of degradation are believed to be hydrolysis and aerobic metabolism. Degradation products formed that may result in exposure in aquatic environments include THPI, TCMT, THPAm, THCY, inorganic sulfur and chlorine, and thiophosgene (Table 83). Captan is slightly mobile, however THPI and THPAm have significantly greater mobility and are more likely to be transported to aquatic habitats through runoff (EPA, 2003a).

Table 83. Metabolites and degradates of captan.

Laboratory study	Degradates/metabolites of captan
Hydrolysis	TCMT THPI
Soil photolysis	THPI (21.3% of applied) THCY (9.4% of applied) CO_2(41.7% of applied)
Aerobic soil metabolism	THPI THPAm THCY CO^2 Thiophosgene Inorganic sulfur Chlorine
Anaerobic soil metabolism	THPI THPAm THCY
Aquatic fate study	THPI (81.2% of applied) THPAm (27% of applied) THPI epoxide (9.4% of applied) THPI (81.2% of applied)

(EPA, 2003a)

Laboratory fate studies with chlorothalonil reveal a number of degradates and metabolites that may be formed in, or transported to, aquatic environments (Table 84). The major degradate of chlorothalonil in the soil under aerobic conditions is SDS-3701. This degradate appears to be more persistent and mobile than chlorothalonil based on both groundwater detections and estimates using a structural analysis model (EPA, 1999b, 2007b). Substantial amounts of SDS-3701 could be available for runoff for longer periods than chlorothalonil (EPA, 1999b). SDS-3701 can also persist in the aquatic environment. Anaerobic aquatic metabolism studies showed SDS-3701 reached maximum concentrations 1-2 months after introduction and remained at a near constant levels (30-40% of applied) until the end of a 4-month study in silt loam soil (EPA, 1999b). An aerobic metabolism study conducted under nonstandard test conditions, which included continuous agitation and aeration and high concentrations of suspended sediment resulted in different metabolite profiles and degradation rates than seen in other laboratory environmental fate studies. These conditions may not reflect quiescent bodies of water with lower amounts of suspended sediment and would give higher rates of reaction than would be expected under natural conditions (EPA, 1999b).

Table 84. Metabolites and degradates of chlorothalonil.

Laboratory study	Degradates/metabolites of captan
Hydrolysis	SDS-19221 (~50% of applied) [2] SDS-3701 (~20% of applied)[2]
Aqueous photolysis	SDS-3701 (4-10% of applied)[1,2]
Aerobic soil metabolism	SDS-3701 (32% of applied) [1] SDS-19221[2] SDS-46851[2] SDS-47523/ SDS-47524[2] SDS-47425[2]
Anaerobic soil metabolism	SDS-3701 (43% of applied)[1]
Anaerobic aquatic metabolism	SDS-3701 (30-40% of applied)[2] SDS-19221 (7% of applied)[2] SDS-46851 (3% of applied)[2] SDS-47523/ SDS-47524 (9% of applied)[2] SDS-47425 (4% of applied)[2]
Aerobic aquatic metabolism	SDS-67042 (25-30% of applied)[2] SDS-67042 sulfoxide (15% of applied)[2] SDS-3701(5-10% of applied)[2] SDS-66432[2] SDS-66382[2] SDS-13353[2]

1- (EPA, 2007b)
2- (EPA, 1999b)

Exposure of salmonid habitats to the stressors of the action

Co-occurrence associated with pesticide uses.

Rights-of-way uses are authorized for 2,4-D, triclopyr BEE, diuron, and linuron. These use sites are the most difficult to analyze as they are not tied to a particular landuse class. EPA classifies three specific kinds of rights-of-way: highway, railroad, and utility (including pipeline) (EPA, 2003c). By definition, they are tied to the transportation of goods and services, which cross urban, agricultural, and wilderness areas alike. Highways and utilities are ubiquitous and rights-of-way applications are likely to occur during the freshwater residence of all of the listed Pacific salmonids (*Appendix 6*). As such, we make the reasonable assumption that they are present in all ESUs to varying degrees. These uses likely have less of an impact in less populated areas and for species that spend less time in freshwater habitats. This approach is consistent with EPA's Biological assessments, "Non-agricultural usages cannot be broken down by state or county as can agricultural usages. However, because they are major uses of 2,4-D, we must assume that they are likely to occur in salmonid Evolutionarily Significant Units (ESUs) in the Pacific Northwest and California" (EPA, 2004a).

We evaluated co-occurrence of listed salmonids with other uses of the six pesticides by comparing the spatial and temporal distribution of salmon (*Appendix 5* and *6*) with potential use of pesticides based on label specifications. To evaluate areal extent of application sites near salmon-bearing waters, NMFS used a GIS overlay containing landuse classifications and salmon distributions to determine overlap. Because cropping patterns and registered use sites may change over time, landuse classifications (agricultural, forestry, urban/developed) are used rather than specific crops. Details of the GIS analysis and the maps are provided in *Appendix 5*. A summary of our findings is presented in Table 85. "NA" denotes uses that are not applicable because they are not authorized through labeling. "Y" indicates both spatial and temporal overlap of potential pesticide use with species presence. "N" denotes labeled uses are authorized but spatial or temporal overlap with the species is lacking. Most species are present in freshwater year-round in some lifestage. The only exceptions include the two Chum ESUs and California Coastal Chinook salmon; these species occur in freshwater 9 – 11 months of the year. Additionally, all of the ESUs/DPSs contained pesticide use sites within the watersheds where the species spawn and rear. Considering that all listed Pacific salmonid ESUs/DPSs use watersheds where the use of the six a.i.s are authorized and that these pesticides are permitted for use in close proximity to salmonid habitats, we expect all listed Pacific salmonid ESUs/DPSs and their designated critical habitats may be exposed to the stressors from one or more of these authorized uses.

Table 85. Co-occurrence of listed Pacific salmonids with potential application of pesticides to use sites within the salmons freshwater distribution.

		Temporal overlap of ESU and labeled use of pesticide					
Pesticide Use Site	**Spatial coverage within species spawning and rearing habitat**	**2,4-D**	**Triclopyr BEE**	**Diuron**	**Linuron**	**Captan**	**Chloro-thalonil**
Puget Sound Chinook Salmon							
Aquatic	17.3%	Y	NA	NA	NA	NA	NA
Cropland	0.7%	Y	Y	Y	Y	Y	Y
Pasture/ Rangeland	3.3%	Y	Y	NA	NA	NA	NA
Forest	50.4%	Y	Y	NA	NA	NA	Y
Wetland	3.2%	Y	NA	NA	NA	NA	NA
Residential/ Industrial	14.8%	Y	Y	Y	NA	Y	Y
Lower Columbia River Chinook Salmon							
Aquatic	4.6%	Y	NA	NA	NA	NA	NA
Cropland	2.0%	Y	Y	Y	Y	Y	Y
Pasture/ Rangeland	3.9%	Y	Y	NA	NA	NA	NA
Forest	56.8%	Y	Y	NA	NA	NA	Y
Wetland	4.3%	Y	NA	NA	NA	NA	NA
Residential/ Industrial	13.3%	Y	Y	Y	NA	Y	Y
Upper Columbia River Spring Run Chinook Salmon							
Aquatic	1.0%	Y	NA	NA	NA	NA	NA
Cropland	3.5%	Y	Y	Y	Y	Y	Y
Pasture/ Rangeland	1.8%	Y	Y	NA	NA	NA	NA
Forest	44.9%	Y	Y	NA	NA	NA	Y
Wetland	0.8%	Y	NA	NA	NA	NA	NA
Residential/ Industrial	4.7%	Y	Y	Y	NA	Y	Y
Snake River Fall-Run Chinook Salmon							
Aquatic	0.6%	Y	NA	NA	NA	NA	NA
Cropland	14.6%	Y	Y	Y	Y	Y	Y
Pasture/ Rangeland	0.2%	Y	Y	NA	NA	NA	NA
Forest	49.2%	Y	Y	NA	NA	NA	Y
Wetland	0.2%	Y	NA	NA	NA	NA	NA
Residential/ Industrial	1.4%	Y	Y	Y	NA	Y	Y
Snake River Spring-Summer Run Chinook Salmon							
Aquatic	0.4%	Y	NA	NA	NA	NA	NA
Cropland	6.6%	Y	Y	Y	Y	Y	Y
Pasture/ Rangeland	0.8%	Y	Y	NA	NA	NA	NA

		Temporal overlap of ESU and labeled use of pesticide					
Pesticide Use Site	**Spatial coverage within species spawning and rearing habitat**	**2,4-D**	**Triclopyr BEE**	**Diuron**	**Linuron**	**Captan**	**Chloro-thalonil**
Forest	47.7%	Y	Y	NA	NA	NA	Y
Wetland	0.3%	Y	NA	NA	NA	NA	NA
Residential/ Industrial	1.7%	Y	Y	Y	NA	Y	Y
Upper Willamette River Chinook Salmon							
Aquatic	0.6%	Y	NA	NA	NA	NA	NA
Cropland	10.6%	Y	Y	Y	Y	Y	Y
Pasture/ Rangeland	16.2%	Y	Y	NA	NA	NA	NA
Forest	49.0%	Y	Y	NA	NA	NA	Y
Wetland	2.3%	Y	NA	NA	NA	NA	NA
Residential/ Industrial	9.0%	Y	Y	Y	NA	Y	Y
California Coastal Chinook Salmon							
Aquatic	0.6%	Y	NA	NA	NA	NA	NA
Cropland	1.0%	Y	Y	Y	Y	Y	Y
Pasture/ Rangeland	0.9%	Y	Y	NA	NA	NA	NA
Forest	63.2%	Y	Y	NA	NA	NA	Y
Wetland	0.3%	Y	NA	NA	NA	NA	NA
Residential/ Industrial	5.5%	Y	Y	Y	NA	Y	Y
Central Valley Spring-Run Chinook Salmon							
Aquatic	1.4%						
Cropland	21.3%	Y	NA	NA	NA	NA	NA
Pasture/ Rangeland	3.2%	Y	Y	Y	Y	Y	Y
Forest	20.4%	Y	Y	NA	NA	NA	NA
Wetland	3.1%	Y	Y	NA	NA	NA	Y
Residential/ Industrial	10.8%	Y	NA	NA	NA	NA	NA
Sacramento River Winter-Run Chinook							
Aquatic	1.4%	Y	NA	NA	NA	NA	NA
Cropland	21.3%	Y	Y	Y	Y	Y	Y
Pasture/ Rangeland	3.2%	Y	Y	NA	NA	NA	NA
Forest	20.4%	Y	Y	NA	NA	NA	Y
Wetland	3.1%	Y	NA	NA	NA	NA	NA
Residential/ Industrial	10.8%	Y	Y	Y	NA	Y	Y
Columbia River Chum Salmon							
Aquatic	5.8%	Y	NA	NA	NA	NA	NA
Cropland	1.9%	Y	Y	Y	Y	Y	Y
Pasture/	4.7%	Y	Y	NA	NA	NA	NA

		Temporal overlap of ESU and labeled use of pesticide					
Pesticide Use Site	**Spatial coverage within species spawning and rearing habitat**	**2,4-D**	**Triclopyr BEE**	**Diuron**	**Linuron**	**Captan**	**Chloro-thalonil**
Rangeland							
Forest	50.8%	Y	Y	NA	NA	NA	Y
Wetland	2.5%	Y	NA	NA	NA	NA	NA
Residential/ Industrial	14.9%	Y	Y	Y	NA	Y	Y
Hood Canal Chum Salmon							
Aquatic	15.5%	Y	NA	NA	NA	NA	NA
Cropland	0.0%	N	N	N	N	N	N
Pasture/ Rangeland	1.4%	Y	Y	NA	NA	NA	NA
Forest	61.0%	Y	Y	NA	NA	NA	Y
Wetland	2.6%	Y	NA	NA	NA	NA	NA
Residential/ Industrial	8.9%	Y	Y	Y	NA	Y	Y
Lower Columbia River Coho Salmon							
Aquatic	4.0%	Y	NA	NA	NA	NA	NA
Cropland	2.1%	Y	Y	Y	Y	Y	Y
Pasture/ Rangeland	4.0%	Y	Y	NA	NA	NA	NA
Forest	59.3%	Y	Y	NA	NA	NA	Y
Wetland	3.6%	Y	NA	NA	NA	NA	NA
Residential/ Industrial	11.7%	Y	Y	Y	NA	Y	Y
Oregon Coast Coho Salmon							
Aquatic	0.7%	Y	NA	NA	NA	NA	NA
Cropland	0.2%	Y	Y	Y	Y	Y	Y
Pasture/ Rangeland	3.1%	Y	Y	NA	NA	NA	NA
Forest	70.8%	Y	Y	NA	NA	NA	Y
Wetland	1.8%	Y	NA	NA	NA	NA	NA
Residential/ Industrial	6.6%	Y	Y	Y	NA	Y	Y
Southern Oregon and Northern California Coast Coho Salmon							
Aquatic	0.4%	Y	NA	NA	NA	NA	NA
Cropland	1.0%	Y	Y	Y	Y	Y	Y
Pasture/ Rangeland	1.6%	Y	Y	NA	NA	NA	NA
Forest	67.6%	Y	Y	NA	NA	NA	Y
Wetland	0.4%	Y	NA	NA	NA	NA	NA

		Temporal overlap of ESU and labeled use of pesticide					
Pesticide Use Site	**Spatial coverage within species spawning and rearing habitat**	**2,4-D**	**Triclopyr BEE**	**Diuron**	**Linuron**	**Captan**	**Chloro-thalonil**
Residential/ Industrial	4.2%	Y	Y	Y	NA	Y	Y
Central California Coast Coho Salmon							
Aquatic	1.5%	Y	NA	NA	NA	NA	NA
Cropland	2.2%	Y	Y	Y	Y	Y	Y
Pasture/ Rangeland	0.1%	Y	Y	NA	NA	NA	NA
Forest	55.6%	Y	Y	NA	NA	NA	Y
Wetland	0.4%	Y	NA	NA	NA	NA	NA
Residential/ Industrial	9.4%	Y	Y	Y	NA	Y	Y
Ozette Lake Sockeye Salmon							
Aquatic	10.3%	Y	NA	NA	NA	NA	NA
Cropland	0.0%	N	N	N	N	N	N
Pasture/ Rangeland	0.0%	N	N	NA	NA	NA	NA
Forest	56.8%	Y	Y	NA	NA	NA	Y
Wetland	3.3%	Y	NA	NA	NA	NA	NA
Residential/ Industrial	1.1%	Y	Y	Y	NA	Y	Y
Snake River Sockeye Salmon							
Aquatic	1.4%	Y	NA	NA	NA	NA	NA
Cropland	0.0%	Y[1]	Y[1]	Y[1]	Y[1]	Y[1]	Y[1]
Pasture/ Rangeland	0.9%	Y	Y	NA	NA	NA	NA
Forest	57.2%	Y	Y	NA	NA	NA	Y
Wetland	3.8%	Y	NA	NA	NA	NA	NA
Residential/ Industrial	1.1%	Y	Y	Y	NA	Y	Y
Puget Sound Steelhead							
Aquatic	17.3%	Y	NA	NA	NA	NA	NA
Cropland	0.7%	Y	Y	Y	Y	Y	Y
Pasture/ Rangeland	3.3%	Y	Y	NA	NA	NA	NA
Forest	50.4%	Y	Y	NA	NA	NA	Y
Wetland	4.5%	Y	NA	NA	NA	NA	NA

		Temporal overlap of ESU and labeled use of pesticide					
Pesticide Use Site	**Spatial coverage within species spawning and rearing habitat**	**2,4-D**	**Triclopyr BEE**	**Diuron**	**Linuron**	**Captan**	**Chloro-thalonil**
Residential/ Industrial	14.8%	Y	Y	Y	NA	Y	Y
Lower Columbia River Steelhead							
Aquatic	1.9%	Y	NA	NA	NA	NA	NA
Cropland	2.5%	Y	Y	Y	Y	Y	Y
Pasture/ Rangeland	4.6%	Y	Y	NA	NA	NA	NA
Forest	61.1%	Y	Y	NA	NA	NA	Y
Wetland	2.6%	Y	NA	NA	NA	NA	NA
Residential/ Industrial	12.2%	Y	Y	Y	NA	Y	Y
Upper Willamette River Steelhead							
Aquatic	0.5%	Y	NA	NA	NA	NA	NA
Cropland	14.6%	Y	Y	Y	Y	Y	Y
Pasture/ Rangeland	20.0%	Y	Y	NA	NA	NA	NA
Forest	40.3%	Y	Y	NA	NA	NA	Y
Wetland	2.6%	Y	NA	NA	NA	NA	NA
Residential/ Industrial	10.1%	Y	Y	Y	NA	Y	Y
Middle Columbia River Steelhead							
Aquatic	0.8%	Y	NA	NA	NA	NA	NA
Cropland	17.1%	Y	Y	Y	Y	Y	Y
Pasture/ Rangeland	1.2%	Y	Y	NA	NA	NA	NA
Forest	26.6%	Y	Y	NA	NA	NA	Y
Wetland	0.7%	Y	NA	NA	NA	NA	NA
Residential/ Industrial	3.3%	Y	Y	Y	NA	Y	Y
Upper Columbia River Steelhead							
Aquatic	1.5%	Y	NA	NA	NA	NA	NA
Cropland	13.1%	Y	Y	Y	Y	Y	Y
Pasture/ Rangeland	1.8%	Y	Y	NA	NA	NA	NA
Forest	33.3%	Y	Y	NA	NA	NA	Y
Wetland	0.8%	Y	NA	NA	NA	NA	NA

		Temporal overlap of ESU and labeled use of pesticide					
Pesticide Use Site	**Spatial coverage within species spawning and rearing habitat**	**2,4-D**	**Triclopyr BEE**	**Diuron**	**Linuron**	**Captan**	**Chloro-thalonil**
Residential/ Industrial	4.4%	Y	Y	Y	NA	Y	Y
Snake River Basin Steelhead							
Aquatic	0.4%	Y	NA	NA	NA	NA	NA
Cropland	8.2%	Y	Y	Y	Y	Y	Y
Pasture/ Rangeland	0.6%	Y	Y	NA	NA	NA	NA
Forest	52.1%	Y	Y	NA	NA	NA	Y
Wetland	0.3%	Y	NA	NA	NA	NA	NA
Residential/ Industrial	1.6%	Y	Y	Y	NA	Y	Y
Northern California Steelhead							
Aquatic	0.6%	Y	NA	NA	NA	NA	NA
Cropland	0.1%	Y	Y	Y	Y	Y	Y
Pasture/ Rangeland	1.0%	Y	Y	NA	NA	NA	NA
Forest	68.2%	Y	Y	NA	NA	NA	Y
Wetland	0.3%	Y	NA	NA	NA	NA	NA
Residential/ Industrial	4.4%	Y	Y	Y	NA	Y	Y
Central California Coast Steelhead							
Aquatic	8.4%	Y	NA	NA	NA	NA	NA
Cropland	2.9%	Y	Y	Y	Y	Y	Y
Pasture/ Rangeland	0.2%	Y	Y	NA	NA	NA	NA
Forest	28.7%	Y	Y	NA	NA	NA	Y
Wetland	2.6%	Y	NA	NA	NA	NA	NA
Residential/ Industrial	22.1%	Y	Y	Y	NA	Y	Y
California Central Valley Steelhead							
Aquatic	1.2%	Y	NA	NA	NA	NA	NA
Cropland	26.9%	Y	Y	Y	Y	Y	Y
Pasture/ Rangeland	5.0%	Y	Y	NA	NA	NA	NA
Forest	15.9%	Y	Y	NA	NA	NA	Y
Wetland	2.1%	Y	NA	NA	NA	NA	NA

		Temporal overlap of ESU and labeled use of pesticide					
Pesticide Use Site	**Spatial coverage within species spawning and rearing habitat**	**2,4-D**	**Triclopyr BEE**	**Diuron**	**Linuron**	**Captan**	**Chloro-thalonil**
Residential/ Industrial	9.7%	Y	Y	Y	NA	Y	Y
South-Central California Coast Steelhead							
Aquatic	0.7%	Y	NA	NA	NA	NA	NA
Cropland	7.1%	Y	Y	Y	Y	Y	Y
Pasture/ Rangeland	1.1%	Y	Y	NA	NA	NA	NA
Forest	19.9%	Y	Y	NA	NA	NA	Y
Wetland	0.9%	Y	NA	NA	NA	NA	NA
Residential/ Industrial	9.6%	Y	Y	Y	NA	Y	Y
Southern California Steelhead							
Aquatic	0.7%	Y	NA	NA	NA	NA	NA
Cropland	4.1%	Y	Y	Y	Y	Y	Y
Pasture/ Rangeland	0.8%	Y	Y	NA	NA	NA	NA
Forest	9.3%	Y	Y	NA	NA	NA	Y
Wetland	0.6%	Y	NA	NA	NA	NA	NA
Residential/ Industrial	33.9%	Y	Y	Y	NA	Y	Y

1-Although cropland does not occur within spawning and rearing habitat of this species, exposure from agricultural uses is expected given authorized uses within the species migration corridor.

Modeling: Estimates of Exposure to the six a.i.s

EPA exposure estimates

EPA's BEs indicate that pesticides containing the six a.i.s are approved for a variety of uses (Table 86). All are approved for use on agricultural crops. Some are also approved for use on other sites such as forestry, rights-of-way, golf courses, nurseries, parks, residential areas, noncrop agricultural lands, aquatic habitats, and rangeland.

Table 86. Summary of use sites approved on active labels.

Active Ingredient	Aquatic	Cropland	Pasture/ Rangeland	Forestry	Residential/ Industrial	Rights of Way
2,4-D	Yes	Yes	Yes	Yes	Yes	Yes
Triclopyr BEE	No	Yes[1]	Yes	Yes	Yes	Yes
Diuron	No[2]	Yes	No[3]	No	Yes	Yes
Linuron	No	Yes	No	No	No	Yes[4]
Captan	No	Yes	No	No	Yes[5]	No
Chlorothalonil	No	Yes	No	Yes	Yes[6]	No

1- Commercially grown turf, ornamentals, and abandoned orchards
2- Approved use sites include ornamental ponds and dry irrigation ditches
3- Approved use sites include hay and fallow land, alfalfa, and general weed control in noncrop and nontimber areas
4- Approved use sites include roadsides and fence rows
5- Approved use sites include golf course, ornamentals, and in paint formulations
6- Forestry uses approved for Washington state

The BEs for the six a.i.s evaluated some, but not all registered uses of the compounds (Table 87). In general, the BEs provided few estimates of exposure given the number and variety of uses currently authorized.

Table 87. Examples of current registered uses of the six a.i.s and the exposure method used by EPA in salmonid BEs.

Active Ingredient	Examples of Registered Uses	Exposure Methods Applied in BEs
2,4-D	Crops: cereal grains, field and pop corn, sweet corn, sorghum, soybeans, sugarcane, rice, apples, pears, cherries, peaches, plums, apricots, nut orchards, pistachios, filberts, potatoes, asparagus, hopps, strawberries, blueberries, grapes, cranberries, citrus, clover, cottonwood and poplar trees grown for pulp,	PRZM-EXAMS for apples, corn, filberts, wheat RICE Model estimates for rice
	Other use sites: forestry, pastures,rangeland, fallow land and crop stubble, grass grown for seed or sod, irrigation ditch banks, abandoned orchards, grasslands not in agricultural production, ornamental turf, tree and brush control, non-cropland such as fencerows, hedgerows, roadsides, ditches, rights-of-way, utility power lines, railroads, airports, industrial sites, and other non-crop areas, and aquatic herbicide uses	PRZM-EXAMS estimates for pastures and turf; direct application for aquatic herbicide uses
Triclopyr BEE	Crops: sod/turf farms	GENEEC Ornamental lawns and turf
	Other use sites: range and pastures; golf course and residential turf; rights-of-way; industrial areas; noncrop agricultural areas such as abandoned orchards, around farm buildings, fence rows, roads,	GENEEC: agricultural and nonagricultural use sites; DIRECT APPLICATION:

Active Ingredient	Examples of Registered Uses	Exposure Methods Applied in BEs
	and ditch banks; forestry;	assumed for forestry applications
Diuron	Crops: alfalfa, apple, artichoke, asparagus, barley, blueberry, caneberry, gooseberry, blackberry, boysenberry, dewberry, loganberry, raspberry, citrus, corn, cotton, filberts, grape, grass seed, oats, olives, papaya, peas, peach, pear, pecan, peppermint, spearmint, red clover, sorghum, tree planting, walnut, wheat, lily bulbs, triticale.	GENEEC: grape, citrus, alfalfa, peaches, walnuts, apples, pears, pecans, cotton PRZM-EXAMS: grass seed, walnuts, apple, alfalfa, citrus
	Other use sites: irrigation and drainage systems when water is not present, impervious surfaces, fence lines, rights-of-way (pipelines, powerlines, railway lines, roads), footpaths, in timber yards and storage areas, around commercial, industrial and farm buildings, electrical substations, and petroleum storage tanks. It has some use as an algaecide in ornamental ponds, fountains, and aquaria, but not natural water bodies. It may be used as a mildewicide in paints used on buildings and structures.	GENEEC: rights-of-way, irrigation and drainage ditches
Linuron	Crops: asparagus, bulbs, carrots, celery, corn, kenaf, marigolds seed, parsley seed, parsnips, sorghum, soybeans, wheat, post-harvest crop stubble and fallow lands	Carrots
	Other use sites: Non-crop areas such as roadsides and fence rows	None reported
Captan	Crops: alfalfa, clover, lespedeza, trefoil, almond, apple, apricot, artichoke, azalea, barley, beans, beets, begonias, blackberry, blueberry, blue grass, bassica, cabbage, canola, cauliflower, camellias, cantaloupe, cucumber, carnation, carrot, cherry, chrysanthemum, cilantro, cole crops, cauliflower, conifers, cotton, collard, corn, cowpeas, crucifer, cucurbits, dewberries, dichondra, eggplant, flax, gladiolus, ginseng, honeydew, kale, lentils, lettuce, milo, mustard, nectarines, oats, onion, okra, peach, plum, prune, peanut, pea, pepper, potato, radish, raspberry, rye, rutabaga, strawberry, Swiss chard, soybean, spinach, squash, watermelon, pumpkin, muskmelon, safflower, sunflower, sesame, sorghum, sugar beets, tomatillo, tomato, turnip, wheat	PRZM-EXAMS for almond, apple, peaches, prune, cherry, and blueberry
	Other use sites: ornamental grasses, golf course turf, paint additive, roses, greenhouse soil	GENEEC: turf grass
Chlorothalonil	Crops: almonds, apricot, asparagus, beans, blueberry, broccoli, brussels sprouts, cabbage, cucumber, cantaloupe, carrot, celery, chayote, cherry, chickpeas, Chinese waxgourd, corn, cranberry, eggplant, filberts, flowering bulbs, ginseng, grass grown for seed, groundcherry, honeydew melon, horseradish, lentil, lupine, mango, mint, muskmelon, okra, onion, papaya, parsnips, passionfruit, peach, peanut, peppers, pepino,	PRZM-EXAMS/GENEEC: cucurbits, peanuts, potatoes, tomatoes, cherries, papaya, cranberries

Active Ingredient	Examples of Registered Uses	Exposure Methods Applied in BEs
	persimmon, pistachio, plum, potato, prune, rhubarb, soybean, squash, strawberry, sugar beets, tomato, yam, watermelon	
	Other use sites: ornamentals, wood preservative (stains and paints), golf course, forestry, lawns around commercial and industrial buildings, collegiate and professional athletic fields, and landscape areas around residential, institutional, public, commercial and industrial buildings, parks, recreational areas and athletic fields.	PRZM-EXAMS: turf

EPA's BEs provided Estimated Environmental Concentrations (EECs) for the six a.i.s in surface water. These EECs were generated using the PRZM-EXAMS model and used as expected concentrations of the six a.i.s for all aquatic habitats where listed salmonids and their prey reside (Table 88). However, no exposure estimates were provided for other identified stressors of the action including inert/other ingredients, other active ingredients with formulations, and for all of the toxic degradates identified. These missing estimates introduce substantial uncertainty into the exposure analysis. The PRZM-EXAMS model generates pesticide concentrations for a generic "farm pond". The pond is assumed to represent all aquatic habitats including rivers, streams, floodplain habitats, estuaries, and near shore ocean environments. EPA's BEs indicate that the PRZM-EXAMS scenarios provide "worst-case" estimates of salmonid exposure and EPA "believes that the EECs from the farm pond model do represent first order streams, such as those in headwaters areas" used by listed salmon. However, listed salmonids use aquatic habitats with physical characteristics that would be expected to yield higher pesticide concentrations than would be predicted with the "farm pond" based model. Juvenile salmonids rely upon a variety of floodplain habitats that are critical to rearing. All listed salmonids use shallow, low flow habitats at some point in their life cycle (Table 71). Below, we discuss the utility of the EECs for the current consultation. NMFS presents information that indicates the EECs do not represent worst-case environmental concentrations to which listed Pacific salmonids may be exposed. Finally, NMFS provides additional modeling estimates to evaluate potential exposure in floodplain habitats used by salmonids.

Table 88. PRZM-EXAMS exposure estimates from EPA's salmonid BEs.

Use Site Scenario: crop, state (surrogate scenario)	Application: rate (lbs a.i./A)/ method/ number of applications	Acute EEC peak (µg/L)	Chronic EEC 60-d average (µg/L)
2,4-D			
Pasture, CA (CA alfalfa)	2.88[1]/aerial/4[7]	23[1]	18.5[1]
Pasture, OR (OR wheat)	2.88[1]/aerial/2[7]	23.9[1]	19.2[1]
Turf, CA (OR grass seed)	2.85[1]/aerial/2[7]	14.2[1]	9.4[1]
Turf, OR (OR grass seed)	2.85[1]/aerial/2[7]	19.3[1]	15.6[1]
Wheat, CA (CA alfalfa)	2.88[1]/aerial/1[7, 8]	3.7[1]	3.0[1]
Wheat, OR	1.25[1]/NR/1[8]	9.0[1]	7.5[1]
Corn, CA	1.00[1]/NR/3[8]	9.7[1]	8.2[1]
Filberts, OR	1.00[1]/NR/4	8.8[1]	7.4[1]
Apples, OR	2.00[1]/NR/2	12.2[1]	9.9[1]
Rice[2]	1.5[1]/NR/1[8]	1431[1]	NR[3]
Triclopyr BEE			
Pastures and Rangeland (GENEEC)[4]	1/ground/1; 1/aerial/1[8]	19;20	NR[3]
Turf and ornamental lawns (GENEEC)[4]	2/ground spray/1[8]	38	NR[3]
Nonagricultural areas (GENEEC)[4]	8/ground/1; 8/aerial/1	152; 152	NR[3]
Forests (direct application)[5]	2-8/direct application/1[7, 8]	1,468-5,872	NR[3]
Drainages system, channeled water (direct application)[5]	1/direct application/1; 8/direct application/1	734; 5,872	NR[3]
Diuron			
Grape, citrus, apples, pears, pecan (GENEEC)[4]	3.2/ ground/1[8]	110	62
Alfalfa (GENEEC)[4]	3.2/ ground/1; 3.2/aerial/1[7]	110; 116	62; 66
Peaches, walnuts (GENEEC)[4]	4/ground/1[7, 8]	137	78
Cotton (GENEEC)[4]	2/ ground/1; 2/aerial/1[7]	69; 73	39; 41
Rights-of-way, irrigation drainage ditches (GENEEC)[4]	12/ ground/1; 12/arial for railroad applications/1	412; 437	234; 248
Grass seed, OR	3.2/ground/1[7]	16	NR[3]
Apples, CA	3.2/ground/1[8]	10	NR[3]
Apples, OR	3.2/ground/1[8]	43	NR[3]
Apples, NY	3.2/ground/1[8]	42	41

Use Site Scenario: crop, state (surrogate scenario)	Application: rate (lbs a.i./A)/ method/ number of applications	Acute EEC peak (μg/L)	Chronic EEC 60-d average (μg/L)
Alfalfa, CA	3.2/aerial/1[7]	22	NR[3]
Citrus, CA	3.2/ground/1[8]	3	NR
Citrus, FL	3.2/ground/1[8]	69	65
Grape, CA	3.2/ground/1[8]	13	12
Walnuts, CA (Almonds, CA)	4.0/ground/1[7]	3	NR[3]
Walnuts, OR	4.0/ground/1[7]	21	NR[3]
Linuron			
Carrots (Index Reservoir)	1.0/ground/2[8]	31.3	NR
Captan			
Almonds, CA	4.5/airblast/5[7]	19.8	3.3
Apples, NY	4.0/airblast/8	10.6	2.9
Peaches, CA	4.0/airblast/8	19.5	6.0
Prunes, CA	3.0/airblast/9[7]	13.1	3.5
Cherries, CA	2.0/airblast/7	1.1	0.97
Blueberries, MI	2.5/airblast/14	1.7	1.6
Turf (GENEEC)	4.0/ground/8[7]	43.4	0.8
Chlorothalonil			
Cucurbits (state NR)	1.75-6.25/NR/1-8[7, 8]	18 - 33	0.6 – 3.6
Peanuts (state NR)	1.125/NR/6-9	18 - 26	0.7 – 2.8
Potatoes (state NR)	1.125/NR/6-10	6 - 9	0.4 – 1.5
Tomatoes (state NR)	1.75-2.25/NR/5-8	26 - 44	0.7 – 3.5
Cherries (GENEEC)	3.6-4.1/NR/4-6[7]	106 - 122	2 - 8
Papaya (GENEEC)	3/NR/5[7]	83-88	2 - 6
Cranberries[6] (GENEEC)	5.3/NR/3	82	1 - 5
Turf (GENEEC)	4 – 22.7/NR/1-10[7,8]	48 - 363	4 - 26

1- 2,4-D values reported in acid equivalents (a.e.) rather than a.i.
2- This estimate derived with EPA RICE Model, not PRZM-EXAMS
3- Not reported
4- EPA standard scenario, Generic Estimated Environmental Concentration
5- Assumed direct application to 6 inches of water (EEC in μg/L =lbs ae applied * 734). However, products containing triclopyr BEE cannot be applied to wetlands, marshes, drainage ditches or other aquatic habitats when surface water is present.
6- Estimated concentration in discharge from bog.
7- Exceeds current label maximum in number of applications or application rate
8- Less than current label maximum in number of applications or application rate

Very few non-crop uses of pesticides were evaluated in the salmonid BEs. However, NMFS also reviewed aquatic exposure estimates developed by EPA within the red legged frog BEs. Although these estimates were specific to registered uses in California only, they provided surface water estimates for pesticides authorized for non-crop uses that were not included in the BEs for listed salmon (Table 89).

Table 89. Estimates of pesticide concentrations in surface waters from California red legged frog BEs.

a.i.	PRZM-EXAMS exposure estimates for surface water (µg/L)[1]		
		Non-Crop	Crops
2,4-D acid/salt	peak	Aquatic weeds: 4,000 Ornamentals, forestry, rights-of-way: 6 - 47	0.08 – 33 (rice:1431)
	60-d avg	Aquatic weeds: 2,610 Ornamentals, forestry, rights-of-way: 5 - 39	0.07 - 27
2,4-D Esters	peak	Pastures, forestry, ornamentals, other: 1.3 - 13	0.55 - 5.5
	60-d avg	NR[2]	NR[2]
Triclopyr BEE[3]	peak	Noncrop agricultural: 65 - 990[3] Pastures: 33 - 395[3] Ornamentals: 5 - 35[3] Golf course turf: 270[3] Residential lawns: 75 - 1499[3]	21 - 165[3]
	60-d avg	Noncrop agricultural: 50 - 794[3] Pastures: 25 - 322[3] Ornamentals: 4 - 29[3] Golf course turf: 220 [3] Residential lawns: 61 - 1172[3]	16 - 134[3]
Diuron	peak	Noncrop agriculture, industrial areas, irrigation and drainage systems, ornamentals, rights-of-way, paved areas: 37- 4911	5 - 140
	60-d avg	Noncrop agriculture, industrial areas, irrigation and drainage systems, ornamentals, rights-of-way, paved areas: 24 - 3428	3 - 103
Linuron	peak	Rights-of-way, impervious surfaces: 60 - 337	2.6 - 41
	60-d avg	Rights-of-way, impervious surfaces: 39 - 211	1.8 - 31
Captan	peak	Golf course turf, ornamental grasses: 3.6 - 29	<0.001 - 21.6
	60-d avg	Golf course turf, ornamental grasses: 0.08 – 1.09	<0.001 – 0.06
Chlorothalonil	peak	Conifers, golf course and general turf, ornamentals: 19 - 274	3 - 69
	60-d avg	Conifers, golf course and general turf, ornamentals: 12 - 146	2 - 43

1- 2,4-D and triclopyr values expressed in a.e./L
2- NR = not reported
3- These values are based on the application rates approved on active labels of triclopyr BEE and triclopyr TEA on October 19, 2009. Some of these labels did not conform to the 1998 triclopyr RED. Additionally, the maximum use rate of triclopyr TEA on some use sites is 9 lbs a.e./A versus 8 lbs a.e./A for triclopyr BEE.

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

As described in the *Approach to the Assessment* section, our exposure analysis begins at the organism (individual) level of biological organization. We consider the life stage and life histories of the individuals likely to be exposed. This scale of assessment is essential as adverse effects to individuals may result in population-level consequences, particularly for populations of extremely low abundance, (*i.e.* threatened and endangered species). Characterization of impacts to an individual's fitness is necessary to assess potential impacts to populations, and ultimately to the species. To assess risk to individuals, we must consider the highest concentrations to which any individuals of the population may be exposed. Several lines of evidence discussed below suggest that EECs in the BEs underestimate exposure of some listed salmonids and designated critical habitat.

Although EPA characterized these exposure estimates as "worst case" in the BEs, it has also acknowledged that measured concentrations in the environment sometimes exceed PRZM-EXAMS EECs (EPA, 2007a). EPA has subsequently clarified that rather than providing worst case estimates, PRZM-EXAMS estimates are high end estimates for the vast majority of applications and aquatic habitats (EPA, 2007a). NMFS agrees that the model is designed to produce upper bound exposure estimates for some, but not all aquatic habitats.

Recent formal consultation and reviews of EPA informal consultations by the Services found that concentrations measured in surface water sometimes exceed peak concentrations predicted with PRZM/EXAMS modeling (NMFS, 2007c, 2008c, 2009d; USFWS, 2008). These findings demonstrate that the EECs generated using PRZM-EXAMS can underestimate peak concentrations of active ingredients that occur in salmonid habitats. Consequently, underestimation of exposure and subsequent risk to species is likely. Below, we discuss the primary reasons why EPA's exposure estimates do not represent worst case exposures to salmonid habitats.

Model outputs are 90th percentile time-weighted averages and are not worst case exposures for all salmonid habitats. It is important to recognize that the PRZM-EXAMS model predicts concentrations based on site-specific assumptions (*e.g.*, rainfall) and that environmental concentrations provided for the estimate do not represent the highest aquatic concentrations predicted given the assumptions. The exposure estimates provided in the BEs are reported as peak concentrations or time-weighted average concentrations (*e.g.,* 21 d and 60 d). Peak concentrations do not represent the maximum concentration predicted by the model. Rather, they represent the 90th percentile of the estimates derived for the given scenario (Lin, Hetrick, & Jones, 1998). Although NMFS agrees this is a relatively protective approach for evaluating exposure in some aquatic habitats, it does not represent the possible "worst case" exposure in many salmonid habitats.

Model inputs did not use maximum application rates, maximum number of applications, and minimum application intervals. We compiled maximum use rates (single and seasonal), maximum number of applications, and minimum application intervals from current labels in the *Description of the Proposed Action.* Several of the PRZM-EXAMS inputs within the BEs (Table 88) were not consistent with the maximum application rates and maximum number of applications allowed.

Few application scenarios were assessed relative to the number of approved uses. The salmonid BEs provide exposure estimates for only a portion of the labeled uses (Table 87). For example, chlorothalonil is approved for use on more than 50 agricultural commodities yet the BE provides estimates for only 6 agricultural crops. Estimates are provided for turf but not for several other types of uses such as applications to ornamentals and the use of chlorothalonil as a wood protectant.

Crop scenarios provided are not representative of the entire action area. The regional scale that the modeled scenarios are intended to represent is unclear. Some of the scenarios were conducted for states outside the distribution of listed salmonids. Others did not provide information on geographic locations simulated (*e.g.,* county, state, region, etc.). The amount of rainfall and other site-specific input assumptions can have large impacts on predicted exposure. For example, assumed site-specific inputs resulted in differences in predicted aquatic concentrations that exceed 20-fold, even when the same application rates and methods of application were evaluated (Table 88). Large differences in site-specific variables that influence transport of pesticides to aquatic habitats exist across the states of California, Idaho, Oregon, and Washington. The relatively small numbers of simulations are not likely representative of all of the pesticide use sites throughout these states. For example, the 2,4-D BE assessed scenarios developed as representative for California and Oregon while no simulations were provided for Idaho or Washington. NMFS also questions whether input values were adequate to represent the geographic variability throughout the action area. Site-specific meteorological and soil conditions vary greatly throughout the four states where listed salmonids are distributed and crops are grown. The BEs did not indicate how sites were selected and how well scenarios represented the range of conditions throughout the four states. Without a description of EPA's scenario selection, it is difficult to determine the representativeness of scenario estimates for the complete range of crop uses.

Crop scenarios do not consider application of more than one pesticide. All six a.i.s are formulated with other a.i.s to produce end-use pesticide products. More than 50 of the pesticide formulations involved in this consultation contain multiple pesticidal a.i.s. For example, a single 2,4-D product may contain as many as four a.i.s., and several captan formulations also contain malathion and/or carbaryl, neurotoxic compounds that both inhibit cholinesterase and whose use has previously been determined to jeopardize or adversely affect listed salmonids (NMFS, 2008c, 2009d). Additionally, a majority of the labels provide recommendations for co-application (tank mixtures) of more than one pesticide to improve product efficacy. Yet, the BEs did not provide exposure estimates for multiple a.i.s associated with either product mixtures or tank mixtures.

NMFS exposure estimates for floodplain habitats

Model inputs used in BEs are not representative of most vulnerable salmonid habitats. The EECs within EPA's BEs were derived primarily using the PRZM-EXAMS model. The EPA "farm pond" scenario is likely a poor surrogate of many habitats used by listed salmonids that are more susceptible to higher pesticide concentrations given their physical characteristics. Small streams and some floodplain habitats represent examples of habitats used by salmonids that can have a lower capacity to dilute pesticide inputs than the farm pond. The PRZM-EXAM estimates assume that a 10-hectare (approximately 25 acres) drainage area is treated and the aquatic habitat is assumed to be static (no inflow or outflow). Pesticide treatment areas of 10-hectares and larger occur frequently in agricultural crops, particularly under pest eradication programs. Additionally, aquatic habitats used by salmon vary in volume and recharge rates and consequently have different dilution capacities to spray drift and runoff events. The assumed drainage area to water volume ratio (100,000 m^2: 20,000 m^3) is easily exceeded for small water bodies. For example, a one-acre pond with an average depth of 1 m would exceed this ratio for treated drainage areas of approximately five acres in size and larger. The assumed aquatic habitat and size of the treated area for the PRZM-EXAMS scenarios suggest that exposure is underestimated for listed salmonids that use relatively small aquatic habitats with low dilution capacities.

Direct over-spray of pesticides to aquatic habitats

To estimate potential exposure of salmon to pesticides in floodplain and other shallow-water habitats, we first determined the initial average concentrations that will result from a direct overspray of shallow surface water (Table 90). When pesticides are applied directly to aquatic habitats the resulting initial concentration is a function of the amount applied and the volume of the water body. The active labels for the six a.i.s do not authorize direct application of pesticides to surface water. The only exceptions include the use of 2,4-D products on aquatic habitats for weed control and applications to rice. Labels recommend treatment rates for aquatic weed control that target surface water concentrations of up to 4,000μg a.e./L. Considering first order degradation of 2,4-D, this would result in a 60-d average concentration of 2610 μg a.e./L for the farm pond scenario (EPA, 2004a). EPA provided EECs resulting from use on rice. EPAs model assumes uniform application of the pesticide to a rice paddy and partitioning of the pesticide between water and the upper 1 cm of sediment (EPA, 2009a). Modeling the maximum seasonal application rate of 1.5 pounds a.e./A results in an estimated concentration in the paddy of 1431 μg a.e./L (EPA, 2004a). The concentrations in salmon habitat resulting from discharged water from treated paddies will be influenced by the volume of discharge and the physical properties of the habitats (e.g. volume and flow).

Table 90. Average initial concentration of any a.i. in surface water resulting from a direct overspray of aquatic habitat.

Application Rate (lbs a.i. / acre)	Water Depth (meters)	a.i. Concentration in Surface Water (μg/L)
0.25	2	14
0.5	2	28
1	2	56
3	2	168
10	2	560
0.25	1	28
0.5	1	56
1	1	112
3	1	336
10	1	1,121
0.25	0.5	56
0.5	0.5	112
1	0.5	224
3	0.5	673
10	0.5	2,242
0.25	0.3	93
0.5	0.3	187
1	0.3	374
3	0.3	1,121
10	0.3	3,736
0.25	0.1	280
0.5	0.1	560
1	0.1	1,121
3	0.1	3,363
10	0.1	11,208

Application of pesticides to adjacent terrestrial habitat

Some chlorothalonil products specify a no-application buffer of up to 150 feet to marine and estuarine habitats (EPA Registration No. 66222-149). However, all of the a.i.s may be applied at the immediate edge of freshwater habitats utilized by the listed salmonids. Primary drift is a likely transport mechanism for pesticides applications that occur immediately adjacent to aquatic habitats including shallow floodplain habitats where juvenile salmonids rear and shelter. We derived exposure estimates for floodplain habitats that incorporated label-specified application requirements (Table 91). These estimates were derived using the AgDrift model and estimate downwind deposition from pesticide drift (Teske, 2001). This method does not incorporate additional contributions that may occur through the runoff pathway. The drift estimates derived represent average projected drift. Although AgDrift reasonably predicts drift, drift is highly variable and is influenced by site-specific conditions and application equipment (Bird, Perry, Ray, & Teske, 2002). Our simulations assumed an aquatic habitat that was 0.1 m deep and 2 m wide. These dimensions are consistent with some of the smaller, and potentially more vulnerable floodplain habitats used by salmonids.

Table 91 Estimated average initial pesticide concentrations in a floodplain habitat that is 2m wide and 0.1m deep using AgDrift.

a.i.	Use Site	Simulation: Rate in lbs a.i./A	Buffer (feet)	Average Initial Concentration in Surface Water (μg/L)
2,4-D amines, acids, salts	Cropland	Ground[1]: 0.07-2 Air[2]: 0.07-2	0	13-368 33-956
	Pasture/Rangeland	Ground[1]: 1-2 Air[2]: 1-2	0	184-368 478-956
	Forest	Ground[1]: 4 Air[2]: 4	0	736 1,912
	Residential/Industrial	Ground[1]: 1.5	0	276
	Rights-of-way	Ground[1]: 2-4 Air[2]: 2-4	0	368-736 956-1,912
2,4-D esters	Cropland	Ground[1]: 0.05-2 Air[2]: 0.05-2	0	9-368 24-956
	Pasture/Rangeland	Ground[1]: 1-2 Air[2]: 1-2	0	184-368 478-956
	Forest	Ground[1]: 4 Air[2]: 4	0	736 1,912
	Residential/Industrial	Ground[1]: 1.5	0	276
	Rights-of-way	Ground[1]: 2-4 Air[2]: 2-4	0	368-736 956-1,912
Triclopyr BEE[3]	Cropland	Ground[1]: 2-8[4] Air[2]: 2-8	0	368 – 1,471
	Pasture/Rangeland	Ground[1]: 2 Air[2]: 2	0	368 956
	Forest	Ground[1]: 6 Air[2]: 6	0	1,103 2,868
	Residential/Industrial	Ground[1]: 1-8	0	184 – 1,471 478 – 3,824
	Rights-of-way	Ground[1]: 8 Air[2]: 8	0	1,471 3,824
Diuron	Cropland	Ground[1]: 1.2 - 4 Air[2]: 1.2 – 2.4	0	221 – 736 574 – 1,147
	Pasture/Rangeland	NA	NA	NA
	Forest	NA	NA	NA

a.i.	Use Site	Simulation: Rate in lbs a.i./A	Buffer (feet)	Average Initial Concentration in Surface Water (μg/L)
	Residential/Industrial	Ground[1]: 12	0	2,207
	Rights-of-way	Ground[1]: 12	0	2,207
Linuron	Cropland	Ground[1]: 1 - 4	0	184 - 736
	Pasture/Rangeland	NA	NA	NA
	Forest	NA	NA	NA
	Residential/Industrial	NA	NA	NA
	Rights-of-way	Ground[1]: 3	0	552
Captan	Cropland	Ground[1]: 2– 4.5 Air[2]: 2– 4.5	0	368 – 828 956 – 2,151
	Pasture/Rangeland	NA	NA	NA
	Forest	NA	NA	NA
	Residential/Industrial	Ground[1]: 1	0	184
	Rights-of-way	NA	NA	NA
Chlorothalonil	Cropland	Ground[1]: 1– 7.3 Air[2]: 1– 7.3	0	184 – 1,343 478 – 3,490
	Pasture/Rangeland	NA	NA	NA
	Forest	Ground[1]: 4.1 Air[2]: 4.1	0	754 1,960
	Residential/Industrial	Ground[1]: 11.3	0	2,078
	Rights-of-way	NA	NA	NA

1 – Tier 1 ground, Low ground boom spray, ASAE fine to medium/course distribution, 50th percentile estimate
2 – Tier 1 aerial spray, ASAE medium to course droplet distribution
NA – Spray drift calculation is not applicable because use on site not authorized

NMFS exposure estimates for pesticide mixtures

All six of the a.i.s are formulated into pesticide products that contain other a.i.s. More than 50 of the pesticide formulations subject to this consultation contain multiple a.i.s. As an example of potential exposure to pesticide mixtures, we evaluated the use of one of these products, EPA Reg. No. 228-654. This pesticide contains 62.22% diuron and 7.78% imazapyr, another photosynthetic-inhibiting herbicide. It is approved for use on utility and pipeline rights-of-way, highway rights-of-way, railroads, fence-rows, non-irrigation ditchbanks, farmyards, non-crop areas around farm buildings, and other industrial non-crop areas. It can be applied by aerial and ground application methods on these sites at single application rates up to 19 lbs of product per acre (12 lbs diuron and

1.5 lbs imazapyr/acre). The label also recommends that the product be tank mixed with nonionic surfactants, methylated seed oils, silicone-based surfactants, or fertilizer/surfactant blends. Additionally, the label provides specifications for simultaneously applying this product with other herbicides including Roundup®, Oust®, Garlon® Finale®, MSMA, Banvel®, Plateau®, and Arsenal®.

Simulations using the AgDrift model were run according to label specifications to account for potential drift of the 2 a.i.s in the formulation to aquatic habitats. A 2004 Court order for injunctive relief requires implementation of no-spray buffers to certain water containing listed salmon in California, Oregon, and Washington (*Washington Toxics Coalition v. EPA*, C01-132C (W.D. Wash. 1/22/2004)). Buffers of 60 feet for ground applications and 300 feet for aerial applications are in effect until EPA completes its consultion. Consequently, we evaluated potential loading via drift with and without the injunctive relief buffers. Results are presented below in Table 92. To simulate a linear right-of-way application, a single aerial swath with a coarse droplet size was assumed. Other input values are reported in *Appendix 7*.

Table 92. AgDrift estimated concentrations of pesticides in surface water adjacent to aerial application at the maximum labeled use rate for EPA Reg. No. 228-654 (12 lbs diuron/A and 1.5 lb imazapyr/A).

Chemical	Buffer	Average initial concentration (μg/L)	
	Ft	EPA-defined pond	NMFS-defined floodplain habitat
With injunctive relief buffers			
diuron	300	1.002	30.8
imazapyr	300	0.125	3.85
Without aquatic habitat buffers			
diuron	0	25.0	2,789
Imazapyr	0	3.12	349
Label prohibits use of product in California			

Monitoring Data: Measured Concentrations of Parent Compounds and Degradates in Surface Waters

We reviewed two types of pesticide monitoring data: 1) ambient data that measure concentrations of pesticides and other contaminants in surface waters, but are not targeted at the field level with any specific pesticide application, and 2) data from more targeted studies (frequently found in published scientific literature and gray literature), which collected samples in waters near where the pesticide of interest was used. We evaluated data from three central sources: USGS' NAWQA database, state databases maintained by California and Washington and targeted monitoring studies which may not be included in monitoring databases. Neither Oregon nor Idaho currently maintains a state database. Data from Washington include studies conducted by the Washington State Departments of Ecology and Agriculture. The NAWQA data typically are general monitoring data, and sampling stations are distributed across a range of land uses, although some data may be from investigations into specific uses. The California and Washington databases contain data from studies that fall into both categories.

Diuron and linuron may persist in the aquatic environment for several weeks depending on site-specific characteristics that contribute to dissipation. 2,4-D, trichlopyr, captan, and chlorothalonil have relatively short aquatic half-lives in the range of a few days to a few weeks, and therefore the detection of these compounds is less likely, and more greatly influenced by sample design (e.g. timing of sampling relative to the timing of application or runoff events). In the following section we describe study design considerations for assessing the utility of monitoring data for evaluating exposure of pesticides to salmon.

Monitoring data considerations

Surface water monitoring can provide useful information regarding real-time exposure and the occurrence of environmental mixtures. A primary consideration in evaluating monitoring data is whether the study design is sufficient to address exposure in a qualitative, quantitative, or probabilistic manner. The available monitoring studies were conducted under a variety of protocols and for varying purposes. General water quality monitoring conducted in larger streams and rivers frequently does not capture "peak" concentrations because it is not correlated with applications and/or storm events following those applications and not all habitats types are sampled. This is one of the reasons NMFS did not use available monitoring data for probabilistic modeling (*i.e.*, it likely does not contain the complete range of possible concentrations). Additionally, the monitoring sampling designs and sites do not represent many salmonid species' ranges (see Figure 61).

Of the monitoring programs discussed, only the Washington State Department of Ecology program was specifically designed to evaluate potential exposure to listed salmonids. This study monitored selected urban and agricultural areas that do overlap with some listed Pacific salmonid habitats in Washington State. This sampling program was intended to evaluate pesticide occurrence in a limited number of salmonid-bearing streams during the pesticide application seasons (A. Johnson & Cowles, 2003). The study design included sampling during the pesticide application season but did not target specific applications of pesticides nor did it target salmonid habitats that would be expected to produce the highest concentrations of pesticides (*e.g.,* shallow off-channel habitat in close proximity to pesticide applications). Sampling was generally conducted on a weekly basis, so it is likely peak concentrations associated with drift and runoff events were not captured. This monitoring program is discussed in more detail in *Monitoring Data from Washington State.*

Other available monitoring data are also applicable to assessing exposure in listed salmon, but to varying degrees. Common aspects that limit the utility of the available monitoring data as accurate depictions of exposure within listed salmonid habitats include: 1) protocols were not designed to capture peak concentrations or durations of exposure in habitats occupied by listed species; 2) limited utility as a surrogate for other non-sampled surface waters; 3) lack of representativeness of current and future pesticide uses and conditions; and 4) lack of information on actual pesticide use to correlate with observed surface water concentrations.

Protocols not designed to capture peak exposure. The NAWQA monitoring studies contain the largest data set evaluated. However, these studies were designed to evaluate trends in water quality and were not designed to characterize exposure of pesticides to listed salmonids (Hirsch, 1988). The NAWQA design does not result in an unbiased representation of surface waters, which limits the ability to make statistical extrapolations to unsampled waters. Also, some agricultural activities and related pesticide uses that may be very important in a particular region may not be represented in the locations sampled. Sampling from the NAWQA studies and other studies reviewed was typically not conducted in coordination with specific applications of the six a.i.s addressed in this Opinion. Similarly, sampling was not designed with consideration to salmon distribution or to target the salmonid habitats most likely to contain the greatest concentrations of pesticides. Given the relatively rapid dissipation of these pesticides in flowing water habitats, it is not surprising that pesticide concentrations from these datasets were generally much lower than those predicted by modeling efforts.

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

Representativeness of current and future uses. Pesticide use varies annually depending on regulatory changes, market forces, cropping patterns, and pest pressure. The use of the six a.i.s in California over the recent decade has either shown a general decrease or remained relatively stable. However, pesticide use patterns change annually and may result in either increases or decreases in use of pesticide products for specific uses. There is considerable uncertainty regarding the representativeness of monitoring conditions to forecast future use of products containing these a.i.s. Prediction of future use is complicated by climate change that may affect agriculture uses and pest pressures.

Lack of information on actual use to correlate with observed concentrations. A common constraint in the monitoring data was lack of information on actual use of pesticides containing the six a.i.s. For example, the ability to relate surface water monitoring data to the proposed action was severely hampered because information on application rates, setbacks/buffers, and application methods associated with the monitoring were generally not reported. In most cases, the temporal and spatial aspect of pesticide use relative to sampling was not reported, further limiting the utility of the information.

Data Described in USEPA's Biological Evaluations

EPA summarized monitoring data in the BEs, derived mostly from the same sources we have considered. As we considered information from these databases, including the more recent data, we do not reiterate the BE summaries herein.

USGS NAWQA Data for California, Idaho, Oregon, and Washington

We obtained updated data from the USGS NAWQA database to evaluate the occurrence of 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil in surface waters monitored in California, Idaho, Oregon, and Washington. Triclopyr BEE (which rapidly converts to the triclopyr acid/anion) and captan are not on the USGS list of analytes. However, we obtained the available data for some of the identified degradates including the triclopyr acid, a degradate of 2,4-D (2,4-DCP) and two degradates of diuron (3,4-DCA and CPMU). No information was available for captan or its degradates. Land uses associated with the sampling stations included agriculture, forest, rangeland, urban, and mixed use. The database query resulted in approximately 5,400 samples in which one or more of the a.i.s or degradates was an analyte. Approximately 360 unique sampling locations were represented, with sample sites located in 11 NAWQA basins distributed throughout California, Idaho, Oregon and Washington (Figure 61). Some waterbodies and/or basins in this dataset do not contain listed salmonids and several of the species have had no sampling within their freshwater and coastal habitats (Table 93). Most notable are those ESUs/DPSs along the coasts of Oregon and California as well as listed salmonid habitats within Idaho. Available data included samples collected from 1991-2010. More than one third of the stations were sampled only once during the span of 19 years, and a relatively small number of sites accounted for the majority of the data; approximately 75% of the data was collected from 36 sites. The temporal and spatial distribution of sampling is inconsistent with temporal and spatial aspects of salmonid distribution. Consequently, we do not expect the data set to be representative of exposure distributions for listed salmonids.

Table 93. Number of NAWQA sample sites within the distribution of listed Pacific salmonids.

Species	ESU	Kilometers of Stream Inhabited	Sites in Spawning and Rearing Habitat	Sites in Migratory Corridor
Chinook	Puget Sound	3,639.65	39	NA
	Lower Columbia River	2,443.29	15	NA
	Upper Columbia River Spring - Run	1,646.75	0	5
	Snake River Fall - Run	1,370.44	0	3
	Snake River Spring/Summer - Run	5,288.23	0	3
	Upper Willamette River	3,013.85	43	3
	California Coastal	2,422.44	0	NA
	Central Valley Spring - Run	2,212.94	6	0
	Sacramento River Winter - Run	546.84	6	0
Chum	Hood Canal Summer - Run	141.89	2	NA
	Columbia River	1,162.18	14	NA
Coho	Lower Columbia River	3,307.78	18	NA
	Oregon Coast	10,220.00	0	NA
	Southern Oregon and Northern California Coast	5,619.58	0	NA
	Central California Coast	1,287.78	0	NA
Sockeye	Ozette Lake	70.98	0	0
	Snake River	1,493.94	0	3
Steelhead	Puget Sound	3,849.64	39	NA
	Lower Columbia River	4,302.03	17	1
	Upper Willamette River	3,063.07	26	3
	Middle Columbia River	10,196.80	84	2
	Upper Columbia River	2,143.15	8	2
	Snake River	13,423.40	0	3
	Northern California	5,324.31	0	NA
	Central California Coast	4,620.72	0	NA
	California Central Valley	4,273.66	49	0
	South-Central California Coast	5,104.56	0	NA
	Southern California	3,015.86	2	NA

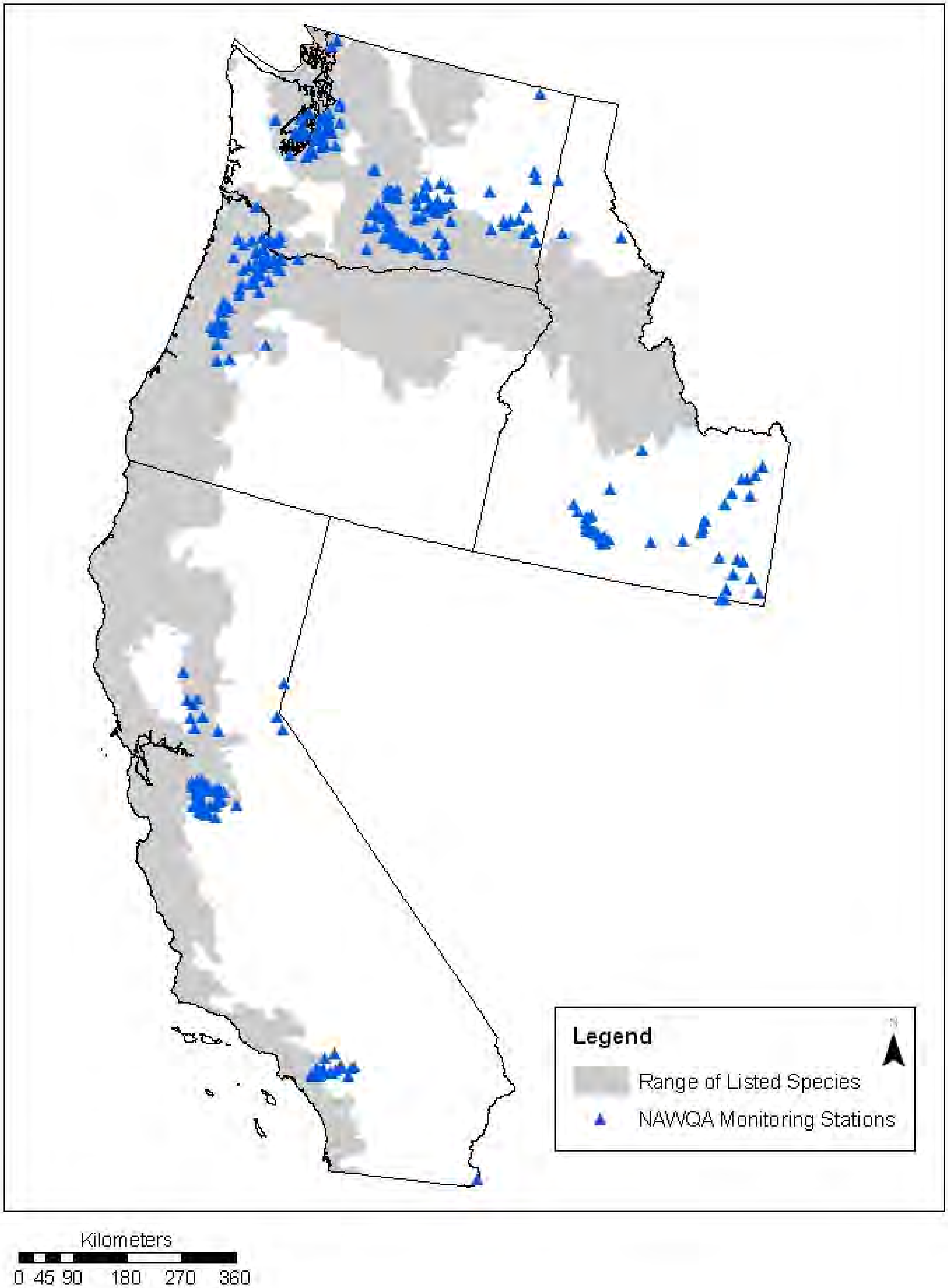


Figure 61. Distribution of NAWQA monitoring sites and listed Pacific salmonids.

The frequency of detection is a combination of the actual occurrence of pesticides in the water and the sampling intensity. NAWQA surface water detections represent the dissolved phase, as the water sample is filtered through a 0.7 micron glass fiber filter. Chemicals transported primarily in the particulate phase would be underreported in this data set. No sediment or tissue data were available from USGS for the six a.i.s. This is also a recognized uncertainty and compromises our ability to determine toxicity of contaminated sediments. Because the USGS monitoring program does not generally coordinate sampling efforts with specific pesticide applications or runoff events, detected concentrations are likely to be lower than actual peak concentrations adjacent to pesticide application sites that occur immediately following drift or a runoff event.

Summary information for quantifiable concentrations of pesticides addressed in this Opinion (Table 94) and their degradates (Table 95) are presented below. In the USGS database, non-detects are reported as less than (“<”) the laboratory reporting level (LRL) for that sample. Other than total number of samples (n), summary statistics were calculated on samples not designated as (“<”). The LRL ranges reported were estimated based on “<”-qualified data. Many of the concentrations that could be quantified were designated as “E,” meaning the concentrations were estimated. These data are included in the summary statistics.

Sampling intensity varied considerably among the available analytes and ranged from 5 – 4327 samples collected during the 19 year sampling period. The most commonly occurring were diuron (46%, range 0.002-23.3 μg/L, median 0.111 μg/L), the 3,4-DCA degradate of diuron (41%, range 0.01-0.7 μg/L, median 0.013 μg/L), and 2,4-D (27%, range 0.007-7.66 μg/L, median 0.085 μg/L). Triclopyr (8.7%, range 0.09-3.35 μg/L, median 0.119 μg/L) and CPMU (4.9%, range 0.0037-0.1701 μg/L, median 0.0231 μg/L) were also detected on a frequent basis, while linuron (0.92%, range 0.007-0.68 μg/L, median 0.022 μg/L) and chlorothalonil were not (0.08%, 0.29 μg/L).

Table 94. Concentrations of Parent Pesticides in NAWQA Water Samples for California, Idaho, Oregon, and Washington.

Statistic	2,4-D	Diuron	Linuron	Chlorothalonil
Samples	1274	1364	4327	1188
Percent detections	27.00	45.60	0.92	0.08
LRL range (μg/L)	0.0218 - 0.27	0.009 - 1.25	0.014 - 1.1	0.035 - 0.48
Minimum concentration (μg/L)	0.007	0.002	0.007	0.29
Maximum concentration (μg/L)	7.660	23.3	0.68	0.29
Median concentration	0.085	0.111	0.022	0.29

Table 95. Concentrations of Degradates in NAWQA Water Samples for California, Idaho, Oregon, and Washington.

Statistic	2,4-DCP	Triclopyr Acid	3,4-DCA	CPMU
Samples	5	1366	1309	526
Percent Detections	0	8.71	41.33	4.94
LRL range (μg/L)	5	0.05 – 0.68	0.004 – 0.023	0.0242 – 0.0915
Minimum concentration (μg/L)	ND	0.09	0.01	0.0037
Maximum concentration (μg/L)	ND	3.35	0.7	0.1703
Median concentration	ND	0.119	0.013	0.0231

Monitoring Data from California Department of Pesticide Regulation

We evaluated monitoring data available from the CDPR, which maintains a public database of pesticide monitoring data for surface waters in California. Entries in the database (www.cdpr.ca.gov/docs/emon/surfwtr/surfdata.htm) are from multiple sources, including monitoring conducted by CDPR, USGS (data from the NAWQA program, as well as other studies), state, city, and county water resource agencies; and some non-governmental or inter-governmental groups such as Deltakeeper. The CDPR requires a formal QA/QC protocol for data submitted or does a separate QA/QC review, thus only data subject to appropriate QA/QC procedures are included in the surface water database. Unlike the USGS NAWQA data set, the CDPR database may contain whole water samples as well as filtered samples. If whole water concentrations are reported for compounds that sorb significantly to the particulate phase, concentrations would appear higher than in a filtered sample, which represents only the dissolved phase. The majority of the studies, which are described in metadata available from CDPR, are not targeted at correlating water concentrations with specific application practices. The database, last updated in June 2008, consists of approximately 270,000 data records. Each record reports a specific sampling site, date, and analyte. The number of records associated with a particular compound is indicative of monitoring intensity rather than actual occurrence in surface waters. In this database, detections below the LOQ are reported as 0 μg/L. Summary statistics were calculated on samples with values above the LOQ.

Some data were available for all a.i.s considered in this opinion, although the number of samples was variable among the analytes. Summary information is provided below in Table 96. Diuron, triclopyr, and 2,4-D were the most frequently detected compounds (14-47% frequency of detection). Samples were analyzed for linuron most frequently and were detected in about 1% of the samples. Captan and chlorothalonil were not detected in any of the samples, but sampling for these compounds was relatively limited compared to the other analytes.

Table 96. Surface Water Concentrations of Pesticides in CDPR Database.

Statistic	2,4-D	Triclopyr	Diuron	Linuron	Captan	Chlorothalonil
Samples	809	583	1971	2805	220	400
Percent Detections	14.0	17.5	46.6	0.93	0	0
LOQ range (μg/L)	0.0218-0.27	0.0224-0.68	0.009-1.25	0.002-5	0.1	0.035-0.578
Minimum concentration (μg/L)	0.0244	0.0251	0.0035	0.0073	0	0
Maximum concentration (μg/L)	4.42	14.5	160	1.6	0	0
Median concentration	0.16	0.7	0.275	0.275	0	0
Dates	1991-2006	1993-2006	1992-2006	1992-2006	1994-2006	1993-2005
# of Studies	9	7	21	12	2	3

Monitoring Data from Washington State

Data from monitoring studies conducted in the state of Washington are included in Department of Ecology's Environmental Information Management (EIM) database (http://www.ecy.wa.gov/eim/). Data in the database are from multiple sources, including state agencies, and may contain whole water samples as well as filtered samples. The EIM requires a formal QA/QC protocol for data submitted or does a separate QA/QC review, thus only data subject to appropriate QA/QC procedures are included. Some of the studies contained in this database may be targeted with respect to specific pesticide uses, while others are more generalized water quality surveys. Some data for all pesticides considered in this Opinion were available, and are shown in Table 97.

The procedure for reporting in the EIM database includes reporting non-detects as the reporting limit for that particular sample, and adding a "U" data qualifier. The reporting limit was not specified in the data accessed by NMFS, thus LOQ ranges were estimated based on "U"-qualified data. Summary statistics were calculated on samples with values above the LOQ (*i.e.*, not qualified with a "U"). Data with a "REJ" qualifier did not meet quality control standards and were not considered. Statistics include data qualified with a "J" (analyte positively identified, resulting value an estimate) and data qualified with an "NJ" (analyte tentatively identified, resulting value an estimate).

In the complete dataset, sample sets consisted of 1282-2370 samples for each of the analytes of interest. 2,4-D acid (48%) , diuron (22%), and triclopyr acid (16%) were the most frequently detected pesticides. Maximum concentrations for these three compounds were 6.57 μg/L, 4.1 μg/L, and 1.3 μg/L, respectively. Linuron, captan, and chlorothalonil were detected <3% of the time with maximum concentrations ranging from 0.12 – 0.6 μg/L. The database indicates captan was detected at a concentration of 0.6 μg/L on 23 occasions. However, the accuracy of these particular samples is unknown because there were no entries in the "Report Data Qualifier" field. If we include the samples, the detection frequency for captan was 1.28%. Captan was not detected at concentrations at or above the LRL in other samples that were considered acceptable.

Table 97. Concentrations of Pesticides in Washington EIM Database.

Statistic	2,4D acid	Triclopyr acid	Diuron	Linuron	Captan	Chlorothalonil
Samples	2040	1984	2370	1282	1802	2080
Percent Detections	47.55	16.13	21.90	3.28	0-1.28	1.88
LRL range (µg/L)	0.03 - 1	0.063-0.25	0.0045 - 2.9	0.0054-5.433	0.033-2.1	0.0056 – 19
Minimum concentration (µg/L)	0.001	0.0028	0.0022	0.03	0.6	0.0027
Maximum concentration (µg/L)	6.57	1.3	4.1	0.12	0.6	0.36
Median concentration (µg/L)	0.063	0.0415	0.043	0.12	0.6	0.12
Dates	1992-2009	1992-2009	1992-2009	1992-2009	1992-2009	1992-2009
# of Studies	31	25	29	8	22	26

Included in the EIM database is a subset of recent monitoring efforts conducted by the Washington State Departments of Ecology and Agriculture in some of Washington's salmon-bearing streams. Final reports for 2003-2008 seasons are publically available on the internet (http://agr.wa.gov/PestFert/natresources/SWM/default.htm). Monitoring was conducted in 2009, and is also provided in the database. A separate summary of data from those investigations is provided below. Water samples are not filtered, and thus concentrations reported include pesticides in both dissolved and particulate phases, although the sampling protocol specifies an attempt to avoid collection of excessive particulates (A. Johnson & Cowles, 2003). Whole water concentrations for compounds that sorb significantly to the particulate phase will appear higher than those for a filtered sample, which represents only the dissolved phase.

The Washington program sampled between 0 and 16 sample stations for the six analytes from 2003 - 2009 (Table 98). Sampling stations were located primarily in agricultural-dominated watersheds (Figure 62). A single watershed, the Cedar-Sammamish (Thornton Creek) represented the urban sites. Three sites were sampled in Thornton Creek in 2003, and 2 sites from 2004-2009. Agricultural sites were distributed in four watersheds (Lower Yakima, Skagit/Samish, Wenatchee and Entiat), but only the Lower

Yakima sites have been sampled since 2003. Sites in the Skagit/Samish watershed were added in 2006 and sites in the Wenatchee and Entiat were added in 2007. Sampling favored the detection of multiple pesticides, rather than peak concentrations in some habitats used by listed salmonids as sampling was not coordinated with pesticide applications at the field scale. Generally, samples were taken weekly between March and September at the various sites, but the specific sampling design has changed somewhat over the years. The limited number and spatial distribution of samples sites does not reflect the distribution of listed ESUs/DPSs in the state and are not expected to represent the full range of habitats and potential exposure of listed salmonids to pesticides.

Table 98. Number of stations sampled for the presence of pesticides by the Washington Department of Ecology.

Sample collection year	2,4D acid	Triclopyr acid	Diuron	Linuron	Captan	Chlorothalonil
2003	9	9	9	0	9	9
2004	6	6	6	0	5	6
2005	6	6	6	0	6	6
2006	11	11	11	11	11	11
2007	16	16	16	16	16	16
2008	16	16	16	16	16	16
2009	16	16	16	16	16	16

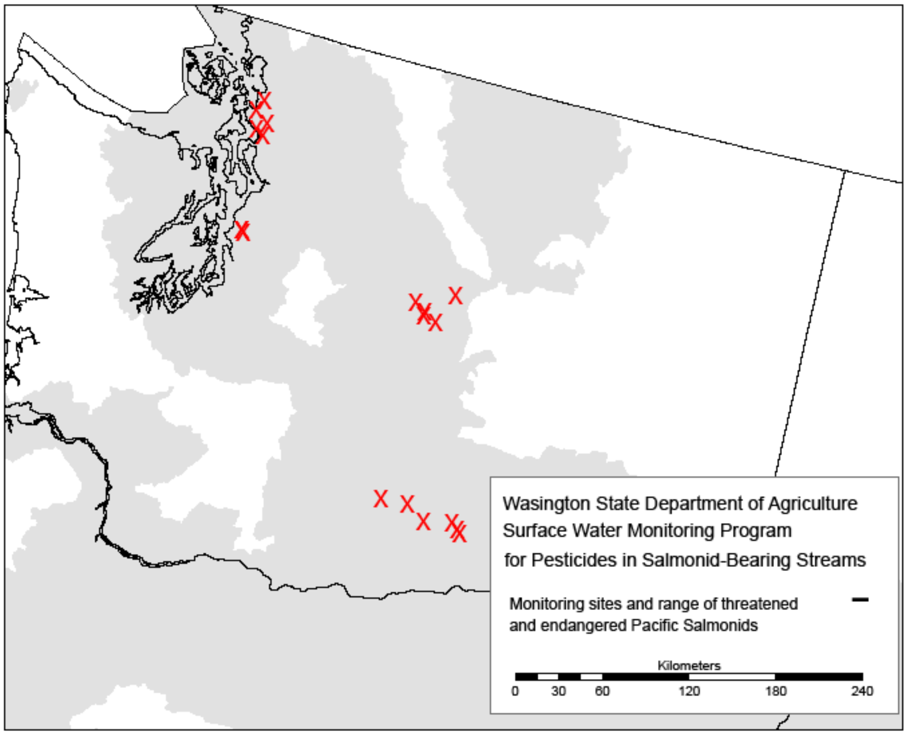


Figure 62. Distribution of Washington Department of Ecology sample stations compared to the distribution of listed salmon ESUs/DPSs.

Table 99. Washington Department of Ecology sample sites within the distribution of listed Pacific salmonids.

Species	ESU	Kilometers of Stream Inhabited	Sites in Spawning and Rearing Habitat	Sites in Migratory Corridor
Chinook	California Coastal	2,422.44	NA	NA
	Central Valley Spring - Run	2,212.94	NA	NA
	Lower Columbia River	2,443.29	0	0
	Upper Columbia River Spring - Run	1,646.75	5	0
	Puget Sound	3,639.65	8	0
	Sacramento River Winter - Run	546.84	NA	NA
	Snake River Fall - Run	1,370.44	NA	0
	Snake River Spring/Summer - Run	5,288.23	0	0
	Upper Willamette River	3,013.85	NA	0
Chum	Columbia River	1,162.18	0	0
	Hood Canal Summer - Run	141.89	0	0
Coho	Central California Coast	1,287.78	NA	NA
	Lower Columbia River	3,307.78	0	0
	Southern Oregon and Northern California Coast	5,619.58	NA	NA
	Oregon Coast	10,220.00	NA	NA
Sockeye	Ozette Lake	70.98	0	0
	Snake River	1,493.94	NA	0
Steelhead	Central California Coast	4,620.72	NA	NA
	California Central Valley	4,273.66	NA	NA
	Lower Columbia River	4,302.03	0	0
	Middle Columbia River	10,196.80	6	0
	Northern California	5,324.31	NA	NA
	Puget Sound	3,849.64	8	0
	Snake River	13,423.40	NA	0
	South-Central California Coast	5,104.56	NA	NA
	Southern California	3,015.86	NA	NA
	Upper Columbia River	2,143.15	5	0
	Upper Willamette River	3,063.07	NA	0

The data provided by the Washington State Departments of Ecology and Agriculture was largely reflective of the results of the larger EIM dataset as it accounted for more than 84-96% of the data for each of the 6 analytes (Table 100). 2,4-D acid (44%), diuron (22%), and triclopyr acid (12%) were the most frequently detected pesticides. Maximum concentrations for these three compounds were 6.57 μg/L, 4.1 μg/L, and 1.3 μg/L, respectively. Chlorothalonil and linuron were detected <1% of the time at maximum concentrations of 0.03 and 0.5, respectively. Captan was not detected.

Table 100. Concentrations of pesticides detected in recent studies by Washington Department of Ecology (2003-2009)[1].

Statistic	2,4-D acid	Triclopyr acid	Diuron	Linuron	Captan	Chlorothalonil
Samples	1708	1708	2011	1229	1526	1738
Percent Detections	43.74	12.30	21.73	0.16	0	0.35
LRL range (μg/L)	0.076-0.17	0.079-0.15	0.19 - 1	0.0054-0.5	0.015-0.44	0.0056 – 0.41
Minimum concentration (μg/L)	0.0047	0.0028	0.0022	0.03	0	0.0071
Maximum concentration (μg/L)	6.57	1.3	4.1	0.5	0	0.03
Median concentration (μg/L)	0.058	0.0325	0.04	0.042	0	0.0144

1 Data in this table are a subset of data used to create Table 97.

Monitoring Data from Oregon

Data from monitoring studies conducted in Oregon are available in the Oregon Department of Environmental Quality's (ODEQ) Laboratory Analytical Storage and Retrieval (LASAR) database (http://deq12.deq.state.or.us/lasar2/). All data contained in LASAR are reviewed, verified, validated, and qualified by the Laboratory and Environmental Assessment Division at ODEQ. Studies of particular relevance in the database include monitoring conducted by ODEQ through its Pesticide Stewardship Partnerships (PSPs). The primary objective of these partnerships is to identify and improve water quality problems through voluntary adaptive management. The approach has been used since 1999, with initial surface water monitoring focused on twelve pesticides and some related degradates in a few watersheds and sub-basins where water quality standards had been frequently exceeded. In recent years the monitoring program has been expanded to include additional pesticides and areas. Currently, seven sub-basins are included in the program, with the Amazon Creek Watershed near Eugene, OR added in 2011. Since 2009, the PSP monitoring program has monitored 100 pesticides, including 2,4-D, triclopyr, diuron, linuron, and chlorothalonil. Captan has not been monitored. The sample locations for the study areas overlap with spawning and rearing habitat of several listed salmonids (Figure 63).

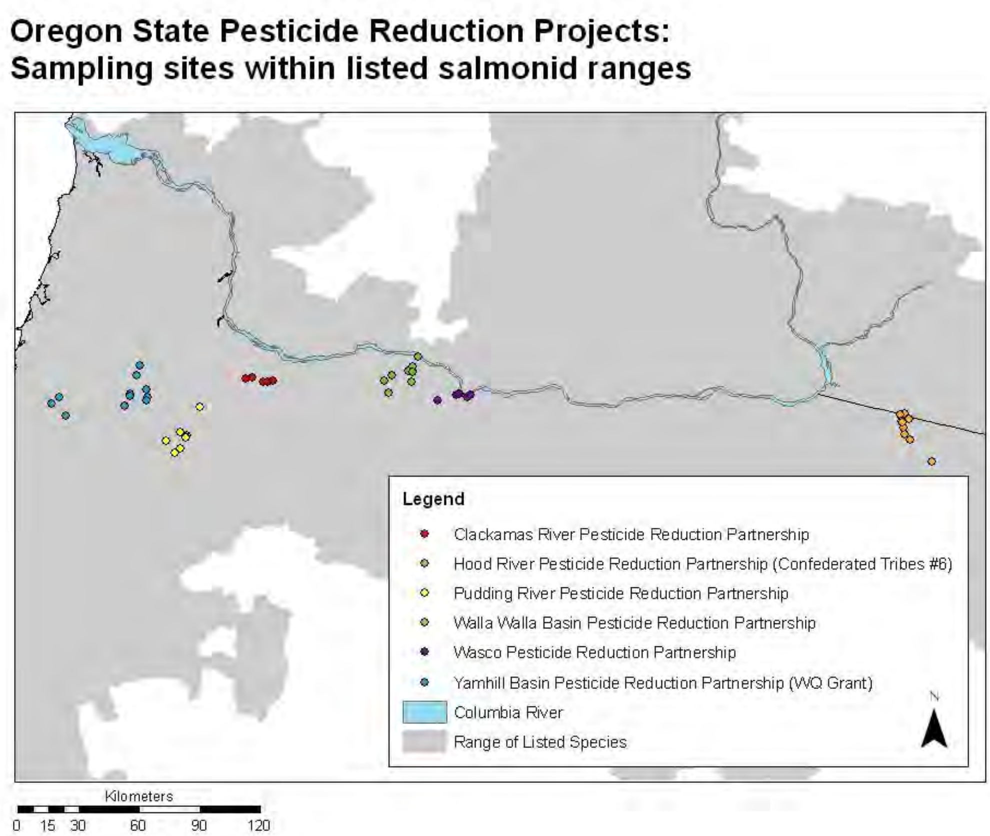


Figure 63. Distribution of Oregon Department of Environmental Quality sample stations compared to the distribution of listed ESU/DPSs.

The study locations and timing of sampling events were chosen considering pesticide use patterns based on local knowledge. Sampling stations are primarily located at publicly accessible sites. The spatial and temporal relationship of sampling to actual pesticide use is unknown. The study design allows for evaluation of ambient water quality trends within the monitored sub-basin (Masterson, 2011).

A summary of the monitoring results based on LASAR queries is provided below in Table 101. Sample sets consisted of 643-703 samples for each pesticide. Diuron was the most frequently detected analyte with a maximum concentration of 26.9 μg/L. Linuron and chlorothalonil were dected in <2% of the samples with maximum concentrations of 0.104 μg/L and 0.688 μg/L, respectively. 2,4-D and triclopyr were not detected during the two years of monitoring although their detections limits (10 μg/L) where much greater compared to the other analytes.

Table 101. Concentrations of pesticides detected in recent studies by Oregon Department of Environmental Quality (2009-2010).

Statistic	2,4D acid	Triclopyr acid	Diuron	Linuron	Chlorothalonil
Samples	703	703	681	681	643
Percent Detections	0	0	64.17	1.62	0.78
LRL range (μg/L)	10	10	0.002-0.410	0.002-0.039	0.018-0.056
Minimum concentration (μg/L)	ND	ND	0.0041	0.0042	0.051
Maximum concentration (μg/L)	ND	ND	26.90	0.104	0.688
Median concentration (μg/L)	ND	ND	0.0687	0.0112	0.232
Dates	2009 - 2010	2009 - 2010	2009 - 2010	2009 - 2010	2009 - 2010
# of Study areas	6	6	6	6	6

Targeted Monitoring Studies

In some cases, EPA documents including the salmonid BEs, California Red-legged Frog (RLF) BEs, and EPA RED documents reported targeted monitoring studies where samples are collected adjacent to the site of pesticide application and water concentrations or the percentage of runoff is associated with particular application rates and/or methods. We describe those in this section, along with other information available on targeted monitoring within the open literature.

2,4-D

The salmonid BE, RLF BE, and EPA RED documents do not identify any targeted surface water monitoring data for 2,4-D. However, a recent study reports 2,4-D and chlorothalonil concentration in water discharged from a golf course (King & Balogh, 2010). Inflow and outflow concentrations of the two chemicals were measured in April through November from 2003 to 2008. Information on chlorothalonil is discussed in the corresponding section below. Total annual 2,4-D used on the golf course ranged from 0.4 – 1.1 lbs/A. However, this rate is not directly comparable to the labeled maximum rate of 4 lbs/A because it's an aerial weighted average that includes treated (primarily golf course roughs) and nontreated areas on the course (e.g. greens). The mean annual loading of 2,4-D was 4.9 g per hectare or 0.5% of applied. The maximum 2,4-D concentration observed was 67.1 μg/L.

Triclopyr BEE

The salmonid BE (EPA, 2004f) reports a single targeted study that resulted in a maximum surface water concentration of 350 μg/L. Triclopyr BEE is not permitted for direct application to aquatic habitats. However, Thompson et al. (1991) investigated the environmental fate of triclopyr BEE in a stream to characterize potential aquatic exposure in the event of an accidental overspray or drift from an aerial forest application. A nominal application rate of 3.4 lbs a.i./A was assessed versus a maximum 6 lbs a.i./A single application rate, which is permitted on labels. The stream depth ranged from 0.5 – 2 m with a velocity of 16 cm/s. Concentrations of triclopyr BEE, triclopyr acid, and pyridinol metabolite (PYR) were evaluated. Peak concentrations of triclopyr BEE measured were 230 and 350 μg/L in the treatment area and downstream sample sites, respectively. The pH level in the stream (7 – 7.5) was conducive to hydrolysis of triclopyr BEE and dissipation of triclopyr BEE was relatively rapid. The time-weighted average concentrations of triclopyr BEE during the first 12-14 hrs post application were 50 μg/L in samples from the treated area, and 110 μg/L in samples from a downstream sample site. Average concentrations of triclopyr BEE declined to less than 1 μg/L 72 h post-application. The maximum concentration of triclopyr acid detected was 140 μg/L. PYR residues did not exceed the level of quantification, which was relatively high (50 μg/L). A spike in total triclopyr residues of 90 μg/L was measured in the stream following a rain evident that occurred 2 days post treatment. Sampling following additional storms at 4 and 31 days post treatment did not reveal quantifiable concentrations of total triclopyr (> 50 μg/L). Although triclopyr cannot be directly applied to surface water, the information in this report is pertinent as accidental oversprays of small water bodies are expected to occur with aerial applications in forested environments. Additionally, initial concentrations resulting from the application rate studied (3.4 lbs per acre) are comparable to concentrations that are expected in shallow water bodies from aerial applications that lack buffers (Thompson et al., 1991). No other targeted monitoring studies were reported in the EPA documents or found in the open literature.

Diuron

The diuron BEs report four studies that investigated concentrations in surface water runoff, although the maximum labeled application rate of 12 lbs a.i./A was not evaluated (Table 102). All studies reported relatively high concentrations of diuron present in runoff water. The highest concentrations detected were those associated with runoff from rights-of-way treatments at approximately 3 lbs diuron/A (3.59 Kg/Ha). Powell et al.(1996) treated 2.4 meter wide strips and measured diuron concentrations in surface water runoff. Detected diuron concentrations in runoff ranged from 144 – 1770 μg/L following a simulated rainfall and 46 – 2849 μg/L following natural rain. The largest amount removed during any sample period was 8.4% of the applied material in a 28-hr period (Powell, et al., 1996). Assuming this material is deposited in an adjacent waterway that runs the length of the treatment area and is 2 meters wide and 0.5 meters deep would result in a diuron concentration of approximately 72 μg/L.

Table 102. Concentrations of diuron detected in surface water runoff.

Use Site	Application Rate lbs diuron/A	Concentrations Detected µg/L	Source
Vineyards	1.8	<0.3 - 467	(Andrieux, Lennartz, Louchart, & Voltz, 1997)
Citrus	Variable:1-4	3.1 - 891	(Braun & Hawkins, 1991)
Citrus	1.6	600 – 1700	(Spurlock, Garretson, & Troiano, 1997)
Rights-of-way	3.2	46 – 2849	(Powell, et al., 1996)

Chlorothalonil

The BEs indicate that surface water monitoring for chlorothalonil that is coordinated with specific use patterns of the pesticide were not available. However, a recent study discussed above, reports 2,4-D and chlorothalonil concentration in water discharged from a golf course(King & Balogh, 2010). The median outflow concentration of chlorothalonil (0.58 µg/L) was significantly greater than the inflow concentration, which was below the detection limit (0.07 µg/L). The mean annual loading of chlorothalonil was 10.5 g per hectare or 0.3% of applied. The maximum chlorothalonil concentration observed was 48.1 µg/L. During the months of April – October 90th percentile concentrations for chlorothalonil ranged from 2 – 4 µg/L, but reached approximately 11 µg/L during November. Another golf course study found a peak concentration of 15µg/L (Ryals, Genter, & Leidy, 1998). Additionally, a small plot study in peanuts reported average runoff concentrations of 95-260 µg/L (Wauchope, Johnson, & Sumner, 2004). These targeted studies are discussed in greater detail in the *Risk Characterization* section.

Linuron and Captan

No targeted monitoring studies were described in either the BE, RED chapter or the RLF effect determinations. Additionally, we found no targeted studies in the open literature that evaluated surface water concentrations of these two pesticides related to field level use of pesticides.

Summary of Monitoring Data

NMFS did not locate many edge-of-field studies for the compounds addressed in this Opinion. However, when targeted monitoring studies were available maximum concentrations detected exceeded maximum concentrations in the monitoring databases reviewed (NWQA, CDPR, and EIM; Table 103). The open literature that was evaluated and the general state of knowledge regarding field runoff from pesticide applications lead us to anticipate the following:

- edge-of-field runoff concentrations will be higher than concentrations measured in waterbodies with substantial diluting volume,
- low-flow or runoff-dominated systems likely contain the highest concentrations (approaching or exceeding modeled concentrations), and
- measured concentrations in general monitoring programs are likely to be lower than peak runoff concentrations, as sampling may not coincide with initial application and/or runoff events.

Table 103. Monitoring Data Concentrations.

Chemical	From Databases[1]				From Targeted Studies	
Parent compounds	Min Conc. µg/L	Monitoring Database	Max Conc. µg/L	Monitoring Database	Max Conc. µg/L	Study Reference
2,4-D	0.11	EIM	7.66	NWQA	67.1	(King & Balogh, 2010)
Triclopyr BEE	NE	NA	NE	NA	350	(Thompson, et al., 1991)[2]
Diuron	0.002	NWQA	160	CDPR	2849	(Powell, et al., 1996)[3]
Linuron	0.003	EIM	1.6	CDPR	NE	NA
Captan	0.6	EIM	0.6	EIM	NE	NA
Chlorothalonil	0.0027	EIM	0.36	EIM	48.1	(King & Balogh, 2010)
Degradates						
2,4-DCP	ND	NWQA	ND	NA	NE	NA
Triclopyr acid	0.0028	EIM	14.5	CDPR	140	(Thompson, et al., 1991)[2]
3,4-DCA	0.01	NWQA	0.7	NWQA	NE	NA
CPMU	0.0037	NWQA	0.1703	NWQA	NE	NA

[1]Minimum and maximum based on detected values
[2]Monitoring from direct overspray of forest stream
[3]Concentration in surface runoff water
ND Not detected
NE Not evaluated
NA Not applicable

Exposure to Other Action Stressors

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

Metabolites and degradates of the six a.i.s

EPA documents identified major degradates and degradates of toxicological concern of the six a.i.s (see *Summary of the Chemical Fate of the Six Active Ingredients*). However, estimates quantifying potential exposure of listed salmonids and their habitat to these transformation products were not provided and remain a considerable source of uncertainty. In general, failure to consider exposure to these breakdown products increases the likelihood that risk is underestimated.

Other ingredients in formulated products

Registered pesticide products containing the six a.i.s include other ingredients such as carriers, surfactants, and synergists. NMFS reviewed many of the active labels of the six a.i.s and found pesticide products commonly contain multiple a.i.s. Examples of some of the formulations that contain multiple a.i.s are presented in Table 104. Several other products are also formulated with petroleum distillates and other solvents, but did not indicate the concentration in the formulation as needed to estimate potential exposure in aquatic environments. Other ingredients in the formulation were frequently not specified.

Table 104. Examples of pesticide product ingredients.

EPA Product Registration Number	Active Ingredients %	Other Ingredients %
62719-260, 228-552	2,4-D 34.4, triclopyr BEE 16.5	49.1
228-317	MCPA 56.14, triclopyr BEE 5.00, dicamba 3.60	35.26
2217-920	triclopyr BEE 8.4, sulfentrazone 0.73, 2,4-D 31.82, dicamba 2.43	56.62
62719-477	triclopyr BEE 25, fluroxypyr 8.6	66.4
352-505	diuron 40, bromacil 40	20
352-618	diuron 46.8, hexazinone 13.2	40
228-386	diuron 4, bromacil 2	94
228-654	diuron 62.22, imazapyr 7.78	30
228-678	diuron 6, thidiazuron 12	82
352-660	linuron 20.3, diuron 20	59.7
264-949	captan 46, PCNB 15, carboxin 10	29
264-998	captan 50, trifloxystrobin 2, thiophanate-methyl 13.6, metalaxyl 0.8	33.6
400-561	captan 25, carboxin 12.5, metalaxyl 3.75	58.75
4-122	captan 12, malathion 0.246, carbaryl 0.30	81.7
400-568	captan 20, carboxin 20, imidacloprid 4	56
66330-48	captan 4.8, fenhexamid 14.3	30.9
100-1347	propiconazole 2.9, chlorothalonil 38.5	58.6
100-1315	chlorothalonil 45.0, azoxystrobin 3.0	52
100-1231	chlorothalonil 29.9, propiconazole 4.7, fludioxonil 1.2	64.2
100-800	chlorothalonil 72, mefenoxam 4.5	23.5

Nonylphenol (NP) and nonylphenol polyethoxylates are “other ingredients” that may be part of a pesticide product formulation and are common adjuvant ingredients added during pesticide applications. NP and nonylphenol polyethoxylates are also ingredients in detergents, cosmetics, and other industrial products and are a common wastewater contaminant from industrial and municipal sources. NP has been linked to endocrine disrupting effects in aquatic systems (Koplin et al., 2002). A national survey of streams found that NP was among the most ubiquitous organic wastewater contaminants in the U.S., detected in more than 50% of the samples tested. The median concentration of NP in streams surveyed was 0.8 μg/L and the maximum concentration detected was 40.0 μg/L (Table 105). Related compounds were also detected at a relatively high frequency (Kolpin et al., 2002).

Table 105. 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 six a.i.s and/or are added prior to application. Inert ingredients are often not specified on product labels. Additionally, NP and NP-ethoxylates represent a very small portion of the more than 4,000 inert ingredients that EPA permits for use in pesticide formulations (Koplin, et al., 2002). Many of these inerts are known to be hazardous in their own right (*e.g.*, xylene is a neurotoxin and coal tar is a known carcinogen). Several permitted inerts are also registered a.i.s (*e.g.*, copper, zinc, chloropictrin, chlorothalonil). Inerts can be more than 50% of the mass of pesticide products, and millions of pounds of these products are applied to the landscape each year (Koplin, et al., 2002). This equates to large contaminant loads of inerts that may adversely affect salmon or their habitat. Uncertainty regarding exposure to these ingredients will be qualitatively incorporated into our analysis.

Tank Mixtures

Several pesticide labels authorize the co-application of other pesticide products and other materials in tank mixes, thereby increasing the likelihood of exposure to multiple chemical stressors (Table 106). In some cases specific tank mixtures with other pesticide products or adjuvants are recommended. In all cases, tank mixtures are authorized unless specifically prohibited on the product label. These ingredients and the other inert ingredients in these products are considered part of the action because they are authorized by EPA's approval of the FIFRA label. Exposure to, and risk associated with, potential ingredients in tank mixtures were not addressed in EPA's BEs and remain a significant source of uncertainty.

Table 106. Examples of label recommended tank mixtures.

Pesticide Products containing	Tank mixture recommendation
2,4-D	Mix with other herbicide products, including Accord® (glyphosate) to increase weed control. Mix with surfactants (*e.g.* LI700) and drift retardants (*e.g.* Compadre ®) to increase efficacy and reduce drift.
Triclopyr BEE	Mix with other herbicides, including 2,4-D products, and liquid fertilizers. Mix product with agricultural surfactants and drift control agents.
Diuron	Mix with other suitable herbicides (*e.g.* OUST XP®, 2,4-D, hexazinone, Sinbar®, glyphosate, trifluralin, ammonium sulfate, DSMA, MSMA, bromoxanil, Arsenal®, and others) and adjuvants (*e.g.* non-ionic surfactants and crop oil concentrates).
Linuron	Mix with other herbicides (*e.g.* metribuzin DF®, alachlor, metolachlor, pendamethalin, paraquat, glyphosate), surfactants, and fertilizers.
Captan	Combine with other fungicides or insecticides at recommended rates.
Chlorothalonil	Compatible with many commonly used insecticides, fungicides, and spray adjuvants.

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 & Grube, 1999). Typically 10 to 20 new a.i.s are registered each year (Aspelin & 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, Donaldson, & Grube, 2004). Pesticide contamination in the nation's freshwater habitats is ubiquitous and pesticides usually occur in the environment as mixtures (R.J. Gilliom, et al., 2006). "More than 90% of the time, water from streams with agricultural, urban, or mixed-landuse watersheds had detections of two or more pesticides or degradates, and about 20% of the time they had detections of 10 or more," (R.J. Gilliom, et al., 2006). The likelihood of exposure to multiple pesticides throughout a listed salmonids' lifetime is great, considering their migration routes and habitats occupied for spawning and rearing. In a three-year monitoring study conducted by the Washington DOE, pesticide mixtures were found to be common in both urban and agricultural watersheds (Burke, Anderson, & Dugger, 2006). An average of three pesticides was found in each sample collected from urban sampling sites, with as many as nine pesticides found in a single sample. Agricultural sites averaged three to five pesticides per sample, with as many as 14 pesticides being detected in a single sample (Burke, et al., 2006). Mixtures of chemicals that share a common mode or mechanism of action are of particular concern to NMFS.

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

Exposure Conclusions

Pacific salmon and steelhead use a wide range of freshwater, estuarine, and marine habitats and many migrate hundreds of miles to complete their life cycle. Many of the a.i.s and degradates addressed in this Opinion, especially 2,4-D, triclopyr, diuron and its degradates, are detected frequently in freshwater habitats within the four western states where listed Pacific salmonids are distributed. Because the action of registering the six a.i.s for the next 15 years authorizes a number of the same uses, they will continue to be present in the action area. Additionally, all of the a.i.s are used when listed species are present in freshwater habitats. Therefore, we expect some individuals within all the listed Pacific salmon and steelhead ESUs/DPSs will be exposed to these chemicals and other stressors of the action. Given variable use of these pesticides across the landscape, and variable temporal and spatial distributions of listed salmonids, we expect exposure is also highly variable among individuals and populations of listed salmon. However, defining exposure and distributions of exposure among differing life stages of each independent population is complicated by several factors. Paramount among these is the uncertainty associated with the use of pesticide products containing these a.i.s. More specifically:

- EPA-authorized labels contain language that frequently does not provide clear distinctions on product use (*e.g.*, many labels do not specify the maximum number of applications, application interval, or maximum annual application rate);

- Product labels authorize the application of chemical mixtures that are not specified or not clearly defined (*e.g.*, the ingredients of pesticide formulations are not fully disclosed, labels recommend tank mixture applications with other pesticides and adjuvants, and tank mixtures with other pesticides are permitted unless specifically stated otherwise);

- Defining actual use of these products is highly uncertain. Historical use information is limited and may not reflect future use.

A major limitation of these assessments is that the majority of monitoring data used was not designed to determine exposure to listed salmonids. Studies conducted by Washington State Departments of Ecology and Agriculture were an exception, but those studies were not designed to evaluate peak exposure or exposure distributions in listed salmon. Therefore, caution should be exercised in using these data for that purpose especially when conducting probabilistic assessments.

Additionally, the assessments lack uncertainty analyses of the monitoring and toxicity data used, which limit the confidence in the given estimates (Warren-Hicks & Moore, 1998). Given the complexity and scale of this action, we are unable to accurately define exposure distributions for the chemical stressors. We assume the highest probability of exposure occurs in freshwater, and nearshore estuarine/marine environments in close proximity to areas where pesticide products containing 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil are applied. We considered several sources of information to define the range of potential exposure to action stressors. These sources are summarized in (Table 107). Ranges of concentrations for the monitoring data, EECs generated by EPA in the salmonid BEs and California Red-legged Frog (RLF) BEs, and NMFS generated spray drift estimates for characterizing initial concentrations in floodplain habitats are given (Table 107). Typically, the estimates for the floodplain habitat are higher than or near the high end of the range of EECs generated by EPA's PRZM-EXAMS modeling. Estimates for the pertinent degradates were generally not evaluated. The highest concentrations detected in surface waters were those associated with applications directly to aquatic habitats and from targeted monitoring studies that sampled surface water immediately adjacent to pesticide application sites.

Table 107. Chemical Concentration Ranges in Monitoring Data and Modeling.

Chemical	Monitoring Data		EPA Estimates		NMFS Estimates
Parent compounds	Range[1] µg/L (Database)	Max Conc. µg/L (Targeted)	Salmonid BE Conc. Range µg/L	RLF BE Conc. Range µg/L	Spray Drift µg/L
2,4-D	0.11-7.66	67.1	3.7 -1431	0.08 – 4,000[4]	13 - 1912
Triclopyr BEE	NE	350[2]	20 – 5,872	21 - 1499	184 - 3824
Diuron	0.002-160	2849[3]	3 - 437	5 - 4911	221 - 1147
Linuron	0.003-1.6	NE	31.3	2.6 - 337	552
Captan	0.6-0.6	NE	1.1 – 43.4	0.001- 29	184
Chlorothalonil	0.0027-0.36	48.1	6 - 363	3 - 274	754 - 2078
Degradates					
2,4-D					
2,4-DCP	ND	NE	NE	NE	NE
1,2,4-benzenetriol; 2,4-DCA; 4-chlorophenol;2-chlorophenol; chlorophenoxyacetic acid; chlorohydroquinone	NE	NE	NE	NE	NE
triclopyr BEE					
Triclopyr acid	0.0028-14.5	140[2]	NE	NE	NE
TCP; TMP; oxamic acid	NE	NE	NE	NE	NE
diuron					
3,4-DCA	0.01-0.7	NE	NE	NE	NE
CPMU	0.0037-0.1703	NE			NE
DCPMU; MCPDMU DCMU; DCPU; TCAP;	NE	NE	NE	NE	NE
linuron					
3,4-DCA	0.01-0.7	NE	NE	NE	NE
DCPMU; DML; DCPU; 3-(3-chloro-4-hydroxyphenyl)-1-methoxy-1-methylurea; 3,4-ichlorophenylurea; 3-(3,4-dichlorophenyl)-1-methylurea;	NE	NE	NE	NE	NE

Chemical	Monitoring Data		EPA Estimates		NMFS Estimates
Parent compounds	Range[1] µg/L (Database)	Max Conc. µg/L (Targeted)	Salmonid BE Conc. Range µg/L	RLF BE Conc. Range µg/L	Spray Drift µg/L
norlinuron; desmethyl linuron; desmethoxy linuron					
captan					
TCMT; THPI; THCY; THPAm; Thiophosgene; inorganic sulfur;	NE	NE	NE	NE	NE
chlorothalonil					
SDS-19221; SDS-3701; SDS-46851; SDS-47523/ SDS-47524; SDS-47425; SDS-67042; SDS-66432; SDS-66382; SDS-13353	NE	NE	NE	NE	NE

NE – Not estimated
1- Minimum and maximum based on detected values in NWQA, CDPR, and EIM databases
2- Direct overspray of forest stream
3- Surface water runoff
4- Target concentration for control of aquatic weeds

Inherent in the modeling estimates is the assumption that the pesticide is applied in a location next to or draining into salmon-bearing waters. Monitoring data may reflect pesticide applications proximate to the waterbody (i.e. values derived from targeted monitoring), or resulting from more distant uses in the watershed or airshed. We assume that the exposure estimates provided by EPA in the BEs and additional modeling and monitoring information provided above represent realistic exposure levels for some individuals of the listed species. Further, we assume the distribution within the range of exposures is a function of pesticide use and the duration of time listed salmonids spend in these habitats. All listed Pacific salmon and steelhead occupy habitats that could contain high concentrations of these pesticides at one or more life stages. However, the time spent in these habitats varies among species. Adult salmon and steelhead spend weeks to several months in freshwater habitats during their migration and spawning activities. Immediately after emerging from the gravel substrate and transitioning from alevins to fry, salmonids move to habitats where they can swim freely and forage. At this point in their development most salmon occupy freshwater habitats. Chum salmon are an exception. They immediately migrate downstream following emergence to nearshore environments in estuaries near the mouth of their natal stream. Upon arrival in the estuary the chum salmon fry inhabit nearshore areas at a preferred depth of 1.5-5 m. In Puget Sound, WA, surveys indicate chum salmon fry are distributed extremely close to the shoreline and concentrated in the top 15 cm of water. Therefore, chum salmon fry are less likely to be exposed to high concentrations of pesticides than other salmonids given they quickly migrate to larger estuaries with greater dilution potential. They may reside immediately next to the shore in estuaries for as little as one or two weeks before moving offshore or into deeper-water habitats within the nearshore environment. Sockeye salmon fry most frequently rear in lakes, where they distribute in the littoral zones. They initially occupy shoreline habitats of only a few centimeters in depth before moving further off-shore and taking on a more pelagic existence. Coho salmon, Chinook salmon, and steelhead fry typically select the stream’s nearshore zone and floodplain habitats associated with their natal rivers and streams. These species are most likely to experience higher pesticide exposures given their use of shallow freshwater habitats for juvenile rearing. Coho salmon and steelhead have a greater preference for the shallow habitats and rear in freshwater for more than a year. Coho salmon fry rear in lower gradient river channels and often rear in pools of the river channels. They may also rear in ponds and lakes. Steelhead juveniles use riffles and faster flowing waters more than coho salmon, and are often found in steeper gradient channels. Coho salmon juveniles may make extensive migrations in fall to overwinter in floodplain habitats such as ponds, sloughs, oxbows, flooded wetlands, and other seasonally connected and inundated habitat. Spring foraging in these habitats often provides substantial growth before smoltification and juveniles in these habitats can grow significantly larger than mainstem overwintering coho salmon juveniles. Steelhead do not use channels with organic bottom substrate for overwintering and often seek refuge under larger stones in the flowing river as protection from strong winter flows. Chinook salmon commonly spawn and rear in larger rivers and tributaries than the other *Oncorhynchus* species. Juvenile Chinook salmon in California, with the exception of the Central Valley spring-run Chinook salmon, spend less than six months in freshwater and out-migrate as fry or sub-yearlings (ocean type). Juveniles in the Columbia River basin and in Puget Sound may out-migrate as fry/sub-yearlings (ocean type) or as yearlings (stream type), depending on race and the river basin of origin. Fry of the ocean type life history typically rear in estuarine shallow waters, tidal wetlands, and sloughs for days to weeks before entering the ocean while yearling or older juveniles spend less time and use deeper water in the estuary.

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

Response Analysis

In this section, we identify and evaluate toxicity information for the stressors of the action and organize the information under assessment endpoints relating to both individual and habitat responses (Figure 64). The assessment endpoints are biological attributes that, when adversely affected, may reduce fitness of individual salmonids or degrade PCEs (*e.g.*, prey abundance, water quality, and suitability of habitat).

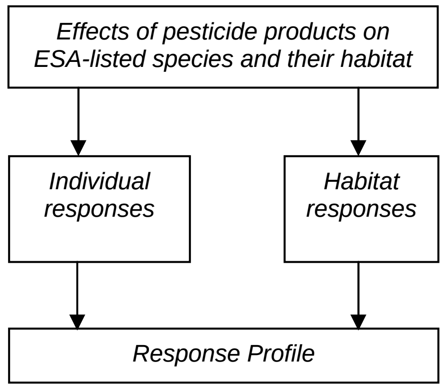


Figure 64. Response analysis.

We begin the response analysis by describing the toxic mode and mechanism of action of the a.i.s, then summarize information associated with relevant assessment endpoints. Toxicity information is derived primarily from action agency documents, but we relied on other sources as well. We used EPA's salmonid BEs, REDS, IREDs, California red-legged frog BEs, and EFED Science Chapters for the six a.i.s. As the California red-legged frog BEs were generally the most recent and comprehensive compilation of toxicity data, we relied heavily on them for this Opinion. In order to provide readers quicker reference to the data complied by EPA, we have included the red-legged frog "Ecological Effects" appendix for each a,i. in this document as *Appendix 10*. We also used some toxicity data developed specifically for this Opinion by our Northwest Fisheries Science Center (NWFSC, included as *Appendix 8*), data from open literature, and data provided by applicants. In some cases (noted in relevant text) we went back to the original source documents referenced by EPA to confirm values. The information provided by EPA addressed aspects of survival, growth, and reproduction of aquatic species (freshwater and saltwater), and provided some discussion on other information found in the open literature, such as results from some field experiments and experiments that evaluated sublethal effects. Under Section 7 of the ESA and its implementing regulations, NMFS evaluates all direct and indirect effects of an action. We therefore evaluate all aspects of an action that may reduce fitness of individuals or appreciably reduce PCEs of designated critical habitat. The evaluation includes information that EPA provided on survival, growth, or reproduction, but also encompasses a broader range of endpoints including behaviors, endocrine disruption, and other physiological alterations such as impairment of olfactory-mediated behaviors.

The information we evaluated is derived from published, peer-reviewed scientific journals, government agency reports (federal, state and local), theses and dissertations, books, information and data provided by the registrants identified as applicants, and independent reports. NMFS scientists evaluate the quality and applicability of all documents used, although unlike EPA, we do not develop formal data evaluation reports (DERs) for the sources we review. Typically, the most relevant study results are those which directly measure effects to an identified assessment endpoint and are derived from experiments with salmonids. Studies with listed Pacific salmonids or hatchery surrogates are preferable, but we also include data from closely-related species in our “salmonid” endpoint summaries. We present data from other fish species as well. Often, there is not a complete suite of information relating to effects on fish, especially for some of the sublethal endpoints. Where appropriate, we include information from studies on other taxa, recognizing and noting where there may be significant interspecies extrapolation. Likewise, we consider information from studies on chemicals that are structurally similar to the a.i.s addressed in this Opinion.

EPA’s ecological risk assessments and BEs primarily summarize acute and chronic toxicity data from “standardized toxicity tests” submitted by pesticide registrants during the registration process, or tests from government laboratories available in EPA databases, or from published, peer-reviewed scientific publications (books and journals). The assessment endpoints from these tests for an individual organism generally include only survival (death), reproduction, and growth measured in laboratory dose-response experiments conducted on a single a.i. Survival is typically measured in both acute (48-96 h) and chronic (21-60 d) tests. Fish reproduction and growth are generally measured using chronic tests (21-60 d). Population-level endpoints and analyses were generally absent in the BEs, other than a few measurements of fish and aquatic invertebrate reproduction and adverse effects to organisms were not translated into consequences to populations. For Biological Opinions, NMFS evaluates the range of effects on individual salmonids to determine potential population-level consequences.

Survival Endpoints

Survival of individual fish is typically measured by incidences of death following 96 h exposures to the a.i. (acute test). Survival data may also include incidences of death following longer exposures (21 or more days, known as chronic tests) which are intended to evaluate effects on growth and reproductive endpoints. Tests are conducted on a subset of freshwater and marine fish species reared in laboratories under controlled conditions (temperature, pH, light, salinity, dissolved oxygen, *etc.,)* (EPA, 2004d). Lethality of the pesticide (a technical product or formulated product) is usually reported as the median lethal concentration (LC_{50}), the statistically-derived concentration sufficient to kill 50% of the test population. For aquatic invertebrates it may be reported as a median effective concentration (EC_{50}), because death of these organisms may be too difficult to confirm and immobilization is considered a terminal endpoint. An LC_{50} is derived from the number of surviving individuals at each concentration tested following a 96 h exposure and is typically estimated by probit or logit analysis and recently by statistical curve fitting techniques. In FIFRA guideline tests, LC_{50}s are typically calculated by probit analysis. If the data are not sufficient for a probit analysis, than either a moving average or binomial is used, resulting in no slope being reported. To maximize the utility of a given LC_{50} study, the slope of the dose-response curve, the 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.*, should be reported. The slope of the observed dose-response relationship is particularly useful in estimating the magnitude of death at concentrations below or above an estimated LC_{50}. The variability of an LC_{50} often given by a 95% confidence interval (95% CI) or statement of standard deviation or standard error. These variability measures provide the degree of confidence associated with a given LC_{50} estimate, and the smaller the range of uncertainty the higher the confidence in the estimate. Survival experiments are most useful when conducted with the most sensitive life stage of the listed species or a representative surrogate. In the case of ESA-listed Pacific salmonids, several surrogates are available for toxicity testing, including hatchery reared coho salmon, Chinook salmon, steelhead, and chum salmon, as well as rainbow trout[4]. Rainbow trout data are often available, as they are a preferred species in toxicological testing.

Toxicity data available for this consultation included some for salmonids. Unfortunately, slopes, estimates of variability for an LC_{50}, and experimental concentrations frequently were not reported. In our review of the salmonid BEs, we did not locate any reported slopes of dose-response curves, although some of this information was presented in some of the corresponding Science Chapters and the CRLF BEs. Death of individuals affects abundance, and may affect distribution of populations.

Growth Endpoints

Growth of individual organisms is an assessment endpoint derived from standard chronic fish and invertebrate toxicity tests summarized in the BEs. It is difficult to translate the significance of reduced growth derived from a guideline study on fish growth in aquatic ecosystems. The health of the fish, availability and abundance of prey items, and the ability of the fish to adequately feed are not assessed in standard chronic fish tests. These are important factors affecting the survival of wild fish. Typically, size or weight of fish is measured several times during an experiment. The test fish are usually fed twice daily, *ad libitum*, (i.e., an over abundance of food is available to the fish). Therefore, any reductions in size are a result of fish being affected to such an extent that they are not feeding or are unable to metabolize food even when presented with an abundance of food. Subtle changes in feeding behaviors or availability of food would not be detected from these types of experiments. If growth is affected in these experiments, it is highly probable that growth of fish in natural aquatic systems would be severely affected. Reductions in juvenile growth may affect survival at sea and susceptibility to predation.

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

Removal of the smaller juveniles from the population would affect abundance, and possibly distribution.

Reproduction Endpoints

Reproduction, at the scale of an individual, can be measured by the number of offspring per female (fecundity), and at the scale of a population by measuring the number of offspring per females in a population over multiple generations. The BEs summarized reproductive endpoints at the individual scale from chronic freshwater fish experiments where hatchability and larval-juvenile survival is measured. In biological opinions, NMFS also considers many other assessment measures of reproduction, including egg size, spawning success, sperm and egg viability, gonadal development, reproductive behaviors, and hormone levels, as these endpoints can have considerable effect on wild populations. Many of these endpoints are not measured in standardized toxicity assays used in pesticide registration, thus we often use data from other sources to evaluate these endpoints. Reproductive rate, along with abundance and distribution is a key determinant of species viability.

In order to have more data on sensitive lifestages such as the egg and embryo NMFS/OPR requested the NWFSC conduct toxicity tests on these lifestages. Zebrafish (*Danio rerio*) are commonly used in these types of tests because their early development is well documented and features are easily observable. Although not closely related to salmonids, the zebrafish was selected as the test organism due to the existing body of data available for this species. The testing report is included in the Opinion as *Appendix 8*. Fertilized eggs were exposed to the a.i.s for 5 days at concentrations ranging from 1 - 10,000 μg/L. Percent survival was noted. Surviving fish were measured and scanned for developmental abnormalities. Results are reported in the discussions of specific a.i.s. Developmental abnormalities and/or smaller size can reduce the ability of the individual to forage, avoid predation, and in some cases, to reproduce normally. Survival of the embryos, size, and developmental abornormalities may affect abundance, distribution, or reproduction.

Sublethal Endpoints

Sometimes qualitative observations of sublethal effects are summarized from 96 h lethality dose-response bioassays in EPA's risk assessments. These observations generally were limited in the BEs, and when noted, pertained to impaired swimming behaviors such as disorientation, and resting on the bottom. None of these behaviors were rigorously measured and therefore are of limited value in assessing the effects of these pesticides on Pacific salmonids. We do, however, note a few of the observations when they pertained to a relevant assessment endpoint, such as impaired swimming. Some BEs presented toxicity information on degradates, metabolites, and formulations. Toxicity information on other or "inert" ingredients found in pesticide formulations was usually not presented.

Sublethal endpoints encompass a variety of physiological and biochemical measurements. NMFS is concerned about effects which reduce the ability of the fish to successfully complete its lifecycle and produce a subsequent generation (*i.e.*, a reduction in fitness). Types of sublethal effects expected and information regarding vary widely from chemical to chemical. Sometimes sublethal effects are not investigated for fish or aquatic invertebrates, but there may be information available regarding these effects for mammals. When appropriate, we extrapolate this information to salmon. Some sublethal endpoints may affect abundance or distribution, and others may affect reproduction.

Multi-species (Micro- and Mesocosm) Studies

Results from multiple species tests, called microcosm and mesocosm studies, were also discussed in the BEs to a varying degree. These types of experiments are likely closer approximations of potential ecosystem-level responses such as interactions among species (predator-prey dynamics), recovery of species, and indirect effects of pesticides on fish. However, the interpretation of results is complicated by how well the results represent natural aquatic ecosystems, and how well the studies apply to salmonid-specific assessment endpoints and risk hypotheses. These studies typically measure individual responses of aquatic organisms to contaminants in the presence of other species. Some studies are applicable to questions of trophic effects and invertebrate recovery, as well as providing pesticide fate information. The most useful mesocosm study results for this Opinion are those that directly pertain to identified assessment endpoints and risk hypotheses. We discuss study results in the context of salmonid prey responses, emphasizing the capacity of prey taxa to rebound following death of individuals as well as shifts in community structure. For herbicides, we also consider modifications in the plant communities in and around the waterbody. One of the notable limitations of most micro- and mesocosm studies is they do not typically represent real world aquatic ecosystems which are degraded from various stressors.

Results from aquatic field studies were generally not discussed in great detail within the BEs. We discuss field studies that evaluated assessment endpoints, particularly those which address salmonid prey responses in systems with ESA-listed salmonids.

Potential effects of herbicides on salmonids and their critical habitats

In previous Opinions, we have addressed organo-phosphorus (OP) and carbamate insecticides. Although used to control insects, these pesticides have a mode of action (cholinesterase inhibition) expected to directly affect salmon and other non-target organisms, such as aquatic and terrestrial invertebrates that provide a forage base for the salmon. The pesticides addressed in this Opinion are herbicides (4 a.i.s) and fungicides (2 a.i.s). Given their intended use, we assumed they may have less of a direct effect on salmon, but were concerned about how their use might modify the ecosystems on which salmon depend. Thus, NMFS surveyed available literature regarding herbicide effects in the environment. A broad range of herbicides were considered, including ones not addressed in this Opinion. A summary of this survey, and conceptual models based on information gleaned from this survey are presented below.

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

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

Fish can consume a very high proportion of the invertebrate secondary production in aquatic habitats (Huryn, 1996, 1998). Juvenile salmonids, often at high densities and growing quickly, are predators and consume a wide range of invertebrates, including those from all functional feeding groups. Changes in the production of any of these groups could change prey availability for these fish; for example, a reduction in periphyton production on rocks in a stream could reduce invertebrate grazer production. Likewise, a change in the quantity or quality of terrestrial leaf litter falling from a riparian buffer could alter the production of invertebrate shredders downstream. In addition to being the ultimate source of food for much of the invertebrate community, plants also provide habitat for invertebrates and fish, including but not limited to substrate for them to shelter on and under (*e.g.,* macrophytes, root wads). Plants and other photosynthetic organisms within and adjacent to salmonid freshwater and estuarine habitats are therefore essential components of productive salmonid habitats. Actions that affect the diversity, biomass and/or the production of primary producers in and around salmonid habitats may limit or alter secondary production within those systems (Figure 65).

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

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

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

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

Effects of herbicides on non-target aquatic communities

Potential effects of herbicides on aquatic and riparian communities are illustrated in Figure 65 and in Figure 66. The range of effects includes direct effects (primarily negative) on photosynthetic organisms and water quality parameters as well as indirect effects (positive and negative) on multiple trophic levels and water quality. Generally, if an herbicide exposure is great enough to reduce primary production within or adjacent to aquatic habitats, there may be effects on multiple higher trophic levels, including fish. There are a number of factors that determine the magnitude of the effects as well as the direction of effects, but it is often difficult to predict those patterns from one system to the next.

Numerous studies using standard toxicity tests have demonstrated that herbicides reduce the growth and biomass of non-targeted as well as targeted photosynthetic organisms. As expected, plants and photosynthetic organisms are more sensitive to herbicides than invertebrates and fish because of the herbicides' various mechanisms of action. For example, Brock and others (T. C. M. Brock et al., 2004) determined HC_5s (hazardous concentrations for 5% of the species) for two herbicides (metribuzin and metamitron) on a variety of taxa, and found not surprisingly that the algae and macrophytes were >100 to >1000x more sensitive than invertebrates and fish. Similarly, Van den Brink et al. (Van den Brink, Blake, Brock, & Maltby, 2006) found that herbicides varied in their toxicity, but the relative sensitivities (based on short-term toxicity tests) of the taxonomic groups included were as follows: algae≥macrophytes>invertebrates>vertebrates. For some herbicides, algae and macrophytes were similar in their sensitivities (e.g., for atrazine and diquat, (Van den Brink, et al., 2006)), but for others, such as 2,4-D (an auxin simulator), macrophytes were significantly more sensitive than all of the algae taxa included in the analyses (Van den Brink, et al., 2006). In their extensive review of herbicides, Brock et al. (T.C.M. Brock, Lahr, & Van den Brink, 2000) also concluded that auxin simulators like 2,4-D were generally more toxic than other photosynthesis inhibitors to macrophytes.

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

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

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

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

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

Brock et al. (T.C.M. Brock, et al., 2000) concluded in their review of herbicides that indirect effects of photosynthetic inhibitors on consumers and predators occur at concentrations around the EC_{50} for standard algae taxa; these impacts on consumers and predators are likely due to reduced availability of food resources and the effects may be delayed relative to the exposure event. Other effects on the ecosystem (*e.g.* blooms of insensitive algae) can occur at lower concentrations (e.g. 0.1 of the EC_{50} of standard algae), and these effects may also be delayed. When macrophytes are impacted, organisms using those macrophytes as habitat are immediately impacted. Some studies published after the Brock et al. (T.C.M. Brock, et al., 2000) review note indirect effects at surprisingly low concentrations, but generally papers published since their review corroborate their findings.

Challenges in scaling up effects and making predictions across salmonid habitats

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

Juvenile salmonids are generally opportunistic drift-feeders, and are therefore sensitive to factors that influence the general quantity and quality of invertebrate prey items. If for instance there were reductions in the production of invertebrate grazers or the inputs of invertebrate prey from riparian vegetation, salmonids may be forced to alter their foraging behavior (*e.g.,* take more risks, select less energy-rich prey) (as shown in Figure 65 and in Figure 66). Alternatively, if there were shifts rather than reductions in the abundances and composition of the prey community within riparian and aquatic habitats, indirect impacts on salmonids may be minimal if foraging behaviors were not altered. Whether or not production of prey decreases or shifts (or increases) after exposure to herbicides will depend in part on the composition of the community (structure and function) and the relative sensitivities of those taxa. Multiple experiments conducted in mesocosms have demonstrated that the particular composition of the community at the time of exposure influences the magnitude of the impact as well as the trajectory of the recovery (D. G. Jenkins & Buikema, 1998; Pesce, et al., 2006; Relyea, 2009; Rohr & Crumrine, 2005), and this would likely be the case as well in salmonid habitats.

Abiotic conditions may also affect how herbicides directly and indirectly affect salmonids and their habitats. For instance, herbicides can affect water quality parameters that may indirectly affect aquatic communities. Austin *et al.* (1991) suggest that increased algal production in oligotrophic systems after exposure to glyphosate may be due to the addition of phosphorous (in the glyphosate), and they suggest this could lead to eutrophication of salmonid habitats. Likewise, total phosphorous increased by eight-fold in earthen mesocosms treated with glyphosate (Perez et al., 2007). In forested watersheds in the southeastern United States, nitrogen concentrations were elevated in streams for two years after herbicides were applied (Neary, Bush, & Michael, 1993). This effect was likely due to the increased leaching from the terrestrial environment and/or reduction in uptake within the stream. Regardless of how nutrients become elevated (from the herbicide itself or from changes in biogeochemical cycles within the watershed) elevated nitrogen and phosphorous concentrations can stimulate periphyton growth in nutrient-limited systems and consequently affect higher trophic levels. Indirect effects from herbicides may also include an increase in stressful water temperatures due to reduced shading and long-term reductions in woody debris used for cover by salmonids from loss of riparian vegetation. If herbicides were used to reduce plant growth over a large area within a watershed, instream flow dynamics may be impacted enough to affect salmonids and their habitats (*e.g.*, (Likens, Bormann, Johnson, Fisher, & Pierce, 1970)). Finally, changes such as increased turbidity (due to reduced bank stability) or decrease dissolved oxygen could have impacts on primary producers as well as consumers within salmonid habitats (Figure 65 and Figure 66).

In addition to the uncertainties associated with variable and diverse communities and the range of sensitivities they have to various abiotic conditions, there are uncertainties about how herbicides may affect aquatic systems affected by other stressors. When experiments are used to examine multiple stressors, the results are often variable and again (like simpler experiments) often depend on the abiotic and biotic conditions at the time. In a series of experiments, Rohr *et al*. (Rohr et al., 2004) found few interactions among food availability, drying conditions and atrazine (at 4 concentrations) on a streamside salamander, but they did find that the lethality of atrazine varied by year and may be condition dependent. These types of experiments reveal that effects may be significant, even if hard to predict. Figure 65 and Figure 66 illustrate the direct and indirect effects stemming from herbicide exposure, but they do not attempt to capture the complex web of interactions that may arise when multiple stressors affect a system.

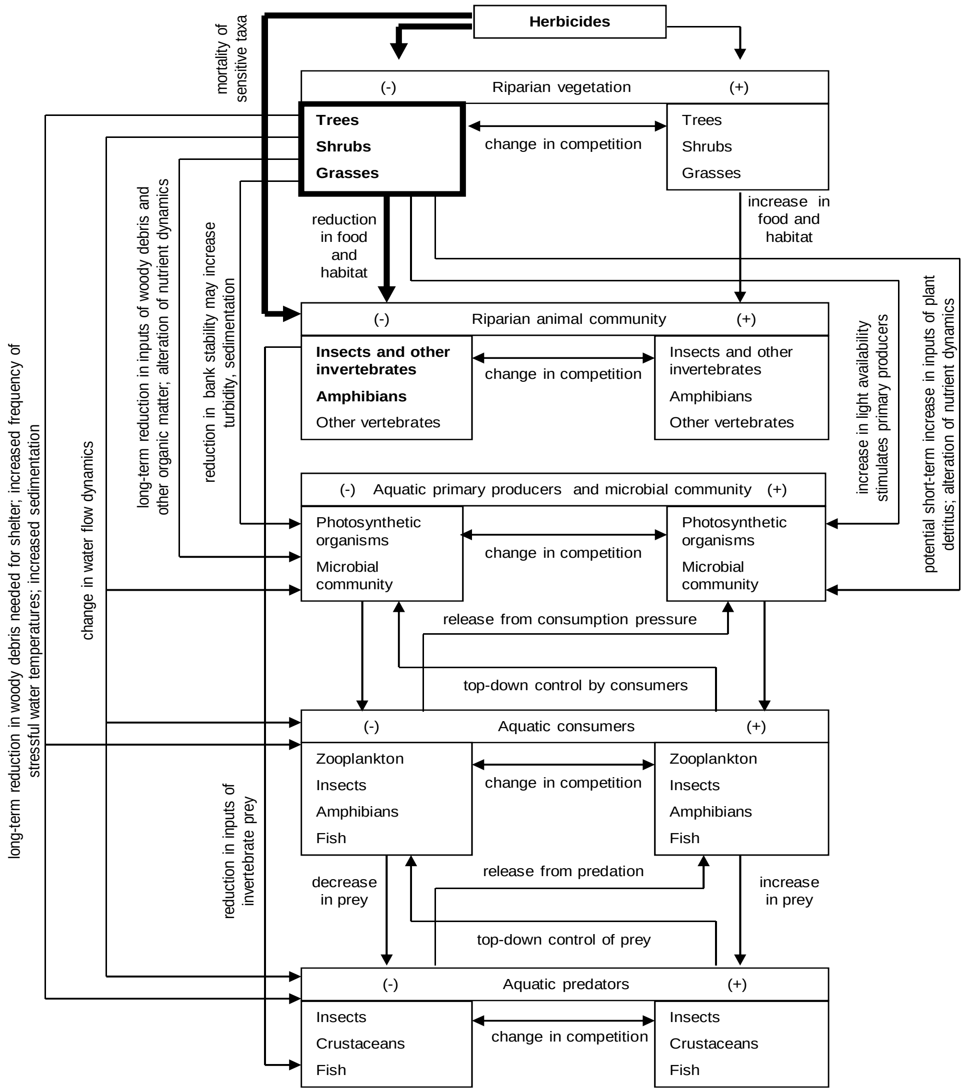


Figure 65. Part I of a conceptual model of potential effects of herbicides aquatic communities. This figure focuses on potential effects of herbicides applied to riparian areas adjacent to salmonid habitats. Bolded arrows and text note those effects that are most likely to occur based on the frequency that they are reported in the literature.

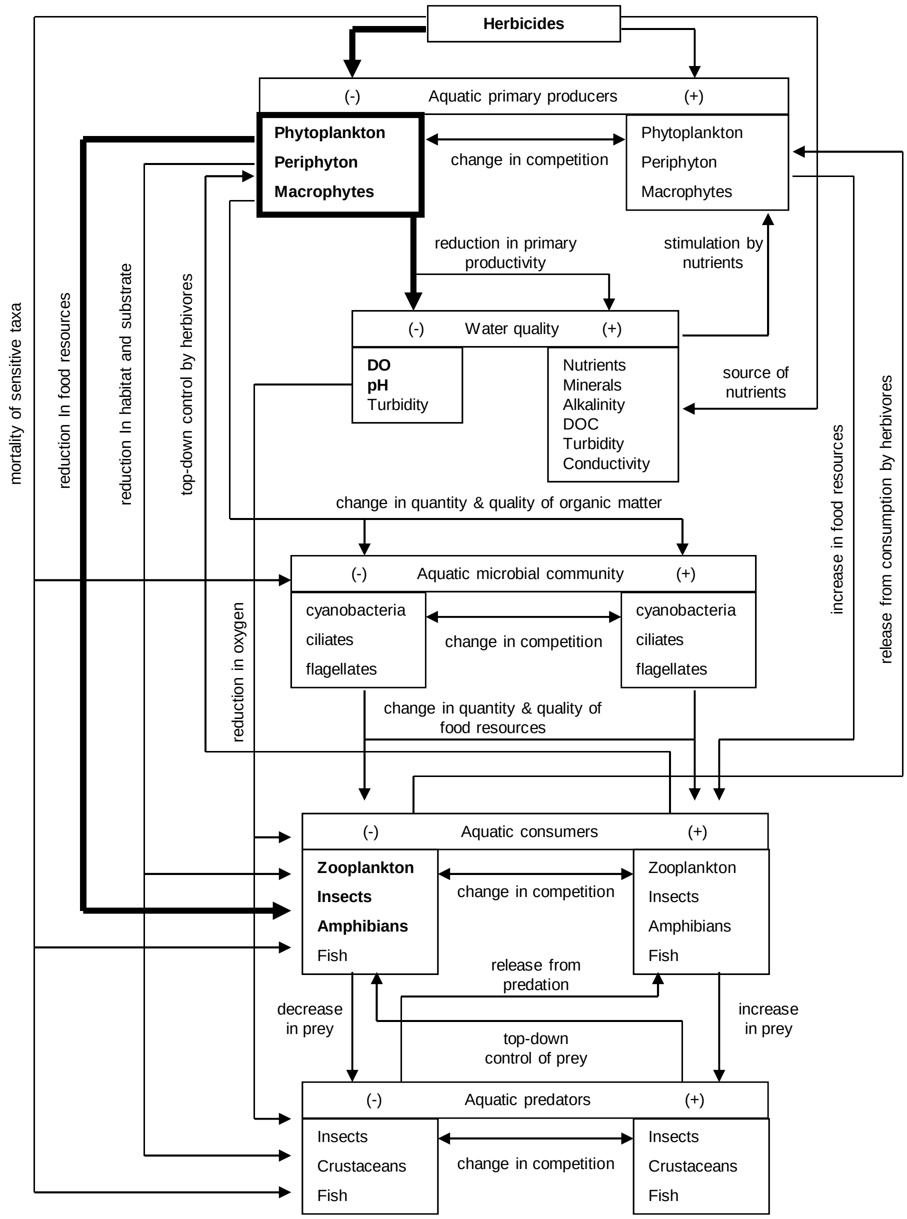


Figure 66. Part II of a conceptual model of potential effects of herbicides aquatic communities. This figure focuses on potential effects of herbicides that are applied to or otherwise reach salmonid habitats. Bolded arrows and text note those effects that are most likely to occur based on the frequency that they are reported in the literature.

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

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

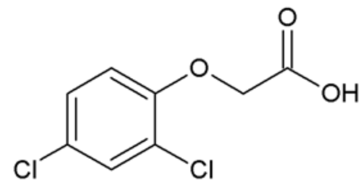
These uncertainties make it difficult to predict how herbicides will affect salmonids and their critical habitats, but they do not change NMFS' determination there may be an adverse impact.

2,4-D and Triclopyr (Synthetic Auxins)

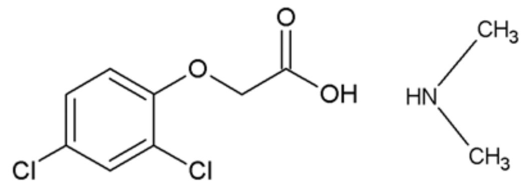
Mode of Action

2,4-D, a phenoxyacetic herbicide, is an auxin inhibitor (Cremlyn, 1991). Auxin (also known as indole-3-acetic acid, or IAA) is a plant hormone which regulates cell elongation. The phenoxyacetic herbicides produce the same reaction as IAA in the plants, but are not broken down by the plant and cause it to grow in an unregulated manner eventually resulting in death (Cremlyn, 1991). The phenoxyacetic herbicides are also known as phenoxyalkanoic acids.

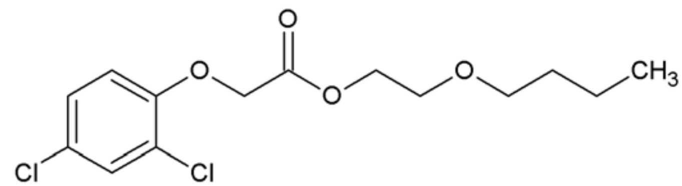
2, 4-D contains an acid group (-COOH), and is marketed in a variety of forms, including a sodium salt, various amines, and a number of esters. The acid structure and examples of an amine and ester are shown in Figure 67. The salt, amines, and ester forms have different environmental fate and toxicity properties. The salt and amines rapidly break down to the acid form, especially in water, and the toxicity of these forms is more similar to the acid. The esters break down more slowly, and their toxicity is greater (sometimes orders of magnitude) than the acid or salts. Greater toxicity of the esters of 2,4-D and similar compounds appears to be related to faster uptake by the organisms (Barron, Mayers, Murphy, & Nolan, 1990). Auxins are specific to plants, and toxic effects in fish appear to be a general narcosis effect (Barron, et al., 1990).



2,4 D acid



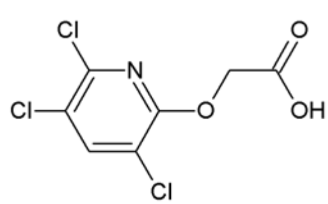
2,4 D dimethyl amine (DMA) salt



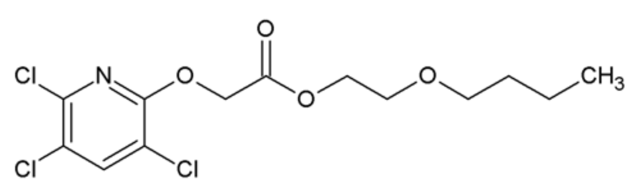
2,4 D butoxy ethyl ester (BEE)

Figure 67. 2,4-D acid, and example amine and ester forms.

Triclopyr is a pyridine herbicide, and is also an auxin inhibitor. In the basic form, it is an acid (-COOH), and like 2,4-D is marketed as both an amine (triclopyr TEA) and an ester (triclopyr BEE). However, it is available in less forms than 2,4-D, and by definition of the settlement agreement, we are only addressing the butoxy ethyl ester (BEE) form in this Opinion. The triclopyr acid and triclopyr BEE structures are shown in Figure 68. Similar to 2,4-D, the ester form is more persistent in the environment and more toxic than the amine form.



Triclopyr acid



Triclopyr butoxy ethyl ester (BEE)

Figure 68. Triclopyr acid and triclopyr BEE.

Evaluating Toxicity: Using acid versus ester data

We present environmental concentrations (in the *Exposure Section*) and toxicity data in terms of acid equivalents (a.e.s). Toxicity values for both 2,4-D and triclopyr are presented for the acid, amine, and ester forms if available. Given the short half-life of the triclopyr ester form, the registrant for triclopyr BEE has suggested evaluating the toxicity of the ester in the environment is not relevant, and that toxicity values for the acid should be used (Dow, 2010). In the triclopyr salmonid BE, EPA estimated acute EECs for the BEE form and used the toxicity data for the BEE form (EPA, 2004f). They did not evaluate chronic effects. Concentrations were expressed as mg or μg a.i./L, and not converted to a.e. In the more recent California red-legged frog assessment, EECs were estimated based on the fate properties of the acid form, and other than the chronic fish endpoint, toxicity values for the BEE were used (EPA, 2009c). Water concentrations were expressed in a.e.s.

Based on a pharmacokinetic study done in coho salmon, the BEE form of triclopyr was taken up by the fish extremely rapidly (Barron, et al., 1990). Essentially all uptake for the BEE occurred in the first 12 hours of exposure, and the majority occurred within the first 6 hours. The uptake rate for the acid form is much slower. This is consistent with what might be predicted based on the log K_{ow}s for the two forms. The fish quickly metabolized the ester to the acid form. Barron *et al* (1990) postulated the apparent difference in toxicity between the two forms of triclopyr was associated with accumulation of the a.i. (*i.e.* differences in effective dose). We concur with this assessment. Thus, for short-term (acute) exposures, we use the toxicity values from the BEE form, and for longer-term (chronic) exposures we use toxicity values for the acid (preferable) or amine (if acid value is not available) form.

Although we have not reviewed a pharmacokinetic study on 2,4-D esters, given the differences in log K_{ow}s for these compounds (IDE 3.81, BEE 4.35, and EHE 5.78 (EPA, 2009c) as compared to the log K_{ow} for the acid (2.81(EPA, 2009c)), we believe it is reasonable to assume the differences in toxicity between the 2,4-D amines and esters are also related to the uptake by organisms. Hughes (1973) tested a 78% butyl ester of 2,4-

D, with results reported in terms of a.i. Both striped bass larvae and fingerlings were tested, and author reports LC_0, LC_{50}, and LC_{100} for both age classes (Hughes, 1973). At 96 h those values were 100 μg/L, 150 μg/L, and 250 μg/L, respectively for the larvae, and 2,000 μg/L, 3000 μg/L, and 4,000 μg/L for the fingerlings. Authors also provide all three endpoints at 24 h, 48, h and 72 h. For both larvae and fingerlings, LC_{xx} values at 24 h and all other time periods were the same as at 96 h, confirming the rapid uptake. Thus, as with triclopyr BEE, for the ester forms we use the ester toxicity values to evaluate short-term (acute) exposures, and acid (preferable) or amine (if acid value is not available) form to evaluate longer-term (chronic) exposures.

Temperature and toxicity

We located no information indicating temperature specifically affects the toxicity of 2,4-D or triclopyr. However, we do note higher water temperatures can affect salmonids in two ways, regardless of specific chemical effects. Higher water temperatures will increase the metabolic rate for fish, thus increasing the rate at which they process the toxicant. Depending on the chemical, this may be either beneficial or detrimental. Higher than optimum water temperatures increases general physiological stress for salmonids, making them more susceptible to other stressors.

pH and toxicity

We located no information indicating pH specifically affects the toxicity of 2,4-D or triclopyr.

Toxicity of 2,4-D and Triclopyr (Assessment Endpoints)

Direct Effects to Salmonids

We evaluated potential effects to salmonids based on toxicity information included in the salmonid BEs (EPA, 2004a, 2004f), the more recent California red-legged frog assessments (EPA, 2009a, 2009c), and the REDs (EPA, 1998b, 2005). These data are presented in the tables, separated by acid, amine, and ester forms. We also searched open literature for endpoints not reported in EPA documentation, and data available since their publication. These are discussed qualitatively.

Survival

Survival is evaluated primarily based on guideline tests used to determine the LC_{50} of a population of test fish (Table 108).

Table 108. Acute toxicity data for fish

Species		96-h LC_{50} Median Concentration (mg a.e./L)	Range (mg a.e./L)	n	Source
2,4-D[1]					
Salmonids					
Rainbow trout	*Oncorhynchus mykiss*	Acid 358 Amines 1,203 Esters 1.05	NA 162-2,244 0.45-14.5	1 2 7	(EPA, 2009a) (Appendix F)
Other fish					
Bluegill	*Lepomis macrochirus*	Acid 292 Amines 234 Esters 0.69	263-320 101-1,722 0.26-11.9	2	(EPA, 2009a) (Appendix F)
Fathead minnow	*Pimepheles promelas*				
Triclopyr[2]					
Rainbow trout	*Oncorhynchus mykiss*	Acid 117 TEA 79.2 BEE 0.47	NA	1	(EPA, 2009c) (Appendix A)
Bluegill sunfish	*Lepomis macrochirus*	Acid 148 TEA 155.4 BEE 0.26	NA	1	(EPA, 2009c) (Appendix A)

[1]Median of values in Appendix if more than one available, [2]Most sensitive values in Appendix.

Reproduction and Growth

Data presented in Table 109 are based on guideline study information included in the EPA BEs. For triclopyr, we also located an open literature study on the effects of triclopyr TEA to fathead minnow (Mayes, Dill, Bodner, & Mendoza, 1984). It describes both acute tests, and a 28-d embryo-larval test. For the acute tests, they note most toxicity occurs within the first 24 h. Chronic exposure concentrations were 13 to 112 mg ai/L, selected based on acute toxicity values. Authors noted in this test that they saw little evidence of effects on hatchability, development, and growth. In 5-day static exposure studies on triclopyr BEE, zebrafish (*Danio rerio*) exposed to 1mg/L exhibited edema. There was 100% mortality of embryos exposed to 10 mg/L (NMFS, 2011 – included as

Appendix 8). 2,4-D acid produced no significant differences in abonormalities or survival of the zebarafish embryos at the concentrations tested (maximum 10 mg/L). As the BEE breaks down quickly in the environment, we have used data for the TEA to evaluate chronic effects. The NOAEC and LOAEC for the zebrafish are higher than the existing NOAEC and LOAEC for the rainbow trout.

Table 109. Chronic toxicity data for fish.

Species		NOAEC LOAEC (mg a.e/L)	Endpoint Affected (Exposure duration)	Source
2,4-D[1]				
Fathead minnow	*Pimephales promelas*	Acid 63.4 Amines 17 (n=2) Esters 0.0792 Acid 102 Amines 45.1 (n=2) Esters 0.1452	Reduction in growth Reduction in larval survival (Not given)	(EPA, 2009a) (Appendix F)
Zebrafish	*Danio rerio*	Acid 10 Acid ND	Reduced length Developmental abonormalities (5 d)	(NMFS, 2011)
Triclopyr[2]				
Rainbow trout	*Oncorhyncus mykiss*	Acid ND TEA ND BEE 0.019 Acid ND TEA ND BEE 0.034	Reduction in growth (Not given)	(EPA, 2009c) (Appendix A)
Fathead minnow	*Pimephales promelas*	Acid ND TEA >32.2 BEE ND Acid ND TEA <50.2 BEE ND	Reduction in growth (Not given)	(Mayes, et al., 1984)
Zebrafish	*Danio rerio*	BEE 0.1 BEE 1	Edema at 1mg/L 100% mortality at 10 mg/L (5 d)	(NMFS, 2011)

[1]Median of values in Appendix if more than one available, [2]Most sensitive values in Appendix.

Swimming

During a 12 d exposure to 400 mg/L 2,4-D acid, (Gomez et al., 1998) noted irregular swimming behavior, hemorrhaging, and pale gills. Hemorrhagic fluid appeared within 24 h of exposure. We located no other reports specifically of impaired swimming behavior.

We located no data regarding effects of triclopyr on swimming.

Olfaction

A study tested serine-evoked electro-olfactogram (EOG) responses in coho salmon parr (*O.kisutch*) exposed to 1, 10, and 100 mg/L of 2,4-D acid. No response was noted for the two lower concentrations, but exposure to 100 mg/L quickly (within 2 minutes) eliminated the EOG response (Tierney, Ross, Jarrard, Delaney, & Kennedy, 2006). EOG response partially recovered after 30 minutes of exposure, but not to the level of control fish. Authors cite some other research indicating salmon may have an avoidance response for 2,4-D (Tierney, et al., 2006).

We located no data regarding effects of triclopyr on olfaction.

Cellular-level Effects and Carcinogenicity

The phenoxy herbicides, including 2,4-D, have been identified as peroxisome proliferators (Ackers, Johnston, & Haasch, 2000). Peroxisome proliferation causes oxidative damage in cells, especially liver hepatocytes (Rakitsky, Koblyakov, & Turusov, 2000). This damage may induce cancer by causing uncontrolled growth, rather than via modifications in genes (*i.e.,* an epigenetic mechanism rather than genotoxicity.) Epigenetic mechanisms are considered threshold-based, and are frequently regulated for human health risk based on No Observable Effect Levels (NOELs) and safety factors (Rakitsky, et al., 2000). Currently, OPP considers no cancer-based endpoints for ecological risk.

Ackers et al (2000) evaluated peroxisome proliferation in mummichog (*Fundulus heteroclitus*) with *in vivo* exposures. Authors concluded 2,4-D does induce peroxisome proliferation in mummichog at a concentration of 1 mg/L 2,4-D acid. The exposure duration was 21 days, but increases in the protein they measured (PMP70) were noted after 7 days of exposure. Some increase was seen in the 0.01 mg/L exposure group, but it did not occur in a dose-response pattern. In addition to potential tumor induction, authors cite other research indicating peroxisome proliferation is associated with modified sex-steroid metabolism, lipid metabolism, and development of young.

Gomez et al (1998) evaluated 2,4-D induced lesions in tench (*Tinca tinca*) exposed *in vivo* to a concentration of 400 mg/L for 12 days. Symptoms appeared within the first 24 h of exposure, and included hemorrhage, enlargement of the kidney parenchyma, and fluid in the cranial cavity. Microscopic observations of the kidney showed increased presence of vacuoles, increased phagocytic activity, and necrosis (Gomez, et al., 1998). Their work indicates even short exposures may have marked sublethal effects, depending on exposure concentrations. Authors selected test concentration based on a tench 96-h LC_{50} of 800 mg/L (Gomez, et al., 1998).

Cope et al (1970) conducted a long-term (5 months) outdoor study of bluegill (*L. macrochirus*) exposed to the propylene glycol butyl ether ester of 2,4-D. Nominal exposure concentrations were 0.1, 0.5, 1.0, 5.0 and 10.0 mg/L. There was some mortality in the 5 and 10 mg/L treatment groups, and these groups also were delayed in spawning by two weeks. Number of nests and offspring produced in the ponds were similar, even with the delayed spawning in the higher treatment groups. Predators and competitors were excluded from the ponds. Some residues of 2,4-D were detected in the pond for six weeks, but concentrations declined markedly after the first two weeks. Liver lesions were present in fish exposed to concentrations of $\geq$1 mg/L. The number of lesions peaked at about 2 weeks, and then declined in most groups. They were still measurable in the 5 and 10 mg/L groups at 4 weeks, and could be measured in the 10 mg/L group at 9 weeks. Based on this study, it appears that liver effects are closely correlated with the delayed spawn in the 5 mg/L and 10 mg/L groups. As concentrations of 2,4-D decline (either by partitioning to the sediment, or possibly conversion to the acid form, which was not measured), some of the groups appeared to recover liver function (Cope, Wood, & Wallen, 1970).

McBride et al (1981) noted physical changes in the gill, liver, and interrenal tissue of sockeye fry (*O.nerka*) exposed to 0.7 and 1.0 mg/L of the butoxyethanol ester (BEE) of 2,4-D. Damage to the interrenal occurred within 48 h of exposure. Following 96-h of exposure, damage was noted at 0.3 mg/L. Sockeye smolts were slightly more resistant, displaying damage only to the interrenal, and at slightly higher concentrations (0.7 – 1.0 mg/L). Following exposure to uncontaminated water, the damage at the higher concentration was reversed (McBride, Dye, & Donaldson, 1981). Authors cite a preexisting study (Rogers & Stalling, 1972), which indicates uptake of the BEE form of 2,4-D is rapid, and peak tissue accumulation occurs within the first 6 h of exposure.

We located no data regarding effects of triclopyr on cellular damage.

Endocrine Disrupting Effects

2,4-D is on the EPA list to be evaluated for endocrine disrupting effects. We did not locate any studies evaluating 2,4-D specifically in fish, but did locate a study regarding anti-androgen effects in alligators (Crain, Guilette Jr., Rooney, & Pickford, 1997). Authors considered the endpoints of plasma hormone, gonadal-adrenal mesonephros (GAM) aromatase activity, and gonadal histopathology. Test procedures were effective, as they did detect differences for their positive control and some of the other chemicals tested, but 2,4-D did not affect any of the parameters tested.

Triclopyr is not on the EPA list to be evaluated for endocrine disrupting effects. We did not locate any studies evaluating such effects for fish.

Indirect Effects to Salmonids (Prey and Habitat Modifications)

Indirect effects on salmon include reductions in prey base (aquatic invertebrates), disruptions in primary productivity in the stream (phytoplankton and macrophytes), and effects on riparian vegetation.

Aquatic Invertebrates (Acute and Chronic Toxicity)

Values presented in Table 110 and Table 111 summarize aquatic invertebrate data tabulated in the appendices of the California red-legged frog BEs (EPA, 2009a, 2009c).

Table 110. Acute toxicity data for aquatic invertebrates.

Species		48 h EC_{50} Median Concentration[1] (mg a.e./L)	Range (mg a.e./L)	n	Source
2,4-D[1]					
Waterflea (Cladoceran)	*Daphnia magna*	Acid 25 Amines 400.5 Esters 4.19	NA 153-642.8 2.2-11.9	1 4 4	(EPA, 2009a) (Appendix F)
Triclopyr[2]					
Waterflea (Cladoceran)	*Daphnia magna*	Acid 132.9 TEA 346 BEE 0.25	NA	1	(EPA, 2009c) (Appendix A)

[1]Median of values in Appendix if more than one available, [2]Most sensitive values in Appendix.

Table 111. Chronic toxicity data for aquatic invertebrates.

Species		NOAEC LOAEC (mg/L)	Endpoint Affected	Source
2,4-D[1]				
Waterflea (Cladoceran)	*Daphnia magna*	Acid 79 Amines 16.05 Esters 0.2 Acid 151 Amines 25.64 Esters 0.483	Acid: number of young Amines: survival and reproduction Esters: survival and reproduction	(EPA, 2009a) (Appendix F)
Triclopyr[2]				
Waterflea (Cladoceran)	*Daphnia magna*	Acid ND TEA 25 BEE ND Acid ND TEA 46.2 BEE ND	Reduction in young and total brood size	(EPA, 2009c) (Appendix A)

[1]Median of values in Appendix if more than one available
[2]Most sensitive values in Appendix.

Aquatic Plants (Phytoplankton and Vascular Plants)

Given that 2,4-D and triclopyr are both herbicides, we anticipate the most sensitive receptors in salmon habitat will be photosynthetic organisms. Instream plants include various types of algal species and vascular plants. Generally the phytoplankton provide an energy source for the stream and the macrophytes are a structural component, providing attachment sites for other organisms and refugia for juvenile fishes. Reductions in primary productivity or modifications in community structure via removal of sensitive species can result in “bottom-up” trophic cascades which may adversely affect salmonids. Loss of structure provided by macrophytes may result in decreased population of aquatic invertebrates or increased predation on juvenile salmonids. Table 112 below summarizes toxicity data for aquatic plants.

Table 112. Toxicity data for aquatic plants.

Species		EC_{50} Median Concentration (mg a.e./L)	Range (mg a.e./L)	n	Source
2,4-D[1]					
Green algae	*Selenastrum capricornutum*	Acid 14.24 Amines 41.69 Esters 1.28 NOAEC Acid ND Amines 27.89 Esters 0.92	2.08-26.4 3.88-156.5 0.066-17.14 NA 0.34-78.89 0.062-8.6	2 10 7 0 12 8	(EPA, 2009a) (Appendix F)
Blue-green algae	*Anabaena flos-aquae*				
FW diatom	*Navicula pelliculosa*				
SW diatom	*Skeletonema costatum*				
Vascular plant	*Lemna gibba*	Acid 0.695 Amines 0.48 Esters 0.3637 NOAEC Acid 0.0581 Amines 0.23 Esters 0.1015	NA 0.2992-1.28 0.33-0.3974 NA 0.0476-1.28 0.062-0.141	1 3 2 1 3 2	(EPA, 2009a) (Appendix F)
Triclopyr[2]					
Green algae	*Selenastrum capricornutum*	Acid 29.8 TEA 12.1 BEE 2.5	NA	1	(EPA, 2009c) (Appendix A)
FW Diatom	*Navicula pelliculosa*	Acid ND TEA 10.6 BEE 0.07	NA	1	(EPA, 2009c) (Appendix A)
Blue-green algae	*Anabaena flos-aquae*	Acid ND TEA 4.1 BEE 1.42	NA	1	(EPA, 2009c) (Appendix A)
Duckweed	*Lemna gibba*	Acid ND TEA 6.1 BEE 0.86	NA	1	(EPA, 2009c) (Appendix A)

[1]Median of values in Appendix if more than one available, [2]Most sensitive values in Appendix.

2,4-D and Triclopyr Degradate Toxicity

In this assessment, we have primarily focused on the acid forms of both 2,4-D and triclopyr as these are the forms most commonly found in the aquatic environment. EPA indicates they have no ecological risk concerns for 2,4-D degradates, and provide no data for any (EPA, 2009a). We have located no information that would cause us to draw a different conclusion. The 2,4-D assessment also discusses potential concerns related to dioxin (2,3,7,8-TCDD) impurities which may occur in 2,4-D and concludes there are no ecological risk concerns (EPA RLF 2009, pg 101). In soil, triclopyr also breaks down to 3,5,6-trichloro-2-pyridinol (TCP) and 2,5,6-trichloro-2-methoxypyridine. Limited data were available, but values presented in the California red-legged frog assessment show TCP to be less toxic than the BEE on an acute basis, but more toxic than the TEA (EPA, 2009c). EPA did not evaluated the TCP, stating "toxicity data for the degradate (Table 113) indicates that when converted to acid equivalent TCP is less toxic than the most sensitive endpoint for triclopyr" ((EPA, 2009c), p 24). The most sensitive endpoint is commonly used in the EPA assessments.

Table 113. Toxicity data for triclopyr degradate TCP.

Species		LC_{50} or EC_{50} Concentration (mg a.e./L)	Range (mg a.e./L)	n	Source
Rainbow trout	*Oncorhynchus mykiss*	1.9	NA	1	(EPA, 2009c) (Appendix A)
Bluegill sunfish	*Lepomis macrochirus*	16.1	NA	1	(EPA, 2009c) (Appendix A)
Aquatic invertebrate	*Daphnia magna*	13.4	NA	1	(EPA, 2009c) (Appendix A)
Green algae	*Selenastrum capricornutum*	2.3	NA	1	(EPA, 2009c) (Appendix A)
Blue-green algae	*Anabaena flos-aquae*	2.3	NA	1	(EPA, 2009c) (Appendix A)

Microbial Community Effects (Sediment, Soil, and Water Column)

Given the nature of 2,4-D and triclopyr and their specificity as auxin inhibitors, we do not anticipate adverse effects on the microbial community.

Riparian Vegetation

Riparian vegetation is important for providing shade to the stream, stabilizing the stream banks, reducing sedimentation, and providing allochthonous input, both in terms of plant material and terrestrial insects. Generally there is not good data regarding the effects of herbicides on wild plants, other than weed species, but EPA requires submission of crop effects data as part of the registration process. We believe this provides a reasonable basis for evaluating effects on herbaceaous plants. Based on typical uses, for 2,4-D we expect that woody shrubs and trees are likely to be more resistant. Triclopyr is known to be effective on woody shrubs and trees. Although we did not locate specific toxicity data for those types of plants, we make the conservative assumption that sensitivity of these plants is similar to the tested crops. Guideline studies determine EC_{25}s of end-use products on the endpoints of vegetative vigor and seedling emergence. We present the most sensitive endpoints for each plant type (monocots and dicots) in Table 114 based on data summarized in EPA California red-legged frog assessments (EPA, 2009a, 2009c).

Table 114. Terrestrial plant data.

Test	Monocot EC_{25}[1,2] (lb a.e./A)Test	Dicot EC_{25} (lb a.e./A)	Source
	2,4-D[1]		
Vegetative vigor	Acid <0.0075 Amines 0.04 Esters 0.2016	Acid 0.0075 Amines 0.003 Esters 0.02	(EPA, 2009a) (Appendix F)
Seedling emergence	Acid 2.1 Amines 0.203 Esters 0.218	Acid 0.033 Amines 0.273 Esters 0.037	(EPA, 2009a) (Appendix F)
	Triclopyr[2]		
Vegetative vigor	Acid ND TEA 0.0114 BEE 0.063	Acid ND TEA 0.005 BEE 0.006	(EPA, 2009c) (Appendix A)
Seedling emergence	Acid ND TEA >0.23 BEE 0.053	Acid ND TEA ND BEE 0.045	(EPA, 2009c) (Appendix A)

[1]Median of values in Appendix if more than one available, [2]Most sensitive values in Appendix.

Summary of Toxicity Data

Assessment endpoints and associated concentrations are summarized in Table 115 for 2,4-D and in Table 116 for triclopyr.

Table 115. Assessment Endpoints and Measures for 2,4-D.
Note: Units in this table are μg ae/L, other tables are mg ae/L

Assessment Endpoint		Assessment Measure	Median Concentration[1,2] (μg ae/L)	Range (μg ae/L)	n
Direct Effects on Salmonids	Survival	*Ester* Salmonid Acute LC_{50}[1] Other Fish Acute LC_{50}[1]	1,050 690	450-14,500 260-11,900	7 2
	Growth Reproduction	*Acid* Chronic NOAEC[1] Chronic LOAEC[1]	63,400 102,000	NA	1
	Swimming	We located no data regarding effects of 2,4-D on swimming.			
	Olfaction	Modifications in EOG response at 100 mg/L (Tierney, et al., 2006).			
	Endocrine Disruption	We located no data regarding effects of 2,4-D on endocrine disruption in fish. One study of 2,4-D did not cause endocrine disruption in alligators (Crain, et al., 1997).			
	Cellular Damage, Carcinogenicity	*Acid* Peroxisome Proliferation (Ackers, et al., 2000) Ester Interrenal damage (McBride, et al., 1981)	1,000 300	NA	1 1
Effects on Prey (Aquatic Invertebrates)	Survival	*Ester* Acute Invert EC_{50}[1]	3,400	2,200-11,900	
	Growth Reproduction	*Acid* Chronic NOAEC[1] Chronic LOAEC[1]	79,000 151,000	NA	1
Effects on Primary Productivity,	Biomass & Abundance	*Ester* Algal EC_{50}[1]	1,280	66-17,140	7
Effects on Submerged and Emergent Vegetation	Biomass & Abundance	*Ester* Vascular Plant EC_{50}[1]	364	330-397	2
Effects on Ecosystem Functioning	Community Metabolism	No data located			
Effects on Riparian Vegetation	Biomass & Abundance	*Amine & Ester* Vegetative Vigor[1,2]	Monocot Amine 0.40 Ester 0.202 Dicot Amine 0.003 Ester 0.020	NA 0.190-0.218 NA 0.004-0.020	1 3 1 3

[1]If more than one value was available. If only one value was available, the actual number given.
[2] Terrestrial plant endpoints given in lb ae/A
NA Not applicable, only one value available

Table 116. Assessment Endpoints and Measures for Triclopyr.
Note: Units in this table are µg ae/L, other tables are mg ae/L

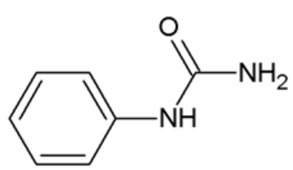
Assessment Endpoint		Assessment Measure	Median Concentration[1,2] (µg ae/L)	Range (µg ae/L)	n
Direct Effects on Salmonids	Survival	*BEE* Salmonid Acute LC_{50}[1] Other Fish Acute LC_{50}[1]	470 260	NA NA	1 1
	Growth	TEA Chronic NOAEC[1] Chronic LOAEC[1]	>32,200 <50,200	NA	1
	Reproduction	BEE Chronic NOAEC[1] Chronic LOAEC[1] (NMFS, 2011)	100 1,000	NA	1
	Swimming	We located no data regarding effects of triclopyr on swimming.			
	Olfaction	We located no data regarding effects of triclopyr on olfaction.			
	Cellular Damage	No specific studies on cellular damage located			
Effects on Prey (Aquatic Invertebrates)	Survival	*BEE* Acute Invert EC_{50}[1]	3,400	2,200-11,900	3
	Growth	*TEA* Chronic NOAEC[1] Chronic LOAEC[1]	No data available		
	Reproduction	*TEA* Chronic NOAEC[1] Chronic LOAEC[1]	25,500 46,200	NA	1
Effects on Primary Productivity, Submerged and Emergent Vegetation	Biomass & Abundance	Algal and Vascular Plant EC_{50}[1]	1,140	70-2,500	4
Effects on Ecosystem Functioning	Community Metabolism	No data located			
Effects on Riparian Vegetation	Biomass & Abundance	*BEE* Vegetative Vigor EC_{25}[2,3] Seedling Emergence EC_{25}[2,3]	Monocot 0.053 Dicot 0.006	NA	1

[1]If more than one value was available. If only one value was available, the actual number is given.
[2] Terrestrial plant endpoints given in lb ae/A
[3] Most sensitive plant values as given in RLF assessment
NA Not applicable, only one value available

Diuron and Linuron (Photosystem II Inhibiting Herbicides)

Mode of Action

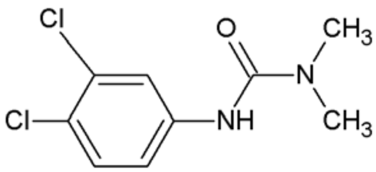
Diuron and linuron are members of a class of pesticides known as phenyl ureas (Figure 69). The basic structure for this class includes an aromatic ring (-phenyl) attached to the urea molecule (-NH_2CONH_2). One or more positions on the phenyl ring are often substituted, as are one or more –R groups on the terminal nitrogen atom (Kamrin, 1997). Diuron and linuron are very similar in structure, with two chlorine atoms substituted on the phenyl ring. The difference between the two is that diuron has two methyls substituted on the terminal nitrogen, whereas linuron has one methyl group and one methoxy group. Other phenyl ureas currently registered in the U.S. include fluometuron, siduron, tebuthiuron, and thiadizauron. It appears monouron used to be registered in the U.S. but was cancelled during the re-registration process (www.epa.gov/pesticides/reregistration/status.htm).



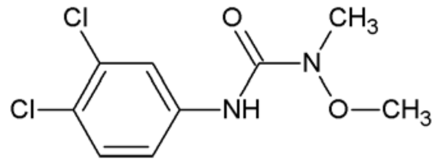
Phenylurea

Figure 69. Basic phenylurea structure.

Diuron and linuron (Figure 70) are systemic photosynthesis inhibitors, and are often used for controlling both annual and perennial grasses (Kamrin, 1997). Phenyl ureas bind to active sites in the plant chloroplasts, interfering with the photosystem II pathway that typically fixes CO_2 and produces energy. An additional effect of the inhibition is production of reactive peroxides, which break down cellular structures in the plant, including the cell membrane. Visible manifestations of the inhibition and cellular breakdown include spotting and browning of the leaves.



Diuron



Linuron

Figure 70. Structures of diuron and linuron.

Temperature and toxicity

We located no information indicating temperature specifically affects the toxicity of diuron or linuron. However, we do note higher water temperatures can affect salmonids in two ways, regardless of specific chemical effects. Higher water temperatures will increase the metabolic rate for fish, thus increasing the rate at which they process the toxicant. Depending on the chemical, this may be either beneficial or detrimental. Higher than optimum water temperatures increases general physiological stress for salmonids, making them more susceptible to other stressors.

pH and toxicity

We located no information indicating pH specifically affects the toxicity of diuron or linuron.

Toxicity of Diuron and Linuron (Assessment Endpoints)

Direct Effects to Salmonids

We evaluated potential effects to salmonids based on toxicity information included in the salmonid BEs (EPA, 2003c, 2004b), the more recent California red-legged frog assessments (EPA, 2008, 2009b), and the REDs (EPA, 1995, 2003d). We also searched open literature for endpoints not reported in EPA documentation, and data available since their publication.

Toxicity data for diuron were reported differently in the salmonid BE (EPA, 2003c) and the California red-legged frog assessment (EPA, 2009b) and we had difficulty reconciling the two, even after reference to the RED (EPA, 2003d). It appears much of the toxicity data is derived either from Mayer and Ellersieck 1986, a compilation of toxicity test data from a U.S. FWS laboratory, or open source publications rather than registrant-submitted guideline tests. Sometimes the sources are referred to by MRID number, and sometimes by author. In some cases, neither MRID number nor author is listed in the bibliography of either the main document or appendices. Thus, wherever possible, we obtained original sources, and in some cases, recalculated genus or species medians. Sources we used or reference sources as cited in EPA documentation are noted at appropriate locations in the data tables. As with the other a.i.s addressed in this Opinion, the ecological effects appendices from the California red-legged frog assessments, which are generally EPA's most recent and comprehensive listing of toxicity data, are included in *Appendix 10*. Original data from Mayer and Ellersieck is also included in *Appendix 10*.

Based on what was presented in the EPA documents, and what we located in literature, there are less data available for linuron than diuron, especially for chronic (reproduction and growth) endpoints. As with diuron, we used original sources whenever possible.

Both diuron (http://www.epa.gov/endo/pubs/prioritysetting/draftlist2.htm) and linuron (http://www.epa.gov/endo/pubs/prioritysetting/finallist.html) have been placed on EPA's list for evaluation of endocrine-disrupting compounds (EDCs). According to EPA's website, screening tests for initial a.i.s are due in February 2011, so we do not anticipate any data from those tests will be available for this consultation. We did locate some open literature studies on endocrine-disrupting properties of linuron. Information from those studies are included in the toxicity data. Other chemicals on the evaluation list for endocrine effects are 2,4-D, captan, and chlorothalonil (http://www.epa.gov/endo/pubs/prioritysetting/finallist.html.

Survival

Survival is evaluated primarily based on guideline tests used to determine the LC_{50} of a population of test fish.

Diuron

Mayer and Ellersieck (1986) report a range of tests on diuron for several salmonid species, including cutthroat trout and rainbow trout, which are of the same genus (*Oncorhynchus*) as the salmonids considered in this Opinion, and lake trout, which are also salmonids, but of a different genus (*Salvelinus*). They also report a number of tests on the bluegill sunfish, a warm-water fish. Values below were derived from the 96 h tests with 95% technical a.i. (Mayer & Ellersieck, 1986). Raw data downloaded from the USGS website are contained in *Appendix 10*. Medians for each species were calculated using an Excel spreadsheet. Overall, salmonids (96 h LC_{50} range 700-7,700 μg/L) appear slightly more sensitive than most of the other fish tested (96 h LC_{50} range 500-14,200 μg/L). On EPA's qualitative scale, diuron is moderately toxic to fish.

The single exception to that is the LC_{50} reported for the striped bass. In one EPA document it was reported as 400 μg/L (EPA, 2009b), and in another it was reported as 500 μg/L (EPA, 2003c). In the salmonid BE (EPA, 2003c) the 500 μg/L value is associated with a test of an 80% a.i. formulated product, and appears to have been conducted on larvae. Another reported data point for fingerlings is 6,000 μg/L also with the 80% a.i. formulated product, which is more consistent with other standardized test data. In the original source (Hughes, 1973) the test material is described as an 80% a.i. product Karmex. The 96 h LC_{50} for larvae is 500 μg/L, and for fingerlings it is 6,000 μg/L. Thus, it appears striped bass larvae are more sensitive than fingerlings, and this likely applies to all fish species.

Linuron

Less data were available for linuron. Mayer and Ellersieck (1986) reported one test with a 95% technical a.i. on channel catfish (96 h LC_{50} 2,900 μg/L). Tests for rainbow trout

(96 h LC_{50} 3,000 μg/L, MRID 40445501) and bluegill (96 h LC_{50} 9,600 μg/L, MRID 40354201) were reported in all three EPA documents (EPA, 1995, 2004b, 2008) and summarized in Table 117. Complete references for these MRIDs were not provided in any of the documents, but we assume they are registrant-submitted guideline studies. On EPA's qualitative scale, linuron is moderately toxic to fish.

Table 117. Acute toxicity data for fish.

Species		96-h LC_{50} Median Concentration[1] (μg/L)	Range (μg/L)	n	Source
		Diuron			
Salmonids					
Cutthroat trout	*Oncorhynchus clarkii*	1,800[1]	710-2,200	10	Mayer & Ellersieck 1986
Lake trout	*Salvelinus namaycush*	2,200[1]	1,100-2,700	12	Mayer & Ellersieck 1986
Rainbow trout	*Oncorhynchus mykiss*	6,700[1]	4,900-7,700	5	Mayer & Ellersieck 1986
Other fish					
Striped bass	*Morone saxatilis*	Larvae 500 Fingerlings 6,000 Karmex formulation (80% a.i.)	NR	NR	Hughes 1973
Bluegill	*Lepomis macrochirus*	8,500[1]	2,800-10,400	12	Mayer & Ellersieck 1986
Fathead minnow	*Pimepheles promelas*	14,200[2]	NA	1	Call et al 1987
		Linuron			
Rainbow trout	*Oncorhynchus mykiss*	3,000	NR	NR	MRID 40445501 as cited in RLF
Channel catfish	*Ictalurus punctatus*	2,900	NA	1	Mayer & Ellersieck 1986
Bluegill sunfish	*Lepomis macrochirus*	9,600	NR	NR	MRID 40354201 as cited in RLF

[1] 96 hr test data on 95% technical only. Most data from static tests, although one flow-through test is included in cutthroat trout data set, and one in lake trout data set.
[2] 96 hr test endpoint calculated by authors
NR Not reported

Reproduction and Growth

Reproduction and growth endpoints are typically evaluated in guideline tests that expose fish to the a.i. and then measure effects on a number of growth and reproductive parameters. Neither diuron nor linuron appeared to have these guideline studies, although other work appeared to have been conducted under contract to EPA, and is likely a guideline test or very similar protocol (D. Call, Brooke, & Kent, 1983). We have not reviewed the original source of Call et al (1983), although we have reviewed the original source of a later publication which appears to be based on the same work (D. J. Call et al., 1987). We have no specific explanation for the discrepancies between the two sets of values. In addition to the data reported in EPA documentation, we also located another work, which reports a reproductive effect not typically considered in the standard test (Gagnon & Rawson, 2009), and is based on a shorter exposure duration at the egg stage. The most sensitive endpoint and the shortest exposure is for the pink snapper (NOAEC 5 μg/l, LOAEC 50 μg/l) as shown in Table 118. The range between the NOAEC and LOAEC in this test overlaps with the endpoints derived from the two Call et al (1983 and 1987) studies. The Nebeker and Schytema (1998) work, which was also with fathead minnow, resulted in NOAECs that are two orders of magnitude higher than the other three tests (Nebeker & Schuytema, 1998). We have no particular explanation for this discrepancy, other than perhaps the growth endpoints they measured are less sensitive than the reproduction endpoints measured in the other studies. In 5 d static exposures studies on zebrafish *(Danio rerio)* embryos 100% of embryos exposed to 10,000 μg/L died. Of the survivors, 9% exhibited curvature and edema ((NMFS, 2011) included as Appendix 8).

There appears to be virtually no chronic fish data available for linuron. The salmonid BE (EPA, 2004b) gives an NOAEC of <42 μg/L, and notes an additional test has been requested since effects were observed at the lowest concentration tested and a NOAEC could not be established. No MRID is given for the study. Neither the RED (EPA, 1995) nor the California red-legged frog assessment (EPA, 2008) give any data. The California red-legged frog assessment used an acute-to-chronic ratio (ACR) calculation. We did not locate any chronic data in the open literature. In 5 d static exposures studies on zebrafish *(Danio rerio)* embryos conducted specifically for this consultation, 51% of embryos exposed to 1,000 μg/L died. Of the survivors, 9% exhibited curvature and edema ((NMFS, 2011) included as Appendix 8). Solutions of 10,000 μg/L produced 100% mortality in the embryos. As effects on the zebrafish were noted at concentrations slightly above the existing NOAEC of <42 μg/L, we have used the existing NOAEC in our evaluation.

Table 118. Chronic toxicity data for fish.

Species		NOAEC LOAEC (μg/L)	Endpoint Affected (Exposure duration)	Source
Diuron				
Pink snapper	*Pagrus auratus*	5 50	Increase in spinal deformities (36 h exposure in egg stage)	(Gagnon & Rawson, 2009)
Fathead minnow	*Pimepheles promelas*	26 62	Number of survivors (Not given)	MRID 00141636, as cited in RLF, (D. Call, et al., 1983)
Fathead minnow	*Pimepheles promelas*	33.4 78.0	Increase in dead or abnormal fry post hatch (64 d exposure)	(D. J. Call, et al., 1987)
Fathead minnow	*Pimephales promelas*	<3,400 3,400	Decrease in juvenile growth (10 d exposure)	(Nebeker & Schuytema, 1998)
Fathead minnow	*Pimephales promelas*	4,200 8,300	Decrease in embryo-larval growth (10 d exposure)	(Nebeker & Schuytema, 1998)
Zebrafish	*Danio rerio*	1,000 10,000	Survival (5 d)	(NMFS, 2011)
Linuron				
Rainbow trout	*Oncorhynchus mykiss*	<42 42	Decreased growth (Not given)	As given in salmonid BE, no MRID referenced
Zebrafish	*Danio rerio*	100 1,000	Survival Edema, curvature (5 d)	(NMFS, 2011)

Endocrine Disrupting Effects

Linuron has been identified by EPA as an endocrine disrupting compound, functioning as "competitive androgen receptor agonist" ((EPA, 2008), Appendix J). Diuron has not been specifically identified as an endocrine disruptor by EPA, although it is on the list of potential endocrine disrupting chemicals, and will be subject to further evaluation by the EPA. NMFS makes no assertion regarding whether diuron is or is not an endocrine disruptor. However, given structural similarities between the two a.i.s, and lack of available data regarding diuron, we have made the conservative assumption that diuron acts in a fashion similar to linuron, acknowledging there may be differences in potency. Thus, in absence of data showing diuron not to be an androgen agonist, or to be a weaker androgen agonist than linuron, we have opted to use available data on linuron to evaluate endrocrine effects for both chemicals. During the comment period for this Opinion, NMFS received preliminary data regarding the effects of linuron and diuron on young male rats from a researcher at the National Health and Environmental Effects Laboratory (NHEERL)[5] (Gray, Jr 2011). Using the USEPA and OECD Hershberger Assay Test Guideline, the researcher found diuron to have antiandrogenic activity but to be a weaker antiandrogen thatn linuron. Diuron significantly increased adrenal weights in young rats, and significantly reduced one of the five androgen-dependent tissues tested. Linuron significantly reduced growth of all five tissues tested.

Lambright et al (2000) examined the effects of linuron both *in vitro* and *in vivo*, and concluded that *in vitro* linuron binds both human and rat androgen receptors and *in vivo*, affects sexual differentiation in rats via this mechanism (Lambright, et al., 2000). It is difficult to extrapolate dietary data from rats to aquatic organisms, given differences in exposure routes (dietary versus gill uptake from water) and metabolism (homeotherms versus poikilotherms), but we did locate two studies evaluating androgenic effects of linuron in aquatic species. Jolly et al (2009) evaluated effects of linuron and several other contaminants both *in vitro* and *in vivo* using the three-spined stickleback (*Gasterosteus aculeatus*). Using production of an androgen-induced protein in kidney cells as a measure, they found concentration dependent effects in both *in vitro* and *in vivo* tests. *In vivo*, linuron caused a significant decrease in protein production at a water concentration of 100 μg/L after a 21-day exposure. Effects were not significant at 25 μg/L (Jolly, et al., 2009). Kashian and Dodson (2001) evaluated effects on the sex determination, fecundity, and growth rate for *D. magna* following a 6 d exposure. There

[5] Gray, Jr. LE, 2011. Hershberger Assay Study with Linuron and Diuron in young male rats. (Unpublished data, submitted in letter to NMFS on June 7, 2011). National Health and Environmental Effects Research Laboratory (NHEERL), U.S. EPA, Research Triangle Park, NC.

were no discernable effects on any of these endpoints at the highest exposure concentration of 100 μg/L (Kashian & Dodson, 2001).

Indirect Effects to Salmonids (Prey and Habitat Modifications)

Indirect effects on salmon include reductions in prey base (aquatic invertebrates), disruptions in primary productivity in the stream (phytoplankton and macrophytes), and effects on riparian vegetation.

Aquatic Invertebrates (Acute and Chronic Toxicity)

Survival - Diuron

Mayer and Ellersieck (1986) report acute EC_{50} data on diuron for 5 separate species (Table 119). We located additional references that reported data on three more species (Hernando, Ejerhoon, Fernandez-Alba, & Chisti, 2003; Martins, Saker, & Teles, 2007; Nebeker & Schuytema, 1998; Neuwoehner, Zilberman, Fenner, & Escher, 2010). Although the dataset is weighted heavily towards cladocerans (5 of the 9 values), it does include other functional groups, such as a stonefly, amphipods, and an isopod. EC_{50} values range from 160-19,400 μg/L, with an overall median of 8,600 μg/L. Based on EPA's qualitative scale, diuron is highly toxic to moderately toxic to aquatic invertebrates.

Survival - Linuron

Based on available data, linuron appears to be more toxic to aquatic invertebrates than diuron (Table 119). Three data points for *D. magna* were available, ranging from 120 μg/L (MRID 0014932) to 1,900 μg/L (MRID 43996501). Applicant contends the study producing the 120 μg/L values was flawed, and recommends the use of the newer 1,900 μg/L based on a better documented study, and better agreement with chronic data (MRID42153401). NMFS has used the value as stated in the EPA documentation. The median of the available values for *D. magna* is 270 μg/L. Mayer and Ellersieck (1986) also report an acute EC_{50} for midge fly larvae (*Chironomid sp*) of 2,900 μg/L. Median value for invertebrate data is 1,085 μg/L. Using EPA's qualitative scale, linuron is highly toxic to moderately toxic to aquatic invertebrates.

Table 119. Acute toxicity data for aquatic invertebrates.

Species		48 h[1] or 96-h[2] EC_{50} Median Concentration[1] (μg/L)	Range (μg/L)	n	Source
Diuron					
Amphipod	*Gammarus fasciatus*	160[2]	NA	1	(Mayer & Ellersieck, 1986)
Stonefly	*Pteronarcys californica*	1,200[2]	NA	1	(Mayer & Ellersieck, 1986)
Waterflea (Cladoceran)	*Daphnia pulex*	1,400[1]	NA	1	(Mayer & Ellersieck, 1986)
Waterflea (Cladoceran)	*Simocephalus serrulatus*	2,000[1]	NA	1	(Mayer & Ellersieck, 1986)
Isopod	*Asellus brevicaudus*	15,500[2]	NA	1	(Mayer & Ellersieck, 1986)
Waterflea (Cladoceran)	*Daphnia magna*	8,600[1]	NA	1	(Hernando, et al., 2003)
Waterflea (Cladoceran)	*Daphnia magna*	11,655	NA	1	(Neuwoehner, et al., 2010)
Waterflea (Cladoceran)	*Daphnia pulex*	17,900	NA	1	(Nebeker & Schuytema, 1998)
Amphipod	*Hyalella azteca*	19,400	NA	1	(Nebeker & Schuytema, 1998)
Waterflea (Cladoceran)	*Daphnia magna*	3,000 (Reduced O_2 consumption, 15 min exposure)	NA	1	(Martins, et al., 2007)
Linuron					
Waterflea (Cladoceran)	*Daphnia magna*	120	NA	1	MRID 0014932 as cited in RLF
Waterflea (Cladoceran)	*Daphnia magna*	270	NA	1	(Mayer & Ellersieck, 1986)
Waterflea (Cladoceran)	*Daphnia longispina*	360	NA	1	Stephenson & Kane 1984 as cited in (Cuppen, Van den Brink, Van der Woude, Zwaardemaker, & Brock, 1997) NOT CONFIRMED, NOT IN MEDIAN CALCULATION
Waterflea (Cladoceran)	*Daphnia magna*	1,900	NA	1	MRID 43996501 as cited in Pyxis 2010
Midge larvae	*Chironomus sp*	2,900	NA	1	(Mayer & Ellersieck, 1986)

Reproduction and Growth - Diuron

There are little chronic data for aquatic invertebrates for either diuron or linuron, although one study by Nebeker & Schuytema (1998) did evaluate the chronic effects of diuron on a range of aquatic species (Table 120). In their study (7-10 d exposure, depending on species) NOAECs ranged from 1,800-13,400 μg/L, with a median of 4,000 μg/L (Nebeker & Schuytema, 1998). In some cases, they noted mortality at the LOAEC in addition to effects on reproduction and growth. A single study, cited in the salmonid BE as TN2418, gives an NOAEC of 200 μg/L for *D. magna* (EPA, 2004b). However, we were unable to find a complete reference for TN2418 in any of the EPA documents, so we cannot confirm this endpoint from the original source. The BE did not report endpoints affected or a LOAEC.

Reproduction and Growth - Linuron

For linuron there was one chronic study on *D. magna*, which gives the NOAEC as 130 μg/L and a LOAEC of 240 μg/L. This value was reported in both the salmonid BE and the RED (MRID 42153401, (EPA, 1995, 2004b)). Neither reports the endpoints affected. We have not reviewed the original source or DER. Given the fact that the current chronic endpoint is actually greater than the lowest available acute endpoint, EPA opted to calculate an ACR for the California red-legged frog assessment (EPA, 2008).

Table 120. Chronic toxicity data for aquatic invertebrates.

Species		NOAEC LOAEC (μg/L)	Endpoint Affected	Source
Diuron				
Waterflea (Cladoceran)	*Daphnia magna*	200 Not given	Not given	TN2418 as cited in RLF
Waterflea (Cladoceran)	*Daphnia pulex*	4,000 7,700	Mortality, reproduction (7 d exposure)	(Nebeker & Schuytema, 1998)
Amphipod	*Hyalella azteca*	7,900 15,700	Mortality, reduced growth (10 d exposure)	(Nebeker & Schuytema, 1998)
Midge larvae	*Chironomus tentans*	3,400 7,100	Mortality, growth (10 d exposure)	(Nebeker & Schuytema, 1998)
Annelid worm	*Lubriculus varigatus*	1,800 3,500	Reduced growth (10 d exposure)	(Nebeker & Schuytema, 1998)
Snail	*Physa gyrina*	13,400 22,800	Reduced growth (10 d exposure)	(Nebeker & Schuytema, 1998)
Linuron				
Waterflea (Cladoceran)	*Daphnia magna*	130 240	Based on existing data, chronic effects appeared to have occurred at same concentrations as acute effects Calculated as ACR for RLF	MRID 42153401 as cited in RLF Also given in salmonid BE

Aquatic Plants (Phytoplankton and Vascular Plants)

Given that diuron and linuron are both herbicides, we anticipate the most sensitive receptors in salmon habitats will be photosynthetic organisms. Table 121 summarizes toxicity data for aquatic plants exposed to diuron or linuron. Instream plants include various types of algal species and vascular plants. Generally the phytoplankton provide an energy source for the stream and the macrophytes are a structural component, providing attachment sites for other organisms and refugia for juvenile fishes. Reductions in primary productivity or modifications in community structure via removal of sensitive species can result in "bottom-up" trophic cascades which may adversely affect salmonids. Loss of structure provided by macrophytes may result in decreased populations of aquatic invertebrates or increased predation on juvenile salmonids.

EPA documents for diuron reported EC_{50}s for a number of algal species and we located some additional data in the open literature. Within the source, these were grouped by generic classification (*e.g.*, green algae, diatoms) and if multiple values were available, we calculated median values using Excel. Based on the data available, all aquatic plants appear much more sensitive to diuron than either fish or invertebrates. EC_{50}s for green algae range from 2.4-51 μg/L. One test result for a red alga was reported. With an EC_{50} of 24 μg/L, it falls within the range of the green algae. Diatoms appear only slightly more tolerant of diuron, with a range of EC_{50}s from 18-95 μg/L. Calculation of a single median value based on all phytoplankton tests resulted in a phytoplankton EC_{50} of 28 μg/L. The one test we located for vascular plants gave a 7-day EC_{50} of 15 μg/L for duckweed.

For linuron, we located only three species specific endpoints for plant species, all of which were reported in the California red-legged frog assessment. We have not been able to confirm these data from the original source. These data include one endpoint each for a green alga (67 μg/L), a diatom (13.7 μg/L), and a vascular plant (2.5μg/L). Median value for all plant data is 13.7 μg/L.

However, we did locate a body of work examining the effect of linuron in various types of microcosms (Cuppen, et al., 1997; Daam, Rodrigues, Van den Brink, & Nogueira, 2009; Daam & Van den Brink, 2007; Daam, Van den Brink, & Nogueira, 2009). These studies evaluated a range of ecological effects including changes in functional parameters (*e.g.*, DO and pH), changes in phytoplankton communities, changes in macrophytes, changes in the zooplankton community, and sometimes changes in macroinvertebrates. Based on changes in these parameters, authors derived an NOAEC for ecosystem functioning. Separate studies, Cuppen et al (1997) and Daam and Van den Brink (2007), based on very similar experiments on indoor microcosms, concluded the ecosystem NOAEC for linuron was 0.5 μg/L. The most sensitive parameters were DO and pH. In both cases, LOAEC was 5.0 μg/L. Daam et al (2009) followed up on this concept with an additional outdoor microcosm experiment and a review of various linuron microcosms.

Table 121. Toxicity data for aquatic plants.

Species		72 h[1] or 96-h[2] EC_{50} Median Concentration (µg/L)	Range (µg/L)	n	Source
Diuron					
Green algae	*Selenastrum capricornutum*	2.4[2]	NA	1	MRID 42218401, as cited in EPA RLF and RED, MRID reference not located
Green algae	*Psuedokirchneriella subcapitata*	EC_{54}=5[3]	NA	1	(Knauer, Sobek, & Bucheli, 2007)
Green algae	*Scendesmus vacuolatus*	12[4]			(Neuwoehner, et al., 2010)
Green algae	*Chlorella sp* *Chlorcoccum sp* *Chlamydomonas sp* *Platymonas sp* *Neochloris sp*	19[1]	10-28		MRID 40228401,1986 as cited in EPA RLF appendix, reference not located
Green algae	*Psuedokirchneriella subcapitata*	51[4]	NA	1	(Neuwoehner, et al., 2010)
Red algae	*Porphyridium cruentum*	24[1]	NA	1	MRID 40228401,1986 as cited in EPA RLF appendix, reference not located
Diatoms	*Nitzschia closterium* *Amphora exigua* *Stauronoides amphoroides* *Achnanthes brevipes* *Cyclotella nana* *Monochrysis lutheri* *Thalassiosira fluviatilus* *Navicula incerta*	35[1]	18-95	8	MRID 40228401,1986 as cited in EPA RLF appendix, reference not located
Duckweed	*Lemna perpusilla*	15 (7 day)	UNK	UNK	ECOTOX 8628 as cited in RLF
Linuron					
Green algae	*Selenastrum capricornutum*	67	NA	1	MRID 42086801 as cited in RLF
Diatom	*Navicula pelliculosa*	13.7	NA	1	MRID 43992302 as cited in RLF
Western waterweed or Anacharis	*Elodea nuttali*	2.5	NA	1	ECOTOX 18629 as cited in RLF

Species		72 h[1] or 96-h[2] EC_{50} Median Concentration (μg/L)	Range (μg/L)	n	Source
Ecosystem	Community metabolism	NOEAC 0.3 LOAEC 3.0	NA	NA	(Daam, Rodrigues, et al., 2009)

[3] Author did not calculate EC_{50}, and data were only presented graphically [4] 24 h tests

experiments under different exposure regimes (single-peak, pulsed, and constant) (Daam, Rodrigues, et al., 2009). Two of the single peak exposure studies produced higher NOAECs based on community metabolism (15 μg/L and <20 μg/L) but other studies corroborated the 0.5 μg/L NOAEC. They then applied a toxic unit (TU) approach suggested by another body of work (T.C.M. Brock, et al., 2000) to calculate ecosystem threshold values for a range of other photosynthesis inhibitors, including diuron (Daam, Rodrigues, et al., 2009). They concluded an ecosystem function threshold for diuron was 0.2 TU, which we calculated as 3 μg/L based on TU data for diuron presented in the paper.

Diuron and Linuron Degradate Toxicity

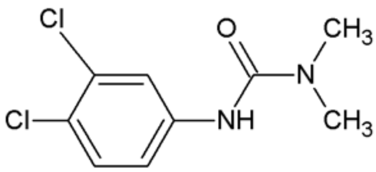
Both diuron and linuron are relatively resistant to abiotic degradation (hydrolysis and photolysis) in the environment and are primarily broken down by microbial action in either soil or water (EPA, 1995, 2003d; Tixier, Sancelme, Bonnemoy, Cuer, & Veschambre, 2001). Neither is broken down by hydrolysis, and when photolytic reactions occur, they primarily result in a hydroxyl group substitution for one of the chlorines on the phenyl ring. In contrast, the microbial-mediated reactions occur at the terminal nitrogen on the urea moiety. Although there are slight differences between the a.i.s as to how the degradation occurs, it generally follows the pathway of removal of the methyl and/or methoxy groups, followed by eventual cleavage of the molecule at the carbonyl atom to produce 3,4-dichloroaniline (DCA) (Figure 71). Metabolic studies on mammals indicate a similar pathway occurs in vertebrates, with initial removal of the methyl groups attached to the urea, and eventually complete processing to DCA. In linuron, the methoxy group appears to be removed preferentially, followed by the methyl group. Metabolites are excreted primarily in urine ((EPA, 1995), Geldmacher et al 1971, as cited in (D. J. Call, et al., 1987), (Menzie, 1969), (Gosselin, Smith, & Hodge, 1984)).

Call et al (1987) evaluated metabolism of diuron in rainbow trout and concluded metabolic products produced by the fish were DCA and two demethylated compounds, likely DCPMU and DCPU.

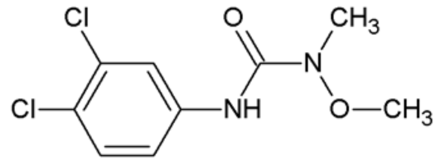
Thus, aquatic organisms may be exposed to the following degradates of diuron and linuron (Table 122, Figure 71).

Table 122. Degradates of Diuron and Linuron.

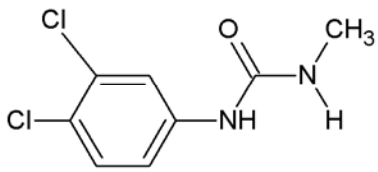
Degradate	Chemical Name	Parent
DCPMU Desmethoxy-linuron	*1-(3,4,dichlorophenyl)-1-methylurea*	Diuron Linuron
DCPU Norlinuron	*3,4-dichlorophenylurea*	Diuron Linuron
DCA	*3,4-dichloroaniline*	Diuron Linuron
Desmethyl-linuron	*1-(3,4,dichlorophenyl)-1-methoxyurea*	Linuron



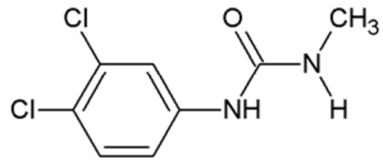
Diuron



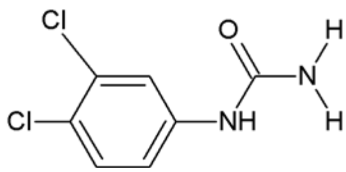
Linuron



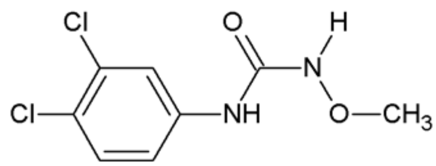
DCPMU



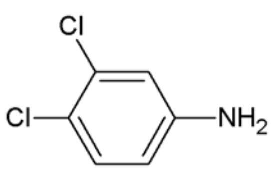
Desmethoxy linuron



DCPU



Desmethyl linuron



DCA (3,4 dichloroaniline)

Figure 71. Toxic degradates of diuron and linuron.

We did not locate a great deal of information on the toxicity of the degradates, although we did locate one highly relevant study that considered a number of the degradates, and their effects on both algae and aquatic invertebrates (*D. magna*) (Neuwoehner, et al., 2010). Another study also did a toxicity analysis, but they used a bacterium, *Vibrio fischeri* in a Microtox® test (Tixier, et al., 2001). We are uncertain how to extrapolate those results to our organisms of concern, thus we do not present those data.

The Neuwoehner *et al* (2010) data is below. Although they considered the degradates of diuron, linuron also has several of the same degradates (Table 123), so we consider this information for both a.i.s. Based on their data, they concluded that loss of one or both methyl groups made the degradates less toxic to the algal species, and more toxic to *D. magna.* The final biotransformation product, DCA, was approximately 2 orders of magnitude more toxic to *D. magna* than parent diuron. However, total removal of the urea moiety[6] resulted in it being approximately 3 orders of magnitude less toxic to the algal species. The less commonly occurring phototransformation products, hydroxyl group substitutions on the phenyl ring, were less toxic than the parent to both algal species and *D. magna.*

Neither the linuron salmonid BE (EPA, 2004b) nor the linuron California red-legged frog assessment (EPA, 2008) included any degradate toxicity data for any species. No degradate toxicity data for any species are presented in either of the diuron assessments (EPA, 2003c, 2008). The diuron salmonid assessments does note DCA as a degradate, but states it was not included in the risk analysis as it is anticipated to be less than 1% of applied a.i. (EPA, 2003c).

[6] Moiety means molecular fragment, in this case urea ($-(NH_2)_2CO$)

Table 123. Diuron degradate toxicity data for *D. magna* and algae.

Species		24 h[1] or 48 h[2] EC_{50} Concentration (µg/L)	Source
Diuron			
Waterflea	*Daphnia magna*	11,655[2]	(Neuwoehner, et al., 2010)
Green algae	*Psuedokirchneriella subcapitata*	51[1]	(Neuwoehner, et al., 2010)
Green algae	*Scendesmus vacuolatus*	12[1]	(Neuwoehner, et al., 2010)
1-(3,4,dichlorophenyl)-1-methylurea			
Waterflea	*Daphnia magna*	10,734[2]	(Neuwoehner, et al., 2010)
Green algae	*Psuedokirchneriella subcapitata*	70[1]	(Neuwoehner, et al., 2010)
Green algae	*Scendesmus vacuolatus*	15[1]	(Neuwoehner, et al., 2010)
3,4,dichlorophenylurea			
Waterflea	*Daphnia magna*	2,871[2]	(Neuwoehner, et al., 2010)
Green algae	*Psuedokirchneriella subcapitata*	11,277[1]	Neuwoehner et al 2010
Green algae	*Scendesmus vacuolatus*	3,286[1]	(Neuwoehner, et al., 2010)
3,4-dichloroaniline			
Waterflea	*Daphnia magna*	227[2]	(Neuwoehner, et al., 2010)
Green algae	*Psuedokirchneriella subcapitata*	24,303[1]	(Neuwoehner, et al., 2010)
Green algae	*Scendesmus vacuolatus*	13,286[1]	(Neuwoehner, et al., 2010)
1-(3-chloro,4-hydroxyphenyl)-1,1-dimethylurea and 1-(3-hydroxy,4-chlorophenyl)-1,1-dimethylurea (phototransformation products)			
Waterflea	*Daphnia magna*	77,474[2]	(Neuwoehner, et al., 2010)
Green algae	*Psuedokirchneriella subcapitata*	278[1]	(Neuwoehner, et al., 2010)
Green algae	*Scendesmus vacuolatus*	103[1]	(Neuwoehner, et al., 2010)

Microbial Community Effects (Sediment, Soil, and Water Column)

Given the nature of diuron and linuron and their specificity for photosystem II, we do not anticipate adverse effects on the microbial community. We did locate one study (Mukherjee et al., 2006) evaluating the effects of linuron on soil microorganisms in a cropped field. The authors concluded application of linuron did not affect soil microfauna.

Riparian Vegetation

Riparian vegetation is important for providing shade to the stream, stabilizing the stream banks, reducing sedimentation, and providing allochthonous input, both in terms of plant material and terrestrial insects. Generally there are not good data regarding the effects of herbicides on wild plants, other than weed species, but EPA requires submission of crop effects data as part of the registration process. We believe this provides a reasonable basis for evaluating effects on herbs and forbs, and based on the mode of action for diuron and linuron, expect that woody shrubs and trees are likely to be more resistant. Guideline studies determine EC_{25}s of end-use products on the endpoints of vegetative vigor and seedling emergence. We calculated medians for the range of crop data presented inthe EPA assessments, which are presented in Table 124.

Table 124. Terrestrial plant data.

Plants	Test	EC_{25} Median (lb ai/A)	Range (lb ai/A)	n	Source
Diuron					
Rapeseed Sorghum Soybean Sugarbeet Tomato Wheat	Vegetative vigor	0.0164	0.00017-0.0753	6	(EPA, 2009b) (Appendix L)
Sorghum Sugarbeet Tomato	Seedling emergence	0.092[1]	0.074-0.81	3	(EPA, 2009b) (Appendix L)
Linuron					
No guideline study data available					(EPA, 2008)

[1] Some studies had indeterminate endpoints, which were not included in the calculation.

Summary of Toxicity Data for Diuron and Linuron

Assessment endpoints and associated concentrations are summarized in Table 124 for diuron and in Table 125 for linuron.

Table 125. Assessment Endpoints and Measures for Diuron.

Assessment Endpoint		Assessment Measure	Median Concentration[1] (μg/L)	Range	n
Direct Effects on Salmonids	Survival	Salmonid Acute LC_{50} Other Fish Acute LC_{50}	2,325 10,450	710-23,800 6,700-14,200	8 2
	Growth	Chronic NOAEC Chronic LOAEC	No data available		
	Reproduction	Chronic NOAEC Chronic LOAEC	30 70	5-4,200 62-8,200	4
	Swimming	We located no data regarding effects of diuron on swimming.			
	Olfaction	No effect 1,000 μg/L (Tierney 2007)			
	Endocrine Disruption	Androgen Protein NOAEC Androgen Protein LOAEC (Jolly, et al., 2009)	25 100	NA	1
Effects on Prey (Aquatic Invertebrates)	Survival	Acute Invert EC_{50}	8,600	160-19,400	9
	Growth	Chronic NOAEC Chronic LOAEC	5,650 11,400	3,400-13,400 3,500-22,800	4
	Reproduction	Chronic NOAEC Chronic LOAEC	4,000 7,700	NA	1
Effects on Primary Productivity, Submerged and Emergent Vegetation	Biomass & Abundance	Algal and Vascular Plant EC_{50}	28	10-95	17
Effects on Riparian Vegetation	Biomass & Abundance	Vegetative Vigor	0.0164 lb ai/A	0.00017-0.0753lb ai/A	6

[1] If more than one value was available. If only one value was available, the actual number is given.
[2] Data for linuron, not diuron. Used as a conservative estimate given no data for diuron available for this endpoint.
NA Not applicable, only one value available

Table 126. Assessment Endpoints and Measures for Linuron.

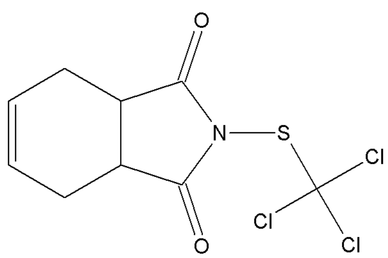
Assessment Endpoint		Assessment Measure	Median Concentration[1] (µg/L)	Range	n
Direct Effects on Salmonids	Survival	Salmonid Acute LC_{50} Other Fish Acute LC_{50}	3,000 6,250	NA 2,400-9,600	1 2
	Growth	Chronic NOAEC Chronic LOAEC	<42 42	NA	1
	Reproduction	Chronic NOAEC Chronic LOAEC	No data available		
	Swimming	We located no data regarding effects of linuron on swimming.			
	Olfaction	No effect 1mg/L (Tierney, Ross, & Kennedy, 2007)			
	Endocrine Disruption	Androgen Protein NOAEC Androgen Protein LOAEC (Jolly, et al., 2009)	25 100	NA	1
Effects on Prey (Aquatic Invertebrates)	Survival	Acute Invert EC_{50}	1,085	120-2,900	4
	Growth	Chronic NOAEC Chronic LOAEC	No data available		
	Reproduction	Chronic NOAEC Chronic LOAEC	No data available		
Effects on Primary Productivity, Submerged and Emergent Vegetation	Biomass & Abundance	Algal and Vascular Plant EC_{50}	13.7	2.5-67	3
Effects on Riparian Vegetation	Biomass & Abundance	Vegetative Vigor	No data available		

[1]If more than one value was available. If only one value was available, the actual number is given.
NA Not applicable, only one value available

Fungicides (Captan and Chlorothalonil)

Captan Mode of Action

Captan is a non-systemic, phthalimide fungicide which inhibits the normal cell division of microorganisms and fungi (Cremlyn, 1991; EPA, 2007c; Krieger & Krieger, 2001). The parent captan molecule (Figure 72) cleaves at the bond between the nitrogen and sulfur atoms, and the trichloromethylthio (TCMT) reacts with thiol groups in various enzymes and cellular structures (Figure 73). The reaction of TCMT with the thiol groups releases thiophosgene, which also reacts with thiol groups (Bernard & Gordon, 2000; Cremlyn, 1991; Krieger & Krieger, 2001; Lukens, 1966).



Captan

Figure 72. Structure of Captan

H-S-C-Cl$_3$ + 2R'-S-H → H$_2$ + Cl-C-S-Cl + R'-S-S-R' + H-Cl

Where R' is a thiol group

Figure 73. Captan Mode of Action

There are two other phthalimide fungicides with similar chemical structures, folpet and captafol (Figure 74). U.S. registrations of captafol were cancelled during re-registration (http://www.epa.gov/pesticides/reregistration/status_page_c.htm, Case #0116). Folpet is still an active registration, but appears to be registered in the U.S. only for use on avocadoes (EPA, 1999a). At the time the RED was written, the avocado use appeared to be limited to Florida (EPA, 1999a). Although we searched the NPRS and PPLS systems (11/29/10 pdd), we were unable to locate a label for folpet to confirm this information.

Folpan Folpet Technical (Reg. No. 11678-18) is still listed as registered, with a label date of 08/02/04.

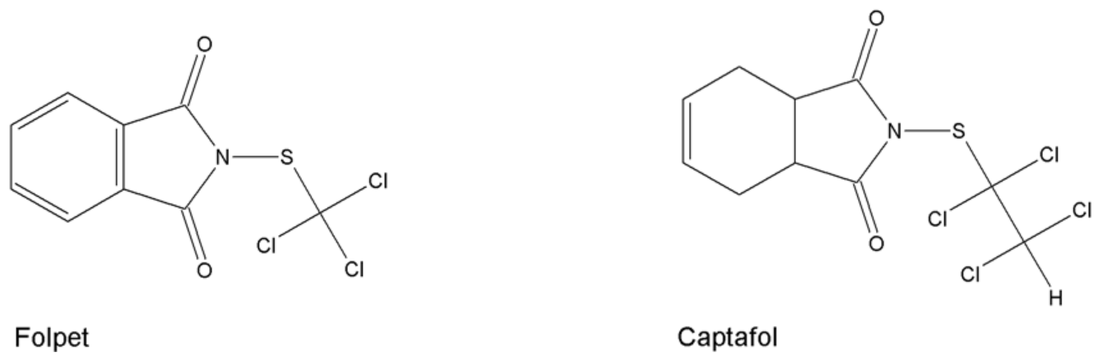


Figure 74. Structures of Folpet (left) and Captafol (right)

Bernard and Gordon (2000) evaluated both captan and folpet, concluding they had a common mechanism of action (Bernard & Gordon, 2000). Although they did not consider captafol, we believe it is reasonable to assume it also has the same mechanism, given structural similarities. Thus, in cases where folpet may be used in salmonid habitat, we assume additivity for captan and folpet. We also believe it is reasonable to consider data and information regarding the toxicological effects of both folpet and captafol in this assessment, while recognizing that the severity of effects will vary dependent on the specific a.i.

Parent captan produces two primary degradates, TCMT (the toxic moiety), and tetrahydrophthalimide (THPI), the phthalimide ring structure. Data presented by EPA in various assessments show THPI to be orders of magnitude less toxic than parent captan for all taxa, which is not unexpected given the loss of the thiochloromethyl group. We do not consider THPI further in this assessment, although we have noted toxicity data provided by EPA. However, the EPA documents (EPA, 2003a, 2004e, 2007c) do not provide toxicity data or environmental half-lives for either TCMT or thiophosgene, other than noting:

> "TCMT moiety degrades moderately rapidly to rapidly by aerobic soil metabolism to CO_2, thiophosgen, and organic sulfur and chlorine. Thiophosgene dissipation is expected to be dependent on volatilization (est. vapor pressure 29.7

mm Hg and estimated Henry's Law Constant 0.00586 atm-$m_3$mole^{-1})." (pg 38, (EPA, 2004e)).

We presume TCMT to be as toxic as parent captan, given it mediates the reaction but are unable to determine how long it may be present in the aquatic environment compared to the parent given the information provided. Thiophosgene is generally recognized to be toxic, primarily as a gas. We located no aquatic toxicity data for TCMT or thiophosgene.

Temperature and toxicity

We located no information indicating temperature specifically affects the toxicity of captan. However, we do note higher water temperatures can affect salmonids in two ways, regardless of specific chemical effects. Higher water temperatures will increase the metabolic rate for fish, thus increasing the rate at which they process the toxicant. Depending on the chemical, this may be either beneficial or detrimental. Higher than optimum water temperatures increases general physiological stress for salmonids, making them more susceptible to other stressors.

pH and toxicity

We located no information indicating pH specifically affects the toxicity of captan, although the pH does affect the hydrolysis rate. The hydrolysis rates increases with increasing pH.

Toxicity of Captan (Assessment Endpoints)

Although captan has been registered and used in commerce since 1951, it does not appear to have been heavily researched from a wildlife perspective, and we did not locate a large number of studies addressing aquatic effects. Both captan and folpet have been identified as probable human carcinogens (EPA, 2004e), although the captan classification was reduced to "not likely" in 2004 (EPA, 2004c). The metabolite and degradate thiophosgene has been identified as a probable common mechanism of action for the two a.i.s. There is a substantial body of literature evaluating carcinogenic and mutagenic effects of folpet and captan in small mammals, thus we find it reasonable to believe similar effects could occur in aquatic organisms.

Direct Effects to Salmonids

Survival

Twenty acute LC_{50} values for fish were presented in the RLF assessment (EPA, 2007c). This was the most complete set of data, and included the information presented in the salmonid BE (EPA, 2003a) and the captan RED (EPA, 2004e). Data tables detailing each end point are located in *Appendix 10*.

We divided the data into salmonids and other fish species. The salmonids group (n=11) includes 5 data points from species in the genus *Oncorhynchus*, the same genus as all of the listed salmonids addressed in this Opinion. Another 5 data points are from fish in the genus *Salvelinus*, and 1 is from a fish in the genus *Salmo*. Most are standard 96 h tests, although some were conducted in static systems, and others were conducted in flow-through systems. For the same species tested in both static and flow-through systems, the LC_{50s} from the flow-through tests are generally lower. Most data are compiled from two documents, which are compendiums of aquatic toxicity data rather than registrant-submitted studies (W. W. Johnson & Finley, 1980; Mayer & Ellersieck, 1986). No sublethal effects are described in any of the EPA documents. For salmonids, the range of LC_{50s}[7] was 26.2-137 μg/L, and the median was 56.5 μg/L. Under the qualitative toxicity classification scale used by EPA, captan is very highly toxic to salmonids on an acute basis. No probit slope data was available for any of these tests.

LC_{50} values were available for other fish (n=9). Three were for the bluegill sunfish (genus *Lepomis*), three were for fathead minnow (genus *Pimopheles*), and there was one each for channel catfish (*Ictalaurus punctatus*), yellow perch (*Perca flavescens*), and the harlequin fish (*Rasbora heteromorpha*). For these fish the range of LC_{50}s[8] was 65-310 μg/L, and the median was 134 μg/L. Under the qualitative toxicity classification scale used by EPA, captan is highly toxic to very highly toxic to these fish on an acute basis. One test for bluegill sunfish resulted in a probit slope of 1.17.

[7] Based on LC50 value, does not include 95% confidence interval.

[8] Based on LC50 value, does not include 95% confidence interval.

Standard 96 hr LC_{50} endpoints were available for the degradates THPI and THPAm, tested with rainbow trout (*Oncorhyncus mykiss*). Both endpoints were non-determinate[9]. The LC_{50} for THPI is >120,000 μg/L and the LC_{50} for THPAm is >126,000 μg/L. Under the qualitative toxicity classification scale used by EPA, these degradates of captan are practically non-toxic to fish on an acute basis. We do not carry them through the remainder of the analysis.

No toxicity data for formulations were presented in EPA documents. In the toxicity to freshwater fish section, the RED notes:

> "The Agency waived the acute formulated product testing with a 50% wettable powder (WP) formulation since the confidential statement of composition for captan 50WP and 80WP end use products showed that the major and minor inerts are not likely to enhance the toxicity of captan (EPA, 2004e)."

Reproduction and Growth

Reproduction and growth endpoints were evaluated in an fish early-life stage toxicity test (MRID 00057846). In this test, effects noted included reductions in adult and larval survival, growth, and overall larval-juvenile development. Other effects included a reduction in the number of eggs laid, and an inability for the juveniles that were exposed to reproduce. The NOAEC established by this test was 16.5 μg/L and the LOAEC was 39.5 μg/L (MRID 0057846). There was no additional discussion in any of the EPA documents regarding reproduction or growth endpoints for fish. In a 5 d static exposure of zebrafish embryos, captan did not affect survival or induce any developmental abnormalities at the concentrations tested (1-1,000 μg/L) (NMFS 2011 – included as Appendix 8).

[9] Only a > value is available, not a specific endpoint.

Carcinogenicity and Mutagenicity

EPA routinely requires carcinogenicity studies in rats and mice to support their human health assessments. There is no correlative assessment of cancer or mutagenicity in wildlife species, although some of the endpoints evaluated in the chronic wildlife assessments might arguably be considered to be expressions of such cellular-level changes (*e.g.*, failure to hatch, or reduced viability of young). In the small mammal studies, captan caused intestinal tumors in mice, and caused renal neoplasms and uterine sarcomas in rats (EPA, 2004e). This was attributed primarily to the thiophosgene metabolite.

One study reviewed evaluated genotoxic effects of captan on the South African clawed frog (*Xenopus laevis*) and the Spanish newt (*Pleurodeles waltl*) (Mouchet, Gauchet, Mailes, Ferriar, & Devaux, 2006). The larval organisms were exposed to captan in water, so the exposure pathway is more relevant to salmonids than dietary tests with mammals would be. Physiologically, the amphibians are also more similar to fish. Genotoxicity was evaluated via the micronucleus test (MNT) and the comet test (CA), both of which have been used previously to demonstrate DNA damage in both fish and amphibians exposed to xenobiotics (Mouchet, et al., 2006). Captan concentrations tested ranged from 15.6 – 2,000 μg/L in both a mineral water (MW, low ionic strength) and reconstituted water (RW, with minerals added, higher ionic strength). The mineral water is similar to water used in most OPPTS guideline tests. Concentrations provided are nominal. Exposure duration was 12 days.

At captan concentrations of ≥500 μg/L, mortality of both test species in both MW and RW solutions was 50% or higher. The EPA reviewer calculated LC50s based on data in the paper (EPA, 2007c). LC50s were 119.4 μg/L(MW) and 353.6 μg/L (RW) for the frog, and 311.1 μg/L (MW) and 500 μg/L (RW) for the newt. These ranges are similar to fish LC_{50}s.

Cardial hepatocytes from the larvae exposed to 15.6 μg/L, 31.25 μg/L, 62.5 μg/L and 125 μg/L of captan in MW were sampled at 1, 2, 4, 8, and 12 days and assayed using MNT and CA (Mouchet, et al., 2006). MNT demonstrated a genotoxic response in the frog at 62.5 μg/L, but not at the lower concentrations. Based on the MNT there was no response at any concentration for the newt. MNT detects cytogenetic damage. For the frog, CA showed genotoxic effects at concentrations ≥ 15.6 μg/L. For the newt, CA showed genotoxic effects at 62.5 μg/L and 125μg/L. Effects appeared as early as 1 day after the start of exposure, although extent of effects varied through the course of the experiment. CA indicates primary DNA damage, and authors note that some DNA lesions could "give rise to fixed mutations."

Based on this study, we conclude that even short-term exposure to captan in aquatic systems may cause cellular and/or DNA damage to salmonids. We note that some of this damage may be repaired by normal cellular damage repair mechanisms, but also that there is potential for some of these changes to cause heritable mutations. We also recognize effects may occur at different (higher or lower) concentrations given phylogenic differences between the test organisms and salmon.

Indirect Effects to Salmonids (Prey and Habitat Modifications)

Aquatic Invertebrates (Acute and Chronic Toxicity)

EPA documents provided standard toxicity test results (both acute and chronic) for *Daphnia magna*, a water flea. *D. magna* is a standard test organism, and for many chemicals, is one of the more sensitive aquatic invertebrates tested. However, when tests are available for high-quality salmonid prey like mayflies or stoneflies, they are frequently more sensitive than *D. magna* for many chemicals. Based on the guideline tests, the EC_{50} for *D. magna* is 8,400 μg/L (95% CI, 7,060-9,960 μg/L, slope 1.19) (MRID GS0120041, (EPA, 2007c)). MRID GS0120041 is a 1979 study conducted by the Laboratory of Terrestrial and Aquatic Biology in Beltsville, MD, and may not have been conducted in accordance with current guideline test protocols. Based on these test results, captan is considered moderately toxic to aquatic invertebrates on an acute basis using EPA's qualitative scale. The salmonid BE (EPA, 2003a) and the RED (EPA, 2004e), report some additional tests for *D. magna*. Two of these tests did not determine the EC_{50}, as it was greater than the concentrations tested. A third reported a 1,300 μg/L EC_{50} (MRID 00002875). It was not reported in the RLF assessment, presumably because the test duration was only 26 h (EPA, 2004e).

Effects noted in the chronic test included reductions in parental and juvenile growth, decreased survival for both parents and juveniles, and a reduction in neonates produced. Based on this test, the NOAEC for *D. magna* is 560 μg/L and the LOAEC is 1,000 μg/L (MRID 44148801, (EPA, 2007c)). The assessment notes risk may be underestimated based on these values, because concentrations used were nominal, not measured and the test material was reported as "unstable in water". NMFS concurs with this evaluation. The assessment also provided an EC50 of > 113,000 μg/L for THPI with *D. magna* as the test organism.

Aquatic Plants (Phytoplankton and Vascular Plants)

Standard toxicity test data for 3 species of phytoplankton and one species of aquatic vascular plants were provided in the most recent EPA assessment (EPA, 2007c). The RLF document does not report specifically which endpoints were affected in the plant tests, nor does the RED. Aquatic plant tests are not reported in the salmonid BE. The 3 phytoplankton species tested were *Scenedesmus subspicatus* (a green alga), *Selenastrum capricornutum* (a green alga), and *Anabaena flos-aque* (a blue-green alga). EC_{50}s for these phytoplankton were 320 μg/L, 1,770 μg/L, and 1,200 μg/L, respectively. All tests were conducted with ≥90% a.i. The vascular plant tested was *Lemna gibba* (duckweed), and EC_{50} for this test was >12,700 μg/L. The RLF document notes the data for *Scenedesmus subspicatus* may underestimate risk, as concentrations were nominal, not measured, and the EC_{50} may in fact be lower. *Selenastrum capricornutum* was also tested with the degradate THPI, and the EC_{50} was >180,000 μg/L. Based on a statement in the RLF assessment (Appendix A, footnote, (EPA, 2007c)) marine/estuarine algal species appear to have been tested (MRID 40228401) but that information was not provided in any of the documents.

NMFS has not located any additional literature documenting aquatic plant endpoints.

Microbial Community Effects (Sediment, Soil, and Water Column)

Although EPA does not typically evaluate the effects of registered a.i.s on microbial processes in water, soil, or sediment, NMFS believes these are a relevant endpoint to consider for fungicides, which are targeted at organisms such as mycorrhizae, which serve important functions in soil.

We located one study evaluating the effects of captan on microbial community respiration, microbial community biomass, bacterial activity, and denitrification (Widenfalk, Svensson, & Goedkoop, 2004). The experiments were conducted in microcosms. Captan was tested at 0.013 μg/kg dwt sed, 1.3 μg/kg dwt sed, and 130 μg/kg dwt sed. Test concentrations were derived from a Maximum Permissible Concentration (MPC) standard of 0.11 μg/L (Crommentuijn, Sijm, de Bruijn, van Leeuwen, & van de Plassche, 2000) using equilibrium partitioning theory. Of the 4 parameters tested, captan affected only bacterial activity. The response was not dose-dependent, as a significant difference from controls was only noted for the 1.3 μg/kg dwt sed test concentration. This particular test evaluated only an 8 h interval, although some of the other parameters were tested for longer periods of time. Based on the data provided, it cannot be determined how long the effects might last past the test period. We note that given the reducing nature and presence of sulfur compounds in most natural sediment, it seems unlikely that captan's sulfur-containing degradates would remain active for very long.

We located no studies evaluating effects of captan on the water column microbial community .

Riparian Vegetation

No terrestrial plant guideline data are available to evaluate effects on terrestrial plants, as EPA waived those studies (EPA, 2004e). In the RLF assessment, EPA notes a literature search resulted in some data regarding captan use as a seed treatment on germination of the test species. We do not believe this type of application is particularly relevant for riparian vegetation, and do not repeat it here. They also noted most papers they located describing effects of captan used as a foliar spray did not describe “any lasting phytotoxic effects” on the plants. One study did have an lb a.i./A rate associated with temporary phytotoxic effects. This study, on highbush blueberries, found an application of captan end-use products (Captan 80WP and Captec 4L) at a rate of 2.4 lb a.i./A, caused mild phytotoxicity, although bushes recovered by harvest (EPA, 2007c). Assessment endpoints and associated concentrations for captan are summarized in Table 127.

Table 127. Assessment Endpoints and Measures for Captan

Assessment Endpoint		Assessment Measure	Median Concentration[1] (μg/L)	Range	n
Direct Effects on Salmonids	Survival	Salmonid Acute LC_{50}[2] Other Fish Acute LC_{50}[2]	56.5 134	26.2-137 65-310	11 9
	Growth	Chronic NOAEC[2] Chronic LOAEC[2]	16.5 39.5	NA	1
	Reproduction	Chronic NOAEC[2] Chronic LOAEC[2]	16.5 39.5	NA	1
	Swimming	We located no data regarding effects of captan on swimming or olfaction.			
	Olfaction				
	Genotoxicity	DNA Change NOAEC DNA Change LOAEC (Mouchet et al 2006)[3]	23.4 46.9	15.6-31.3 31.3-46.9	2
Effects on Prey (Aquatic Invertebrates)	Survival	Acute Invert EC_{50}[2]	8,400	NA	1
	Growth	Chronic NOAEC[2] Chronic LOAEC[2]	560 1,000	NA	1
	Reproduction	Chronic NOAEC[2] Chronic LOAEC[2]	560 1,000	NA	1
	Abundance (?)	Meso/microcosm Data, if available and applicable	None located as of 12/6/10		
Effects on Primary Productivity	Biomass & Abundance	Algal EC_{50}[2]	1,200	320-1,770	3
Effects on Submerged and Emergent Vegetation	Biomass & Abundance	Vascular Plant EC_{50}[2]	>12,700	NA	1
Effects on Riparian Vegetation	Biomass & Abundance	Terrestrial Plant Phytotoxicity (Polavarapu 2000)	2.4 lb a.i./A	NA	NA

[1]If more than one value was available. If only one value was available, the actual number is given.
[2]From guideline/standardized test data presented in EPA documents (primary source most recent BE on RLF)
[3]Data from amphibian species, water exposure, NOAEC and LOAEC based on 1 d exposure
NA Not applicable, only one value available

Chlorothalonil Mode of Action

Chlorothalonil is classed as an aromatic fungicide. There are a number of other chemicals in this class, including some that also have chloro- and/or nitro- sidechains on the benzene ring, but chlorothalonil is unique in having the triple-bonded nitrogen atoms on the benzene (Figure 75).

Chlorothalonil interacts with sulfhydryls via substitution of the (R-S-) group at the chlorine atom in the 4 or 6 position on the ring (Figure 76) (Long & Siegel, 1975). "The fungitoxic action of chlorothalonil has been attributed to its ability to deplete cellular glutathione reserves (Tillman et al, 1973 as cited in (P.E. Davies, 1985a))." Following depletion of those reserves, it can then interact with sulfhydryl groups in other proteins or enzymes, interfering with cellular processes. In fish, Davies and White (1985) found chlorothalonil interacted with glutathione in a range of tissues, including internal organs, gills, muscle and blood, but that the primary site of accumulation was in the gall bladder and bile (P.E. Davies & White, 1985).

Glutathione, a tripeptide consisting of glutamic acid, cysteine, and glycine, is present at high concentrations in all animal tissues ((Lehninger, 1975), pg 795). It is involved in the amino acid transport cycle ((Lehninger, 1975), pg 795), and is also involved in phase II xenobiotic transformation in the liver ((Parkinson, 1996), pg 177). Glutathione protects cells by binding electrophiles, and reducing the effects of oxidizing agents on other cellular components ((Parkinson, 1996), pg 189). Gluthione conjugations may occur enzymatically (catalyzed by glutathione-*S*-transferase, or nonenzymatically ((Parkinson, 1996), pg 177). Glutathione-*S*-transferase is inducible ((Parkinson, 1996), pg 177, (P.E. Davies, 1985b)). It was noted there was induction of hepatic glutathione and glutathione-*S*-transferase at sublethal concentrations of chlorotahlonil (10 μg/L) in rainbow trout, but not at lethal (30 μg/L) concentrations (P.E. Davies, 1985b). Dröge and Breitkreutz (2000) discuss the importance of cellular glutathione in immune response, noting that levels too low or too high are associated with autoimmune diseases like human immunodeficiency virus (HIV). Also, it was clearly shown that chlorothalonil reacts first with cellular glutathione, and then reacts with glyceraldehyde 3-phosphate dehydrogenase (GADPH) (Tillman, Siegel, & Long, 1973). GAPDH is an important component of the glycolysis cycle, and also protects cells against oxidative stress. Thus chlorothalonil's depletion of glutathione and GAPDH contributes to suppression of the organism's immune response.

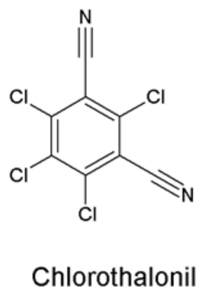


Figure 75. Structure of Chlorothalonil

Figure 76. Chlorothalonil substitutions at 4 and 6 position (-SH represents any –S R group)

Comparison of Captan and Chlorothalonil Modes of Action

Long and Siegal (1975) compare the modes of action for chlorothalonil (isophthalonitrile) with that of captan and folpet (trichloromethyl sulfenyls) and note in their discussion (p 392-393):

> "Both the isophthalonitrile and trichloromethyl sulfenyl fungicides are considered to act as general or non-specific toxicants. These fungicides have certain other similar reaction characteristics. They react with low and high molecular weight cellular thiols with toxicity ultimately residing with the inactivation of thiol containing proteins. . . There are, however, major differences in reactivity between these groups of fungicides. They differ in their rate of reaction,

> complexity of reaction, and the type of binding sites involved in the reaction. While the mode of action of chlorothalonil resembles that of the trichloromethyl sulfenyl fungicides in certain general characteristics, pronounced differences can be noted in their chemical mechanism of reaction with cellular constituents (Long & Siegel, 1975)."

Based on their analysis, although captan and chlorothalonil both interact with sulfhydryl groups, we believe it is inappropriate to consider it additivity. We do note, however, that based on mode of action for both of these a.i.s, it is likely organisms exposed to one or both will be more susceptible to some types of other toxicants, especially those which cause oxidizing reactions.

Temperature and toxicity

Davies and White (1985) varied the water temperature from 10°C -16°C in the studies they performed on rainbow trout. In these experiments, LC_{50s} did decrease as temperature increased (P.E. Davies & White, 1985). Three trials at 10°C, 14°C, and 16°C produced 96-hr LC_{50}s of 18.0 μg/L, 17.1 μg/L, and 10.5 μg/L, respectively. However, the test conducted at 16°C also had lower DO. Additionally, two of the tests (at 14°C and 16°C) were conducted as flow-through, and the 10°C test was static, which can cause variability in results between the tests. No measure of variability (95% CI) was provided for the LC_{50}s in the paper, so we cannot determine if these temperature-toxicity differences are significant.

We also note higher water temperatures can affect salmonids in two ways, regardless of specific chemical effects. Higher water temperatures will increase the metabolic rate for fish, thus increasing the rate at which they process the toxicant. Depending on the chemical, this may be either beneficial or detrimental. Higher than optimum water temperatures increases general physiological stress for salmonids, making them more susceptible to other stressors.

pH and toxicity

We located no information indicating pH specifically affects the toxicity of chlorothalonil.

Toxicity of Chlorothalonil (Assessment Endpoints)

A number of LC_{50} values for a range of fish species were available for chlorothalonil. Additionally, studies by Davies ((P.E. Davies, 1985a, 1985b, 1985c) and Davies and White (P.E. Davies & White, 1985) provide significant insight into the biochemistry and metabolism of chlorothalonil in fish including rainbow trout. Other researchers (Baier-Anderson & Anderson, 1998) have also investigated effects at a cellular level. Tierney (Tierney, et al., 2006), considered the effects of chlorothalonil on salmon olfaction.

Direct Effects to Salmonids

Survival

In Davies and White (P.E. Davies & White, 1985), authors made behavioral observations in addition to determining LC_{50s}. Dependent on temperature and dissolved oxygen (DO) concentrations, 96 hr LC_{50s} for *Oncorhynchus mykiss* (referred to by its older Latin name of *Salmo gairdneri*), ranged from 10.5 μg/L to 18.0 μg/L. All temperature and DO variations were in acceptable (although not optimal) ranges for salmonids. They noted all fish were lethargic when exposed to chlorothalonil. At concentrations below 8.7 μg/L *O. mykiss* exhibited normal startle reactions. However, at higher concentrations they observed the following.

> "In *S. gairdneri*, loss of startle reaction was followed by a reduction of activity and station holding at the surface or bottom. A gradual bronzing of skin colour occurred especially in facial patches and around the lateral-line. Reddening at fin bases was also observed. Dead fish were a pronounced bronze colour. No fin collapse occurred [although it did for *Galaxias* species tested]."

To develop the LC_{50} median and range for fish we took the data from the EPA RLF assessment (EPA, 2007b), which contains the most recent comprehensive analysis of aquatic chlorothalonil toxicity data, and cross-referenced it with the salmonid BE (EPA, 2003b) and the review document provided by the applicant (Brain, Perkins, & Bang, 2010).

There was a range of test durations, beginning at 24 h and extending to 10 d. In several cases, data sets from individual studies included measurements at multiple time intervals. In most cases, longer exposure times produced lower LC_{50s}. The 10 d LC_{50s} were indeterminate, so it is difficult to tell precisely how they relate to the rest of the tests, which are typically 24-96 h. To ensure comparability across the data, we used the 96 h tests to calculate median and range for salmonids and other fish.

We divided the data into salmonids and other fish species. The salmonids group (n=5) includes only the genus *Oncorhynchus*, the same genus as all of the listed salmonids addressed in this Opinion. All are for *Oncorhynchus mykiss*, the rainbow trout. For salmonids, the range of LC_{50s} was 18-42 μg/L, and the median was 18 μg/L. Under the qualitative toxicity classification scale used by EPA, chlorothalonil is very highly toxic to salmonids on an acute basis. No probit slope data was available for any of these tests.

LC_{50} values were available for other fish (n=9). Three were for fish of the genus *Galaxias,* a trout species common in Australia and New Zealand. Although not closely related to the Pacific salmonids considered in this opinion, it is a cool-water fish, and occupies a similar ecological niche. Some of the rainbow trout data is from the same study (P.E. Davies & White, 1985). For *Galaxias* species, the range of LC_{50s} was 16.3-29.2 μg/L, and the median was 18.9 μg/L, indicating they are similar in sensitivity to salmonids in regards to chlorothalonil.

In a recent study looking at zebrafish mortality, developmental abnormalities, and body length, embryos exposed to increasing concentrations of chlorothalonil had increased mortality (NMFS 2011 – included as *Appendix 8*). At 0.1 mg/L, 75.6% of the zebrafish had died, and mortality was 100% at 1.0 mg/L

Additionally, there were three 96 h LC_{50s} for the bluegill sunfish (genus *Lepomis*), and one each for fathead minnow (*Pieopheles promelas*), channel catfish (*Ictalurus punctatus*), and the threespine stickleback (*Gasterosteus aculeatus).* For these fish the range of LC_{50s} was 23-84 μg/L, and the median was 55.5 μg/L. Under the qualitative toxicity classification scale used by EPA, chlorothalonil is highly toxic to very highly toxic to these fish on an acute basis.

Degradates

EPA documentation (EPA, 2007b) included toxicity data for the 4-hydroxy chlorothalonil degradate, although only acute LC_{50} data for bluegill sunfish. These two studies (MRID 0029415 and MRID 00030393) resulted in values of 45,000 μg/L and 15,000 μg/L, respectively (median 30,000 μg/L). Based on results from the a.i. and the general observation that salmonids are typically more sensitive to pesticides, we estimated an LC_{50} for salmonids based on the available data. Using median values of available data for bluegill only (30,000 μg/L for degradate (n=2) and 60 μg/L for a.i. (n=3)), the degradate appears to be 500x less toxic than the a.i. Applying the 500x factor to the median LC_{50} for salmonids (18 μg/L, n=5) results in an estimated LC_{50} of 9,000 μg/L for salmonids. Acute toxicity data (EC_{50}s) were also available for the aquatic invertebrate *D. magna* for both the a.i. and the degradate. Two guideline studies were available for the a.i., MRID 00068754 and MRID 45710221, with EC_{50}s of 68 μg/L and 54 μg/L respectively, resulting in a median of 61 μg/L. Comparing this to the 26,000 μg/L EC_{50} of the degradate (MRID 0030394) results in a similar ratio of 426. Using EPA's qualitative scale, the 4-hydroxy degradate is classified as slightly toxic to fish and aquatic invertebrates based on available data.

Based on available data, the 4-hydoxy degradate seems less toxic to aquatic organisms on an acute basis. However, there are few data available to evaluate acute effects, and we located no data to evaluate chronic effects, either in EPA documentation or the open literature. Thus we considered the mechanism of the glutathione interaction with the chlorothalonil molecule to provide additional insight into potential toxicity of 4-hydroxy chlorothalonil. Based on the work of Long and Siegal (Long & Siegel, 1975), chlorothalonil interaction with glutathione produces 2 derivatives in fungal cells. The sulfur atom of the cysteine subunit of glutathione substitutes at the 4- or 6- position on the benzene ring of chlorothalonil. Davies (P.E. Davies, 1985a) further characterized the conjugates in rainbow trout, noting that these conjugates were mono-, or di-conjugates, with the first substitution occurring at the 4- site, and the second substitution at the 6-site. He also identified a less commonly occurring tri-conjugate, with a substitution at the 2- site. Long and Siegal (Long & Siegel, 1975) noted the glutathione reaction proceeded quickly in fungal cells, indicating parent chlorothalonil is highly reactive, especially at the 4- site. Hydroxyl (-OH groups) are strong electron donors to the ring structure, and chlorines (-Cl) are strong electron withdrawing groups from the ring. Thus the substitution of the –OH at the 4- site would also change the reactivity of the secondary conjugation site. Based on this analysis, the available data are likely a good predictor of the toxicity of the 4-hydroxy degradate compared to the parent.

No data were located regarding the toxicity of the diamide chlorothalonil degradate or the 1-amide 4-hydroxy chlorothalonil degradate. Based on the structural analysis, the 1-amide 4-hydoxy degradate is likely similar in toxicity to the 4-hydroxy degradate, as the more reactive 4- site has again been substituted. The diamide substitution would behave in a fashion similar to parent chlorothalonil, although perhaps less strongly.

No data were available regarding potential chronic effects of the 4- hydroxyl or other degradates, however we elected to make an estimate using an acute-to-chronic ratio (ACR) for fathead minnow, and applying the degradate factor derived from the bluegill. Based on the LC_{50} of 23 μg/L and the NOAEC of 3 μg/L for the fathead minnow (EPA, 2007b), the calculated ACR is 7.67 (unitless). Applying this to the salmonid median LC_{50} results in a estimated chronic NOAEC of 2.4 μg/L for salmonids. Then multiplying it by the fish degradate factor of 500x, the anticipated chronic toxicity of the 4-hydroxy degradate would be 1,174 μg/L.

Thus, although fate properties indicate the 4-hydroxy degradate is more persistent than parent chlorothalonil, we anticipate it would have to occur in concentrations of ~1,000 μg/L to manifest effects. Based on data we located, we are unable to estimate toxic concentrations of the diamide degradate, although a conservative assumption is toxicity equivalent to the parent.

Formulations

EPA included some test data for formulations of Bravo in the both the salmonid assessment (EPA, 2003b) and the RLF assessment (EPA, 2007b). Because these data are more clearly delineated in the RLF assessment, we have opted to use them from that source. Only acute toxicity data were provided, and only the Bravo formulations were tested. Some studies are from the early 1970s, and tested a 75% a.i. product listed as Bravo TM W-75 (MRIDs 0087304, 0087303, and 00087258). Only two are 96 h tests, and we do not present data from the 48 h test (MRID 0087304). For rainbow trout, the formulation LC_{50} was 103 μg/L (MRID 0087303). When expressed in μg/L a.i., it is given as 77.2 μg/L a.i. For bluegill, the formulation LC_{50} was 167 μg/L (MRID 0087258). When expressed in μg/L a.i., it is given as 125 μg/L a.i. More recent studies (~1990s) on the formulation Bravo-720 were also available, and test species included rainbow trout (MRID 43302101), bluegill (MRID 42433804), and *D. magna* (MRID 42433806). For rainbow trout, the formulation LC_{50} was 61 μg/L and when expressed in μg/L a.i., it is given as 33.2 μg/L a.i. For bluegill, the formulation LC_{50} was 49 μg/L and when expressed in μg/L a.i., it is given as 26.2 μg/L a.i. For *D. magna*, the formulation LC_{50} was 180 μg/L and when expressed in μg/L a.i., it is given as 97 μg/L a.i. No chronic tests were reported.

Based on the Bravo data, using the toxicity values from TGAI tests would be protective for those formulations when assessing acute lethality. One applicant provided acute toxicity data for a number of additional formulations (discussed under *Mixtures*, in the *Risk Characterization*). Analysis of the additional data also indicates that using a value for the TGAI would be protective for formulation currently marketed by that applicant. We note formulations may change over time and for specific uses.

Reproduction and Growth

Other than a somatic growth measurement in Davies et al. (P. E. Davies, Cook, & Goenarso, 1994), we located no further information on reproduction or growth except the guideline studies reported in EPA documentation. Davies et al. (P. E. Davies, et al., 1994) do not report any values for growth, thus we assume there was no significant effect compared to the controls. One early life-stage test, on fathead minnow, was submitted ((EPA, 2007b), MRID 00030391). Endpoints affected were hatching success, with reduction of ~10% compared to controls at 6.5 μg/L, and survivability of F_0 generation, with a reduction of ~85% compared to controls at 16 μg/L. At 16 μg/L, hatching success was reduced by ~44%. Based on this test, the NOAEC was 3.0 μg/L, and the LOAEC was 6.5 μg/L.

In a 5 d static exposure study on zebrafish embryos, chlorothalonil produced increased mortality in the embryos as compared to the controls at 10 μg/L, although at this concentration it was not statistically significant ((NMFS, 2011) – Included as *Appendix 8.)* At 100 μg/L, mortality of the embryos was 76%, and at 1,000 μg/L, mortality was 100%. At the 100 μg/L test concentration, >90% of the surviving embryos exhibited fin deformities, and extensive erosion of skin tissue. These values correspond with existing chronic data ((EPA, 2007b), MRID 00030391). Existing endpoints are slightly lower, and we have used them as the assessment endpoint.

Olfaction

Tierney et al. (2006), evaluated the effects chlorothalonil, along with endosulfan, glyphosate acid, iodocarb (IPBC), trifluralin, and 2,4-D on juvenile coho salmon electro-olfactograms (EOGs). Chlorothalonil was tested at 1 mg/L (1,000 μg/L), which is actually above the water solubility of the compound. Authors provide a water solubility value of 0.6 mg/L, but EPA documentation lists it as 0.8 mg/L (EPA, 2007b). Acetone was used as a solvent to achieve the test concentration. Reported concentrations are nominal, but authors note solutions were prepared daily and report no precipitation. At the test concentration of 1 mg/L, chlorothalonil did not modify the EOG amplitude.

Authors state test concentration is “well above the 96 h LC_{50} range of 14.3 to 17.1 μg/L, so chronic effects or other sublethal effects are likely to develop before olfactory toxicity.”

Although the specific mechanism of salmon olfactory inhibition has not been well established, authors (Tierney, et al., 2006) do state:

> “Most pesticides known to inhibit salmonid olfaction such as the organophosphorus insecticide diazinon, atrazine, and the carbamate fungicide IPBC have acetylcholinesterase-inhibiting properties. . . Considering that elevated peripheral AChE may lead to increased mucous production, it is not unreasonable to hypothesize that olfaction could be diminished by an anticholinesterase mucous plug, which is exactly what Jarrad, et al. (Jarrard, Delaney, & Kennedy, 2004) hypothesized.”

Based on the hypothesized mechanism, and the existing data, we do not consider olfactory effects of chlorothalonil a significant issue for salmon.

Biochemical and Immunological Effects

Although EPA routinely requires carcinogenicity and developmental studies in rats and mice to support their human health assessments, they do not regularly consider immunological effects for either human or wildlife risk assessments. Generally, for wildlife, the guideline chronic tests evaluating reproduction (a sensitive endpoint) are thought to provide reliable estimates of no effect concentrations for other endpoints, such as immunological effects. To our knowledge, this assumption has not been empirically tested, but may be appropriate in some instances for some chemicals. However, it may not be appropriate for other cases, especially where effects appear only after prolonged exposure or during a recovery period. Perturbation of metabolic cascades can have far-reaching consequences, including lowered resistance to pathogens, increased susceptibility to other chemical toxicants, or simply a diverting of energetic resources that might otherwise be used for growth or reproduction. Linkage of such effects at a cellular and biochemical level to effects on whole organisms is difficult, and is still an evolving science. Some evidence of such effects from chlorothalonil exists in open literature, and we present it here, although at this point we are unable to quantitatively link it to organism or population level effects.

Several different research groups have considered sublethal effects. Some of these studies evaluated the mechanism *in vitro*, but others exposed whole organisms to chlorothalonil. Baier-Anderson and Anderson (1998) used an exposure of striped bass (*Morone saxatilis*) phagocytes to chlorothalonil to investigate effects on cell viability, phagocytosis, production of reactive oxygen species (ROS) and production of NADPH. Impairment of these cellular and metabolic processes increases susceptibility to infections (Baier-Anderson & Anderson, 1998). They concluded chlorothalonil impairs ROS and NADPH production in the cells at ≥ 250 μg/L. As this is an exposure, we note there is no clear correlation to actual water concentrations that might lead to this level of cellular exposure.

Davies et al. (P. E. Davies, et al., 1994) exposed live fish (*O. mykiss*, *G.maculatus*, and *Pseudaphritis urvilli*) to concentrations of chlorothalonil ranging from 0.3-8.2 μg/L for 10 d. Authors measured a number of parameters. Oxygen consumption (an indicator of stress) for *P.urvilli* increased significantly at 0.3 μg/L and for *O. mykiss* at 1.4 μg/L. Increases in liver glutathione and glutathione-*S*-transferase were noted for *O. mykiss* at 1.4 μg/L and for *P.urvilli* at 8.2 μg/L. DO consumption and glutathione increases occurred in a dose-dependent manner. *P.urvilli* also showed a significant decrease in leucocrit at 4.4 μg/L, but this effect did not appear to occur in a dose-dependent manner. Based on their evaluation, they considered “consistent” indicators to show a NOAEC of 0.8 μg/L, and a LOAEC of 1.4 μg/L for sublethal effects. Effects on glutathione and glutathione-*S*-transferase liver enzymes would likely occur prior to effects on immunological endpoints.

Shelley et al. (Shelley, Balfry, Ross, & Kennedy, 2009), evaluated effects of chlorothalonil on the immune responses of rainbow trout. They included a 28 d exposure period and a 14 d recovery period in their experiment, and exposed the fish at concentrations of 0.1 μg/L, 0.2 μg/L, 0.5 μg/L, and 1.0 μg/L. In addition to measuring biochemical immune response parameters, they also challenged the exposed fish with a marine pathogen, *Listonella anguillarum* during the recovery period. Evaluations of the fish during exposure showed no significant difference in parameters measured (lymphocyte and granulocyte counts, respiratory burst, number of phagocytic cells, and phagocytic capacity) during the exposure period. However, they did see significant effects on respiratory burst (0.1 μg/L) and percent phagocytic cells (0.1 μg/L, 0.2 μg/L) during the recovery period. Chlorothalonil appeared to have no significant effect on susceptibility to the pathogen. The authors concluded "Our demonstration of immunomodulation following exposure to environmentally relevant concentrations of chlorothalonil and pentachlorophenol suggests the potential sensitivity of the immune system to the effects of these CUPs [current use pesticides]." Additionally, they illustrated the concern for lingering effects on the organism after the stressor is removed (Shelley, et al., 2009).

Overall, evidence for immunocompromise is inconclusive, although we do note that sublethal effects have been documented in *Oncorhyncus* species at concentrations below the chronic NOAEC (3.0 μg/L) for fish based on guideline studies submitted to EPA by the registrants.

Indirect Effects to Salmonids (Prey and Habitat Modifications)

Aquatic Invertebrates (Acute and Chronic Toxicity)

EPA described guideline test data for aquatic invertebrates in both the salmonid assessment (EPA, 2003b) and the RLF assessment (EPA, 2007b). Because these data are more clearly delineated in the RLF assessment, we have opted to use them from that source. Two studies evaluated the acute EC_{50}s for *D. magna*. EC_{50} values were 68 μg/L (MRID 00068754) and 54 μg/L (MRID 45710221), resulting in a median of EC_{50} of 61μg/L. Two chronic studies were also conducted, with one reporting a NOAEC of 0.6 μg/L and a LOAEC of 1.8 μg/L (MRID 45710222). Another reported a NOAEC of 39 μg/L and a LOAEC of 79 μg/L (MRID 45710222). The NOAEC of 0.6 μg/L and LOAEC of 1.8 μg/L (MRID 45710222) are more consistent with the fish data (NOAEC of 3.0 μg/L, MRID 00030391), and the fact that *D. magna* are generally more sensitive to aquatic toxicants than fish. Although EPA noted (EPA, 2007b) that there was some uncertainty in the measured concentrations of chlorothalonil in the study, values derived from it were used as endpoints in their risk assessment. Given that the LOAEC (79 μg/L) established in the study producing the higher value (MRID 45710222) is actually greater than the EC_{50} of 61μg/L established by the acute tests, we have opted to disregard the higher values in evaluating chronic effects to aquatic invertebrates.

Davies et al. (1994), evaluated mortality in four species of Australian aquatic invertebrates: the giant Tasmanian freshwater crayfish (*Astacopsis gouldi*), the freshwater aytid shrimp (*Partaya australiensis*), an amphipod (*Neoniphargus sp. A*), and an isopod (*Colubotelson chiltoni minor*). Authors calculated EC_{50}s for all species following exposure durations of 4 d and 7 d, which is longer than the typical 48 h (2 d) acute tests for *D. magna*, but not as long as the chronic (21 d) tests. They were unable to calculate a determinate EC_{50} for the isopod and amphipod, giving it only as >40 μg/L for both species. However, they were able to determine a 4 d and 7 d for the crayfish (16.0 μg/L and 10.9 μg/L, respectively) and the shrimp (12.0 μg/L and 3.6 μg/L), respectively. Authors also measured sublethal endpoints, such as whole body glutathione and glutathione-S-transferase concentrations/activities and AChE inhibitions. Significant changes in glutathione were observed at 0.3 μg/L for the crayfish, and in both glutathione and glutathione-S-transferase at 1.8 3 μg/L for the crayfish. Thus the NOAEC for glutathione (the most sensitive endpoint) for the crayfish was not established, but is something < 3 μg/L.

In the Davies et al. study (1994), the authors also considered AChE inhibition, likely because they were also evaluating organophosphates. They did see significant inhibition in tail muscle of the crayfish at a chlorothalonil concentration of 6.7 μg/L. The declining inhibition trend continued in the next highest test concentration (17.5 μg/L) and there was mortality at the highest concentration. We have located no other discussion in literature regarding effects of chlorothalonil on AChE.

Aquatic Plants (Phytoplankton and Vascular Plants)

EPA reported aquatic plant data from both guideline studies and open literature in both the salmonid assessment (EPA, 2003b) and the RLF assessment (EPA, 2007b). Because these data are more clearly delineated in the RLF assessment, we have opted to use them from that source.

Guideline tests for freshwater phytoplankton included one for *Selenastrum capricornutum*, a green alga, and *Navicula pelliculosa*, a freshwater diatom. The EC_{50} and NOAEC for *S. capricornutum*were 190 μg/L and 50 μg/L, respectively. Typically, the green alga is the most sensitive of the phytoplankton species tested, but for chlorothalonil, the diatom was more sensitive. The EC_{50} and NOAEC for *N. pelliculosa* were 14 μg/L and 3.9 μg/L, respectively. The only vascular plant data reported is for duckweed (*Lemna gibba*). The NOAEC was 290 μg/L and the LOAEC was 630 μg/L based on a reduction in biomass (dry weight).

Microbial Community Effects (Sediment, Soil, and Water Column)

Although EPA does not typically evaluate the effects of registered a.i.s on microbial processes in water, soil, or sediment, NMFS believes these are a relevant endpoint to consider for fungicides, which are targeted at organisms such as mycorrhizae, which serve important functions in soil. We located no relevant studies on these endpoints

Riparian Vegetation

EPA reported terrestrial plant data from both guideline studies and open literature in the RLF assessment (EPA, 2007b). They located no additional applicable plant information in the open literature. The seed germination/seedling emergence NOAEL and vegetative vigor NOAEL are both 16 lb a.i./A. Assessment endpoints and associated concentrations for chlorothalonil are summarized in Table 128 below.

Table 128. Assessment Endpoints and Measures for Chlorothalonil

Assessment Endpoint		Assessment Measure	Median Concentration[1] (µg/L)	Range	n
Direct Effects on Salmonids	Survival	Salmonid Acute LC_{50}[2] Galaxias Acute LC_{50}[2] Other Fish Acute LC_{50}[2]	18 18.9 55.5	10.5-42.3 16.3-29.2 23-84	5 3 6
	Growth	Chronic NOAEC[2] Chronic LOAEC[2]	No values reported		
	Reproduction	Chronic NOAEC[2] Chronic LOAEC[2]	3.0 6.5	NA	1
	Swimming	We located no data regarding effects of chlorothalonil on swimming.			
	Olfaction	EOG NOAEC (Tierney et al 2006)	>1,000	NA	1
	Immune Response	GSH Inhibition NOAEC GSH Inhibition LOAEC (Davies et al 1994)	0.8 1.4	NA	NA
Effects on Prey (Aquatic Invertebrates)	Survival	Acute Invert EC_{50}[2]	61	54-68	2
	Growth	Chronic NOAEC[2] Chronic LOAEC[2]	No values reported		
	Reproduction	Chronic NOAEC[2] Chronic LOAEC[2]	0.3 1.8	NA	1[3]
Effects on Primary Productivity	Biomass & Abundance	Algal EC_{50}[2]	14	6.8-190	3
Effects on Submerged and Emergent Vegetation	Biomass & Abundance	Vascular Plant NOAEC[2] Vascular Plant EC_{50}[2]	290 630	NA	1
Effects on Riparian Vegetation	Biomass & Abundance	Terrestrial Plant NOAEL	16 lb ai/A	NA	NA

[1]If more than one value was available. If only one value was available, the actual number is given.
[2]From guideline/standardized test data presented in EPA documents (primary source most recent BE on RLF)
[3]Two studies available, but only one used. See text
NA Not applicable, only one value available

Evaluation of Data Available forResponse Analysis:

We summarize the available toxicity information by assessment endpoint in Table 129. Data and information reviewed for each assessment endpoint were assigned a qualitative ranking of either “low”, “moderate”, or “high.” A high confidence ranking was given if the information stemmed from direct measurements of an assessment endpoint, was 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 criteria were not met and a low ranking was assigned if two criteria were not met. Evidence of adverse effects to assessment endpoints for salmonids and their habitat from the 6 a.i.s was generally available regarding acute lethality to salmonids and aquatic invertebrates. Availability of information for other assessment endpoints was highly variable.

Table 129.. Summary of assessment endpoints and effect concentrations

Assessment Endpoint	Evidence of adverse responses (yes/no)	Concentration range of observed effect or concentrations tested showing absence of effect (µg/L)	Degree of confidence in effects (low, moderate, high)
2,4-D (amines/salts)		(µg ae/L)	
Fish:			
-survival (LC_{50})	yes	162,000-2,244,000	high
-growth (LOAEC)	yes	45,100	moderate
-reproduction (LOAEC)	yes	45,100	moderate
-swimming	-	-	-
-olfactory-mediated behaviors (**acid)**	yes	100,000	high
-endocrine disruption	-	-	-
-cellular damage/carcinogenicity	yes	1,000	moderate
Habitat:			
-prey survival (LC50) (**acid)**	yes	4,970	high
-primary productivity	yes	3,880-156,500	high
-submerged and emergent vegetation	yes	299-480	high
(LOAEC)	yes	0.003-0.273 lb ae/A	
-riparian vegetation (EC_{25})			
2,4-D (esters)			
Fish:	yes	450-14,500	high
-survival (LC_{50})	yes	145	moderate
-growth (LOAEC)	yes	145	moderate
-reproduction (LOAEC)	-	-	-
-swimming	-	-	-
-olfactory-mediated behaviors	-	-	-
-endocrine disruption	yes	300	moderate
-cellular damage/carcinogenicity			
Habitat:	yes	3,400	high
-prey survival (LC_{50})	yes	66-17,140	high
-primary productivity (LOAEC)	yes	330-397	high
-submerged and emergent vegetation	yes	0.02-0.218 lb ae/A	moderate
(LOAEC)			
-riparian vegetation (EC_{25})			

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)
Triclopyr BEE		(µg ae/L)	
Fish:			
-survival (LC_{50})	yes	470	high
-growth (LOAEC)(TEA)	yes	<50,200	moderate
-reproduction (LOAEC)	yes	1,000	high
-swimming	-	-	-
-olfactory-mediated behaviors	-	-	-
-endocrine disruption	-	-	-
-cellular damage/carcinogenicity	-	-	-
Habitat:			
-prey survival (LC_{50})	yes	2,200-11,900	high
-primary productivity,submerged and	yes	70-2,500	high
emergent vegetation (EC_{50})	yes	70-2,500	high
-riparian vegetation	yes	0.006-0.053 lb ae/A	moderate
Degradate toxicity	yes	1,900-16,100	high
Diuron			
Fish:			
-survival (LC_{50})	yes	710-23,800	high
-growth	-	-	-
-reproduction (LOAEC)	yes	62-8,200	moderate
-swimming	-	-	-
-olfactory-mediated behaviors	no	1,000	high
-endocrine disruption	-	-	-
Habitat:			
-prey survival (LC_{50})	Yes	160-19,400	high
-primary productivity,submerged and			
emergent vegetation (EC_{50})	yes	10-95	high
-riparian vegetation	yes	0.00017-0.0753 lb ai/A	moderate
Linuron			
Fish:			
-survival (LC_{50})	yes	3,000	high
-growth (LOAEC)	yes	42	moderate
-reproduction	-	-	-
-swimming	-	-	-
-olfactory-mediated behaviors	no	1,000	high
-endocrine disruption (LOAEC)	yes	100	moderate
Habitat:			
-prey survival (LC_{50})	yes	120-2,900	high
-primary productivity, submerged and emergent vegetation (EC_{50})	yes	2.5-67	high
-riparian vegetation	-	-	-
Captan			
Fish:			
-survival (LC_{50})	yes	26-137	high
-growth (LOAEC)	yes	39.5	moderate
-reproduction	yes	39.5	moderate
-swimming	-	-	-
-olfactory-mediated behaviors	-	15 – 1000	-
-genotoxicity	-	-	-
Habitat:		-	
-prey survival (LC_{50})	yes	8,400	high
-primary productivity (EC_{50})	yes	320-1,770	high
-submerged and emergent vegetation (EC_{50})	yes	>12,700	high
-riparian vegetation	yes	2.4 lb ai/A	moderate

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)
Chlorothalonil			
Fish:			
-survival (LC_{50})	yes	10.5-42.3	high
-growth	-	-	-
-reproduction (LOAEC)	yes	6.5	moderate
-swimming	-	-	-
-olfactory-mediated behaviors	no	1,000	high
-Immune response (LOAEC)	yes	1.4	high
-cellular damage/carcinogenicity	yes	3 – 7.2	high
Habitat:			
-prey survival (LC_{50})	yes	54-68	high
-primary productivity (EC_{50})	yes	6.8-190	high
-submerged and emergent vegetation (EC_{50})	yes	630	high
-riparian vegetation (NOAEL)	yes	16 lb ai/A	moderate

Dash (-) indicates no information

Risk Characterization

In this section we integrate our exposure and response analyses to evaluate the likelihood of adverse effects to individuals and populations (Figure 77). We combined the exposure analysis with the response analysis to determine the likelihood of salmonid and habitat effects occurring from the stressors of the action. Next we evaluate the evidence presented in the exposure and response analyses to support or refute risk hypotheses. We then evaluate the effects to specific ESUs and DPSs in the *Integration and Synthesis* section.

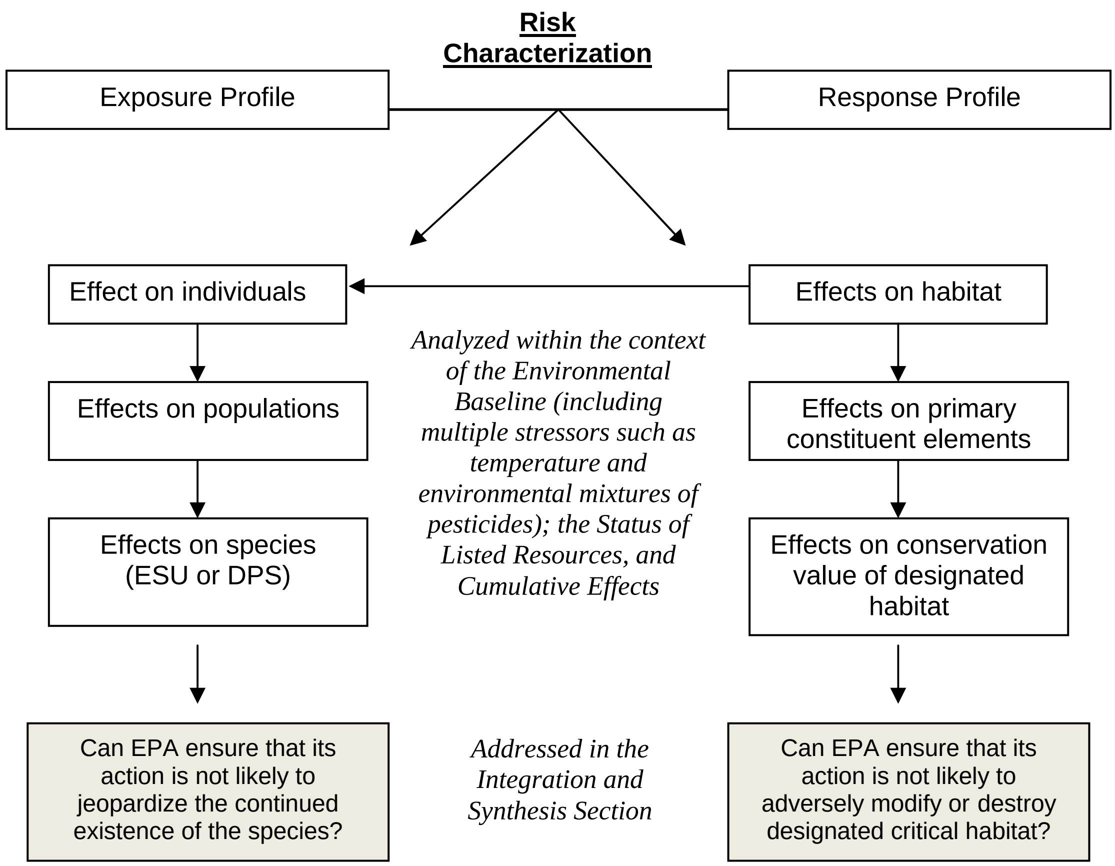


Figure 77. Schematic of the Risk Characterization Phase.

Integration of Exposure and Response

In Table 130 through Table 136, we compare the estimated environmental exposures (EECs) and measured environmental concentrations (MECs) for the 6 a.i.s with concentrations determined to affect assessment endpoints in the toxicity data we reviewed. This portion of the analysis is based strictly on a.i., and does not take into account other stressors of the action that may contribute to toxicity, and/or that other a.i.s may be present, creating additive or synergistic toxicity.

The tables show the exposure concentration ranges (minimum–maximum values) gleaned from the three sources of exposure data we analyzed: EPA's estimates presented in the BEs, NMFS' modeling estimates for floodplain habitats; and surface water monitoring data from ambient monitoring programs and targeted monitoring. In addition to the salmonid BEs submitted to NMFS, we also considered the exposure estimates developed by EPA in the BEs for the California red-legged frog. Some, although not all, of the red-legged frog BEs considered non-crop uses which were not included in the salmonid BEs submitted to NMFS. For some of the a.i.s evaluated in this Opinion, non-crop uses are an important market. Because of application rates and intervals, location, and/or extent of impervious surface associated with non-crop uses, EECs may be very different than they are for agricultural uses. In the summary tables, we grouped EECs related to specific use sites into generic landuse categories to better compare the overlap of use sites with landuse information for the various ESUs.

The effect concentrations are based on the toxicity data reviewed in the *Response Analysis Section*. For the survival assessment endpoint, effect concentrations are generally LC_{50}s, the concentration at which 50% of the test organisms die. Fish may die at higher or lower concentrations, depending on individual sensitivity. When a 95% confidence interval (CI) or slope of the dose-response curve is provided, we can get a better estimate of what the range may be, or when an individual fish might be affected. However, the 95% CI and/or slope data are often not provided. The concentration at which death occurs for the first individual (*i.e,* the most sensitive individual in the group) often cannot be determined from the data provided in standard toxicity tests. Sometimes it can be estimated if the slope of the dose-response curve is included in the data, but because the tests are optimized to determine the LC_{50} estimations of individual effects (essentially an LC_{01}) may not be accurate. Typically, a sensitive life stage (young juvenile fish) is tested, but life stage sensitivity can vary depending on toxicant. Additionally, in the wild fish are often exposed to additional stressors which can influence response.

For some pesticides, lethal effects occur quickly, possibly within hours of introduction of the a.i. to the test system. For other chemicals, sub-lethal effects may not manifest for hours and/or lethal effects may not occur for the first 24 or 48 hours of the test. While NMFS considers the available range of LC_{50} data we also consider factors such as time-to-response, the correlation in sensitivities between the test organisms and the life stage being considered, and the potential for sensitive individuals to die at concentrations below the reported LC_{50}s. In addition to differences in individual and life stage sensitivities, there is some variation between species. For the a.i.s in this opinion, toxicity data regarding survival endpoints were available for one or more salmon species, thus we assume them to be relatively good predictors of response. However, other endpoints are often derived from less closely related species.

This analysis considers both spatial and temporal overlap of salmon and salmon habitat with pesticide applications, but we do not do a crop specific analysis. In some cases, application rates, methods, or frequencies may vary for different landuse categories (*e.g*, agricultural uses, urban/residential uses, forestry uses, ROW uses), and we do take those differences into account where appropriate. For instance, we would not assume a high application rate for a forestry use would occur on agricultural land. Based on a run-timing analysis (*Appendix 6*) and an evaluation of landuse within the ESUs (*Appendix 5*) some sensitive life-stage may be present when the a.i.s are applied. This holds true for all ESUs/DPSs and all a.i.s, although not necessarily all use sites.

The analysis is also predicated primarily on standard toxicity endpoints, as we located only a few studies with ecologically relevant sublethal information, and that information was not available for all a.i.s. This analysis does allow NMFS to systematically address which assessment endpoints are likely to be affected by exposure to the 6 a.i.s. Where uncertainty arises, NMFS highlights the information and discusses its influence on our inferences and conclusions.

2,4-D

The herbicide 2,4-D is marketed in a variety of forms. Although EPA gives these forms different PC codes, they are treated as a single entity for registration and thus, as a single entity for this consultation. In our evaluation NMFS has made a distinction between the ester forms of 2,4-D (2,4-D butoxyethyl ester (BEE), 2,4-D ethylhexyl ester (EHE), and 2,4-D isopropyl ester (IPE)) and the other forms (acid, sodium salt, and amine salts (2,4-

D diethanolamine (DEA) salt, 2,4-D dimethylamine (DMA) salt, 2,4-D isopropylamine (IPA) salt) and 2,4-D triisiolpropanolamine (TIPA) salt). We make this distinction because the ester forms are significantly more toxic than the other forms. Although all forms, including the esters, degrade to the acid in the environment, we believe short-term exposures to the ester forms are relevant, particularly in the case of direct water applications. Thus, we evaluate acute exposures using toxicity data for both the esters and the acid/salt/amines, and we evaluate longer-term (chronic) effects based on the toxicity data for the acid/salts/amines.

Of the a.i.s addressed in this Opinion, 2,4-D has the widest range of uses, and is the only one registered for use as an aquatic herbicide. Forms used in direct water applications include one of the esters (BEE), amines, salts, and the acid, according to the 2,4-D master label. As a direct water application, it is often used to control nuisance weeds, including invasive species such as European watermilfoil. 2,4-D also has urban (*e.g.*, turf, industrial sites) and homeowner uses. Given the wide-range of uses, and the persistence of the acid form in water, we believe it is reasonable to assume all populations in most ESUs are exposed to some 2,4-D during their lifetimes. 2,4-D is one of the most commonly detected a.i.s considered in this Opinion, with 14 - 48% detections in the various monitoring programs. For the 2,4-D analysis we relied heavily on the EECs from the California red-legged frog assessment (EPA, 2009a), as it separated the estimates for the ester and acid forms.

Agricultural Uses

For agricultural uses, 2,4-D application rates are typically in the range of 1-2 lbs ae/A, with 1-2 applications per year. For the amines/salts/acid, there are no exceedences of any endpoint for the fish or prey. Generally, the assessment endpoints are several orders of magnitude higher than EECs or concentrations detected in monitoring. The exception to that is an overlap with the algal and aquatic vascular plant endpoints with the floodplain estimate. For the esters, there is an overlap between the lower end of range of salmon lethality endpoints and the floodplain estimate. This overlap indicates potential for lethal effects in these smaller habitats. There is also an overlap with the algal and aquatic vascular plant endpoints with the floodplain estimate, and an exceedence of the LOAEC for ecosystem functioning (which is related to primary producers). We anticipate there may be a decrease in primary productivity in the floodplain habitats due to agricultural uses of 2,4-D. The floodplain estimates also exceed the endpoint for short term cellular damage associated with the ester form. Available plant toxicity data indicates 2,4-D used in agricultural applications may affect herbaceous plants in the riparian zone, especially dicots.

2,4-D is also authorized for use on rice and cranberries. Rice is grown in water, and cranberries are often flooded for harvest. Only the acid, amines, and salts are authorized for rice. Using an initial concentration of 4,000 μg/L, EPA used the rice model to calculate a chronic concentration of 1,486 μg/L. This concentration does not exceed survival assessment endpoints for fish or prey, although it does exceed the chronic cellular damage endpoint. This concentration would be released in the tailwater, and would be affected by volume of the receiving stream, and potentially by holding times required by the state or locality.

Forestry Uses

Foresty uses of 2,4-D are authorized at a higher rate than the agricultural uses, up to 4 lb ae/A. While the application methods include ground or aerial broadcast sprays, according to the master label, they also include other methods such as basal spray, cut stump, frill, and injection that may reduce the area treated, or the runoff associated with that treatment. Broadcast sprays at the maximum rate of 4 lb ae/A are limited to once every 12 months according to the 2,4-D Master Label.

For the amines/salts/acid, there are no exceedences of any endpoint for the fish or prey. Generally, the assessment endpoints are several orders of magnitude higher than EECs or concentrations detected in monitoring. The exception to that is an overlap with the algal and aquatic vascular plant endpoints with the floodplain estimate. For the esters, there is an overlap between the lower end of range of salmon lethality endpoints and the floodplain estimate. This overlap indicates potential for lethal effects in these smaller habitats. There is also an overlap with the algal and aquatic vascular plant endpoints with the floodplain estimate, and an exceedence of the LOAEC for ecosystem functioning (which is related to primary producers). We anticipate there may be a decrease in primary productivity in the floodplain habitats due to agricultural uses of 2,4-D. The floodplain estimates also exceed the endpoint for short term cellular damage associated with the ester form. Available plant toxicity data indicates 2,4-D used in forestry applications may affect herbaceous plants in the riparian zone, especially dicots.

ROW Uses

ROW uses of 2,4-D range from 2 lb ae/A for control of annual and perennial weeds to 4 lb ai/A for control of woody plants. Multiple applications of 2 lb ae/A have a 30 day interval (2,4-D Master Label).

For the amines/salts/acid, there are no exceedences of any endpoint for the fish or prey. Generally, the assessment endpoints are several orders of magnitude higher than EECs or concentrations detected in monitoring. The exception to that is an overlap with the algal and aquatic vascular plant endpoints with the floodplain estimate. For the esters, there is an overlap between the lower end of range of salmon lethality endpoints and the floodplain estimate. This overlap indicates potential for lethal effects in these smaller habitats. There is also an overlap with the algal and aquatic vascular plant endpoints with the floodplain estimate, and an exceedence of the LOAEC for ecosystem functioning (which is related to primary producers). We anticipate there may be a decrease in primary productivity in the floodplain habitats due to agricultural uses of 2,4-D. The floodplain estimates also exceed the endpoint for short term cellular damage associated with the ester form. Available plant toxicity data indicates 2,4-D used in ROWapplications may affect herbaceous plants in the riparian zone, especially dicots.

Urban/Developed Area Uses

Urban uses of 2,4-D, which include turf uses (1.5 lb ae/A) and other uses (defined in the master label as not cropland) range from 2 lb ae/A for control of annual and perennial weeds to 4 lb ai/A for control of woody plants.

For the amines/salts/acid, there are no exceedences of any endpoint for the fish or prey. Generally, the assessment endpoints are several orders of magnitude higher than EECs or concentrations detected in monitoring. The exception to that is the overlap with the algal and aquatic vascular plant endpoints with the floodplain estimate. For the esters, there is an overlap between the lower end of range of salmon lethality endpoints and the floodplain estimate. This overlap indicates potential for lethal effects in these smaller habitats. There is also an overlap with the algal and aquatic vascular plant endpoints with the floodplain estimate, and an exceedence of the LOAEC for ecosystem functioning (which is related to primary producers). We anticipate there may be a decrease in primary productivity in the floodplain habitats due to agricultural uses of 2,4-D. The floodplain estimates also exceed the endpoint for short term cellular damage associated with the ester form. Available plant toxicity data indicates 2,4-D used in urban applications may affect herbaceous plants in the riparian zone, especially dicots.

Direct Water Applications

Direct water applications for control of aquatic weeds are permitted at 4 lb ae (2,4-D Master Label) , or a concentration equivalent to 4,000 μg/L (label 5481-145). These applications are authorized for the amines/salts/acid and for the BEE. For the amines/salts/acid, there are no exceedences of any endpoint for the fish or prey. There is overlap with the longer term cellular damage endpoint associated with the acid, an overlap with algal plant endpoints, and EECs exceed all vascular plant EC_{50}s. For the BEE, authorized water concentrations exceed the lower end of the range of the LC_{50} data available for salmonids and the median LC_{50} from the data set, although not the upper end of the range. The 4,000 μg/L concentrations also exceeds the prey survival LC_{50}, the effect concentration for short term cellular damage associated with the ester form, vascular plant EC_{50}s and overlaps with the range of algal EC_{50}s.

The LC_{50s} for the ester forms do overlap with estimates for the floodplain habitats and for direct applications of ester formulations to water. We believe there may be cases where the direct water applications of the BEE form will cause fish mortality, but in most other cases we believe lethal effects are unlikely. Peak estimates for the floodplain habitats are particularly relevant for the esters, which are taken up by the fish quickly. Endpoints for cellular damage overlap with concentrations in the floodplain habitats, and from the direct water applications. Overall we believe it is possible but unlikely that any form of 2,4-D will kill fish, and possible that it may cause cellular damage. Use of the ester forms near shallow, low-flow environments creates a particularly vulnerable situation. Based on what data we were able to locate, effects on olfaction appear unlikely.

Plants, especially vascular plants, are sensitive to all forms of 2, 4-D, and plants may suffer mortality, particularly in direct water applications or in the floodplain habitats. Based on the toxicity profile for 2,4-D, we anticipate the most likely effects to be reduction in biomass, abundance, and diversity of riparian vegetation or vascular plants in-stream.

Table 130. Summary of estimated and measured environmental concentrations, and effect concentrations for 2,4-D.

Landuse categories		2,4-D Ester (μg ae/L)	2,4-D Acid, salt, amine (μg ae/L)	All forms, using acid, amine, salt endpoints
		Peak EECs	Peak EECs	Chronic EECs (60 d)
Agricultural 1-2 lb ai/A (Table 3)	Farm pond (Table 89)	0.55-5.5	0.08-33	0.07-27
	Rice Direct Water (Table 89)	Not authorized	4,000	1,486
	Floodplain (Table 91)	9-956	13-956	NA
Forestry 4 lb ai/A (Table 3)	Farm pond (Table 89)	1.3-13	6-47	5-39
	Floodplain (Table 91)	736-1,912	736-1,912	NA
ROW 2-4 lb ai/A (Table 3)	Farm pond (Table 89)	NE	6-47	5-39
	Floodplain (Table 91)	368-1,912	368-1,912	NA
Urban/Developed 1.5-4 lb ai/A (Table 3)	Farm pond (Table 89)	NE	6-47	5-39
	Floodplain (Table 91)	276	276	NA
Direct water applications	Farm pond (Table 89)	1,431-4,000	1,431-4,000	2,610
Monitoring data		Databases: 0.007-7.660 (14-48% detections); Targeted monitoring maximum: 67.1 μg ae/L golf course, 2,343 μg ae/L in rice paddy (acid, amine, salt)		
Range of Effects on Assessment Endpoints (From Table 114)		2,4-D Ester (μg ae/L)	2,4-D Acid, salt, amine (μg ae/L)	All forms, using acid, amine, salt endpoints
		Acute	Acute	Chronic
Salmonid survival (LC_{50})		450-14,500	162,000-2,244,000	NA
Fish reproduction and growth (NOAEC)		NA	NA	17,000-63,400
Olfactory-mediated behaviors (2 min EOG response)		100,000 (based on acid)	100,000	100,000 (based on acid)
Cellular damage		300 (Interrenal, 96h)	ND	1,000 (peroxisome, 7d)
Endocrine disruption		ND	ND	ND
Swimming		ND	ND	ND
Prey survival (LC_{50})		3,400	4,970	NA
Prey reproduction and growth (NOAEC)		79,000	79,000	NA
Primary productivity in-stream Algae EC_{50} Vascular plant EC_{50}		 660-17,140 330-397	 340-156,500 299-480	NA
Riparian vegetation (EC_{25})		Monocot 0.20 lb ai/A Dicot 0.020 lb ai/A	Monocot 0.40 lb ai/A Dicot 0.003 lb ai/A	NA
Ecosystem functioning [a] $NOEC_{eco}$ $LOEC_{eco}$		 10 100	 10 100	NA

ND – No data, NE – Not estimated, NA – Not applicable
[a] (T.C.M. Brock, et al., 2000)

Triclopyr BEE

Agricultural uses of triclopyr include pasture/rangeland, and non-crop areas. One of its primary uses is in forestry, and it is also registered for use on rights-of-way, residential turf, and in industrial areas. Similar to 2,4-D, the ester form breaks down relatively quickly in the environment, but uptake of it by fish is rapid. We use the toxicity data for the BEE to evaluate acute toxicity, and the toxicity data for the TEA to evaluated effects from longer exposures. The salmonid BE (EPA, 2004f) did not provide chronic EECs or evaluate these effects, and it was unclear whether the California red-legged frog BE used fate parameters of the BEE or the TEA. We expect chronic concentrations estimated based on the persistence of the acid or amine forms might be higher. We used some of the estimates provided, along with NMFS-generated floodplain estimates. Floodplain estimates address only short-term (acute) concentrations.

Comparision of EECs

The EPA salmonid BE (EPA, 2004f) and the California red-legged frog BE (EPA, 2009a) provide significantly different EECs for triclopyr. The salmonid BE used EECs from the triclopyr RED (EPA, 1998b). Those EECSs were calculated using GENEEC. Although not specified in the RED, estimates appear to be for a single application. EECs were estimated separately for the BEE and TEA, which have different fate properties. For ground applications of the BEE, EECs for 1.0 lb ae/A, 3.0 lb ae/A, and 8.0 lb ae/A are 19 μg/L, 57 μg/L, and 152 μg/L, respectively (reported in our Table 88). For aerial applications of the BEE, EECs for rates of 1.5 lb ae/A and 8.0 lb ae/A are 30 μg/L, and 160 μg/L, respectively. Due to the rapid breakdown of the BEE to the acid, EECs for the BEE were not estimated for longer time periods. In the RED and salmon BE, EPA also reported EECs for direct application to 6 inches of water. This estimation method assumes a similar water body to the NMFS floodplain estimates Direct water application EECs for current use rates of 1-8 lb ae/A range from 730 -5,870 μg/L.

Applicants for triclopyr (Dow, 2011) and EPA (EPA, 2009c) note that the 1998 RED restricts use of triclopyr BEE to 2 lb ae/A/year on pasture and rangeland, 6 lb ae/A for forestry applications, and 8 lb ae/A/year for all other use sites. Our label analysis indicated the same (Table 4, Description of the Action). In the California red-legged frog assessment, EPA reported higher rates (up to 20 lb ae/A) in their label analysis, with no yearly maximum and used this information in the modeling ((EPA, 2009c), pg 57)). Due to a the lack of a yearly maximum, many of the scenarios (17 out of 32) were modeled with 17 applications (every 21 days for one year) ((EPA, 2009c), pg 62). A number of other scenarios (7) used rates not currently authorized for the BEE, and some (5) were for direct water applications, which are not authorized for the BEE (although they are for the TEA). The text did not specify if fate parameters used were for the BEE or TEA. We found 4 scenarios that appear to be applicable to current labels. EECs from these scenarios, with application rates ranging from 0.76 – 4.5 lb ae/A, range from 5.3 – 250 μg/L. The 4.5 lb ae/A scenario, which produced the 250 μg/L EEC, used the CA impervious RLF parameters. Impervious surfaces typically produce higher runoff than grassy, cropped, or forested areas. None of the scenarios modeled that appear to be applicable included the current maximum use rate.

Based on the current maximum use rates as stated by the applicant, and the modeling in the two BEs (EPA, 2004f, 2009c), it appears the applicable EECs for the EXAMS waterbody are in the range of 150-250 μg/L for high application rates and/or mid-range application rates in high runoff situations. For lower application rates (1-3 lb ae/A), EECs range from 19 μg/L (EPA, 2004f) to 75 μg/L (EPA, 2009c). Concentrations in small waterbodies (EPA direct water applications, NMFS floodplain estimates) are higher, in the range of 184-3,824 μg/L. There is significant uncertainty associated with all estimates.

There was little toxicity data available for triclopyr, either for the BEE or the TEA form, thus there is significant uncertainty associated with our conclusions. The BEE is more toxic, and although it does degrade quickly, uptake by fish also appears to be rapid, and we have used BEE toxicity endpoints for comparisons with the peak concentrations. The

TEA is much less toxic. We consider the TEA acute endpoints, and use the TEA chronic endpoints to evaluate chronic effects. The acid form is relatively persistent in water, and percent detections ranged from 8.7-17.5%, depending on database.

Agricultural Uses

Application rates in agricultural areas are 2 lb ae/A for pasture and rangeland, and up to 8 lb ae/A for around farm buildings. There is no overlap of the peak farm pond estimates with any of the acute assessment endpoints (BEE) except the algal EC_{50}. Floodplain estimates overlap with all acute assessment endpoints available at the 8 lb ae/A rate, and overlap with the prey EC_{50} and algal EC_{50} at the 2 lb ae/A. Longer term concentrations to compare with chronic endpoints were not available, but if the peak concentrations are compared to the TEA assessment endpoints (both acute and chronic) none overlap, and in fact they are typically at least an order of magnitude lower.

Forestry Uses

Application rates for forestry uses are 6 lb ae/A. There is no overlap of the peak farm pond estimates with any of the acute assessment endpoints (BEE) except the algal EC_{50}. Floodplain estimates overlap with all acute assessment endpoints available at the 8 lb ae/A rate. Longer term concentrations to compare with chronic endpoints were not available, but if the peak concentrations are compared to the TEA assessment endpoints (both acute and chronic) none overlap, and in fact they are typically at least an order of magnitude lower.

ROW Uses

ROW application rates are 8 lb ae/A. We note that some ROW applications include some percentage of impervious surfaces (pavement) or less pervious surfaces (gravel or packed dirt), thus there might be greater runoff than would be anticipated from cropped or grassy areas. There is no overlap of the peak farm pond estimates with any of the acute assessment endpoints (BEE) except the algal EC_{50}. Floodplain estimates overlap with all acute assessment endpoints available. Longer term concentrations to compare with chronic endpoints were not available, but if the peak concentrations are compared to the TEA assessment endpoints (both acute and chronic) none overlap, and in fact they are typically at least an order of magnitude lower.

Urban/Developed Area Uses

Application rates in urban/developed area uses range from are 1-8 lb ae/A. There is no overlap of the peak farm pond estimates with any of the acute assessment endpoints except the algal EC_{50}. Floodplain estimates overlap with all acute assessment endpoints available at the 8 lb ae/A rate, but overlap only with the and algal EC_{50} at the 1 lb ae/A application rates. Longer term concentrations to compare with chronic endpoints were not available, but if the peak concentrations are compared to the TEA assessment endpoints (both acute and chronic) none overlap, and in fact they are typically at least an order of magnitude lower.

Overall, we expect that rapid runoff of the BEE into small waterbodies could kill fish or prey, but that it would occur rarely, and under specific circumstances. These circumstances might include application directly next to a small waterbody, or on an impervious or semi-pervious surface that drains into a small waterbody (*i.e.*, a storm drain) if rain occurs soon after that. In other instances, we do not expect lethal or chronic effects on fish or prey given available data. We do note, however, that limited toxicity data is available, and most of the available data is from guideline studies. Sublethal effects do not appear to have been investigated for this chemical. Also, significant uncertainty surrounds the EECs. We do anticipate use of triclopyr BEE will affect plant communities, including those in the riparian zone, either from drift or via direct application. Triclopyr is sometimes used in habitat restoration or invasive weed control programs, and the changes in the riparian community will vary on a site-specific basis.

Table 131. Summary of estimated and measured environmental concentrations, and effect concentrations for Triclopyr BEE.

Landuse categories		Triclopyr BEE (μg ae/L)	Triclopyr TEA (μg ae/L)	Triclopyr TEA (μg ae/L)
		Peak EECs	Peak EECs	Chronic EECs (60 d)
Agricultural Range/pasture 2 lb ai/A Around farm building 8 lb ai/A (Table 4)	Farm pond[1] (Table 88)	Pasture 20-40 Building 160	NA	NE
	Floodplain (Table 91)	368-1,471	NA	NA
Forestry 6-8 lb ai/A (Table 4)	Farm pond[1] (Table 88)	120-160	NA	NE
	Floodplain (Table 91)	1,103-2,868	NA	NA
ROW 8 lb ai/A (Table 4)	Farm pond[1] (Table 88)	160	NA	NE
	Floodplain (Table 91)	1,147-3,824	NA	NA
Urban/Developed 1-8 lb ai/A (Table 4)	Farm pond[1] (Table 88)	20-160	NA	NE
	Floodplain (Table 91)	184-3,824	NA	NA
Monitoring data (Table 102)		No ambient monitoring data for triclopyr BEE or TEA. Max conc in targeted monitoring was 350 μg/L in surface runoff water (*i.e.* not in receiving stream)		
Range of Effects on Assessment Endpoints (From Table 115)		Triclopyr BEE (μg ae/L)	Triclopyr TEA (μg ae/L)	Triclopyr TEA (μg ae/L)
		Acute	Acute	Chronic
Salmonid survival (LC_{50})		470	79,200	NA
Fish reproduction and growth (NOAEC)		NA	NA	>32,200
Olfactory-mediated behaviors		ND	ND	ND
Cellular damage		ND	ND	ND
Endocrine disruption		ND	ND	ND
Swimming		ND	ND	ND
Prey survival (LC_{50})		250	346,000	NA
Prey reproduction and growth (NOAEC)		NA	NA	25,500
Primary productivity in-stream Algae (EC_{50}) Vascular plant (EC_{50})		70-2,500 860	4,100-12,100 6,100	NA
Riparian vegetation (EC_{25}) lb a.i/A		0.006-0.063	0.005-0.23	NA
Ecosystem functioning $NOEC_{eco}$ $LOEC_{eco}$		ND	ND	ND

ND – No data, NE – Not estimated, NA – Not applicable

1 Concentrations from EPA salmonid 2009, GENEEC estimate of 1 lb ae/A produces 20 μg/L from an aerial application. Concentrations for other rates were multiples of this rate, and ones reported in the table were adjusted for current allowable maximums.

Diuron

There is overlap with fish survival endpoints for all estimated concentrations except the farm pond estimates for agricultural uses (Table 132). Diuron is heavily used as right-of-way herbicide, so these higher EECs are of concern. It is also relatively persistent in water, so there may be extended exposure in lower-flow environments such as ponds and lakes, and/or an extended chemical plume in faster-flowing waters such as streams and rivers. Measured concentrations are lower, but it is fairly frequently detected in monitoring (29-47% detections), with more frequent detections in California. We are uncertain of the specific reason for greater detection frequency, but likely it is related to use. There is significant overlap in the estimated concentrations and the lower end of the fish reproduction and growth assessment endpoints. Diuron has not been identified as an endocrine disruptor, but linuron has, and given structural similarities, we believe it is reasonable to use the linuron endocrine disruption endpoint for diuron. There is significant overlap of this concentration and estimates from all uses. That endpoint is also within the range of detected values.

Agricultural Uses

For most agricultural uses rates range from 0.6 -3.2 lb ai/A, although there are some 24(c) uses on lilies and a section 3 use on papaya that are slightly higher (4 lb ai/A). Farm pond EECs do not overlap with salmon or prey survival endpoints, although the high end of the estimates do approach the prey survival endpoint. Floodplain estimates do overlap with the lower end of the range for both salmon and prey assessment endpoints. The 60 d concentrations overlap with the lower end of the fish growth and reproduction endpoints. Neither peak nor chronic estimates from the farm pond overlap with the growth and reproduction endpoint for prey, nor does the floodplain estimate. All estimates do overlap with the olfactory-mediated behavior endpoint and the endocrine disruption endpoint. We note those two endpoints are actually the linuron endpoints, as no data was available for diuron, thus there is uncertainty regarding what this overlap ultimately means, but we do believe it warrants some concern. All estimates overlap with the vascular plant EC_{50}. We anticipate some effects from diuron to in-stream plants, and also some effects on the terrestrial plants in the riparian zone.

Diuron is authorized for non-crop uses in agricultural areas at up 12 lb ai/A. In the California red-legged frog assessment (EPA, 2009b), EPA provided two separate EECs for these types of uses. For irrigation ditches, the peak EEC was 53 μg/L and the 60 d EEC was 37 μg/L. For fence row/hedgerow ROW uses, corresponding values were 688 μg/L and 547 μg/L. The document did not specify the scenarios used in these estimates, but likely the higher values for the ROW uses are due to an assumption of some percentage of impervious surface. Altenatively, the differences may be associated with timing of application.

Farm pond EECs do not overlap with salmon survival endpoints, but the high end of the estimates does approach the low end of the survival endpoint. EECs overlap with the low end of the prey survival endpoints, but do not exceed the high end. The 60 d concentrations overlap with the lower end of the fish growth and reproduction endpoints. Floodplain estimates overlap with the lower end of the range for both salmon and prey survival assessment endpoints. Neither peak nor chronic estimates from the farm pond overlap with the prey growth and reproduction endpoint from the guideline tests, but the floodplain estimate does. All estimates do overlap with the olfactory-mediated behavior endpoint and the endocrine disruption endpoint. We note those two endpoints are actually the linuron endpoints, as no data was available for diuron, thus there is uncertainty regarding what this overlap ultimately means, but we do believe it warrants some concern. All estimates overlap with the vascular plant EC_{50}. We anticipate some effects from diuron to in-stream plants, and also some effects on the terrestrial plants in the riparian zone. Unlike triclopyr BEE, we are unaware of any habitat restoration uses of diuron.

We expect diuron will be unlikely to kill fish, may cause some reduction in prey, and could potentially affect growth, reproduction, or health although we are unable to estimate to what extent.

ROW Uses

ROW uses are authorized at much higher application rates than agricultural uses (12 lb ai/A) and applicants have indicated this is a substantial portion of their market. Farm pond EECs do not overlap with salmon survival endpoints, but do approach the low end of the range of survival endpoints. EECs overlap with the low end of the prey survival endpoints, but do not exceed the high end. The 60 d concentrations overlap with the lower end of the fish growth and reproduction endpoints. Floodplain estimates overlap with the lower end of the range for both salmon and prey survival assessment endpoints. No estimates overlap with the prey growth and reproduction endpoint from the guideline tests. All estimates do overlap with the olfactory-mediated behavior endpoint and the endocrine disruption endpoint. We note those two endpoints are actually the linuron endpoints, as no data was available for diuron, thus there is uncertainty regarding what this overlap ultimately means, but we do believe it warrants some concern. We expect diuron will be unlikely to kill fish, may cause some reduction in prey, and could potentially affect growth, reproduction, or health although we are unable to estimate to what extent. All estimates overlap with the vascular plant EC_{50}. We anticipate some effects from diuron to in-stream plants, and also some effects on the terrestrial plants in the riparian zone.

Urban/Developed Area Uses

We have included the ROW uses and industrial site use EECs provided by EPA (EPA, 2009b) in this category. They are authorized at application rates of 12 lb ai/A. Farm pond EECs for the industrial sites overlap with salmon survival endpoints, and ROW the estimates approach the low end of the range of survival endpoints. EECs overlap with the low end of the prey survival endpoints, but do not exceed the high end. The 60 d concentrations overlap fish growth and reproduction endpoints and approach prey reproduction and growth endpoints. Floodplain estimates overlap with the lower end of the range for both salmon and prey survival assessment endpoints. All estimates overlap with the olfactory-mediated behavior endpoint and the endocrine disruption endpoint. We note those two endpoints are actually the linuron endpoints, as no data was available for diuron, thus there is uncertainty regarding what this overlap ultimately means, but we do believe it warrants some concern. We expect diuron could kill fish in certain circumstances, but expect it to occur infrequently. These uses of diuron may cause some reduction in prey, and could potentially affect growth, reproduction, or health although we are unable to estimate to what extent. All estimates overlap with the vascular plant EC_{50}. We anticipate some effects from diuron to in-stream plants, and also some effects on the terrestrial plants in the riparian zone.

Table 132. Summary of estimated and measured environmental concentrations, and effect concentrations for diuron.

Use categories		Diuron (µg ai/L) Peak	Diuron (µg ai/L) Chronic (60 d)
Agriculture Most crops[1] 0.6-3.2 lb ai/A Non-crop uses 12 lb ai/A	Farm pond (EPA RLF 2009 Table 3.2)	Crop 5-140 Non-crop 53-688	Crop 3-103 Non-crop 38-487
	Floodplain (Table 91)	Crop 221-1,147 Non-crop 4,911	NA
Forest	Farm pond	NA	NA
	Floodplain	NA	NA
ROW 12 lb ai/A	Farm pond (EPA RLF 2009 Table 3.2)	630	476
	Floodplain (Table 91)	2,207	NA
Urban 12 lb ai/A	Farm pond (EPA RLF 2009 Table 3.2)	630-4,911	476-3,428
	Floodplain (Table 91)	2,207	NA
Monitoring data		0.002-160 (29-47% detections) More detections and higher concentrations in CA databases than WA databases; maximum concentrations detected in target surface water runoff: 467 µg ai/L vineyards; 1,700 µg ai/L citrus; 2,849 µg ai/L rights-of-way.	
Assessment Endpoints			
Salmonid survival (LC_{50})		710-23,800	NA
Fish reproduction and growth (NOAEC)		NA	5-4,200
Olfactory-mediated behaviors[2]		10	10
Cellular damage		ND	ND
Endocrine disruption[2]		25	25
Swimming		ND	ND
Prey survival (EC_{50})		160-19,400	NA
Prey reproduction and growth (NOAEC)		NA	4,000
Primary productivity Vascular plant (EC_{50})		10-95	NA
Riparian vegetation (EC_{25})		Monocot 0.0164 lb ai/A	NA

ND – No data, NE – Not estimated, NA – Not applicable
[1] 24(c) on easter lilies in two counties, and section 3 on papaya at 4 lb ai/A
[2]Not determined for diuron, assume similar response as determined for linuron.

Linuron

Linuron is primarily a specialty crop herbicide, approved for use on a relatively limited number of crops. It does have some rights-of-way uses, but these are confined to non-crop agricultural areas and it is not commonly selected for these uses due to cost, according to the representatives for the applicant (expressed verbally at applicant meeting 10/08/2010). Data for some of the standard endpoints was not available for linuron, but based on what data are available, toxicity appears similar to diuron. Also, like diuron, it is persistent in aquatic systems. However, it has a much lower rate of detection than diuron (0.007-1.6% as opposed to 29-47%). This is likely a reflection of differences in usage.

There is no overlap of fish assessment endpoints with estimated or measured uses other than the NOAEC for endocrine disruption. The endocrine disruption endpoint overlaps with all estimates from all uses, although it is higher than measured concentrations (Table 133). Endocrine disruption effects would likely be expressed in lowered reproductive rates, and given linuron's persistence in water, could occur in waterbodies receiving direct runoff. Slow-moving waters such as the floodplain habitats would be most at risk for endocrine effects.

Estimated and measured concentrations overlap slightly with some prey survival endpoints, suggesting some applications may cause lethality to sensitive aquatic invertebrates. Although there were no data on aquatic invertebrate reproduction and growth, based on comparisons with diuron data and structural similarities between the two a.i.s, we do not anticipate estimated and measured concentrations would overlap with them either.

There is significant overlap of primary productivity endpoints (algal and vascular plants) with all uses and estimates, and these endpoints are also within the range of measured concentrations. Estimates for all uses and measured concentrations are also much higher than the most sensitive endpoint of ecosystem functioning (NOAEC 0.3 μg/L). We believe the in-stream plant community of any waterbody receiving linuron will be affected by it. Effects may include reductions in plant biomass, loss of diversity, and shifts in the plant community to more tolerant species. No data were available for terrestrial plant endpoints. However, based on linuron's mode of action, and recommended application rates, we anticipate there will be effects on riparian vegetation if it is applied near these habitats. Effects may include reductions in plant biomass, loss of diversity, and shifts in the plant community to more tolerant species.

Table 133. Summary of estimated and measured environmental concentrations, co-occurrence, and effect concentrations for linuron.

Use categories		Linuron (µg ai/L) Peak	Linuron (µg ai/L) Chronic (60 d)
Ag 1-4 lb ai/A	Farm pond (Table 89)	Crop 2.6-41 Non-crop 60-337	Crop 1.8-31 Non-crop 39-211
	Floodplain (Table 91)	184-736	NA
Forest	Farm pond (Table 89)	NA	NA
	Floodplain (Table 91)	NA	NA
ROW	Farm pond (Table 89)	NA	NA
	Floodplain (Table 91)	NA	NA
Urban	Farm pond (Table 89)	NA	NA
	Floodplain (Table 91)	NA	NA
Monitoring data		0.007-1.6 (0.16-3.2% detections)	
Assessment Endpoints		Linuron (µg ai/L)	Linuron (µg ai/L)
Salmonid survival (LC_{50})		3,000	NA
Fish reproduction and growth (NOAEC)		NA	<42
Olfactory-mediated behaviors		10	10
Cellular damage		ND	ND
Endocrine disruption		25	25
Swimming		ND	ND
Prey survival (EC_{50})		270-2,900	130
Prey reproduction and growth (NOAEC)		ND	ND
Primary productivity Vascular plant EC_{50})		2.5-67	2.5-67
Riparian vegetation (EC_{25})		ND	ND
Ecosystem functioning[1] $NOEC_{eco}$ $LOEC_{eco}$		 0.3[1] 3[1]	 0.3[1] 3[1]

ND – No data, NE – Not estimated, NA – Not applicable
[1]Damm et al 2009
(T.C.M. Brock, et al., 2000)

Captan

Captan dissociates rapidly in aquatic systems, so exposure to the parent molecule is expected to be limited. Parent captan breaks down into a ring structure (THPI) and the toxic moiety, a trichloromethylthio molecule (TCMT). EPA documentation provided virtually no data on TCMT, either from a fate or toxicity perspective, and we did not locate any data regarding it in our literature review. The discussion below is relevant to effects of the parent.

Agricultural Uses

Captan is primarily authorized for agricultural uses at rates ranging from 1-4 lb a.i./A. It is authorized for use on sod farms (1 lb ai/A) which we include in this landuse category. It is also registered for use on a number of flowers, generally as a root or seed dip. Because these seem to be primarily nursery applications, we consider it in this landuse category. There is no overlap of fish survival endpoints with farm pond estimates from agricultural uses although the high end of the EECs approaches the lower end of the fish survival endpoints. Reproduction and growth endpoints, and genotoxicity endpoints overlap with all peak exposure estimates for all uses (Table 134). Information to assess the duration of exposure necessary to elicit these responses was not available for captan. However, the reproduction, growth, and genotoxicity effects may require longer exposure durations to the parent molecule than is likely to occur in a natural environment. We do not anticipate captan will kill fish. It may cause sublethal effects.

There is no overlap of prey survival, reproduction,or growth endpoints with any of the chronic exposure estimates for any uses. There is no overlap of primary productivity endpoints, either algal or vascular plants, with any estimates for any uses. Based on what data are available, there could be effects on riparian vegetation if captan is applied near the waterbody. We do not anticipate captan will reduce prey abundance or diversity. We do not anticipate captan will modify primary productivity or riparian vegetation.

Urban/Developed Area Uses

Urban are uses include golf course turf and ornamental grasses. Farm pond EECs from the ornamental grasses overlap with the lower end of the fish survival endpoints. Turf EECs do not. Reproduction and growth endpoints, and genotoxicity endpoints overlap with all peak exposure estimates for all uses (Table 134). Information to assess the duration of exposure necessary to elicit these responses was not available for captan. However, the reproduction, growth, and genotoxicity effects may require longer exposure durations to the parent molecule than is likely to occur in a natural environment. We do not anticipate captan will kill fish. It may cause sublethal effects.

There is no overlap of prey survival, reproduction,or growth endpoints with any of the chronic exposure estimates for any uses. There is no overlap of primary productivity endpoints, either algal or vascular plants, with any estimates for any uses. Based on what data are available, there could be effects on riparian vegetation if captan is applied near the waterbody. We do not anticipate captan will reduce prey abundance or diversity. We do not anticipate captan will modify primary productivity or riparian vegetation.

Table 134. Summary of estimated and measured environmental concentrations, and effect concentrations for captan.

Use categories		Captan (µg ai/L) Peak	Captan (µg ai/L) Chronic (60d)
Agricultural 1-4 lb ai/A	Farm pond (Table 89)	<0.001-21.6	<0.001-0.06
	Floodplain (Table 91)	184-1,960	NA
Forest	Farm pond (Table 89)	NA	NA
	Floodplain (Table 91)	NA	NA
ROW	Farm pond (Table 89)	NA	NA
	Floodplain (Table 91)	NA	NA
Urban 1-4lb ai/A	Farm pond (Table 89)	3.6-29	0.08-1.1
	Floodplain (Table 91)	184	NA
Monitoring data		Infrequently detected, not included in NAWQA monitoring (0-1.3 % detections)	
Assessment Endpoints			
Salmonid survival (LC_{50})		26.2-137	NA
Fish reproduction and growth (NOAEC)		NA	16.5
Olfactory-mediated behaviors		ND	ND
Cellular damage		15.6-31.3	15.6-31.3
Endocrine disruption		ND	ND
Swimming		ND	ND
Prey survival (EC_{50})		1,200-8,400	NA
Prey reproduction and growth (NOAEC)		NA	560
Primary productivity Vascular plant EC_{50})		1,200 >12,700	1,200 >12,700
Riparian vegetation (EC_{25})		Dicot 2.4 lb ai/A	Dicot 2.4 lb ai/A
Ecosystem functioning $NOEC_{eco}$ $LOEC_{eco}$		ND	ND

ND – No data, NE – Not estimated, NA – Not applicable

Chlorothalonil

Chlorothalonil is highly toxic to aquatic organisms based on standardized laboratory tests. However, fate properties, especially the rates of photolysis and aerobic aquatic metabolism, are an important factor in how toxic it may be in the environment. It degrades rapidly via photolysis, with a half-life of 0.4 d. The half-life for aerobic aquatic metabolism is reported as ranging from 2.5 – 21 d in the chlorothalonil BE for the California red-legged frog, and modeled as 35.2 d (EPA, 2007b). This EPA document is the source of the EECs presented in Table 135. The applicant for chlorothalonil stated that in a later EPA assessment, the aerobic aquatic half-life ranged from 0.1 – 3.1 d, and was modeled as 1.5 d (Syngenta, 2011). NMFS did not review that document, a drinking water assessment (EPA, 2010) prior to development of the draft Opinion. Prior to development of the final opinion, we requested the drinking water assessment (EPA, 2010) from EPA, received it from EPA, and reviewed it. We have incorporated that new data into the discussion of EECs.

Comparision of EECs

Review of the drinking water assessment (DWA) showed that EPA revised several of the parameters used in modeling for chlorothalonil, including the aerobic aquatic metabolism (EPA, 2010). Additionally, EPA modeled total toxic residues (TTR), including the 4-hydoxy degradate (referred to in the document as SDS-3701). EECs from the drinking water document are for the Index Reservoir, and are not directly comparable to the pond used in ecological risk assessments. In addition to the aerobic aquatic metabolism half-life (now 1.5 d, previously 35.2 d), EPA used revised aerobic soil metabolism half-life values (now 33 d, previously 71 d), revised anerobic aquatic metabolism half-life values (now 151 d, previously 15 d), and a different soil partitioning coefficient (now K_{oc} of 554 mL/g, previously K_d 19.5 (unitless)) (EPA, 2007b, 2010).

The applicant document (Syngenta, 2011) does not take a TTR approach, and uses the 1.5 d aerobic metabolism half-life, but a shorter anaerobic aquatic metabolism half-life (23 d versus 33 d in EPA 2010), aerobic soil metabolism half –life (23 d versus 33 d in EPA 2010), and a higher K_{oc} (3,840 mL/g versus 554 mL/g). The differences in model parameters used in the red-legged frog BE (EPA, 2007b) and the applicant model runs cause a difference in EECs large enough to affect some risk conclusions. Much of the differences in EECs are due to the relationship between the degradation half-life modeled (1.5 d versus 35.2 d) and the application interval modeled (7-14 d, 3-14 applications). Using the shorter half-life (Syngenta, 2011), much of the parent chemical in the EXAMS pond will have degraded prior to introduction of runoff from the next application event. Using the longer half-life (EPA, 2007b), very little of the parent chemical in the EXAMS pond will have degraded prior to introduction of runoff from the next application event. This situation causes the chemical to appear to "accumulate" in the pond, which has no outflow parameter, and EECs are consequently higher.

Differences in the K_d value used in the red-legged frog BE (19.5 unitless, (EPA, 2007b) and the K_{oc} value in the applicant document (3,840 mL/g,(Syngenta, 2011)) also account for some of the differences in EECs, as the higher K_{oc} in the applicant document will drive a greater partitioning of the chemical(s) to the soil compartment, lowering water concentrations.

Unlike the EPA and the applicant, NMFS has not reviewed the basic fate studies underlying the model inputs, thus we make no judgement as to the "correctness" of the various input values. However, we do note there seems to be agreement on the 1.5 d aerobic aquatic metabolism half-life value, and also agreement on the 0.4 d photolysis half-life for parent chlorothalonil. Given these two processes, it appears chlorothalonil will degrade rapidly in most natural aquatic systems.

Considering Monitoring Data

In the ambient water quality monitoring programs, chlorothalonil was rarely detected (0-1.9% of samples) and when it was, the concentrations measured ranged from 0.003 – 0.36 μg/L (Table 135). In a USGS monitoring study focused on evaluating chlorothalonil presence in surface water due to use on soybeans, it was detected more frequently (5.0% of samples) than in ambient monitoring programs, but also at relatively low concentrations (0.004 – 0.43 μg/L) (Scribner et al., 2006). The 4-hydroxy degradate was detected more often in this study (23% of samples, 0.002-0.93 μg/L). The 4-hydroxy degradate is less toxic than the parent. In general, based on available data, it appears that chlorothalonil does degrade quickly in the environment, and NMFS has considered that in our analysis. Some targeted monitoring studies conducted near the application site have reported concentrations similar to the peak farm pond estimates for turf uses. These include average detections of up to 260 μg/L of chlorothalonil in surface water runoff from agricultural test plots (Ryals, et al., 1998; Wauchope, et al., 2004) and a maximum detection of up to 48 μg/L in a stream associated with runoff from golf-course turf (King & Balogh, 2010). The golf course monitoring showed monthly average concentrations of chlorothalonil remained relatively consistent throughout the monitoring period (approximately, 0.1-1 μg/L). Chlorothalonil was predominantly applied to the golf course in late October. The 95th percentile concentrations remained less than 4 μg/L during April through September, when applications were infrequent. The 95th percentile concentration increased to approximately 11 μg/L during November following the predominant application period. NMFS also considered this frequency information in the analysis.

Unlike some of the fast-acting neurotoxins considered in the first three biological opinions (NMFS, 2008c, 2009e, 2010) chlorothalonil's mode of action against fungi is via a depletion of glutathione reserves, and the information we have located regarding higher-level organisms indicates similar pathways. Time-to-response data in Davies and White (1985) shows some 24 h LC_{50}s with higher than 96 h LC_{50}s, and notes several hours before the onset of sublethal effects suggesting shorter duration exposures may have less effect. Thus, a brief pulse of chlorothalonil above the 96 h LC_{50} may or may not cause a mortality event.

Although concentrations of chlorothalonil detected in ambient monitoring are generally <1 μg/L, these concentrations are in the range where prolonged exposure may decrease immune response, causing increased susceptibility to other stressors. Chronic EECs (21-60 d) from both EPA and applicant modeling range from 0.33 – 146 μg/L, depending on the use site and number of applications.

Agricultural Uses

Agricultural peak EECs presented in Syngenta (2011) (Table 2, pg 24) range from 2.1-10.6 μg/L for most crops, and 6.6-16.2 μg/L for cranberries. These EECs barely overlap with the lower end of the salmonid lethality endpoints (10.5-42.3 μg/L), and do not overlap with prey survival endpoints. However, they do exceed the endpoint for immune response (0.8 μg/L). They do not overlap with primary productivity endpoints for either algal plants or vascular plants. Agricultural peak EECs from the red-legged frog BE range from 3-69 μg/L. These overlap with and exceed salmon lethality endpoints, prey survival endpoints, the EC_{50} for algal plants and the cellular damage endpoints. EPA peak EECs for sod farms are considerably higher (274 μg/L) and exceed all assessment endpoints except for in-stream vascular plants. Floodplain estimates exceed all assessment endpoints. We note that floodplain estimates do not account for any degradation processes. We do not anticipate agricultural uses of chlorothalonil will cause degradation of riparian vegetation. Based on available data, we do not anticipate olfactory impairment. We do anticipate some immune suppression type responses may occur based on all estimates and monitoring.

Forestry Uses

Chlorothalonil use in forested areas appears to be primarily associated with nursery operations and commercial Christmas trees farms, according to the applicant (Syngenta, 2011). In an email from the applicant to EPA dated April 1, 2011 (forwarded to NMFS on April 4, 2011), the applicant provided final language they will send in for label amendments to clarify use of chlorothalonil relative to forestry use. According to the applicant conifer use will be spelled out as (1) conifer nursery beds; (2) Christmas tree and bough production plantations; (3) tree seed orchards; and (4) landscape situations.[10] Available use data presents a similar picture, and it appears chlorthalonil is not regularly used in any large-scale forestry applicaitons. In meetings, applicants indicated they plan to submit a fast-track amendment to the label to clarify uses.

[10] In an email to the applicant on May 10, 2011, NMFS sought clarification on the 4th use: “landscape situations” as this phrase seemed vague. The applicant responded on May 11, 2011, to clarify that landscape situations was for “specimen trees in a commercial landscape.”

Forestry (conifer) peak EECs presented in Syngenta (2011) (Table 2, pg 24) range from 8.8-15.2 μg/L. These EECs overlap with the lower end of the salmonid lethality endpoints (10.5-42.3 μg/L), and do not overlap with prey survival endpoints. However, they do exceed the endpoint for immune response (0.8 μg/L). They overlap with primary productivity endpoints for either algal plants but not for vascular plants. Forestry peak EEC from the red-legged frog BE was 18 μg/L. This overlaps with but does not exceed salmon lethality endpoints, and exceeds the EC_{50} for algal plants and the cellular damage endpoints. It does not overlap with prey survival or in-stream vascular plant endpoints. The floodplain estimate exceeds all assessment endpoints. We note that floodplain estimates do not account for any degradation processes. We do not anticipate agricultural uses of chlorothalonil will cause degradation of riparian vegetation. Based on available data, we do not anticipate olfactory impairment. We do anticipate some immune suppression type responses may occur based on all estimates and monitoring data.

Urban Uses

Uses in urban and developed areas include a landscape uses on some decorative plant species (*e.g.*, rose, pachysandra), but are primarily turf uses. Authorized use rates are significantly higher than agricultural uses (11.3 lb ai/A), but are generally limited to two applications per year. We note that aerobic soil metabolism half-lives are dependent on application rate, with longer half-lives associated with higher application rates ((EPA, 2010), Supplement A). Thus, high rate turf uses may result in low concentrations of chlorothalonil in runoff water for longer periods of time than the lower rate agricultural uses.

Turf peak EECs presented in Syngenta (2011) (Table 2, pg 24) range from 6.8 – 7.5 μg/L. These EECs do not overlap with any endpoints except for immune response (0.8 μg/L). EPA peak EECs for turf are considerably higher (115-279 μg/L) and exceed all assessment endpoints except for in-stream vascular plants. Floodplain estimates exceed all assessment endpoints. We note that floodplain estimates do not account for any degradation processes. We do not anticipate urban uses of chlorothalonil will cause degradation of riparian vegetation. Based on available data, we do not anticipate olfactory impairment. We do anticipate some immune suppression type responses may occur based on all estimates and monitoring.

Table 135. Summary of estimated and measured environmental concentrations, and effect concentrations for chlorothalonil.

Use categories		Chlorothalonil (µg ai/L) Peak EECs	Chlorothalonil (µg ai/L) Chronic EECs (60 d)
Agricultural 1-3 lb ai/A most crops 7.3 lb ai/A sod farm (Table 8)	Farm pond[1] (Table 89)	Most crops 3-69[1] Sod farm 274	Most crops 2-43 Sod farm 146
	Floodplain (Table 91)	184-3,490	NA
Forestry 4 lb ai/A (Table 8)	Farm pond (pg 61, RLF BE)	19	12
	Floodplain (Table 91)	1,960	NA
ROW	No ROW uses authorized		
Urban/Developed 11.3 lb ai/A Turf (Table 8)	Farm pond (Table 89)	General 115 Golf course 279	61.6 148
	Floodplain (Table 91)	2,078	NA
Monitoring data		0-1.88% detections; 0.003-0.36 µg/L Targeted monitoring maximum 48.1 µg ai/L turf, 260 µg ai/L in agricultural runoff.	
Assessment Endpoints			
Salmonid survival (LC_{50})		10.5-42.3	NA
Fish reproduction and growth (NOAEC)		NA	3.0
Olfactory-mediated behaviors (2 min EOG response)		>1,000	>1,000
Immune response		0.8	0.8
Endocrine disruption		ND	ND
Swimming		ND	ND
Prey survival (LC_{50})		54-68	NA
Prey reproduction and growth (NOAEC)		NA	0.3
Primary productivity in-stream Algae EC_{50} Vascular plant EC_{50}		 14 630	 14 630
Riparian vegetation (EC_{25})		16 lb ai/A	16 lb ai/A
Ecosystem functioning[a] $NOEC_{eco}$ $LOEC_{ecco}$		ND	ND

ND – No data, NE – Not estimated, NA – Not applicable

[1]Crops estimated at between 3 and 10 applications, generally 7-14 d apart. This appears to be based on max rate allowed per season/year and minimum interval, as label does not specify max number of applications.

Analysis of non-agricultural uses of chlorothalonil

More than 40 companies hold active registrations for pesticides containing chlorothalonil and there are more than 100 products containing chlorothalonil that are currently registered with EPA. The majority of agricultural uses limit the maximum single and seasonal application rates to ≤3 lbs and 20 lbs of chlorothalonil/A, respectively. However, non-agricultural uses allow single maximum application rates of up to 11.3 lbs a.i./A and annual applications rates up to 73 lbs a.i./A (Table 136).

Table 136. Summary of labeled use rates of chlorothalonil products

Use	Max. Single Appliciation Rate (lbs a.i./A)	Annual App. Rate (lbs a.i./A)
Agricultural Crops	1-5	3-23
Golf course	11.3	26 – 73[1]
Lawns around commercial/ Industrial buildings, collegiate and professional athletic fields	11.3	26
Christmas tree plantations and forestry applications	4.1	16.5
1- Annual rate (lbs a.i./A) for fairways, tees, and greens is 26, 52, and 73, respectively		

Exposure to non-agricultural uses of chlorothalonil

To evaluate potential exposure of salmonids to non-agricultural uses of chlorothalonil products we considered the available modeling and water quality monitoring data. As described earlier, the fate and transport models used by EPA and NMFS predict all non-agricultural uses of chlorothalonil can produce concentrations well above median lethal concentrations for salmonids (LC_{50} range is 10.5 – 42.3µg/L). We also considered other model estimates that assessed non-agricultural uses of chlorothalonil found in peer-reviewed scientific journals. TurfPQ is a model designed for assessing runoff of pesticide from turf (Haith, 2001). Vincelli (2004) and Haith and Rossi (2003) used TurfPQ to evaluate runoff of chlorothalonil from golf course turf at several locations in the United States. Both papers predict peak and monthly average concentrations of chlorothalonil that greatly exceeded salmonid LC_{50}s.

Although fate and transport models suggest lethal exposure of chlorothalonil is likely from turf uses, ambient water quality monitoring programs infrequently detect chlorothalonil. Additionally, when detected with ambient monitoring it is typically found at concentrations that are < 1 µg/L. A low detection frequency of chlorothalonil is not unexpected; chlorothalonil degrades rapidly under most aquatic conditions. This rapid degradation reduces the likelihood of detection, particularly when sampling is not spatially and temporally coordinated with field-level application of the compound. Although rapid degradation also reduces the likelihood of exposure, targeted monitoring (edge of treated-field) studies suggest exposure near treatment areas may occur frequently and at concentrations that are detrimental to salmonids. Ryals *et al.* (1998) sampled ponds on three North Carolina golf courses every two weeks from January to December 1994. Chlorothalonil was detected in 84% of the 124 samples analyzed with a maximum concentration of approximately 15 ppb. Based on the laboratory toxicity test with chlorothalonil, this concentration could cause direct acute mortality to salmonids. The average annual concentration found on the three golf courses ranged from 0.3 – 0.8 ppb; laboratory studies indicate that chronic concentrations in this range may cause sublethal impacts to fish (oxygen consumption, increases in glutathione and glutathione-S-transferase).

A long-term research project by King and Balogh (2010)evaluated the losses of chlorothalonil from golf course turf and monitored concentrations of chlorothalonil in a small stream that bisected the golf course. During the 7-year study period (April 2003 – November 2009) the application of pesticide products containing chlorothalonil were documented to quantify discharges associated with prevailing golf course practices. Ten different formulations of pesticides containing chlorothalonil were applied. The average annual application rate for the 22 ha study area was 2.85 lbs/A. Although this rate is well below the maximum annual rates for golf course applications, it’s not directly comparable to the label limitations because the average rate includes both treated (greens, tees, and fairways) and untreated (roughs) segments of the study area. Throughout the study period water samples were collected from two locations, the points where the stream entered (inflow) and exited (outflow) the golf course (King & Balogh, 2010).

Samples were collected from April through November each year by a combination of grab samples and automated sample collection based on flow volume. The median chlorothalonil concentration measured at the outlet of the study area (0.58 µg/L) was significantly greater than the median concentration measured at the inlet (0.03 µg/L) suggesting detections were primarily the result of pesticide uses on the golf course. The mean annual loss of chlorothalonil from the golf course was 0.3% of applied. Although this runoff fraction is quite low, it suggests runoff is a relevant exposure pathway given chlorothalonil application rates.

King and Balogh (2010) found that measured concentrations of chlorothalonil in the stream exiting the golf course periodically exceeded fish LC_{50}s. EPA indicates a salmonid 96 h LC_{50} study submitted by registrants and meeting FIFRA registration guidelines reports a median lethal concentration of 18 µg/L for the rainbow trout (MRID# 45710219, *Appendix 10*). Another study with rainbow trout reports a 96 h LC_{50} of 10.5 µg/L (P.E. Davies, 1985b). A complete data set to evaluate the frequency that lethal concentrations were achieved are not available (King & Balogh, 2010). However, the annual peak concentrations monitored in this study are provided in Table 137. On two occasions (2006 and 2007), annual peak concentrations measured were approximately 3-5 fold lower than the two salmonid LC_{50}s suggesting incidents of acute mortality during those years would be unlikely. The maximum annual concentrations observed in the stream flowing out of the treated golf course exceeded, or were near the salmonid LC_{50}s, in 2003, 2004, 2005, 2008, and 2009 suggesting one or more mortality incidents would likely occur if salmonids where present during these peak runoff events.

Table 137. Peak concentration of chlorothalonil in stream at golf course outflow and corresponding predicted salmonid mortality.

Year	Peak concentration[1]	Predicted mortality of salmonids exposed to peak concentration[2]
2003	13.1	24%
2004	47.2	97%
2005	48.1	97%
2006	3.9	0.4%
2007	3.7	0.3%
2008	21.9	67%
2009	11.9	18%

1- Data provided to NMFS by Kevin King, USDA-ARS, March 15, 2011

2- Assumes 18 μg/L LC_{50} and a probit slope of 4.5

Uncertainty exists when comparing results of laboratory toxicity studies conducted under relatively stable exposure conditions to field exposures which may be characterized by rapid dissipation and pulsed exposures, such as would be expected for chlorothalonil. Both concentration and duration of exposure can influence toxicity. Comparing peak concentrations to laboratory toxicity conducted under stable exposure conditions increases the likelihood of overestimating risk. Whereas, comparing field exposures averaged over time (time-weighted-averages) to laboratory toxicity increases the likelihood of underestimating risk because this approach can mask higher concentrations that may be toxicologically relevant at shorter exposure durations. To evaluate the data using both approaches, we also calculated time-weighted-averages incorporating samples collected immediately after the peak concentration was measured on November 12, 2005. Several applications of chlorothalonil were made to the golf course in October, but the last application prior to the measured peak was on October 26, 17 days earlier. The peak concentration was associated with a rainfall event of approximately 1 inch on November 12, 2005 (King, 2011). The measured concentrations in Table 138 show relatively rapid declines in the stream concentrations, followed by a second pulse of chlorothalonil associated with a second rainfall event of approximately 0.5 inches on November 14, 2005 (King, 2011).

Table 138. Concentrations of chlorothalonil detected in golf course stream (48 μg/L)[1]

Sample collected Date (time)	Chlorothalonil concentration in stream μg/L
11/12/2005 1:06	48.10[2]
11/13/2005 3:57	24.96
11/13/2005 6:37	20.60
11/13/2005 9:10	11.25
11/13/2005 12:26	10.16
11/13/2005 16:56	3.63
11/14/2005 0:41	2.32
11/14/2005 13:27	3.11
11/15/2005 5:12	0.68

11/15/2005 17:18	4.10
11/15/2005 22:43	5.84
11/16/2005 23:48	8.24
11/22/2005 1:14	1.03
11/28/2005 6:29	0.14
11/30/2005 23:31	0.25
1- Data provided to NMFS by Kevin King, USDA-ARS, March 15, 2011 2- Peak concentration measured	

Syngenta suggested that this "data confirm rapid dissipation of chlorothalonil based on flow and environmental fate within ~ 3 hrs after the initial peak concentration and ~ 95% dissipation within 48 hrs (Syngenta, 2011)." Syngenta also suggests that this peak "did not represent a biologically relevant exposure to fish and invertebrates (Syngenta, 2011)." NMFS agrees that the dissipation pattern observed was relatively rapid, but does not agree that the potential exposure is not biologically relevant. NMFS does not agree with Syngenta's assertion that the initial peak had rapidly dissipated within 3 h. The first sample obtained following the peak concentration of 48.11 µg/L was collected nearly 27 h later, and still contained 24.96 µg/L; using these two measurements, the average concentration for the 27 h interval was 36.54 µg/L. This value is comparable to the 24 hr LC_{50} reported by Davies and White 1985 (40.1 µg/L,(P.E. Davies, 1985b)), and less than two fold the 24 hr LC_{50} reported in the registrant submitted LC_{50} (61 µg/L, MRID# 45710219). We also calculated a 96 h time-weighted-average using daily mean values for sampling conducted on November 12-15, 2005. The number of samples collected each day during the 4-day period depended on flow volume and ranged from 1-5. The resulting 96 h average concentration was 16.44 µg/L, which is comparable to the two salmonid LC_{50}s (10.5 and 18 µg/L). In the registrant submitted study, exposure to concentrations of 18 µg/L and greater for 96 hrs caused mortality and marked changes in behavior including swimming at the surface, hanging at the surface, and lethargy. Oxygen levels in the water were 10.0 – 10.1 mg/L, which is optimal for salmonids (MRID# 45710219). Davies and White found that lowering the oxygen concentration in the test system from 8.03 to 5.12 mg/L significantly increased the toxicity of chlorothalonil to salmonids, with 96 hr LC_{50}s of 17.1 and 10.5 µg/L, respectively (P.E. Davies, 1985b). During the 96 hr exposure, "all fish showed marked lethargy on exposure to chlorothalonil…. The degree of lethargy increased with time and concentration of exposure (P.E. Davies, 1985b)." Salmonids showed a loss of startle reaction at concentrations ≥8.7 µg/L. "Loss of startle reaction was followed by reduction of activity and holding at the surface or bottom. A gradual bronzing of skin colour occurred especially in facial patches and around the lateral-line. Reddening at the fin bases was also observed. Permanent lethargy was followed by loss of righting ability and death….all species tested showed complete or partial loss of appetite when offered food at the end of experiments (P.E. Davies, 1985b)."

The data suggest that the concentrations observed by King and Balogh (2010) are biologically relevant to salmonids. Exposure patterns are likely to vary significantly between sites depending on site-specific and application-specific conditions, and may be greater or less than those observed by King and Balogh (2010). Overall, the available evidence suggests that golf course use of chlorothalonil could result in lethal and sublethal effects to salmonids. Additionally, other authorized turf uses may also pose significant risk as they allow comparable use rates (*e.g.*, sod farms and ornamental turf associated with commercial and industrial properties). The relatively high concentrations detected in the golf course studies are not surprising considering modeling estimates for turfgrass and other targeted edge-of-field monitoring. Wauchope et al. (2004) detected considerably higher concentrations in surface water runoff from small peanut test plots associated with much lower application rates; the average concentrations detected in runoff from simulated rain events ranged from 95 – 260 µg/L and were associated with application rates of 1.16 lbs a.i./A. King and Balogh (2010) found that runoff losses and stream concentrations where greatest following applications during periods of soil saturation and precipitation excess. Although chlorothalonil is not particularly mobile, they found it was transportable and detectable in the stream after significant periods free from application. This was attributable to chlorothalonil persistence in the turf environment. A similar pattern was observed in the golf course study by Ryals *et al.* (1998); chlorothalonil was applied to a golf course in August and September with "extremely high levels of chlorothalonil" occurring in a golf course pond in October. The maximum concentration detected during this single year study was 17 μg/L. The authors proposed that despite the compound's limited mobility, the concentrations may be explained due to an accumulation of chlorothalonil in the soils associated with repeated applications and the compounds persistence.

Co-occurrence of listed Pacific salmonids and nonagricultural uses of chlorothalonil

As described in the *Effects of the Proposed Action*, we compared distribution of salmon with distribution of pesticide use sites using GIS. Land use classified as forestry (for forestry uses) and urban/residential (for golf course, sporting complex, and ornamental turf) were used to assess co-occurrence of non-agricultural uses of chlorothalonil products. Forestry is the dominant land coverage type for the majority of the ESUs. However, according to one of the registrants of chlorothalonil, forestry applications are primarily for Christmas tree production (Syngenta, 2011). Assuming this is the case suggests a very small portion of forested areas would be treated with chlorothalonil. Excluding forestry, Syngenta indicates that 65% of non-agricultural use of chlorothalonil are for golf course applications with the other 35% for use on landscape, ornamental, nursery, and sod production (Bang, 2011). These uses are expected to occur primarily in the urban/residential landuse types that cover 1 – 34% of the watersheds where listed salmonids spawn and rear. Additional information on the number of golf courses and proximity to salmon bearing streams was provided by Syngenta (Bang, 2011). The number of golf courses reported within the spawning and rearing habitat of each listed salmonids ranged from 0 – 314, with up to 68 golf courses directly intersecting with waterways where listed salmonids are distributed (Table 139). We also used golf-course address information provided by Syngenta to further evaluate the spatial relationship between golf courses and listed Pacific salmonid habitats (Table 140). The number of golf courses within 2.5 km of spawning and rearing habitat ranged from zero (Ozette Lake and Snake River Sockeye ESUs) to 114 (Central California Coast Steelhead DPS).

Table 139. Co-occurrence of listed Pacific salmonids with potential application of nonagricultural use sites of chlorothalonil within the species freshwater distribution.

ESU or DPS	Spatial coverage of pesticide use sites within species spawning and rearing habitat		Golf courses within ESU or DPS distribution[1]	
	Forest	Urban/ Residential	Total	Number that insect with listed salmonid bearing stream
Puget Sound Chinook Salmon	50.4 %	14.8%	177	52
Lower Columbia River Chinook Salmon	56.8%	13.3%	54	16
Upper Columbia River Spring Run Chinook Salmon	44.9%	4.7%	11	3
Snake River Fall-Run Chinook Salmon	49.2%	1.4%	7	3
Snake River Spring-Summer Run Chinook Salmon	47.7%	1.7%	10	5
Upper Willamette River Chinook Salmon	49.0%	9.0%	83	29
California Coastal Chinook Salmon	63.2%	5.5 %	20	4
Central Valley Spring-Run Chinook Salmon	20.4%	10.8%	78	15
Sacramento River Winter-run Chinook	20.4%	10.8%	78	15
Columbia River Chum Salmon	50.8%	14.9%	39	6
Hood Canal Chum Salmon	61.0%	8.9%	9	0
Lower Columbia River Coho Salmon	59.3%	11.7%	55	25
Oregon Coast Coho Salmon	70.8%	6.6%	29	18
Southern Oregon and Northern California Coast Coho Salmon	67.6%	4.2%	36	9
Central California Coast Coho Salmon	55.6%	9.4%	33	6
Ozette Lake Sockeye Salmon	56.8%	1.1%	0	0
Snake River Sockeye Salmon	57.2%	1.1%	0	0
Puget Sound Steelhead	50.4%	14.8%	177	63
Lower Columbia River Steelhead	61.1%	12.2%	50	23
Upper Willamette River Steelhead	40.3%	10.1%	57	30
Middle Columbia River Steelhead	26.6%	3.3%	34	14
Upper Columbia River Steelhead	33.3%	4.4%	14	6
Snake River Basin Steelhead	52.1%	1.6%	11	7
Northern California Steelhead	68.2%	4.4%	8	6
Central California Coast Steelhead	28.7%	22.1%	161	68
California Central Valley Steelhead	15.9%	9.7%	109	38

ESU or DPS	Spatial coverage of pesticide use sites within species spawning and rearing habitat		Golf courses within ESU or DPS distribution[1]	
	Forest	Urban/ Residential	Total	Number that insect with listed salmonid bearing stream
South-Central California Coast Steelhead	19.9%	9.6%	52	19
Southern California Steelhead	9.3%	33.9%	315[2]	17

1- Assumes golf course occupies area within 3000 foot radius of a the golf course physical address (Bang, 2011). Distance of golf courses to listed Pacific salmonid habitats, as depicted by NMFS, Streamnet, and CalFish GIS data layers. These layers were developed using different methodologies and may not include all smaller streams, tributaries, and conveyances.

2- At the time of the original study, the author did not have access to the updated shape file for this DPS (Bang, 2011). The new values were provided in comments received following the second draft of the Opinion..

Table 140. Estimated distance of golf courses to listed Pacific salmonid habitats.

Species	ESU	Golf Courses within Species Range	Distance to Habitat[1]		Courses Within 2.5 km[1]	
			Avg (km)	Range (km)	Spawning/ Rearing Habitat	Migratory Habitat
Chinook	Puget Sound	177	4.081	0.034 – 38.968	100	NE[2]
	Lower Columbia River	54	2.53	0.172 – 6.598	28	NE[2]
	Upper Columbia River Spring - Run	11	7.78	0.131 – 41.93	9	16
	Snake River Fall - Run	7	5.58	0.263 – 22.488	5	13
	Snake River Spring/Summer - Run	10	4.45	0.460 – 14.953	7	13
	Upper Willamette River	103	5.14	0.005 – 22.45	45	4
	California Coastal	20	6.67	0.169 – 20.683	7	NE[2]
	Central Valley Spring - Run	106	14.71	0.029 – 65.656	22	0

Species	ESU	Golf Courses within Species Range	Distance to Habitat[1]		Courses Within 2.5 km[1]	
			Avg (km)	Range (km)	Spawning/ Rearing Habitat	Migratory Habitat
	Sacramento River Winter - Run	77	19.49	0.135 – 57.06	11	0
Chum	Hood Canal Summer - Run	9	5.87	1.649 – 10.024	1	NE[2]
	Columbia River	42	6.22	0.451 – 23.677	13	NE[2]
Coho	Lower Columbia River	55	1.96	0.106 – 6.294	39	NE[2]
	Oregon Coast	29	0.87	0.003 – 3.097	27	NE[2]
	Southern Oregon and Northern California Coast	36	1.89	0.016 – 9.317	28	NE[2]
	Central California Coast	32	9.46	0.256 – 27.724	8	NE[2]
Sockeye	Ozette Lake	0	-	-	0	0
	Snake River	0	-	-	0	17
Steelhead	Puget Sound	177	3.475	0.040 – 39.346	103	NE[2]
	Lower Columbia River	63	2.90	0.139 – 14.588	42	0
	Upper Willamette River	77	2.65	0.036 – 20.39	49	5
	Middle Columbia River	37	2.33	0.108 – 25.919	31	2
	Upper Columbia River	22	2.93	0.131 – 12.03	17	12
	Snake River	11	2.41	0.067 – 13.784	9	13
	Northern California	8	0.62	0.088 – 1.948	8	NE[2]
	Central California Coast	161	2.09	0.015 – 14.133	114	NE[2]
	California Central Valley	129	5.182	0.029 – 32.096	61	0
	South-Central California Coast	52	2.89	0.051 – 10.659	29	NE[2]
	Southern California	314	5.60	0.042 – 19.473	93	NE[2]

Species	ESU	Golf Courses within Species Range	Distance to Habitat[1]		Courses Within 2.5 km[1]	
			Avg (km)	Range (km)	Spawning/ Rearing Habitat	Migratory Habitat

1- Estimate based on distance between physical address of the golf course and nearest stream within the ESUs distribution. See Appendix 5 for how distribution was determined. Distance of golf courses to listed Pacific salmonid habitats, as depicted by NMFS, Streamnet, and CalFish GIS data layers. These layers were developed using different methodologies and may not include all smaller streams, tributaries, and conveyances.

2- The number of golf courses within the freshwater migratory corridors was not evaluated separately from spawning and rearing habitats because the habitat types completely overlap.

The available information suggests the non-agricultural uses of chlorothalonil are likely to have adverse effects to individual salmonids when exposure occurs. The likelihood of population and species level effects depends partially on the distribution and frequency of exposure throughout the species range. These uses, as well as agricultural uses are further considered in the *Integration and Synthesis* section of the opinion.

Mixture Analysis

More than 50 pesticide products are currently registered that contain one of the six a.i.s and a least one other pesticide. Exposure to multiple active ingredients can cause antagonistic and synergistic responses compared to exposure to a single a.i. The potential for interactive effects associated with exposure to multiple a.i.s in formulation mixtures is highly uncertain, and relatively few toxicity studies on currently registered pesticide formulations containing the six a.i.s have been provided to NMFS for review. Syngenta provided laboratory toxicity studies for rainbow trout, daphnia, and algae with five pesticide formulations containing chlorothalonil (Syngenta, 2011). When LC_{50} and EC_{50} values were normalized based on the percent chlorothalonil, the ranges of toxicity observed for the 5 formulations where comparable to the ranges in toxicity that have been seen in fish, invertebrates, and plants for the single a.i. alone. Consequently, these data suggest that neither the other active ingredient, nor other ingredients in the formulations, caused an increase in toxicity compared to the chlorothalonil alone. Although this information is highly useful, considerable uncertainty exists regarding the toxicity of the remaining chlorothalonil formulations. The test material in each toxicity study provided consisted of 33 – 45% chlorothalonil and one other a.i. that accounted for only 3 - 4% of the total formulation. Whereas, toxicity information has not been provide for other a.i. combinations and formulations that contain much greater relative proportions of other a.i.s. For example, considerable uncertainty remains regarding registration numbers 432-961 (30.5% propamocarb), 42519-30 (38.9% potassium phosphite), 83070-2 (16.66% thiophonate methyl), 67071-17 (19% diuron and 6% 2-N-octyl-4-isothiazolin-3-one), 71711-24 (17.2% flutolanil and 1.8% propiconazole), 74075-1 (12.08% diiodomethyl p-tolyl sulfone), 5905-472 (27.25% sulfur), and others. Uncertainty also remains regarding formulations toxicity of product mixtures containing 2,4-D, triclopyr BEE, diuron, linuron, and captan.

As an example of the potential risk of pesticide product mixtures, we more closely evaluate the risk associated with one of these products formulated with two active ingredients, diuron (62.22% of the formulation by weight) and imazapyr (7.78% of the formulation by weight). Both a.i.s are herbicides so we focus on potential changes to the plant and phytoplankton communities in surface water and the riparian zone. This product is approved for use on a variety of rights-of-way and non-crop agricultural lands. It can be applied at rates of 19 lbs of product per acre. To simulate a rights-of-way application we assumed a single aerial swath was applied by air (Refer to *Appendix 7*). We used AgDrift to estimate the initial average concentration in a floodplain habitat, with and without a 300 foot buffer to the aquatic habitat (Table 141). This buffer was chosen because a 2004 order for injunctive relief requires implementation of no-spray buffers to certain water containing listed salmon in California, Oregon, and Washington (*Washington Toxics Coalition v. EPA*, C01-132C (W.D. Wash. 1/22/2004). Buffers of 60 feet for ground applications and 300 feet for aerial applications are in effect until EPA completes its consultation on diuron.

Table 141. AgDrift estimated concentrations of pesticides in surface water adjacent to aerial application at the maximum labeled use rate for EPA Reg. No. 228-654 (12 lbs diuron/A and 1.5 lb imazapyr/A).

Chemical	Buffer	Average initial concentration (μg/L)	
	Ft	EPA-defined pond	NMFS-defined floodplain habitat
With injunctive relief buffers			
diuron	300	1.002	30.8
imazapyr	300	0.125	3.85
Without aquatic habitat buffers			
diuron	0	25.0	2,789
imazapyr	0	3.12	349
Label prohibits use of product in California			

We also used AgDrift to estimate deposition of the two active ingredients on riparian zone habitat at various distances downwind from the targeted treatment site (Table 142).

Table 142. AgDrift estimated point deposition of diuron and imazapyr at various distances downwind from application site at the maximum labeled use rate for EPA Reg. No. 228-654 (12 lbs diuron/A and 1.5 lb imazapyr/A).

Buffer in Feet	Rate (lbs/A)	
distance from edge of treatment area	Diuron	Imazapyr
10	1.74	0.217
100	0.250	0.031
300	0.028	0.004

Next we compared the concentrations estimated for the floodplain and riparian habitats to toxicity values for aquatic and terrestrial plants. We did not do a thorough search of the toxicity data available for imazapyr but considered the toxicity values used by EPA in their 2006 registration eligibility decision for imazapyr (Table 143). The values for aquatic plants suggest imazapyr is comparable to diuron in its toxicity to vascular plants but is much less toxic to nonvascular plants. However, imazapyr has much greater toxicity to both terrestrial monocots and dicots compared to diuron.

Table 143. Toxicity values of diuron and imazapyr to nontarget plants.

	Concentration (μg/L)	
Assessment Endpoint	Diuron[1]	Imazapyr[2]
Aquatic concentration (μg/L)		
Aquatic vascular plant EC_{50}	15	18
Aquatic nonvascular plant EC_{50}	2.4	11,500
Terrestrial rate (lbs/A)		
Terrestrial monocot EC_{25}		

Emergence	0.099	0.0046
Vegetative vigor	0.021	0.012
Terrestrial dicot EC_{25}		
Emergence	0.075	0.0024
Vegetative vigor	0.002	0.0009
1 – (EPA, 2009b) 2 - (EPA, 2006)		

These data suggest impacts to aquatic plants are likely for diuron in habitats comparable to the EPA-defined pond and NMFS-defined floodplain; estimated concentrations exceeded median toxicity thresholds for both vascular and nonvascular plants, with and without the buffer. Concentrations of imazapyr were greater than the median toxicity threshold for vascular plants when there was not a buffer to the aquatic habitat; imazapyr concentrations did not exceed the median toxicity threshold for nonvascular plants in either estimate, or vascular plants when a 300 foot buffer was simulated. The data also suggest impacts to nontarget terrestrial vegetation are also likely for diuron and imazapyr. Downwind deposition of diuron exceeded EC_{25} values for plant emergence and vegetative vigor in both monocots and dicots at distances of 100 feet and less; there was also some overlap with these endpoints at the 300 foot buffer distance suggesting plant impacts throughout the buffer would be likely. The data suggest even greater risks associated with imazapyr to terrestrial vegetation; all EC_{25} values for plant emergence and vegetative vigor were exceeded in both monocot and dicot plants.

When the toxicity values of the individual a.i.s are considered independently of one another they suggest reduction of primary producers in surface waters and riparian habitats are likely, particularly if the buffers to those habitats are reduced. We expect that the combined exposure to these two active ingredients will amplify these potentially harmful responses. Herbicides are frequently combined into pesticide product mixtures because such mixtures improve the spectrum of weeds controlled and/or increase their effectiveness against target weeds. We expect the same is true of nontarget plants in the riparian area and aquatic habitats; exposure to a mixture of diuron and imazapyr is more likely to cause a reduction in the abundance of primary producers than exposure to diuron alone. It should also be noted that this particular product authorized tank mixtures with a number of other herbicides. This increases the likelihood that exposure of salmonids and their habitat to mixtures will occur, and therefore, adverse responses may be amplified. For example, the label advises "for faster burndown or brown-out of target weeds, tank mix this product with Roundup® or Finale®." It also authorizes tank mixtures with several other herbicides including Oust®, Garlon®, MSMA, Banvel®, Plateau®, and Arsenal®. We assume co-application of these products increase the risk of adverse responses to the riparian zone and aquatic habitats. Further evaluation of the effects of herbicide mixtures and herbicide-induced modifications to riparian and aquatic habitats are discussed in the *Risk Hypotheses* section below.

Evaluation of Risk Hypotheses:

In this phase of our analysis we examine the weight of evidence from the scientific and commercial data to determine whether it supports or refutes a given risk hypothesis. This is not a statistical analysis, but rather a qualitative weighing of the available lines of evidence. We also highlight general uncertainties and data gaps associated with the data. In some instances there may be no information specifically related to a given hypothesis. In some cases, if information on a similar endpoint or chemical is available, and it is reasonable to do so, we extrapolate from the available data to fill gaps, recognizing that this may introduce additional uncertainty in the analysis. If the evidence supports the hypothesis we determine whether it warrants an assessment at the population level. Although six a.i.s are addressed in this Opinion, we recognize the modes of action and toxicities of these compounds vary widely, and have considered them separately through the analysis. In some cases, a group of compounds may be discussed together in this section if toxicities are in a similar range, or the toxicity/exposure profiles are similar.

The available information to characterize pesticide exposure included surface water monitoring data and estimates from pesticide transport models. We combine this information with the distribution and life-history characteristics of listed Pacific salmonids. As discussed in the *Exposure Analysis* section above, each source of information has inherent limitations and uncertainties. For example, the pesticide monitoring data were generally not designed to quantify peak exposure concentrations or distributions of exposure in listed Pacific salmonid habitats. Consequently, models were used to supplement monitoring data and together the information was used to describe the potential range of pesticide concentrations in salmonid habitats. The NMFS AgDrift model runs provided estimates for concentrations resulting from drift to a shallow and narrow body of water, such as those found in floodplain habitats used by listed Pacific salmonids. Small streams and many floodplain habitats are more susceptible to higher pesticide concentrations than larger, high flow systems as their physical characteristics provide less dilution.

We recognize that pesticide concentrations will vary greatly among habitats used by salmonids, and exposure durations will be reduced in flowing water systems where higher velocities occur. There is uncertainty as to what the magnitude of response of fish and salmonid prey will be under different environmental dissipation patterns. Standardized toxicity tests for pesticide registration are poor predictors of real world aquatic ecosystems as fish and other test organisms are exposed to relatively constant pesticide concentrations for arbitrary durations (*e.g.* acute of 96 h and chronic of 21 d) that may poorly reflect field exposures, which tend to be repeated pulses. The response of fish and their prey to different durations of exposure, and exposure mimicking different environmentally relevant dissipation patterns of the six a.i.s, is a prominent data gap. We generally did not average exposure concentrations over time, so called 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 lifestage and its life history are important considerations in its use of aquatic habitats. This natural variation is overlaid with the inherent variation of environmental factors including climate (*e.g.,* precipitation patterns), habitat stressors, and land use. Given this biological and environmental variability, it is difficult to predict the precise exposure to the stressors of the action for any one individual let alone for a population or species.

Consequently, we used general life history information to evaluate potential exposure in the myriad aquatic habitats. For example, all listed Pacific salmon and steelhead occupy habitats that could contain high concentrations of these pesticides at one or more life stages. That said, populations show temporal variation in use of those habitats. Most species use shallow floodplain habitats and/or small streams during their freshwater and estuarine rearing period. These periods of development and growth can differ significantly between species and populations (Refer to *Appendix 6*). Coho, steelhead, sockeye, and stream-type Chinook spend much longer in freshwater systems prior to migrating to the ocean, while ocean-type Chinook and chum spend less time rearing in freshwater. Ocean-type Chinook migrate from their natal stream within 2-6 months of hatching and spend several months rearing in floodplain, estuary, nearshore habitats before continuing on to the open ocean. Chum spawn in side channels, tributary streams, and mainstem rivers. The egg and alevin life stages reside at these sites until they approach or reach the fry stage. Swim-up fry immediately migrate downstream to estuarine areas, where they typically reside near the shoreline for one or more weeks. Thus, a chum fry's freshwater residency period is only a few days, compared with more than a year for other species such as steelhead.

To account for the temporal and spatial variation of aquatic habitats across individuals, populations, and species, we evaluated the potential for individual fitness consequences, (*i.e.,* assessment endpoints) by comparing the range in expected exposure concentrations with adverse effect levels in the context of aquatic habitat utilization.   We divided salmonid habitats into two basic groups.

The first group is composed of spawning and rearing habitats. These freshwater aquatic habitats range from first order streams to large mainstem rivers as well as lakes. They are essential for successful reproduction and for the development and growth of young fish.

The second habitat group is composed of migratory corridors, estuaries, and nearshore marine areas. Most salmonid species use some of these habitats to migrate and rear (feed, develop, shelter), prior to moving into open ocean areas. In general, pesticide exposure will likely be less intense in these areas compared to the other freshwater systems given their size, flow, and use by salmonids. Exceptions include estuaries and nearshore marine environments where juveniles are rearing for extended periods (weeks-months) proximate to high pesticide use areas such as rights-of-ways, agricultural operations near tidal areas and stormwater runoff from dense urban centers.

Although we recognize this as a simplification of the diversity in life histories as well as aquatic habitats used by listed Pacific salmonids, the framework allows us to evaluate risk hypotheses based on differences in habitats and their use by salmonids. We explicitly address species differences in the *Integration and Synthesis* section by evaluating the potential for the stressors of the action to jeopardize the continued existence of the species; or for the potential for stressors to adversely modify their designated critical habitat. Ultimately, for each of the risk hypotheses we make a determination of whether fitness of individuals is sufficiently compromised to warrant an analysis at the population level.

Risk Hypotheses

Here we evaluate the available evidence to determine whether each risk hypothesis is supported. If the available information supports a hypothesis, we analyze the effects at the population scale. If the available information does not support a hypothesis, we do not conduct population-level analyses.

Risk hypothesis 1. Exposure to 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil is sufficient to:

A. Kill salmonids from direct, acute exposure

Species' life history information indicates that listed salmonids are at the greatest risk of exposure to acutely toxic concentrations of the six a.i.s during freshwater occupancy. Salmonids that rear in small streams and floodplain habitats are particularly vulnerable to the highest expected concentrations. We found limited survival data comparing the salmonid lifestages (*i.e.*, eggs, fry, smolts, returning jacks, and returning adults) for the six a.i.s. We identified no survival data for estuarine or marine salmonid life stages. The vast majority of lethality data are based on standard toxicity laboratory tests conducted with juvenile salmonids (predominantly rainbow trout) that determine the LC_{50}. These data show that the six a.i.s have a wide range of LC_{50}s, and that salmonid species tended to be among the most sensitive of the freshwater fish species tested. We relied on these data to evaluate whether expected concentrations of the six a.i.s are sufficient to kill individual salmonids.

Of the chemicals assessed, the two fungicides are the most toxic based on salmonid survival data. They are classified as very highly toxic[11] based on LC_{50} ranges of 26.2-137 μg/L and 10.5-42.7 μg/L for captan and chlorothalonil, respectively. We expect concentrations of these fungicides will reach lethal levels based on the range of toxicity and exposure values derived from monitoring data, EPA's modeling estimates, and NMFS modeling estimates. Model estimates indicate concentrations of captan may reach lethal levels in some aquatic habitats associated with turf, and to a lesser degree agricultural uses. Chlorothalonil is expected to reach lethal concentrations for various agricultural, forest, and residential/urban land uses. EPA incident data document six fish kills that may have resulted from chlorothalonil applications. Although infrequently detected with ambient monitoring, targeted monitoring have shown chlorothalonil concentrations as high as 48 μg/L in surface water (King & Balogh, 2010).

Of the herbicides assessed, triclopyr BEE (LC_{50} 470 μg/L) is highly toxic to salmonids, while 2,4-D esters (LC_{50} 450-14,500 μg/L) and diuron (LC_{50} 710-23,800 μg/L) show variable results that range from highly toxic to slightly toxic. EPA incident data include documentation of seven fish kills incidents for which EPA suggested that 2,4-D was the probable cause, and another six incidents where 2,4-D was characterized as the possible cause. Diuron was listed as the probable cause in five fish kills and the possible cause in another four. No fish kill incidents were reported for trichlopyr BEE when used

[11] EPA uses a descriptive scale for acute aquatic effects: very highly toxic (LC50 <100 μg/L), highly toxic (LC50 100-1,000 μg/L), moderately toxic (LC50 >1,000-10,000 μg/L), slightly toxic (LC50 >10,000-100,000 μg/L), and practically non-toxic (LC50 >100,000 μg/L), as published in Kamrin 1997.

according to label specifications. We expect concentrations of triclopyr BEE and diuron to reach lethal levels in some habitats based on estimated concentrations associated with most registered uses. 2,4-D esters may also reach lethal concentrations when pesticides are applied in close proximity to shallow floodplain habitats and when applied directly to water for aquatic weed control (2,4-D Master Label indicates a target treatment rate of 4,000 μg/L for butoxyethyl ester products). However, non-ester forms of 2,4-D are not expected to cause acute lethality in fish as LC_{50}s are >162,000 μg/L.

Direct applications of 2,4-D BEEto aquatic habitats may result in sufficient concentrations to kill adult fish. However, as described in the *Environmental Baseline*, some states have aquatic weed control programs in place requiring spray timing when there is a lower risk of salmon co-occurrence, which reduces the potential for this outcome.

Other formulations and uses of 2,4-D pose less risk. The available monitoring data, if representative of salmonid habitats, indicated that concentrations of these other forms rarely reach LC_{50} values for most of the compounds in freshwaters. However, the monitoring data do not reflect treatment rates for aquatic weed control and it is unlikely that they reflect peak concentrations for other registered uses. As described in the *Exposure Analysis*, monitoring data are limited when compared to the range of habitats used by salmonids. Few data were found that targeted applications and subsequent concentrations in edge of field habitats which typically show much higher concentrations than weekly, monthly, or seasonal monitoring efforts.

The evidence supports evaluating population-level consequences from reductions in salmonid survival for triclopyr BEE, 2,4-D, and diuron.

Linuron is moderately toxic[1] to salmonids (3,000 μg/L). However, neither exposure estimates nor monitoring data suggest linuron will reach concentrations that are lethal to salmonids.

We expect concentrations of some of the a.i.s in salmonid floodplain habitats will reach lethal levels based on exposure concentrations derived from monitoring data, EPA's modeling estimates, and NMFS modeling estimates (see *Exposure Analysis*). The youngest swimming salmonids appear to be the most likely to die from short-term, acutely toxic exposures in these habitats. It is less likely that adults would be killed by acute concentrations in most freshwater aquatic habitats compared to juveniles. However, if adults are present in smaller floodplain habitats during spray applications or severe runoff events, mortality is possible.

In conclusion, the available information on measured and expected concentrations of the a.i.s supports the risk hypothesis that direct, acute exposure is sufficient to kill salmonids for 2,4-D, triclopyr BEE, diuron, captan, and chlorothalonil, but not for linuron.

B. Reduce salmonid survival through impacts to growth or development.

Fish growth can be affected by pesticides in two ways: by a reduction in somatic processes and by behavior modifications that reduce foraging (NMFS, 2008c, 2009e, 2010). Salmonids are at the greatest risk of reduced growth from pesticide exposure during their fry to smolt life stage where rapid growth is needed in order to survive. The longer salmonids remain in freshwater the greater the probability for pesticide exposure. Juveniles rearing in estuaries and nearshore environments are also susceptible to growth impacts. For most of the listed salmonid species, but especially stream-type Chinook and coho, extended periods of growth occur in shallow, low-flow habitats, including floodplain habitats and small streams. Time to first feeding is a critical lifestage transition period for all salmonids. Following the absorption of the yolk sac, fry need adequate prey upon which to feed and the ability to capture them before the onset of starvation.

We did not identify any studies conducted with the six a.i.s that provided a quantitative relationship between growth and fish survival in the lab or field. However, there is abundant ecological literature showing smaller salmonids have reduced first year survival (discussed in NMFS 2008e).

Information to identify a specific threshold for growth impacts of diuron and linuron in fish were not provided or found in the open literature and are consequently highly uncertain. However, the estimated chronic thresholds for standardized toxicity studies that include growth and other endpoints fall well within the expected concentration ranges for these two compounds. Additionally, diuron and linuron are relatively persistent in the aquatic environment. Therefore, we assume impacts to growth may occur. Laboratory studies with 2,4-D, triclopyr, captan, and chlorothalonil either show no significant effects to growth, or impacts at concentrations that are not expected to occur or persist in aquatic habitats occupied by listed salmonids. Therefore impacts to somatic growth from exposure to 2,4-D, triclopyr, captan, and chlorothalonil are less likely.

One mechanism of reproductive and developmental impairment is through endocrine disruption. However, among the six a.i.s this mechanism has only been evaluated for linuron. As discussed in the *Response* section above, linuron acts as an androgen receptor agonist. We expect that salmonid exposure to linuron will reach concentrations that have been shown to cause endocrine-related effects in fish. We did not find any studies evaluating the reproductive and developmental endpoints of linuron in fish. However, the information presented by EPA suggests that these endpoints may be impaired at expected environmental concentrations. Endocrine disruption has not been evaluated in diuron but it may result in similar responses to linuron given structural similarities. Laboratory studies show that early life stages of fish exposed to environmentally relevant concentrations of diuron suffer increases in developmental abnormalities and death.

A number of reproductive or developmental effects have been seen with chlorothalonil, 2,4-D, and captan. Chlorothalonil exposure to fish has been shown to reduce hatching success and survival at concentrations that are expected and have been measured in surface water. Developmental effects observed with 2,4-D (decreased larval survival) and captan (decreased growth, altered larval and juvenile development) in fish are considered to be less likely given estimated and measured environmental concentrations.

The weight of evidence supports the conclusion that fitness level consequences from reduced size and/or impaired development is likely in rearing salmonids exposed to diuron, linuron, and chlorothalonil. Growth and developmental effects are less likely to occur from the uses of 2,4-D, triclopyr BEE, and captan.

C. Reduce salmonid growth through impacts on the availability and quantity of salmonid prey

This hypothesis focuses on rearing juveniles and the amount of prey available to ensure adequate growth and ultimately, size. As mentioned previously, habitats most vulnerable to pesticide contamination are shallow, low flow habitats where salmonids congregate to feed on a variety of terrestrial and aquatic invertebrates. Other aquatic habitats used by rearing salmonids are also vulnerable to reductions in prey, including channel edges along larger streams, rivers, estuaries, and nearshore marine areas.

We address several lines of evidence to determine the likelihood of reduced salmonid growth from impacts to aquatic invertebrate prey. The first line of evidence we evaluated is whether salmonid prey items are sensitive to acute and chronic exposures from expected concentrations of the six a.i.s. This primarily involved evaluating laboratory experimental results reporting on acute toxicity of the pesticides to aquatic invertebrates. Survival estimates were available for all of the active ingredients but we found data with only standard cladoceran test species for most of the a.i.s. Diuron and chlorothalonil were exceptions to this pattern.

We reviewed diuron toxicity data for a range of relevant taxa including cladocerans, an amphipod, isopod, and a stonefly. We expect initial concentrations of diuron will be sufficient to kill the more sensitive species in some habitats. Additionally, surface water concentrations may remain toxic to these species for extended durations where flow or recharge does not play a significant role in dissipation.

We do not expect chlorothalonil to persist in aquatic environments, although peak concentrations of chlorothalonil are sufficient to kill aquatic invertebrates based on the available standardized acute toxicity information with *Daphnia magna*. Tests with other taxa suggest predicted chronic exposures to chlorothalonil may be sufficient to kill taxa including an isopod, amphipod, crayfish, and a shrimp with EC_{50}s ranging from 3.6 μg/L to >40 μg/L for 4 – 7 d exposure durations.

We also expect initial concentrations of triclopyr BEE and linuron to be sufficient to kill aquatic invertebrates in some habitats. The environmental fate parameters of triclopyr suggest receiving waters will not remain toxic to invertebrates for extended durations. Linuron is more persistent although it can be applied to a relatively limited number of use sites. Although only moderately toxic to aquatic invertebrates, use of 2,4-D butoxyethyl ester at the treatment rate of 4,000 μg/L to control aquatic weeds exceeds aquatic invertebrate EC_{50}s and is thus expected to reduce the availability of aquatic invertebrates. However, we do not expect other uses and other forms of 2,4-D to affect the availability of salmonid prey through direct toxicity. Impacts to salmonid prey from captan exposure also appear less likely given the available toxicity and exposure information.

The second line of evidence evaluated is whether field-level reductions in aquatic invertebrates correlate to use of the six pesticides addressed in this Opinion. Data to assess this line of evidence were lacking. Although we located few microcosm or community studies for the pesticides evaluated in this Opinion, data from other pesticides show shifts in benthic communities - from sensitive mayfly, stonefly and caddisfly taxa, the preferred prey of salmonids, to worms and midges - occur in areas with degraded water quality (T. F. Cuffney, et al., 1997; Hall, Killen, & Anderson, 2006). Recovery of salmonid prey communities following acute and chronic exposures to pesticides depends on the organisms' sensitivity, life stage, and length of life cycle, among other characteristics. Univoltine species will take longer than multivoltine species to recover (Liess & Schulz, 1999). Recovery of high quality salmonid prey items such as caddisflies, stoneflies, and mayflies will be slow, given they have long life cycles and infrequent reproduction.

The third line of evidence we evaluated was whether salmonids showed reduced growth in areas of low prey availability. An evaluation of this line is complicated by multiple factors affecting habitat quality (*e.g.*, water quantity, quality, temperature, riparian zone condition, *etc*.), which in turn affects prey items and salmonids. We were unable to locate information attributing reduced growth in salmonids to prey reduction caused by specific exposure to the six pesticides, as most studies focused on measuring direct effects on salmonids or direct effects on invertebrates. However, there are multiple field experiments and studies that demonstrate reduced fish growth resulting from reduced prey availability (Baxter, Fresh, Murakami, & Chapman, 2007; Brazner & Kline, 1990; Metcalfe, Fraser, & Burns, 1999) or document fish growth rates below maximal potential growth rates when prey are limited (Dineen, Harrison, & Giller, 2007).

Collectively, the lines of evidence support the overall hypothesis that 2,4-D, triclopyr BEE, diuron, linuron, and chlorothalonil can reduce salmonid growth through impacts to the availability of prey. The evidence does not support this hypothesis for uses of captan.

D. Reduce survival, migration, and reproduction through impacts to olfactory-mediated behaviors.

Pacific salmonids rely on olfaction to sense environmental cues that facilitate success in mating, locating food, migration, homing, and avoiding predation. Several classes of pesticides, including herbicides and fungicides, are known to impair olfaction in fish and several studies have shown that pesticides and other contaminants can disrupt olfactory processes that are important for survival and reproduction (Tierney, Baldwin, Hara, Ross, & Scholz, 2010).

We located studies that evaluated Pacific salmon olfactory response to 2,4-D, chlorothalonil and linuron. Reduced fitness from olfactory-mediated behaviors associated with exposure to 2,4-D and chlorothalonil appear unlikely given the available toxicity and exposure estimates (Tierney, et al., 2006). However, we expect concentrations of linuron will reach levels sufficient to impair olfactory responses in salmon. Linuron reduced salmonid response to the odorant L-serine 50-80% following a

30 minute exposure, and in more sensitive species response was significantly inhibited within 15 minutes (Tierney, et al., 2007). Reduction in L-serine responses suggest reduced fitness through diminished predator avoidance, food location, and imprinting (Tierney, et al., 2010). Additionally, environmentally realistic concentrations of a mixture of linuron and other pesticides compromised juvenile steelhead's ability to detect changes in odorant concentrations (Tierney, Sampson, Ross, & Kennedy, 2008). Although data are unavailable for diuron while data with linuron support this hypothesis, assuming similar potencies, and giving the benefit of the doubt to the species, NMFS assumes that these similar responses will also occur when salmonids are exposed to diuron.

We located no studies directly measuring olfactory-mediated behavioral responses of fish following exposures to triclopyr or captan. This recognized data gap introduces uncertainty as to whether these a.i.s impair olfaction and, if so, at what concentrations effects might occur.

The weight of evidence supports olfactory-mediated behaviors will be impaired from exposures to linuron and diuron and will reduce fitness for individuals. Impairment of olfactory response is less likely or more uncertain for 2,4-D, triclopyr BEE, captan, and chlorothalonil.

Risk hypothesis 2. Exposure to 2,4-D, triclopyr BEE, diuron, and linuron is sufficient to:

A. Reduce aquatic primary producers thereby affecting salmonid prey communities and salmon

Herbicides such as 2,4-D, triclopyr BEE, diuron, and linuron are used to remove unwanted vegetation. Plants and other primary producers are an essential component of productive salmonid habitats because they provide food resources to aquatic invertebrates and provide shelter for invertebrates and fish. Ecosystem studies show that herbicides have variable effects following reductions in primary producers, however impacts to fish through trophic level interactions can occur, particularly in systems that are dominated by sensitive plants.

We expect 2,4-D, triclopyr BEE, diuron, and linuron will reduce primary producers and have fitness consequences for some salmon given the toxicity information available on ecological responses and the estimated and measured concentrations of these herbicides. However, the response will depend on site-specific conditions and are difficult to predict. In part, the response is dependent on the relative tolerance of the existing primary producers and the corresponding shifts that can occur to more tolerant and potentially less desirable prey species. Ecosystem recovery may occur relatively rapidly where there is functional redundancy with more tolerant species. However, recovery of invertebrates and fish may be prolonged in situations where the dominant vascular plants are sensitive. Diuron and linuron are photosynthetic inhibitors and broad spectrum herbicides that are toxic to a broad range of primary producers. 2,4-D and triclopyr BEE are auxin-simulators, and vascular plants and other macrophytes tend to be more sensitive than other primary producers like algae.

Collectively, the available lines of evidence support the overall hypothesis that the four herbicides cause adverse effects to salmonids and their habitat through reductions in primary production and subsequent reductions in the availability of prey, particularly for the uses of diuron, linuron, and triclopyr BEE. For all forms of 2,4-D, direct water applications to macrophyte-dominated systems are most likely to cause changes to the abundance and composition of primary producers, and consequently changes in the quality and quantity of salmonid prey. As discussed above, whether these responses are detrimental will depend on site-specific conditions. This hypothesis is also supported for some terrestrial applications of all forms of 2,4-D immediately adjacent to shallow habitats.

B. Reduce riparian vegetation to such an extent that stream temperatures are elevated, erosion increases, and reduction in inputs of woody debris and other organic matter occurs.

This risk hypothesis considers aquatic habitat changes due to potential herbicide impacts to riparian vegetation. Possible changes to salmonid habitat associated with modifications to the riparian zone include alterations in terrestrial input of organic matter (including leaf litter, woody debris, and terrestrial insects); increased input of contaminants due to decreased vegetative filtering; reduced maintenance of natural flow dynamics; decreased bank stability and associated increased erosion and sedimentation; and decreased shading and increased stream temperatures.

We are not aware of any studies that specifically evaluated aquatic habitat responses that may correspond with changes to the riparian habitat from these four herbicides. However, we expect 2,4-D, triclopyr, diuron, and linuron will alter riparian vegetation either from direct application to riparian habitats or, to a lesser extent, from transport of the herbicides to riparian habitats. 2,4-D, triclopyr BEE, diuron, and linuron are all systemic herbicides that can be taken up by the roots or leaves of plants and can be spray applied. All four herbicides may be transported to riparian habitats through drift and runoff. 2,4-D and triclopyr BEE are selective herbicides expected to reduce woody vegetation and broadleaf plants. Diuron and linuron are non-selective (broad spectrum) herbicides for control of broadleaf plants, grasses, algae, and moss. Given their broad spectrum action on plants, diuron and linuron are expected to have more severe impacts to riparian habitats compared to 2,4-D and triclopyr BEE. However, riparian zone impacts are most likely to occur from direct applications of herbicides. Although linuron is not permitted for riparian habitat use, diuron applications to riparian habitats may occur as labels allow for "general weed control" to non-crop and non-timber areas. The general weed control use site also allows for direct application of diuron to intermittently flooded marshes, swamps, and bogs after water has receded. We expect diuron to significantly reduce vegetation in riparian habitats and intermittently flooded habitats when applied at the labeled use rate (12 lbs a.i./A).

2,4-D and triclopyr BEE will modify riparian habitats when these chemicals are applied directly to riparian zones. However, changes to the riparian zone habitat may or may not result in negative responses to salmonid habitat. Removal of woody vegetation can reduce shading and consequently increase stream temperatures. It can also decrease bank stability, which may result in increased erosion and sedimentation detrimental to salmonids, and reduce inputs of organic material that are beneficial to the aquatic community. Alternatively, removal of woody vegetation could increase growth of understory vegetation and improve the contaminant filter capacity of the riparian zone which may be beneficial. Whether riparian zone applications of 2,4-D and triclopyr BEE are beneficial or detrimental to the aquatic habitat and salmonids will depend on site specific conditions.

Overall, the weight of evidence suggests that 2,4-D, triclopyr BEE, diuron, and linuron can cause alterations to riparian zones that may result in adverse effects to aquatic communities that affect the individual fitness of salmon.

Risk hypothesis 3. Exposure to mixtures of diuron and linuron can act in combination to increase adverse effects to salmonids and salmonid habitat.

We are not aware of any data that directly assess mixtures of diuron and linuron. However, these compounds are structurally very similar, have a common mode of action, and produce several common degradates. Therefore, we reasonably assume they can act in combination to increase adverse effects to salmonids and salmonid habitat.

Risk hypothesis 4. Exposure to other stressors of the action including degradates, additional active ingredients, and inert/other ingredients in pesticide products and tank mixes cause adverse effects to salmonids and their habitat.

In addition to exposure to the a.i.s, salmonids and their habitat are likely exposed to other stressors of the action, including degradates and additional active ingredients in formulated products and tank mixes. Salmonid habitats may also be exposed to a number of the approximately 4,000 inert ingredients approved for use in end-use pesticide products by EPA, as well as adjuvants, such as surfactants and other products that are applied as tank mixtures. Once the mixture (formulated pesticide or tank mix) is introduced into the environment, physiochemical properties of the various compounds will cause them to move through the environment at different rates and partition into different compartments. We expect some percentage of these other stressors will be present in salmonid habitats from spray drift deposition, and from runoff events following application. Salmon and their habitats exposed to these multiple stressors are expected to show a greater response than laboratory animals exposed only to one a.i, thus available toxicity data generally underestimate the response in a field-applied pesticide mixture.

A. Exposure to degradates of 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil

The BEs identify many of the degradates of the six a.i.s. However, estimates quantifying potential exposure of listed salmonids and their habitat to these transformation products were not provided. Information on the toxicity of these compounds was reported for few of the degradates and remains a considerable source of uncertainty. We found little information to supplement the BEs. Exposure to some of the environmental degradates may increase adverse effects to salmonids and their habitat. Although toxicity and exposure data are limited for many of these degradates, some data suggest these stressors can add substantial risks. For example, results from an experiment in which a mesocosm was exposed to 3,4-dichloroaniline (DCA), a degradate of both diuron and linuron, suggest it may cause persistent toxicity and ecological effects (Maund, et al., 2009). In general, exposure to degradates of the a.i.s is expected to increase adverse effects to salmonids and their habitat.

B. Additional Active Ingredients

As discussed in the *Mixture Analysis* section above, pesticide products containing multiple a.i.s are common. While the a.i.s will move through the environment at different rates, it is reasonable to believe that all a.i.s in a given pesticide formulation will co-occur in receiving waters, especially from drift deposition and in the first runoff from the field following application. Examples of some of the formulation mixtures include up to 4 herbicides or fungicides. The potential interactive relationships that exist among most of these a.i. combinations have not been defined and are uncertain. Others are likely to cause additive or synergistic effects compared to the single a.i. alone. For example, diuron and linuron likely cause additive effects in plants given their similar structure and mode of action (EPA Reg. No. 352-660). Another product contains captan and two cholinesterase inhibitors, malathion and carbaryl. Some of these a.i.s may be more acutely toxic than the a.i.s specifically considered in this Opinion, although they may have a different mode of action. Others may not be as acutely toxic, but may cause reproductive effects, bioaccumulate, or otherwise adversely affect the salmon or their environment in some way.

2,4-D, triclopyr BEE, diuron, linuron, and captan all provide recommendations for tank mixtures that contain additional a.i.s. It is reasonable to assume co-location of a.i.s that are co-applied will occur in the aquatic habitats. Specific interactions between additional a.i.s in products and tank mixes and the a.i.s addressed in this Opinion are mostly unknown, but it is reasonable to assume toxicity of the a.i.s may be enhanced. In general, exposure to other active ingredients in pesticide products and tank mixes is expected to increase adverse effects to salmonids and their habitat.

C. Inert/other ingredients

In addition to a.i.s, pesticide products contain other ingredients which are sometimes referred to as the inert ingredients. Some of these ingredients are toxic to aquatic organisms or increase the toxicity of the active ingredients. As with tank mixes, the likelihood of these compounds co-occurring in the water column is difficult to determine with any specificity, but can reasonably be presumed to occur in spray drift deposition and runoff following applications. The other ingredients may make up the majority of the pesticide formulation, but few are required to be specifically identified by pesticide labels. Examples of these ingredients are the nonylphenol polyethoxylates, which have been linked to endocrine disruption and were addressed at length in previous Opinions on EPA pesticide registrations (NMFS, 2008e, 2009e, 2010). There are a myriad of other ingredients, some of which may increase the toxicity of the a.i.s. The majority of a pesticide formulation is often composed of inert ingredients. Consequently, salmonid exposure to these ingredients may be greater than exposure to the assessed active ingredient. EPA currently has no specific method of accounting for this potential additional toxicity and risk, but it cannot be ignored. NMFS has opted to address the uncertainty associated with these ingredients in a qualitative sense. Collectively, the available lines of evidence support the overall hypothesis that other stressors of the action cause adverse effects to salmonids and their habitat.

From our review of the available information it is not possible to accurately quantify the contribution of other stressors of the action. These stressors include the additional a.i.s and inert/other ingredients in pesticide formulations as well as tank mixes. These stressors of the action are an important consideration when assessing potential effects on listed salmonids and their habitat. Thus, to provide the benefit of the doubt to the species, we assume these stressors of the action will contribute additional, unquantifiable reductions in fitness to individuals beyond that of 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil.

Risk hypothesis 5. Exposure to other pesticides present in the action area can act in combination with the six a.i.s to increase effects to salmonids and their habitat.

Environmental mixtures of pesticides are common. We found few data evaluating the response of aquatic species to mixtures containing the six a.i.s. Toxicity investigations with pesticide mixtures reveal that responses of aquatic species are variable, and will depend on the composition of the mixture, concentrations of the a.i.s, modes of action and duration of exposure. Additionally, sequential exposures from other pesticides in the action area are reasonably expected to increase effects to salmonids and their habitats if and when they impact the same environmental receptors. For example, in addition to 2,4-D, triclopyr BEE, diuron, and linuron there are many other herbicides used in the action area that may further reduce primary production. Therefore, based on the available toxicity and exposure data, we assume exposure to other pesticides present in the action area will act in combination with the six a.i.s to increase the effect to salmonids and their habitat.

Risk hypothesis 6. Exposure to elevated temperatures can enhance the toxicity of the stressors of the action.

We reviewed the available information to determine whether empirical data indicated enhanced toxicity at elevated temperatures for the a.i.s assessed in this opinion. However, we located only one study, with chlorothalonil, that assessed the influence of temperature on the chlorothalonil response (P.E. Davies, 1985c). That study reported lower LC_{50}s at higher temperatures, although the LC_{50}s were relatively similar for the range evaluated (10-16°C). Higher water temperatures can increase the metabolic rate for fish, thus increasing the rate at which they process the toxicant. Depending on the chemical, this may be either beneficial or detrimental. Water temperatures higher than optimum also increase general physiological stress for salmonids making them more susceptible to other stressors. Sufficient data to support or refute this hypothesis are lacking. Therefore we did not assume that elevated temperature would enhance the toxicity of the stressors of the action. A summary of effects on individual fitness for each of the a.i.s is presented in Table 144.

Table 144 Summary of individual-based risk hypotheses.

Risk Hypotheses	Is individual fitness of exposed salmonids compromised?					
	2,4-D	Triclopyr BEE	Diuron	Linuron	Captan	Chloro-thalonil
1.A. Kill salmonids from direct, acute exposure	yes	yes	yes	no	yes	yes
1. B. Reduce salmonid survival through impacts to growth or development	no	no	yes	yes	no	yes
1. C. Reduce salmonid growth through impacts on the availability and quantity of salmonid prey	yes	yes	yes	yes	no	yes
1. D. Reduce survival, migration, and reproduction through impacts to olfactory-mediated behaviors	no	no	yes	yes	no	no
2.A. Reduce aquatic primary producers thereby affecting salmonid prey communities and salmonids	yes	yes	yes	yes	no	no
2.B. Reduce riparian vegetation to such an extent that stream temperatures are elevated, erosion increases, and reduction in inputs of woody debris and other organic matter occurs	yes	yes	yes	yes	no	no
3. Exposure to mixtures of diuron and linuron can act in combination to increase adverse effects to assessment endpoints	no	no	yes	yes	no	no
4. Exposure to other stressors of the action including degradates, additional a.i.s and inert/other ingredients in formulations and tank mixes cause adverse effects to salmonids and their habitat	yes	yes	yes	yes	yes	yes
5. Exposure to other pesticides present in the action area can act in combination with the six a.i.s to increase effects to salmonids and their habitat	yes	yes	yes	yes	yes	yes
6. Exposure to elevated temperatures can enhance the toxicity of the stressors of the action	no	no	no	no	no	no

Risk Characterization: Evaluation of Critical Habitat Risk Hypotheses

We use the toxicity information presented earlier in the *Effects of the Proposed Action* section to evaluate the scientific lines of evidence that support or refute risk hypotheses developed for critical habitats. We determined that freshwater spawning and rearing sites, migration corridors, estuarine areas, and nearshore marine areas within designated critical habitats are likely to be exposed to the stressors of the action over the 15-year registration duration. We will discuss, and when possible, estimate expected concentrations and durations of exposure for these habitats based on pesticide use information, surface water monitoring data, EPA modeling estimates, and NMFS modeling estimates. For each risk hypothesis below we qualitatively weigh the evidence to determine whether the PCE attributes of water quality and/or prey availability are affected. We ultimately determine whether the degradation of water quality and prey availability within freshwater spawning habitat, freshwater rearing habitat, migration corridors, estuarine areas, and nearshore marine areas will rise to the level expected to reduce the intended conservation role of designated critical habitats - evaluated within the *Integration and Synthesis for Designated Critical Habitat* section. The final conclusion of whether EPA's proposed actions are likely to adversely modify or destroy a species' designated critical habitat is provided in the *Conclusion* section.

Risk hypothesis 1. Exposure to the stressors of the action is sufficient to degrade water quality and substrates in freshwater spawning sites.

Freshwater spawning sites require water quality and substrate conditions that support spawning, incubation, and larval development. The degradation of water quality by exposure to the stressors of the action is indicated by the toxic responses in a variety of aquatic organisms including listed salmonids (see section on *Integration of Exposure and Response* above). 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 six 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 are in shallow, low flow systems located in high pesticide use areas such as intensive agricultural or urbanized watersheds, and consequently experience multiple applications of the a.i.s.

Contamination of spawning sites by the stressors of the action can degrade water quality in several ways. Exposure to the stressors of the action can impair spawning behaviors as well as kill spawning adults thereby diminishing the quality of this critical habitat. In vulnerable habitat, expected concentrations of 2,4-D, triclopyr BEE, diuron, captan, and chlorothalonil are sufficient to kill a percentage of spawning adults. Other spawners may experience impaired olfaction from exposure to diuron and linuron which leads to reduced ability to detect spawning olfactory cues. Other ingredients and degradates of the six a.i.s may also result in degraded water quality. Expected concentrations of the four herbicides are sufficient to kill primary producers and reduce primary production. This can result in a number of undesirable water quality conditions including significant increases in nutrient levels, reduced dissolved oxygen, reduced pH, higher alkalinity, and higher turbidity and conductivity (T.C.M. Brock, et al., 2000).

Degradation of substrate is also expected due to likely impacts of the four herbicides to riparian vegetation ranging from alterations to the composition of plant communities up to complete plant removal (*e.g.* removal of woody species or emergent vegetation). Alterations or removal of riparian vegetation typically results in decreased bank stability, increased erosion, and increased deposition of fine sediments into the stream channel. Salmon require well oxygenated gravel substrates for successful spawning. We expect the use of these herbicides in and around riparian habitats may result in degraded substrates by increasing gravel imbeddedness in freshwater spawning sites.

Collectively, the overlap of spawning sites with application areas combined with expected concentrations and toxicity effect thresholds to aquatic organisms and riparian vegetation indicates that degradation of the water quality of the spawning PCE is likely for the six a.i.s and adverse effects to substrate are likely for the four herbicides. We evaluate whether the degradation of this PCE, sites for spawning, in combination with other affected PCEs reduce the conservation value of the 26 designated critical habitats within the *Integration and Synthesis for Designated Critical Habitat* section.

Risk hypothesis 2. Exposure to the stressors of the action is sufficient to degrade water quality, natural cover, and/or reduce prey availability in freshwater rearing sites.

Freshwater rearing sites need to provide good water quality, abundant forage, and cover to support juvenile development. Reductions in any of these attributes can limit the existing and potential carrying capacity of rearing sites and subsequently reduce their conservation value. Recovery of listed salmonid populations is tied closely to the success of juveniles to fully develop, mature, and grow during freshwater residency periods. All species of Pacific salmonids spend some amount of time in freshwater feeding and rearing areas. Chum salmon use 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, and complex sites that can range from small, shallow, intermittent floodplain habitats to channel edges of large river systems. As such, expected concentrations range from some of the highest estimates (via spray drift into floodplain habitats) 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. Expected floodplain concentrations of the six a.i.s are shown in the *Exposure and Response Integration* above. At these levels, water quality would be affected from the six a.i.s based on concentrations that exceed toxicity thresholds for lethality to fish (all except linuron), lethality to aquatic invertebrates (all except captan), impacts to olfactory senses in fish (diuron and linuron), cellular damage in fish (captan and chlorothalonil), and damage to plankton and aquatic macrophytes (2,4-D, triclopyr BEE, diuron, and linuron). As discussed above in *Risk Hypothesis 1*, a number of water quality parameters (*e.g.* nutrient concentrations, DO, pH, *etc.*) will be modified by the instream concentrations of the four herbicides in floodplain habitats. Additionally, we expect herbicide modification to riparian vegetation will result in decreased shading of these habitats and an increase in the occurrence of temperatures that are stressful to salmonids in freshwater rearing sites.

Triclopyr BEE, diuron, linuron, and chlorothalonil are expected to reach levels that reduce prey abundance. The laboratory toxicity data and exposure estimates indicate that salmonid prey would be severely affected if the lower end of the survival range (shown in the *Exposure and Response Integration* above) is representative of salmonid prey communities. Direct application of 2,4-D butoxyethyl ester at the maximum labeled rate of 4,000 μg/L is also expected to reduce prey through direct toxicity, while other uses of 2,4-D are not. We located few data on the response of real world prey communities to the six a.i.s. We assume that many of the salmonid prey items are either as sensitive as or more sensitive than the standard toxicity test organisms for which data exist. This assumption is supported, in part, by comparative data showing that caddisflies, stoneflies, and mayflies are typically more susceptible to pesticide toxicity than laboratory-reared freshwater invertebrates (K. R. Johnson, Jepson, & Jenkins, 2008; J. L. Peterson, Jepson, & Jenkins, 2001).

The IBI and other metrics of aquatic community health were reviewed. In areas of intensive agriculture, where we expect use of the stressors of the action, biological integrity is often significantly reduced (T. R. Cuffney, M. R. Meador, S. D. Porter, & M. E. Gurtz, 1997). Many of the preferred salmonid prey items are present only in low numbers or absent altogether in these areas. We see similar depauperate communities in urban areas. We recognize many other limiting factors contribute to the poor condition of these aquatic communities. However, these six a.i.s and their formulations may be responsible for a portion of these reductions.

We also expect that in some situations, the herbicidal action of 2,4-D, triclopyr BEE, diuron, and linuron will alter characteristics of the plant communities in freshwater rearing sites decreasing the availability of food or cover for salmonids and their prey. Natural cover that would reasonably be reduced from these four herbicides includes shade and aquatic vegetation (based on exposure and toxicity to aquatic and terrestrial plants). In cases where woody vegetation is eliminated from riparian zones to the extent that bank destabilization occurs, altered stream hydrology could affect the availability of other cover including rocks, side channels, and undercut banks. One extreme example suggested that the application of an herbicide to a riparian zone caused major long-term changes to the hydrology of a stream and degraded the fish habitat (http://water.epa.gov/scitech/datait/tools/warsss/streamero.cfm). Herbicide-induced changes to vegetative communities in the riparian zone and aquatic habitat also have implications for the availability of prey as salmonids consume both terrestrial and aquatic insects. As plant communities are modified in riparian zones and aquatic habitats, the species that rely on them will also be affected. Habitat responses in these environments are expected to be variable and will depend on the sensitivity of existing plants to 2,4-D, triclopyr BEE, diuron, and linuron.

Studies of responses in test ecosystems indicate the selection pressure from exposure to herbicides can result in reductions to primary production and changes in the composition of primary producers that can adversely impact higher trophic levels. Brock et al. (2000) concluded in their review of herbicides that indirect effects of photosynthetic inhibitors (e.g. diuron and linuron) on consumers and predators occur at concentrations around the EC_{50} for standard algae taxa. We expect these impacts to occur in freshwater rearing areas given the available exposure and toxicity information. We also expect other effects on the ecosystem such as blooms of insensitive algae that can occur at lower concentrations (e.g. 0.1 of the EC_{50} of standard algae), and can be a delayed response. The Brock et al. (2000) review of ecosystem impacts indicates that the most dramatic ecological responses may occur with auxin-simulators (*e.g.* 2,4-D and triclopyr) because aquatic macrophytes are more sensitive than algae. Vascular plants and other macrophytes are important structural elements that provide important cover to salmonids in their preferred freshwater rearing areas. Aquatic vegetation provides cover to salmonids from avian predation as well as protection from larger fish. In addition, a diverse vascular plant community provides important substrate for an array of insect species upon which young salmon prey. A shift from a macrophyte-dominated community to a plankton-dominated community will reduce the natural cover PCE attribute in freshwater rearing areas and

> "may lead to drastic shifts in the aquatic community by habitat destruction. The organisms for which aquatic plants play an important role include periphyton, crustaceans, aquatics insects (especially larvae), mollusks, but also certain fish and tadpole species. All these groups may disappear or decrease in numbers as a result of

> mortality of macrophytes…[and] the disappearance of the vegetation may in some cases lead to a bloom of phytoplankton (T.C.M. Brock, et al., 2000)."

Although such shifts can lead to a temporary increase in primary production that are advantageous to some species, it is expected to result in considerable indirect effects to macrophyte-associated cover and prey resources important to salmonids in freshwater rearing sites. Reductions and removal of cover and prey biomass in floodplain and other habitats that support rearing may substantially reduce this PCE's role in recovering salmonid populations. Concentrations of the four herbicides in other freshwater habitats that support rearing are also expected to reach levels that reduce both water quality and prey abundance.

Collectively, substantial data indicated that expected concentrations of the six a.i.s are sufficient to adversely affect water quality; 2,4-D, triclopyr BEE, diuron, linuron, and chlorothalonil are sufficient to adversely affect salmonid prey (forage); and concentrations of the four herbicides are expected to degrade the natural cover attribute of freshwater rearing PCEs. Therefore, we evaluate these effects in order to determine whether the conservation value of species' designated critical habitats will be reduced (see *Integration and Synthesis for Designated Critical Habitat* section below).

Risk hypothesis 3. Exposure to the stressors of the action is sufficient to degrade water quality, 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, reduces cover, and/or further impedes the mobility and survival of juveniles and adults. Expected contaminant concentrations associated with the use of products containing 2,4-D, triclopyr BEE, diuron, linuron, and chlorothalonil may limit prey availability in migratory sites where juveniles pause to rest and feed during their migration to the ocean. Additionally, exposure of riparian and aquatic habitats to the four herbicides may reduce the availability of natural cover such as shading and aquatic vegetation.

Rest areas such as undercut banks, side channels, submerged and overhanging large wood, log jams, and beaver dams are 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 natural cover or prey resources due to the degradation of water quality at these rest areas may cause migrating juveniles to continue downstream thus avoiding needed rest and food, ultimately affecting their health and ability to successfully transition to saltwater environments. Many of these rest areas are located in places where water flow is reduced compared to the main channels. Stressors of the action may persist longer in these areas due to reduced flow.

Additionally, channel-edge habitats that are proximate to applications of the stressors of the action are at risk, increasing the probability of exposure to high concentrations from drift and runoff following application events. Many migratory sites overlap with some of the highest use areas for the stressors of the action such as intensive agricultural valleys. Based on the size, flow rate, and proximity to application sites, exposure durations and concentrations within migratory habitats are expected to be highly variable. That said, we expect uses of the six a.i.s will at times lead to concentrations that will reduce salmonid prey, natural cover, and/or degrade water. The degradation of water quality within migratory sites may affect the mobility of juveniles and adults exposed to diuron and linuron by impairing their olfaction. The 6 a.i.s are expected to degrade water quality and the four herbicides are expected to degrade natural cover.

Collectively, the available data indicated that expected concentrations of the six a.i.s are sufficient to adversely affect water quality; concentrations of 2,4-D, triclopyr BEE, diuron, linuron, and chlorothalonil are sufficient to reduce the availability of salmonid prey (forage); and concentrations of the four herbicides are sufficient to reduce the availability of natural cover 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, natural cover, and/or reduce prey availability in estuarine areas.

Estuarine areas require good water quality to support juvenile and adult physiological transitions between fresh water and salt water as well as to provide juvenile and adult cover and prey resources sufficient to support survival, growth, and maturation. Prey resources for Pacific salmonids within estuaries include a diverse group of organisms - from aquatic invertebrates to small fishes depending on the size of the salmonid. The allowable uses of the stressors of the action overlap with estuaries designated as critical habitat.

All of the a.i.s are allowed for use in estuarine-containing watersheds. Contamination of estuaries occurs via drift, runoff, and atmospheric deposition. Streams and rivers flowing into estuaries act as conveyor belts as they transport the stressors of the action from areas higher in watersheds (O. W. Johnson, et al., 1997). We located no estuarine monitoring data specific to the stressors of the action. This is a large data gap, as the available exposure data derived for freshwater habitats (EPA modeling estimates, NMFS modeling estimates and monitoring data) are not representative of estuarine habitats. Pacific estuaries are incredibly variable to one another; size, tidal volume, exchange rate, freshwater input, salinity, watershed land uses, trophic structures, bathymetry, etc., influence and shape estuarine ecosystems (Salo, 1991). Estuaries remain dynamic, complex systems that are not completely understood. As such, predictive models are not available to estimate concentrations and dissipation rates of pesticides within estuaries, and therefore we assume that use of the a.i.s within estuarine-containing watersheds may cause contamination.

The available toxicity information for estuarine and marine organisms for the a.i.s is presented in Table 145. The majority of aquatic toxicity data are from survival assays for the sheepshead minnow (fish) and mysid (estuarine invertebrate). The available studies indicate a similar range of sensitivity to the six a.i.s between the marine and freshwater species test species. Consequently, we expect a similar ecological response when exposure is comparable.

Table 145. Assessment endpoint toxicity values (µg/L) for saltwater aquatic organisms presented in salmonids BEs.

Assessment endpoint	Assessment measure	Range (µg/L)
2,4-D (all forms)		
Fish Survival	estuarine, and marine fish LC_{50} (96 h)	3,200 – 465,000
Fish reproduction or larval survival	NOEC-LOEC	56 – 79 (2,4D-BEE)
Invertebrate survival	estuarine and marine EC_{50}	1,800 – 401,800
Aquatic primary producer	EC_{50} NOEC	660 - 123,300 538 - 80,000
Triclopyr BEE		
Fish Survival	estuarine, and marine fish LC_{50} (96 h)	450 - 760
Fish reproduction or larval survival	NOEC/LOEC	no studies reported
Invertebrate survival	estuarine and marine EC_{50}	320 – 2,470
Aquatic primary producer	EC_{50} NOEC	1,170 209
Diuron		
Fish Survival	estuarine, and marine fish LC_{50} (96 h)	6,300 – 6,700
Fish reproduction or larval survival	NOEC/LOEC	<440
Invertebrate survival	estuarine and marine EC_{50}	1,100 – 5,000
Aquatic primary producer	EC_{50} NOEC	no studies reported
Linuron		
Fish Survival	estuarine, and marine fish LC_{50} (96 h)	890
Fish reproduction or larval survival	NOEC/LOEC	no studies reported

Assessment endpoint	Assessment measure	**Range** (µg/L)
Invertebrate survival	estuarine and marine EC_{50}	890 – 5,400
Aquatic primary producer	EC_{50} NOEC	no studies reported
Captan		
Fish Survival	estuarine, and marine fish LC_{50} (96 h)	1,900
Fish reproduction or larval survival	NOEC/LOEC	no studies reported
Invertebrate survival	estuarine and marine EC_{50}	no studies reported
Aquatic primary producer	EC_{50} NOEC	no studies reported
Chlorothalonil		
Fish Survival	estuarine, and marine fish LC_{50} (96 h)	32
Fish reproduction or larval survival	NOEC/LOEC	no studies reported
Invertebrate survival	Estuarine and marine EC_{50}	3.6 - 320
Aquatic primary producer	EC_{50} NOEC	11 no studies reported

Collectively, the exposure and toxicity information supports that degradation of water quality is expected for 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil. Prey resources for juveniles may also be reduced from pulses of the stressors of the action associated with the use of 2,4-D, triclopyr BEE, diuron, linuron, and chlorothalonil in high risk areas such as distributary channels in tidal flats and channels draining diked agricultural areas where the pesticide products are applied. Applications of the four herbicides adjacent to estuarine habitats are also expected to reduce the available cover, particularly in cases where transport occurs during periods of low flow and neap tides. 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, natural cover, and/or reduce prey availability in nearshore marine areas.

Nearshore marine areas require water quality conditions and forage, including aquatic invertebrates and fishes, to support growth and maturation. Similar to estuarine sites, nearshore marine sites have very few data on the behavior and dissipation rates of the stressors of the action. More information is available on the toxicity to a few saltwater organisms, although it is still insufficient to make a definitive conclusion. The available acute toxicity information shows that the stressors of the action can kill and reduce growth of marine flora and fauna and have comparable sensitivity to freshwater organisms.

There is no doubt that the stressors of the action contaminate nearshore environments based on allowable uses. However, significant uncertainty arises to the persistence and rate of degradation of the stressors. Fundamental environmental fate data are lacking, not to mention experimental results from environmentally realistic exposure scenarios for key salmonid prey taxa including small, forage fish. We found no information on the environmental fate of these materials in nearshore marine habitats. Whether and how frequently the stressors of the action attain toxic levels for sufficient durations within nearshore marine environments remains unknown. We anticipate some level of degradation of water quality of these habitats, particularly for those species with nearshore marine areas of designated critical habitat within Puget Sound. In Puget Sound we expect the greatest deposition and loading from allowable applications as compared to other nearshore marine areas along the California, Oregon, and Washington coasts because of the longer residence time of the water, proximity to rights-of-way uses, stormwater discharges, and inflow from the numerous rivers. For this reason, we discuss effects to the attributes in 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 essential physical and biological features will be reduced in spawning, rearing, migratory, estuarine, and nearshore marine habitats. Next, within the *Integration and Synthesis of Effects to Designated Critical Habitat* section, we evaluate whether these adverse changes to PCEs affect the conservation value of designated critical habitat.

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; 2016) are more likely to occur than longer-term projects (10-years; 2021). Projections are based upon recognized organizations producing best available information and reasonable rough-trend estimates of change stemming from these data. NMFS analysis provides a snapshot of the effects from these future trends on listed ESUs.

The states of the west coast region, which contribute water to major river systems, are projected to have the most rapid growth of any area in the U.S. within the next few decades. California, Idaho, Oregon, and Washington are forecasted to have double digit increases in population for each decade from 2000 to 2030 (USCB, 2005). Overall, the west coast region has a projected population of 72.2 million people in 2010. The U.S. Census Bureau predicts this figure will grow to 76.8 million in 2015 and 81.6 million in 2020.

Although general population growth stems from development of metropolitan areas, growth in the western states is projected from the enlargement of smaller cities rather than from major metropolitan areas. Of the 46 western state metropolitan areas that experienced a 10% growth or greater between 2000 and 2008, only seven have populations greater than one million people. Of these major cities, one and two cities are from Oregon and California, respectively. They include

Portland-Vancouver-Beaverton, OR (1.81% per year), Riverside-San Bernadino-Ontario, CA (3.31% per year), and Sacramento-Arden-Arcade-Roseville, CA (2.18% per year) (USCB, 2009).

As these cities border coastal or riverine systems, diffuse and extensive growth will increase overall volume of contaminant loading from wastewater treatment plants and sediments from sprawling urban and suburban development into riverine, estuarine, and marine habitats. Urban runoff from impervious surfaces and roadways may also contain oil, heavy metals, PAHs, and other chemical pollutants and flow into state surface waters. Inputs of these point and non-point pollution sources into numerous rivers and their tributaries will affect water quality in available spawning and rearing habitat for salmon. Based on the increase in human population growth, we expect an associated increase in the number of NPDES permits issued and the potential listing of more 303(d) waters with high pollutant concentrations in state surface waters. 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 & 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 & Woods, 2007). Impacts from heightened agricultural production will likely result in two negative impacts on listed Pacific salmonids. The first impact is the greater use and application of pesticide, fertilizers, and herbicides and their increased concentrations and entry into freshwater systems. Carbaryl, carbofuran, and methomyl, and other pollutants from agricultural runoff may further degrade existing salmonid habitats. Second, increased output and water diversions for agriculture may also place greater demands upon limited water resources. Water diversions will reduce flow rates and alter habitat throughout freshwater systems. As water is drawn off, contaminants will become more concentrated in these systems, exacerbating contamination issues in habitats for protected species.

The western states are widely known for scenic and natural beauty, and are used recreationally by residents and tourists. Increases in use could place additional strain on the natural state of park and nature areas that are also occupied by protected species. Hiking, camping, and recreational fishing in these natural areas is unlikely to have any extensive effects on water quality.

The above non-federal actions are likely to pose continuous unquantifiable negative effects on listed salmonids addressed in this Opinion. Each activity has negative effects on water quality. They include increases in sedimentation, increased point and non-point pollution discharges, decreased infiltration of rainwater (leading to decreases in shallow groundwater recharge, decreases in hyporrheic flow, and decreases in summer low flows).

Non-federal actions likely to occur in or near surface waters in the action area may also have beneficial effects on the 28 ESUs. They include implementation of riparian improvement measures, fish habitat restoration projects, and best management practices (*e.g.*, associated with timber harvest, grazing, agricultural activities, urban development, road building, recreational activities, and other non-point source pollution controls).

Coupled with EPA's registration of the a.i.s of the past three recent Opinions which include 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; the effects from anthropogenic growth on the natural environment will continue to affect and influence the overall distribution, survival, and recovery of Pacific salmonids in California, Idaho, Oregon, and Washington.

Integration and Synthesis

Analysis for Listed Species

The *Integration and Synthesis* section describes NMFS' assessment of the potential for EPA's registration of 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil to reduce the reproduction, numbers or distribution of listed Pacific salmonids, taking into account status of the species, the environmental baseline, and cumulative effects.

In the *Effects* section we described the effects we anticipate for the salmon themselves due to direct toxicity from the active ingredients. We described anticipated direct effects due to exposure to other stressors of the action and interactions of multiple stressors. We also discussed indirect effects to salmonids via effects on prey, primary productivity, 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 this section we analyze the likelihood that effects of the a.i.s and other stressors of the action will reduce the reproduction, numbers, or distribution of listed Pacfic salmon within the context of the species-specific considerations discussed in the *Environmental Baseline* and *Status of Listed Resources* sections. We evaluate ESU/DPS-specific life history characteristics and distribution of pesticide use sites within their watersheds to determine the likelihood of exposure and probable effects on populations. This is accomplished by considering co-occurrence in both time (detailed in *Appendix 6*) and space (detailed in *Appendix 5*) of use sites. In this section we also consider the impact of site specific restrictions such as federal land management plans and state regulations. Although these restrictions are not part of the federal action under consultation, they do affect how the a.i.s are used in the ecosystems supporting listed Pacific salmonids.

Based on our analysis, we evaluate whether use of the a.i. as registered will likely reduce the reproduction, numbers, or distribution of populations within each ESU/DPS. This likelihood is expressed qualitatively as low, medium, or high for each of the a.i.-ESU/DPS combinations.

Evaluating the Likelihood of Effects on Populations

We link the assessment endpoints and risk hypotheses we considered in the *Effects* section to reduction in the reproduction, numbers, or distribution of populations in the following way. Reductions in reproduction are caused by physiological or behavioral impairments that decrease the number of fish reaching spawning grounds, that cause fish to mate unsuccessfully or not at all, or that reduce the number or viability of eggs or young produced. Reductions in numbers are caused by direct lethality at any life stage, increased mortality due to predation or interaction with other stressors, or inability of the habitat to support normal growth and development of the fish (*e.g.*, decreased prey availability, lack of cover, reduced primary productivity). Unlike a dam or other physical barrier that can clearly be linked to a reduction in distribution because it blocks access, reductions in distribution caused by chemical stressors are more subtle. Reductions in distribution are typically the result of reductions in reproduction, numbers, or some combination thereof to the point the population no longer uses the affected waterbody and/or cannot recolonize it.

We considered the likelihood for appreciable reduction in the reproduction, numbers, or distribution of a population to be low if we expected that the a.i. and other stressors of the action would: rarely or never kill fish; would have minor or transient effects on physiological functions, would be unlikely to reduce reproduction, and would cause little or no reduction in prey availability, primary productivity, or cover.

We considered the likelihood for appreciable reduction in the reproduction, numbers, or distribution of a population medium if we expected: that the a.i and other stressors of the action might kill fish, but that it would occur infrequently; that it will have effects on other physiological functions, but not to the extent the fish is unable to complete life functions; that it will cause minor reductions in reproduction; or that it will cause some reduction in prey availability, primary productivity or cover. If we expected an a.i. whose effects meet the criteria for medium would not often reach salmon-bearing waters in certain ESU/DPSs based on landuse, authorized use sites, and/or other restrictions the likelihood of appreciable reduction in the reproduction, numbers, or distribution of a population was considered low.

We considered the likelihood for appreciable reduction in the reproduction, numbers, or distribution of a population high if we expected that the a.i and other stressors of the action were expected to frequently kill fish; cause impairments of physiological functions to the extent that fish die or are unable to perform necessary life functions such as predator avoidance, foraging, and migration; to be likely to reduce reproduction, or to cause significant reduction in prey availability, primary productivity, or cover. If we expected an a.i. whose effects meet the criteria for high would not often reach salmon-bearing waters in certain ESU/DPSs, landuse, authorized use sites, and/or other restrictions, the likelihood of appreciable reduction in the reproduction, numbers, or distribution of a population may be considered medium or low.

Evaluating the Likelihood of Effects on Species

ESUs/DPSs are made up of discrete population(s) of salmon or steelhead. Each of these populations support the survival and recovery of the species, but may not all be equally affected by the use of an a.i.(s). Some ESUs/DPSs have been reduced to only one or two populations, others have more. However, in some cases, although there are a number of populations, one or two of these populations are particularly important to the species. Taking into account both the unevenness of use of the various a.i.s across the landscape, and the relative importance of various populations to the ESU/DPS, we determine the potential for appreciable reduction in the reproduction, numbers, or distribution of the species. This also is expressed qualitatively as low, medium, or high and summarized in Table 146.

The potential for appreciable reduction in reproduction, numbers, or distribution of the species was considered low in cases where the likelihood for appreciable reduction in reproduction, numbers, or distribution at the population level was low for all populations.

The potential for an appreciable reduction in reproduction, numbers, or distribution of the species was considered medium in cases where the likelihood for an appreciable reduction in reproduction, numbers, or distribution at the population level was medium for all or most populations. If the likelihood for an appreciable reduction in reproduction, numbers, or distribution at the population level was low for some populations, but medium for one or more populations particularly important to the ESU/DPS, likelihood for reductions at the species level was considered medium.

The potential for appreciable reduction in reproduction, numbers, or distribution of the species was considered high in cases where the likelihood for appreciable reduction in reproduction, numbers, or distribution at the population level was high for all or most populations. If the likelihood for appreciable reduction in reproduction, numbers, or distribution at the population level was low or medium for some populations, but high for one or more populations particularly important to the ESU/DPS, likelihood for reductions at the species level was considered high.

Determining Jeopardy

In the *Conclusion* section, we present jeopardy and no jeopardy determinations (Table 148). We believe high potential for reduction in the reproduction, numbers, or distribution of the species will jeopardize the ESU/DPS. We believe a low potential for reduction in the reproduction, numbers, or distribution of the species will not jeopardize the ESU/DPS. A medium potential for reduction in the reproduction, numbers, or distribution of the species sometimes may jeopardize and sometimes may not jeopardize the ESU/DPS, depending on circumstances associated with population(s) at risk, the relative importance of those populations to the ESU/DPS, and the characteristics and uses of the a.i under consideration.

Analysis for Critical Habitat

This section describes NMFS' assessment of the likelihood that EPA's registration of 2,4-D, triclopyr BEE, diuron, linuron, captan and chlorothalonil will destroy or adversely modify designated critical habitat for the 26 ESUs/DPSs that have designated critical habitat covered in this Opinion. Critical habitat has not been designated for the LCR coho salmon and Puget Sound steelhead.

All species addressed in this Opinion have similar PCEs. These PCEs are sites that support one or more life stages and include

1. freshwater rearing sites,
2. freshwater migration corridors,

3. estuarine areas,
4. nearshore marine areas, 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 some of the 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 classifications and salmon distributions to determine overlap of application sites and designated critical habitat. Because cropping patterns and registered use sites may change over time, landuse classifications (agricultural, forestry, urban/developed) are used rather than specific crops. Details of the GIS analysis and the maps are provided in *Appendix 5*.

In the *Effects* 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 *Summary of Individual a.i.s*.

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 Pacfic salmon within the context of ESU/DPS- specific considerations discussed in the *Environmental Baseline* and *Status of Listed Resources* sections. We also consider the impact of site specific restrictions such as federal land management plans and state regulations. Although these restrictions are not part of the federal action under consultation, they do affect how the a.i.s are used in the ecosystems supporting listed Pacific salmonids.

Evaluating the Likelihood of Adverse Effects on PCEs

The likelihood of adverse effects onPCEs was considered low in cases where we did not anticipate reductions or community-level modifications in the in- and near-stream plant communities or reductions in water quality that might impair fish health or cause greater susceptibility to other stressors.

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 plant communities or reductions in water quality that might impair fish health or cause greater susceptibility to other stressors. Reductions or community-level modifications to in-stream plant communities last longer than the duration of the chemical pulse. Plant communities are less diverse and abundant, but still provide sufficient cover and energy base for the system. Changes in riparian vegetation affect amount or type of allochthonous input or reduce shading. Degradation of water quality affects fish health and susceptibility to other stressors, but does not 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 plant communities or reductions in water quality that might impair fish health or cause greater susceptibility to other stressors. Reductions or community-level modifications to in-stream plant communities last longer than the duration of the chemical pulse. Plant communities are less diverse and abundant, and no longer provide sufficient cover and energy base for the system. Changes in riparian vegetation significantly affect amount or type of allochthonous input, significantly reduce shading, increase sedimentation, or destabilize streambanks. Degradation of water quality affects 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 149). Taking into account the uneven use of the a.i.s, and the conservation value of the various watersheds, we determined the likelihood that 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 plant 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. Our conclusions regarding destruction or adverse modification of critical habitat are presented in Table 147. In that table, yes indicates that we consider the proposed action likely to result in the destruction or adverse modification of critical habitat, while no indicates that we do not consider the proposed action likely to result in the destruction or adverse modification of critical habitat.

Summaries of Individual a.i.s

2,4-D acid, amines, and esters

Based on available usage data, labels, and other information from applicants we were not able to completely distinguish separate markets or use sites for the ester and amine forms, thus we have assumed either may be applied on most authorized use sites. As described in the *Response* section of the *Effects* chapter, we believe the greater toxicity of the ester forms is related to the rate of uptake. The master label allows for the application of ester products at many sites, including the direct water application of the butoxyethyl ester (BEE) form to habitats that contain listed salmonids. Due to this fact, we have considered toxicity data from both the ester and amine forms in our evaluation of acute (short-term lethal) effects. Given relatively quick degradation of the ester to acid form in water, we use acid and amine toxicity data in our evaluation of chronic (longer term growth, reproduction, and sub-lethal) effects.

The butoxyethyl ester (BEE) and amine forms are both approved for direct water applications. Direct water applications are a major concern, as the recommended concentrations (4,000 μg/L) in the water exceed all toxicity endpoints for fish, prey, and primary productivity for the ester. For the amines, the recommended concentration exceeds only the toxicity endpoint for vascular aquatic plants. We believe that direct water applications of 2,4-D BEE in accordance with label directions may pose a serious risk to listed salmonids. Aquatic weed management may occur under local, state, or federal resource management plans or permitting systems and is often intended to improve the habitat for various beneficial uses. However, in some cases aquatic weed control may be conducted for the purposes of human activities, such as access to boat docks or beaches, and may not be regulated by the state or governed by a resource management plan. These resource management plans and/or permitting systems are not a part of the federal action (*i.e.*, the label), but are part of the *Environmental Baseline*.

In Washington State, in-water applications of pesticides require a permit under the National Pollutant Elimination Discharge System (NPDES) (http://www.ecy.wa.gov/programs/wq/plants/management/aqua028.html, accessed 02/23/11). Washington has issued general permits for lake applications, and for use in freshwater wetlands, rivers, and estuaries. Both general permits include application timing windows to limit use of 2,4-D during peak periods of salmon use of freshwater habitats for spawning and rearing. While this limits effects, it does not completely eliminate them. Control of weeds in irrigation canals is covered under a different permit, and 2,4-D is not among the pesticides authorized for this use by the state. The website notes that under a court order, 2,4-D BEE may not be used in waters containing listed salmonids. Although not specified on the website, the court order referenced appears to be the injunctions associated with the original *Washington Toxics Coalition* court case, which will be lifted following issuance of the final Opinion for this a.i. We are uncertain if Washington will remove or modify this restriction when the injunction is lifted. There is an individual fisheries management permit issued to the Washington State Department of Fish and Wildlife, but it covers piscicides such as rotenone, not 2,4-D. The current restrictions in place in the state of Washington regarding direct water applications of 2,4-D provide substantial protection to listed salmonids. The restriction against using 2,4-D BEE in waters containing listed salmonids is an important part of that protection.

In California, in-water applications of pesticides require a permit under the National Pollutant Elimination Discharge System (NPDES) (http://www.swrcb.ca.gov/water_issues/programs/npdes/aquatic.shtml, accessed 2/24/11). Direct water applications of 2,4-D is addressed in the California permit. The California permit requires the discharger (pesticide applicator) to develop and implement an Aquatic Pesticide Application Plan (APAP). Under terms of the general permit, the discharger must submit a plan describing their analysis of alternatives, and Best Management Practices (BMPs) they may use to minimize effects to non-treated areas. There is a Maximum Concentration Limit (MCL) of 70 μg/L for 2,4-D for receiving waters with a municipal use (MUN) designation, but no MCL for waters designated as warm or cold water habitat. Nonylphenol-containing surfactants are permitted, but subject to criteria of 6.6 μg/L for freshwater and 1.7 μg/L for saltwater. The only mention of listed species considerations is the following:

> "34. This General Permit does not authorize any take of endangered species. The discharge is prohibited from adversely impacting biologically sensitive or critical habitats, including, but not limited to, habitat of species listed under federal or State endangered species laws. To ensure that endangered species issues are raised to the responsible agencies, the State Water Board has notified the U.S. Fish and Wildlife Service, the National Marine Fisheries Service, and the California Department of Fish and Game of this General Permit." (page 6, Waste Discharge Requirements)

Beyond the language provided above, the California general permit does not contain specific requirements for listed salmonids, but relies instead on the individual applicator to ensure they are not adversely affecting listed species and their habitat. Thus, we do believe the requirements of the California general permit provide little additional protections for listed salmonids beyond what is currently on the FIFRA label.

There is currently no permitting requirement for direct water applications of pesticides in Oregon, although we did locate a fact sheet describing the proposed permitting system being developed coincident with EPA's national level Pesticide General Permit (http://www.deq.state.or.us/wq/wqpermit/docs/general/npdes2300a/ProposedFactSheetPesticideGP.pdf, accessed 2/24/11). While the proposed permit calls for Integrated Pest Management (IPM) techniques, and requires development of a Pesticides Discharge Mangement Plan (PDMP), it makes no mention of any specific considerations or limitations for listed species. For these reasons, we do not believe the requirements of the proposed Oregon general permit provide any additional protections for listed salmonids beyond what is currently on the FIFRA label.

Idaho does not have delegated authority under Clean Water Act to develop NPDES permits, thus any direct water applications are governed at the federal level. EPA is currently developing a pesticide general permit on which NMFS is conducting a section 7 consultation. Idaho does have a noxious weed control plan that includes control of aquatic weeds such as Eurasian watermilfoil (http://www.agri.state.id.us/Categories/PlantsInsects/NoxiousWeeds/Documents/general/Overview%20of%20ISDA%20Noxious%20Weed%20Program%202004.pdf., accessed 2/25/11). There is a state strategic plan for Eurasian watermilfoil, but based on our review, it does not contain any specific provisions for listed salmon or other listed species (http://www.agri.state.id.us/Categories/PlantsInsects/NoxiousWeeds/Documents/Milfoil/EWM%20Strategy%20Final.pdf). As far as NMFS can ascertain, no programs in Idaho provide any additional protections for listed salmonids beyond what is currently on the FIFRA label.

Based on an analysis of state, regional, and national maps showing roads, railroads, electric transmission lines, and pipelines, we concluded ROW uses are most likely to be concentrated in more developed areas. In more remote areas, roads and railroads are often situated along river valleys, and sometimes in close proximity to the stream or river. These types of rights-of-way are generally associated with larger streams (third-order and above), so we assume EECs from

EPA's PRZM-EXAMS modeling are more reflective of concentrations that might be found in these waters than NMFS floodplain EECs. However, unlike triclopyr BEE, evaluation of California's pesticide use reporting and information provided by applicants indicated that while 2,4-D is used for rights-of-way, it is not a primary market. The Washington State Department of Transportation currently prohibits use of 2,4-D amine formulations within 60 ft of water. However, there is no similar prohibition on use of the ester formulations or the salt, which are also authorized for ROW use. We located no similar ROW prohibitions in Oregon, Idaho, or California.

We located restrictions on pesticide uses in forestry applications in Washington, Oregon, and Idaho. Idaho prohibits both ground and aerial application within 100 ft of "Class 1 streams, flowing Class 2 streams, and other areas of open water" (ID Forest Practices Act 20.02.01). The Idaho code does allow hand application, but only to specific targets. Class 1 streams are those that important for spawning, rearing, or migration of fish. Class 2 streams are defined as headwater streams or minor drainages used by few, if any, fish. Oregon prohibits application of any chemicals by aircraft within 60 ft of type F streams, significant wetlands, and large lakes (OR Dept of Forestry, 629-620-0400). Type F streams are those which contain salmonids and/or game fish. Oregon also prohibits ground applications within 10 feet of significant wetlands and aquatic areas of Type F streams, large lakes, other lakes with fish use, and areas of standing open water larger than one quarter acre at the time of application (OR Dept of Forestry, 629-620-0400). Washington has a more complicated method of calculating no-spray zones that is associated with the bankfull width of the stream and various components of the riparian management zone (RMZ) (WAC 222-38-020). However, at a minimum, it appears that ground applications are not permitted within 50 ft of the bankfull edge of salmon-bearing waters in western Washington, and not permitted within 30 ft of the bankfull edge of salmon-bearing waters in eastern Washington. There is a 25 ft buffer around wetlands and other surface waters. A formula based on site characteristics is provided to calculate aerial no-spray zones for western Washington. For eastern Washington, no-spray zones are calculated by adding the "core zone" to the "inner zone". This appears to result in a minimum 75 ft no-spray zone for streams <15ft wide, and a 100 ft no-spray zone for streams >15 ft wide.

Oregon and Idaho also have some restrictions on applications of low- and high-volatility 2,4-D esters (OR State Forest Laws 634.372, and ID Administrative Code 02.03.03.500.01, 02.03.03.500.02, 02.03.03.550.01, 02.03.03550.02). However, these seem to be primarily designed to prevent crop damage from spray drift rather than protect surface waters so we do not discuss them further.

2,4-D is registered for a broad range of uses, and is one of the most commonly used herbicides in the U.S. In addition to agricultural uses, ROW uses, and direct water applications, it is also authorized for turf uses, including in homeowner products. Thus, it could be used in any of the landuse classes we evaluate (agriculture, forested, urban/residential, ROW). We expect the temporal use pattern to vary among the different use sites, however, applications of 2,4-D are likely to occur throughout the period where plants are actively growing (early spring – late fall for most areas within the salmonids freshwater distribution). Thus, 2,4-D is likely to enter water bodies from multiple sources, and we expect these inputs to be frequent occurrences, especially in areas that are either highly developed or heavily agricultural or both. The multiple source, frequent occurrence pattern is a key factor in the risk posed by this a.i.

Overall, for 2,4-D, we anticipate mostly sublethal effects for fish, and little to no direct mortality for terrestrial applications of 2,4-D products. Some of these sublethal effects may manifest essentially as a reduction in available energy for activities such as reproduction, migration, or foraging due to reductions in glycogen storage in the liver. In one field study fish exposed to 2,4-D esters spawned two weeks late (Cope *et al*., 1970). Systems inhabited by salmonids are often temporally distinct, with runs of listed salmonids and other fish spawning and emerging sequentially. A delay in spawning could mean that a typical food source is not present when the fry emerge, and/or a predator not typically present is in the system, or is larger in size. Such effects would vary widely by system, and are relatively unpredictable. However, if of a sufficient magnitude, such an event could cause reductions in abundance. We are especially concerned about this type of effects in ESUs/DPSs that have only one or two viable populations, and for coho, who have temporally distinct brood years. We do not anticipate large reductions in in-stream primary productivity from terrestrial 2,4-D. However, any form of the a.i. applied near a waterbody is likely to affect terrestrial plants, especially herbs and forbs. Riparian vegetation near the waterbody provides shade, bank stabilization, sediment, chemical and nutrient filtering, and provides a niche for the terrestrial invertebrates that are also salmon prey items. We are unable to quantify this effect due to variability in plant susceptibility to the forms of 2,4-D, and differences in species composition and density for various locations. We believe the a.i. will have a detrimental effect on the riparian vegetation. In general, riparian zones with diverse communities of herbaceous plants, shrubs, and trees will be most resilient.

In general, we believe most uses of 2,4-D amines and esters other than the direct water applications of 2,4-D BEE have a low likelihood to the reduce the reproduction, numbers, or distribution of populations of listed salmonids. We believe direct water applications of the BEE form have a medium likelihood to reduce the reproduction, numbers, or distribution of listed salmonids if it is applied to salmon-bearing waters when fish are present. Based on analysis of the federal labels and existing state restrictions, direct water applications of the BEE could occur in any state within the salmon's range other than Washington. We cannot predict with any certainty where or how often such applications might occur. We believe direct water applications of the amine forms have a low likelihood to reduce the reproduction, numbers, or distribution of populations of listed salmonids. In addition to directly affecting the fish, direct water applications of any form of 2,4-D will reduce biomass of vascular plants in the system, resulting in a reduction of cover for the fish and habitat for aquatic invertebrates. Conversely, the applications may improve the overall condition of the habitat by controlling invasive vegetation. We believe terrestrial uses of all forms of 2,4-D (esters and amines) have a low likelihood to the reduce the reproduction, numbers, or distribution of populations of listed salmonids.

Considering all forms and uses of 2,4-D, we believe the a.i. has a medium likelihood to the reduce the reproduction, numbers, or distribution of populations of listed salmonids as currently registered. Due to potential degradation of water quality as measured by effects on primary productivity and fish health, and potential degradation of riparian vegetation we believe it has a medium likelihood of degrading PCEs.

Triclopyr BEE

Based on use data from California's pesticide use reporting, Doane's use data, and information provided by applicants, a primary market for triclopyr BEE is rights-of-way. Main uses are roads and railroads, with some use on electric transmission lines, substations, and similar industrial locations.

Given the amount of relatively impervious surfaces (pavement, gravel, hard-packed) associated with rights-of-way, concentrations of the a.i. may be higher for these uses than for applications on vegetation and/or more loosely packed soil. We also expect wash-off to be rapid, thus exposure of a group of fish to a high concentration pulse of the ester form of triclopyr BEE is the scenario with which we are most concerned. Based on modeled estimates, this pulse appears to be a high enough concentration to kill a group of fish. If it occurred when a group of adult fish were migrating to spawn, when a group of smolts were emigrating, or in a shallow habitat where fry or larval fish were present, it could cause a major decrease in the affected population. We stress that we believe this situation is likely to occur rarely. However, rare catastrophic events may have dire consequences for species subject to small population dynamics, so we have considered this scenario in our ESU/DPS specific evaluation.

Based on an analysis of state, regional, and national maps showing roads, railroads, electric transmission lines, and pipelines, we concluded ROW uses are most likely to be concentrated in more developed areas. In more remote areas, roads and railroads are often situated along river valleys, sometimes in close proximity to the stream, rivers, and floodplain habitats. Rights of way uses are approved for high application rates (8 lbs a.i./A). The Washington State Department of Transportation currently prohibits use of triclopyr ester within 60 ft of water. We located no similar ROW prohibitions in Oregon, Idaho, or California.

A second important use for triclopyr BEE is in forestry. When scaled up to a lb a.i./A rate, direct (basal) applications are extremely high, but we expect these types of applications will result in lower concentrations than those modeled because they will be applied only in small areas (spot treatments). High application rates and multiple applications are also approved for non-crop agricultural areas and ephemeral aquatic habitats (floodplains and marshes) during the dry period. When these applications occur in close proximity to salmon-bearing water they could cause runoff of triclopyr BEE into nearby streams, or degrade the riparian habitat supporting the stream.

A number of salmon ESUs/DPSs spawn and/or rear in forested areas, some of which are managed by other federal agencies such as the U.S. Forest Service, the U.S. Bureau of Land Management, and the National Park Service. When these agencies use pesticides on forested lands supporting listed species, those uses are generally subject to a Section 7 programmatic or site-specific consultation. In the *Environmental Baseline* we describe several of the Biological Opinions based on these consultations. Some permit triclopyr. Several of those which permit use of triclopyr limit it to the TEA form. All have restrictions on the amount of a.i. applied. Many require "no–apply" buffer zones near water containing listed species, and require low drift, low runoff application methods. Based on these Opinions, we have assumed potential for use of triclopyr BEE near salmon-bearing waters is extremely limited when those waters are located in federally managed forests.

We also located restrictions on pesticide uses in forestry applications in Washington, Oregon, and Idaho. Idaho prohibits both ground and aerial application within 100 ft of "Class 1 streams, flowing Class 2 streams, and other areas of open water" (ID Forest Practices Act 20.02.01). The Idaho code does allow hand application, but only to specific targets. Class 1 streams are those that are important for spawning, rearing, or migration of fish. Class 2 streams are defined as headwater streams or minor drainages used by few, if any, fish. Oregon prohibits application of any chemicals by aircraft within 60 ft of type F streams, significant wetlands, and large lakes (OR Dept of Forestry, 629-620-0400). Type F streams are those which contain salmonids and/or game fish. We located no restrictions on ground applications or hand applications. Washington also has forest uses restrictions (WAC 222-38-020, described in the 2,4-D section). Based on our interpretation of the Washington regulations, there is a minimum 50 ft no-spray buffer for ground applications in western Washington, and 30 ft no-spray buffer for ground applications in eastern Washington. For aerial applications, there is a minimum 75 ft no-spray zone for streams <15ft wide, and a 100 ft no-spray zone for streams >15 ft wide in eastern Washington. Based on a comparison with the ground restrictions, the aerial no-spray buffer in western Washington are likely to be slightly larger than the ones in eastern Washington. Thus, within the states of Washington, Oregon, and Idaho, we assumed there will be some type of buffer when triclopyr BEE is used in forested areas governed by the forest practice rules.

Triclopyr BEE may kill salmon if they are present when the a.i. reaches the water while still in the ester form. The BEE hydrolyzes rapidly to triclopyr acid, which is much less toxic, and is not expected to cause mortality if the salmon are exposed to it. We expect there may be times when triclopyr BEE does reach the water in ester form due to right-of-way applications, but that will be infrequent. We do not anticipate use of triclopyr BEE will affect reproduction or sublethal endpoints. We believe triclopyr BEE has a medium likelihood to reduce the reproduction, numbers, or distribution of populations of listed salmonids based on the current registration, other restrictions from federal management plans, section 7 consultations on those plans, and state regulations.

Triclopyr also appears unlikely to cause a reduction in prey availability or instream primary productivity that would affect the salmon. Applications in the riparian zone may modify vegetation. This change could be either beneficial or detrimental to stream and to the salmon which it contains. On federal lands these triclopyr applications should be governed by management plans subject to consultation. Additionally, in Washington, Oregon, and Idaho use in forests is governed by state regulations. We believe use of triclopyr BEE has a low likelihood of adverse effects on PCEs based on the current registration, other restrictions from federal management plans, section 7 consultations on those plans, and state regulations.

Diuron

Diuron is classified as moderately toxic to fish and moderately to highly toxic to aquatic invertebrates. It is persistent in aquatic systems, and is frequently detected in ambient water quality monitoring programs (29-47% detections). There may be extended exposure in slow-flowing waters. In faster flowing waters, there may be an extended chemical plume or increased concentrations in certain areas due to multiple input sources. Effects on fish development and reproduction occur at concentrations that have been reported in the ambient monitoring data.

Linuron, which is structurally similar to diuron, has been identified as an endocrine disruptor in fish. Although diuron has not been identified as an endocrine disruptor, in absence of data we have assumed the linuron endpoints are applicable given those structural similarities. Effects on primary productivity occur in the range of 10-95 μg/L, and the NOAEC for ecosystem functioning is 0.5 μg/L.

Diuron is used heavily on rights-of-way, and is also authorized for a number of agricultural uses. Rights-of-way uses have much higher application rates (up to 12 lb ai/A) than agricultural uses (1-4 lb ai/A). Although rights-of-way applications often occur in narrow strips, given the linear distances on which it might be used, areas of an acre or larger could conceivably be treated, thus we believe EECs calculated using PRZM-EXAMS are relevant. Additionally, linear rights-of-way such as those associated with roads and railways may occur in close proximity to rivers and shoreline habitats running in parallel for several miles, increasing the likelihood of exposure. Simulation of a linear application of diuron using AgDrift showed a single swath treatment could result in concentrations detrimental to salmonids and their habitat (*Risk Characterization*).

Based on use data from California's pesticide use reporting, and information provided by applicants, the main uses for diuron are roads and railroads, with some use on electric transmission lines, substations, and similar industrial locations. Using state, regional, and national maps of roads, railroads, electric transmission lines, and pipelines, we concluded ROW uses are most likely to be concentrated in more developed areas. In more remote areas, roads and railroads are often situated along river valleys, sometimes in close proximity to the stream or river. Road and railroad rights-of-way are generally associated with larger streams (third-order and above), so we assume EECs from EPA's PRZM-EXAMS modeling are more reflective of concentrations that might be found in these waters than NMFS floodplain EECs. The Washington State Department of Transportation currently prohibits use of diuron in western Washington and use within 60 ft of water in eastern Washington. We located no similar ROW prohibitions in Oregon, Idaho, or California.

Agricultural uses are expected to primarily affect primary productivity, but not fish health or prey availability endpoints. We believe ROW uses could occasionally kill fish, and may cause decreased reproduction or endocrine effects. Additionally, it may decrease prey and depress primary productivity. We do not believe it rises to the level where it will cause appreciable reduction in abundance, distribution, or reproduction of listed salmonids in the affected waters.

Diuron is currently registered for use on "intermittent aquatic habitats when not flooded" and "seasonally dry floodplains" at rates of 4 - 12 lb a.i./A. At the applicant meeting held on March 18, 2011, the applicants indicated the maximum rate was typically not used in the U.S. but was used commonly in South America. Usage data from California and Washington show diuron is heavily used for ROW and do not report use on intermittently flooded areas. Some, applicants have indicated a willingness to remove this use from their labels (Drexel, 2011; MANA, 2011) (DuPont, 2011). NMFS has considered the likely use and intended label revision in the analysis of diuron. Because NMFS believes EPA's authorization of such uses are likely to adversely modify critical habitat, the proposed label revision has been included as an element of the RPA for those ESUs/DPSs where NMFS concludes that the use of diuron as currently labeled would adversely modify critical habitat. Because this proposed label revision also minimizes the impact of incidental take, NMFS has also included it as a RPM.

We are concerned about effects on habitat, from a fish health and primary productivity standpoint, in areas that receive frequent inputs or inputs from multiple sources. In some cases, diuron may affect waters downstream from frequent input areas. An example of such an area is the Lower Columbia River, downstream of Willamette Valley. We expect in some cases diuron will degrade riparian habitat and decrease the productivity of receiving waters.

Overall, we believe diuron has a medium likelihood to reduce the reproduction, numbers, or distribution of populations of listed salmonids as currently registered. Due to potential degradation of water quality as measured by effects on in-stream primary productivity, fish health, and potential degradation of riparian vegetation we believe it has a medium likelihood of degrading PCEs .

Linuron

Linuron is structurally very similar to diuron. It has nearly the same fate and toxicity profile, but is registered only for a small number of crops. It has some ROW uses, but those are primarily in an agricultural context. We anticipate all linuron uses will occur in agricultural areas. Based on information provided by the applicant, and confirmation with pesticide usage data in California and Washington, linuron is used in very limited amounts. We expect minimal exposure in most ESUs/DPSs.

We believe linuron has a low likelihood to reduce the reproduction, numbers, or distribution of populations of listed salmonids as currently registered. This expectation is associated primarily with the level of use rather than the properties of the a.i. We believe it has a low likelihood of degrading PCEs. This expectation is associated primarily with the level of use rather than the properties of the a.i.

Captan

Captan dissociates rapidly in aquatic systems, so exposure to the parent molecule is expected to be limited. We do not anticipate captan will kill fish, reduce prey abundance or modify primary productivity or riparian vegetation. Sublethal effects are possible, but unlikely as exposure duration is expected to be limited given the rapid degradation. We do not anticipate exposure will be of sufficient duration to manifest adverse effects.

We believe captan has a low likelihood to reduce the reproduction, numbers, or distribution of populations of listed salmonids as currently registered. We believe it has a low likelihood of adverse effects on PCEs due to rapid dissipation in aquatic systems and minimal effect on riparian vegetation.

Chlorothalonil

Chlorothalonil has two distinct types of uses – on agricultural crops, and on turf. Multiple applications at fairly short intervals (7-14 days) are authorized for both types of uses.

Chlorothalonil is also currently registered generically for use on conifers. The applicant indicated the forestry uses are typically commercial Christmas tree farms and nursery operations (Syngenta, 2011). Usage data from Washington State corroborates this statement, and usage data from California shows little use on any type of forestry. Exposure of salmon to chlorothalonil from these specific uses is more limited than it would be if chlorothalonil were used to control pests in large areas of forest, such as commercial silviculture operations. The applicant plans to submit a fast-track label amendment to EPA to clarify the uses. Proposed label language limits conifer use to (1) conifer nursery beds; (2) Christmas tree and bough production plantations; (3) tree seed orchards; and (4) landscape situations.[12] NMFS has considered likely use, the clarification provided by the applicant , and intended label revision in the analysis of chlorothalonil. Because NMFS believes that EPA's authorization of those uses that the applicant intends to change are likely to jeopardize listed species, NMFS includes the proposed label revision as an element of the RPA applicable for those ESUs/DPSs for which NMFS finds adverse modification. The same proposed label change also minimizes the impact of take, so NMFS also includes it as a RPM.

Although chlorothalonil is classified as very highly toxic to fish, it degrades quickly in aquatic systems via photolysis ($t_{1/2}$=10 h) and aerobic aquatic metabolism ($t_{1/2}$=1.5d). In clear, shallow, or sunlit waters we do not expect to be present for extended time following a specific runoff or overspray event. In more turbid, deeper, or shaded waters it may to be present longer. Chlorothalonil is infrequently detected in ambient monitoring programs (0-2% detections), and when detected, concentrations are low compared to acute endpoints, but in the range of several of the chronic endpoints, such as immune response, and prey reproduction and growth.

However, EECs, which are a better estimator of initial runoff concentrations, do exceed assessment endpoints for salmonids, prey, and primary productivity. EECs for turf uses are generally much higher (~ an order of magnitude) than most crop uses. Some targeted monitoring studies on turf uses show peaks greater than salmonid survival endpoints and chronic

[12] In an email to the applicant on May 10, 2011, NMFS sought clarification on the 4th use: "landscape situations. The applicant responded on May 11, 2011, to clarify that landscape situations was for "specimen trees in a commercial landscape."

concentrations that are expected to cause reductions in available prey.   Thus, we anticipate some lethal events for salmon could occur based on both types of use. We believe the likelihood of such events occurring is greater for turf use than for crop use.

“Turf” encompasses a wide range of sites, including golf courses, recreational fields, and managed landscapes around buildings and in parks. The majority of these use sites are in the urban/developed landuse class. Turf also includes sod farms, but those are categorized in the agricultural landuse class. Based on our analysis, we believe there is some likelihood of turf uses, including golf courses near salmon waters in all ESUs/DPSs. In highly urbanized/developed areas, we anticipate there will be frequent low-level inputs of chlorothalonil from multiple sources, and occasional peak concentrations which exceed salmon survival endpoints from some sources.

In highly agricultural areas, we anticipate there will be frequent low-level inputs of chlorothalonil due to the permitted application intervals, and also that it will be from multiple sources. In situations with frequent low-level inputs, we expect that chronic low concentrations will have reduce primary productivity and/or modify the community structure. We also anticipate these low concentrations may affect fish immune response, making salmon more susceptible to other stressors. Shifts in plant community structure could degrade salmonid habitat by leading to reductions or shifts in invertebrate profiles, affecting salmonid prey base or availability.

Overall, we believe chlorothalonil has a medium likelihood to reduce the reproduction, numbers, or distribution of populations of listed salmonids as currently registered. Additionally, due to potential degradation of water quality as measured by effects on primary productivity and fish health, we believe it has a medium likelihood of adverse effects on PCEs.

ESU/DPS Specific Evaluations for Threatened and Endangered Pacific Salmonids

Below, we summarize the current status of each species, including baseline stressors. VSP parameters (abundance, growth rate, genetic variability, and spatial structure) are presented as a measure of the ESU/DPS's relative health. As exposure to a.i.s during the juvenile life stage is of particular concern, we highlight the length of time juveniles are found in shallow, more vulnerable habitats. The number of extant populations that co-occur with agricultural and urban areas is also given.

Puget Sound Chinook Salmon (Threatened Species)

The Puget Sound ESU is comprised of 22 extant populations. Eleven of these populations have declining productivity; the remaining populations are at replacement value. Current spawner abundance is significantly lower than historical estimates. The spatial structure for this species is compromised by extinct and weak populations that are disproportionately distributed in the mid- to southern Puget Sound and the Strait of Juan de Fuca. The genetic diversity of this ESU has been reduced due to a disproportionate loss of populations exhibiting the early-run life history and past hatchery practices.

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%). The developed areas of this ESU likely have higher concentrations of rights-of-way uses. 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. These areas serve as spawning, rearing, and migration habitat for Puget Sound Chinook. Juveniles generally migrate to marine waters within 6 months of emergence, though some have longer freshwater residences of one or more years. Given their long residency period and use of freshwater, estuarine, and nearshore areas, juveniles and migrating adults have a high probability of exposure to pesticides that are applied near their habitats.

2,4-D: 2,4-D was determined to pose a medium risk to fish. In the a.i. summary, we discuss three separate use patterns: direct water application of esters, direct water application of amines, terrestrial application of both esters and amines. Because the risk of these exposures is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of affecting populations of Puget Sound Chinook as medium. Although effects from terrestrial applications are expected to reduce primary productivity in-stream, affect fish health, and degrade riparian vegetation in some locations, our greatest concern is the potential for salmonid lethality due to direct water applications of 2,4-D BEE. NMFS believes that multiple populations may be exposed to these direct water applications within a single year, resulting in a decrease in population numbers significant enough to affect the entire ESU. Based on risk from all use patterns, we rate the likelihood of 2,4-D affecting the ESU as medium.

Triclopyr BEE: In the preceding a.i. summary, we determined that Triclopyr BEE poses a medium risk to fish. As discussed in the a.i. summary, the medium likelihood is based on our concerns about occasional high exposures resulting in a fish mortality event. If a high concentration of the a.i. was to co-occur with peak fish presence, this could cause a significant drop in the size of population. Because the risk of this exposure is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Puget Sound Chinook as medium. However, in order to decrease lambda of the species, multiple mortality events would have to occur, either within the same population or across several populations within the ESU. Given the uses, fate, and toxicity of the chemical, we do not expect mortality to be a common occurrence. Existing federal and state limitations, described in the a.i. summary in this section, further decrease the likelihood of high exposures. Thus, we believe that this compound has a low likelihood of causing an effect to the species as a whole.

Diuron: Diuron was determined to pose a medium risk to salmon. As discussed in the a.i. summary, use in rights-of-way is our largest concern, as it has the highest application rate and runoff potential. Because the risk of this exposure is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Puget Sound Chinook as medium. In this ESU, diuron use in rights-of-way is strictly limited by state legislation. We expect exposure from other uses to lead to declines in the primary productivity of the system, but not to wide-scale fish-health issues. However, given the use rates, pattern of uses for this chemical, and restrictive state regulations, we believe that there is a low likelihood of diuron causing a negative effect to multiple populations, or repeated negative effect to the same population. Therefore, we conclude that there is a low likelihood of an effect to the species as a whole.

Linuron: Given the limited uses of linuron, we expect exposure to be relatively infrequent. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Puget Sound Chinook as low. As discussed in the a.i. summary, linuron exposure in agricultural areas may have some effect on the primary productivity of a system. We anticipate that the effects of linuron exposure would have a minimal effect on populations within this ESU. Thus, there would be a low likelihood of an effect to the ESU as a whole.

Captan: In the a.i. summary, we concluded that captan has a low likelihood of affecting salmon. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Puget Sound Chinook as low. Due to the fate properties of captan, we stated that water concentrations will rarely be high enough to cause a physiological response. Therefore, there is a low likelihood of captan causing population level, or species level effects.

Chlorothalonil: In the a.i. summary, we determined that there was a medium likelihood of chlorothalonil having a negative effect on salmon. Higher or repeated exposures, particularly those associated with turf uses, may cause fish mortality. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Puget Sound Chinook as medium. The distribution of populations and use sites across the range of the species indicate that it is possible that multiple populations could experience substantial negative effects. Therefore, we believe that there is a medium potential for effects at the ESU level.

Lower Columbia River (LCR) Chinook Salmon (Threatened Species)

The LCR Chinook salmon ESU includes 20 fall- and 2 late-fall runs and 9 spring-run populations. The majority of spring-run LCR Chinook salmon populations are nearly extirpated. Total returns for all runs are substantially depressed, and only one population is considered self-sustaining. The spatial structure for this ESU is relatively intact despite a 35% reduction in habitat. The genetic diversity of all populations (except the late fall-runs) has been eroded by large hatchery influences and low effective population size.

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.

Populations located near the Portland area are expected to have increased exposure to urban uses, while the more Northern populations experience inputs from agricultural and forestry uses. Turf uses, including use on golf courses, are spread throughout the ESU with a higher concentration near Portland and along the mainstem Columbia. We expect that salmonids near Portland will have significant exposure from rights-of-way uses. This area has a high concentration of rights-of-way from rail, road, and utilities. These uses are of greater concern as they tend to be higher use rates with greater probability of runoff. This concern is mediated somewhat by the wide distribution of populations throughout the basin, and the fact that exposure will likely occur in higher volume, higher flow habitats, such as the Columbia River. While agricultural uses should not be discounted, urban uses are of greater concern – particularly for professional lawn applications. 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.

2,4-D: 2,4-D was determined to pose a medium risk to fish. In the a.i. summary, we discuss three separate use patterns: direct water application of esters, direct water application of amines, terrestrial application of both esters and amines. Because the risk of these exposures is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of affecting populations of Lower Columbia River Chinook as medium. Although effects from terrestrial applications are expected to reduce primary productivity in-stream, affect fish health, and degrade riparian vegetation in some locations, our greatest concern is the potential for salmonid lethality due to direct water applications of 2,4-D BEE. NMFS believes that multiple populations may be exposed to these direct water applications within a single year, resulting in a decrease in population numbers significant enough to affect the entire ESU. Based on risk from all use patterns, we rate the likelihood of 2,4-D affecting the ESU as medium.

Triclopyr BEE: In the preceding a.i. summary, we determined that Triclopyr BEE poses a medium risk to fish. As discussed in the a.i. summary, the medium likelihood is based on our concerns about occasional high exposures resulting in a fish mortality event. If a high concentration of the a.i. was to co-occur with peak fish presence, this could cause a significant drop in the size of population. Because the risk of this type of exposure is greater for populations near the Portland area due to development, we rate the likelihood for these populations as medium and the likelihood for the remaining populations as low. Therefore we rate the likelihood to populations of Lower Columbia River Chinook as low to medium. However, in order to decrease lambda of the species, multiple mortality events would have to occur, either within the same population or across several populations within the ESU. Given the uses, fate, and toxicity of the chemical, we do not expect mortality to be a common occurrence. Existing federal and state limitations, described in the a.i. summary in this section, further decrease the likelihood of high exposures. Thus, we believe that this compound has a low likelihood of causing an effect to the species as a whole.

Diuron: Diuron was determined to pose a medium risk to salmon. As discussed in the a.i. summary, use in rights-of-way is our largest concern, as it has the highest application rate and runoff potential. The risk of this type of exposure is greater for these populations, as they are located near the Portland area. Therefore we rate the likelihood to populations of Lower Columbia River Chinook as low to medium. In this ESU, diuron use in rights-of-way is strictly limited by state legislation. We expect exposure from other uses to lead to declines in the primary productivity of the system, but not to wide-scale fish-health issues. Given the use rates, pattern of uses for this chemical, and restrictive state regulations, we believe that there is a low likelihood of diuron causing a negative effect to multiple populations, or repeated negative effect to the same population. Therefore, we conclude that there is a low likelihood of an effect to the species as a whole.

Linuron: Given the limited uses of linuron, we expect exposure to be relatively infrequent. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Lower Columbia Chinook as low. As discussed in the a.i. summary, linuron exposure in agricultural areas may have some effect on the primary productivity of a system. We anticipate that the effects of linuron exposure would have a minimal effect on populations within this ESU. Thus, there would be a low likelihood of an effect to the ESU as a whole.

Captan: In the a.i. summary, we concluded that captan has a low likelihood of affecting salmon. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Lower Columbia Chinook as low. Due to the fate properties of captan, we stated that water concentrations will rarely be high enough to cause a physiological response. Therefore, there is a low likelihood of captan causing population level, or species level effects.

Chlorothalonil: In the a.i. summary, we determined that there was a medium likelihood of chlorothalonil having a negative effect on salmon. Higher or repeated exposures, particularly those associated with turf uses, may cause fish mortality. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Lower Columbia River Chinook as medium. The distribution of populations and use sites across the range of the species indicate that it is possible that multiple populations could experience substantial negative effects. Therefore, we believe that there is a medium potential for effects at the ESU level.

Upper Columbia River (UCR) Spring run Chinook Salmon (Endangered Species)

The UCR Spring-run Chinook salmon ESU is comprised of three extant populations. These populations are affected by low abundances and failing recruitment. The long-term trend for abundance and lambda for all three populations indicate a decline. The ESU's genetic integrity is compromised by periods of low effective population size and a low proportion of natural-origin fish. Spatial structure of this ESU is fairly intact but has been compromised by low summer flows.

While this ESU has very few populations left, we do not expect that there will be much exposure to any of the a.i.s, with the exception of direct water applications of 2, 4-D BEE. There is very little agriculture and urban development within the ESU, and correspondingly less right-of-way. The percentage of agricultural and developed lands that overlap with UCR Chinook salmon habitat is about 5.4% and 4.7%, respectively. Forested lands make up about 45% of the ESU. Much of the forested land is federally owned; any program involving the use of pesticides would be covered under its own ESA consultation. Therefore, we are considering exposure to be minimal in these areas as well. Most exposure will occur during migration along the Columbia River. This exposure is of less concern as it is a high volume, high flow system.

2,4-D: 2,4-D was determined to pose a medium risk to fish. In the a.i. summary, we discuss three separate use patterns: direct water application of esters, direct water application of amines, terrestrial application of both esters and amines. While we expect exposure from most uses to be minimal, we are not able to predict where 2,4-D direct water applications may occur so we cannot rule out this type of exposure. Because the risk of these exposures is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of affecting populations of Upper Columbia Spring Run Chinook as medium. Although effects from terrestrial applications are expected to reduce primary productivity in-stream, affect fish health, and degrade riparian vegetation in some locations, our greatest concern is the potential for salmonid lethality due to direct water applications of 2,4-D BEE. In cases of aquatic weed infestation, 2,4-D can be applied directly to the water. This use can occur anywhere, including areas with low routine pesticide use. NMFS believes that if any of the three populations was exposed to these direct water applications, the resulting decrease in population numbers could be significant enough to affect the entire ESU. Based on risk from all use patterns, we rate the likelihood of 2,4-D affecting the ESU as medium.

Triclopyr BEE: In the preceding a.i. summary, we determined that Triclopyr BEE poses a medium risk to fish. As discussed in the a.i. summary, the medium likelihood is based on our concerns about occasional high exposures resulting in a fish mortality event. However, as stated above, we expect exposure to this a.i. to be minimal, thereby decreasing the likelihood of a negative effect. Because the risk of this exposure is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of potential adverse effects to populations of Upper Columbia Spring Run Chinook as low. Given the uses, fate, and toxicity of the chemical, we do not expect mortality to be a common occurrence. Existing federal and state limitations, described in the a.i. summary in this section, further decrease the likelihood of high exposures. Thus, we believe that this compound has a low likelihood of causing an effect to the species as a whole.

Diuron: Diuron was determined to pose a medium risk to salmon. As discussed in the a.i. summary, use in rights-of-way is our largest concern, as it has the highest application rate and runoff potential. However, as stated above, we expect exposure to this a.i. to be minimal, thereby decreasing the likelihood of a negative effect. Because the risk of this exposure is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of potential adverse effects to populations of Upper Columbia Spring Run Chinook as low. In this ESU, diuron use in rights-of-way is strictly limited by state legislation. Given the use rates, pattern of uses for this chemical, and restrictive state regulations, we believe that there is a low likelihood of diuron causing a negative effect to multiple populations, or repeated negative effect to the same population. Therefore, we conclude that there is a low likelihood of an effect to the species as a whole.

Linuron: Given the limited uses of linuron, we expect exposure to be relatively infrequent. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Upper Columbia Spring Run Chinook as low. As discussed in the a.i. summary, linuron exposure in agricultural areas may have some effect on the primary productivity of a system. We anticipate that the effects of linuron exposure would have a minimal effect on populations within this ESU. Thus, there would be a low likelihood of an effect to the ESU as a whole.

Captan: In the a.i. summary, we concluded that captan has a low likelihood of affecting salmon. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Upper Columbia Spring Run Chinook as low. Due to the fate properties of captan, we stated that water concentrations will rarely be high enough to cause a physiological response. Therefore, there is a low likelihood of captan causing population level, or species level effects.

Chlorothalonil: In the a.i. summary, we determined that there was a medium likelihood of chlorothalonil having a negative effect on salmon. Higher or repeated exposures, particularly those associated with turf uses, may cause fish mortality. While we expect exposure from most uses to be minimal, turf uses may occur throughout the ESU so we cannot rule out this type of exposure. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Upper Columbia Spring Run Chinook as medium. NMFS believes that if any of the three populations was exposed to chlorothalonil as a result of turf uses, the resulting decrease in population numbers could be significant enough to affect the entire ESU. The distribution of populations and use sites across the range of the species indicate that it is possible that populations could experience substantial negative effects. Therefore, we believe that there is a medium potential for effects at the ESU level.

Snake River (SR) Fall-run Chinook Salmon (Threatened Species)

The SR Fall-run Chinook salmon ESU consists of one population that spawns in the lower mainstem Snake River. Its spatial distribution has been reduced to 10 to 15% of the historical range. The annual population growth rate for the population is just over replacement, and the ESU remains highly vulnerable due to low abundance. Genetic diversity has been reduced with the loss of additional populations and influx of hatchery raised spawners.

Pesticide use areas for the 6 a.i.s within this ESU's include evergreen forests (49%), cultivated crops (15%), pastures (1%), and developed lands (1%). The one population remaining in this ESU may experience some exposure to the six a.i.s. There is some developed and some agricultural area in the spawning and rearing areas, though they are generally set back from the river. Further, any exposure would occur in a high flow, high volume habitat which decreases the likelihood of experiencing a high concentration. Given the uses of these a.i.s, there may be adverse effects to some individuals, but we do not expect that population-level impacts will occur, except for 2,4-D BEE direct water applications. As there is only one population, we do not make separate population and species level calls in the following a.i. summaries.

2,4-D: 2,4-D was determined to pose a medium risk to fish. In the a.i. summary, we discuss three separate use patterns: direct water application of esters, direct water application of amines, terrestrial application of both esters and amines. While we expect exposure from most uses to be minimal,in cases of aquatic weed infestation, 2,4-D can be applied directly to the water. This use can occur anywhere, including areas with low routine pesticide use. . Therefore, we rate the likelihood of affecting the remaining population of Snake River Fall Run Chinook as medium.

Triclopyr BEE: In the preceding a.i. summary, we determined that Triclopyr BEE poses a medium risk to fish. As discussed in the a.i. summary, the medium likelihood is based on our concerns about occasional high exposures resulting in a fish mortality event. However, as stated above, we expect exposure to this a.i. to be minimal, thereby decreasing the likelihood of a negative effect. Thus, we believe that this compound has a low likelihood of causing an effect to the one population of Snake River Fall Run Chinook.

Diuron: Diuron was determined to pose a medium risk to salmon. As discussed in the a.i. summary, use in rights-of-way is our largest concern, as it has the highest application rate and runoff potential. However, as stated above, we expect exposure to this a.i. to be minimal, thereby decreasing the likelihood of a negative effect. Given the use rates and pattern of uses for this chemical we believe that there is a low likelihood of diuron causing a negative effect to the remaining population of Snake River Fall Run Chinook.

Linuron: Given the limited uses of linuron, we expect exposure to be relatively infrequent within the range of this ESU. We anticipate that the effects of linuron exposure would have a low effect on the remaining population of Snake River Fall run Chinook.

Captan: In the a.i. summary, we concluded that captan has a low likelihood of affecting salmon. Due to the fate properties of captan, we stated that water concentrations will rarely be high enough to cause a physiological response. Therefore, there is a low likelihood of captan affecting the population of Snake River Fall Run Chinook.

Chlorothalonil: In the a.i. summary, we determined that there was a medium likelihood of chlorothalonil having a negative effect on salmon. However, as stated above, we expect exposure to this a.i. to be minimal, thereby decreasing the likelihood of a negative effect. Therefore, we rate the likelihood of chlorothalonil affecting the remaining population of Snake River Fall Run Chinook as low.

Snake River (SR) Spring/Summer-run Chinook Salmon (Threatened Species)

This ESU includes 31 historical populations. Productivity trends are approaching replacement levels, though most populations are far below their respective interim recovery targets. Many individual populations have highly variable abundance and no positive long-term growth. The genetic diversity and spatial distribution of this ESU are intact.

The percentage of cultivated croplands and developed lands that overlap with SR Spring/Summer-run Chinook salmon habitat are 6.6% and 1.7%, respectively. Juvenile fish mature in fresh water for one year and may migrate from natal reaches into alternative summer-rearing or overwintering areas.

With the exception of direct water applications of 2, 4-D BEE, exposure of the a.i.s to the Snake River Spring-Summer Run populations is likely to be fairly low. As many spawn and rear in U.S. Forest Service lands, any pesticide use would be authorized under additional ESA consultations. Existing Opinions on weed control programs in these areas aim to minimize pesticide runoff and strictly limit the amount of land that can be treated. Given these conditions, we do not believe that populations in these areas will experience adverse effects from any of the a.i.s, except from direct water application of 2, 4-D BEE. Agricultural and urban areas are not common in the watersheds comprising the ESU, and those that are present are clustered mostly around the mainstem Snake and Columbia Rivers. Some populations may experience exposure from agricultural or urban uses, particularly during migration. Since these exposures will occur in a high volume high flow system, we expect population effects to be minimal.

2,4-D: 2,4-D was determined to pose a medium risk to fish. In the a.i. summary, we discuss three separate use patterns: direct water application of esters, direct water application of amines, terrestrial application of both esters and amines. While we expect exposure from most uses to be minimal, we are not able to predict where 2,4-D direct water applications may occur so we cannot rule out this type of exposure. Because the risk of these exposures is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of affecting populations of Snake River Spring-Summer Run Chinook as medium. Although effects from terrestrial applications are expected to reduce primary productivity in-stream, affect fish health, and degrade riparian vegetation in some locations, our greatest concern is the potential for salmonid lethality due to direct water applications of 2,4-D BEE. In cases of aquatic weed infestation, 2,4-D can be applied directly to the water. This use can occur anywhere, including areas with low routine pesticide use. NMFS believes that multiple populations may be exposed to these direct water applications within a single year, resulting in a decrease in population numbers significant enough to affect the entire

ESU. Based on risk from all use patterns, we rate the likelihood of 2,4-D affecting the ESU as medium.

Triclopyr BEE: In the preceding a.i. summary, we determined that Triclopyr BEE poses a medium risk to fish. As discussed in the a.i. summary, the medium likelihood is based on our concerns about occasional high exposures resulting in a fish mortality event. However, as stated above, we expect exposure to this a.i. to be minimal, thereby decreasing the likelihood of a negative effect. Because the risk of this exposure is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of potential adverse effects to populations of Snake River Spring-Summer Run Chinook as low. Given the uses, fate, and toxicity of the chemical, we do not expect mortality to be a common occurrence. Existing federal limitations, described in the a.i. summary in this section, further decrease the likelihood of high exposures. Thus, we believe that this compound has a low likelihood of causing an effect to the species as a whole.

Diuron: Diuron was determined to pose a medium risk to salmon. As discussed in the a.i. summary, use in rights-of-way is our largest concern, as it has the highest application rate and runoff potential. However, as stated above, we expect exposure to this a.i. to be minimal, thereby decreasing the likelihood of a negative effect. Because the risk of this exposure is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of potential adverse effects to populations of Snake River Spring-Summer Run Chinook as low. Given the use rates and pattern of uses for this chemical, we believe that there is a low likelihood of diuron causing a negative effect to multiple populations, or repeated negative effect to the same population. Therefore, we conclude that there is a low likelihood of an effect to the species as a whole.

Linuron: Given the limited uses of linuron, we expect exposure to be relatively infrequent. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Snake River Spring-Summer Run Chinook as low. As discussed in the a.i. summary, linuron exposure in agricultural areas may have some effect on the primary productivity of a system. We anticipate that the effects of linuron exposure would have a minimal effect on populations within this ESU. Thus, there would be a low likelihood of an effect to the ESU as a whole.

Captan: In the a.i. summary, we concluded that captan has a low likelihood of affecting salmon. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Snake River Spring-Summer Run Chinook as low. Due to the fate properties of captan, we stated that water concentrations will rarely be high enough to cause a physiological response. Therefore, there is a low likelihood of captan causing population level, or species level effects.

Chlorothalonil: In the a.i. summary, we determined that there was a medium likelihood of chlorothalonil having a negative effect on salmon. However, as stated above, we expect exposure to this a.i. to be minimal, thereby decreasing the likelihood of a negative effect. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of adverse effects to populations of Snake River Spring-Summer Run Chinook as low. Exposure to chlorothalonil is further restricted by federal and state legislation. As such, there is a low likelihood of multiple populations experiencing adverse effects. Therefore, we believe that there is a low potential for effects at the ESU level.

Upper Willamette River (UWR) Chinook Salmon (Threatened Species)

The UWR Chinook salmon ESU is composed of seven populations. Of these, only the McKenzie population is producing naturally. Abundance is low for all populations, and growth rates are negative. The spatial distribution of this ESU has been dramatically reduced, with 30 to 40% of the total historic habitat blocked by dams. The genetic diversity of this ESU has been compromised by hatchery stocks and mixing between populations.

The percentage of cultivated and developed lands that overlap with UWR Chinook salmon habitat are 10.5% and 9%, respectively. Our GIS analysis indicates all populations in this ESU may be exposed to pesticides applied in agriculture and urban areas. Juveniles rear in the mainstem Willamette River and floodplain wetlands during the inundation period. Residence periods range from 6 months to over a year, with three distinct emigration runs.

We expect that populations within this ESU will be exposed to the six a.i.s due to the high degree of agricultural and developed land classes. Specialty crops are likely to be grown in this area, increasing the likelihood of linuron exposure. Further, while some of the spawning and rearing streams are in forested areas, they are not necessarily in Federal lands. As such, we cannot assume any additional protections from other ESA consultations. The valley is also heavily used by railroads, roads, and electrical transmission lines, increasing the likelihood of rights-of-way applications. We also expect that environmental mixtures will compound the effects of these chemicals. Use of the a.i.s is likely to cause a decrease in primary production, chronic effects, and may result in fish mortality. 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.

2,4-D: 2,4-D was determined to pose a medium risk to fish. In the a.i. summary, we discuss three separate use patterns: direct water application of esters, direct water application of amines, terrestrial application of both esters and amines. Because the risk of these exposures is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of affecting populations of Upper Willamette River Chinook as medium. Although effects from terrestrial applications are expected to reduce primary productivity in-stream, affect fish health, and degrade riparian vegetation in some locations, our greatest concern is the potential for salmonid lethality due to direct water applications of 2,4-D BEE. NMFS believes that multiple populations may be exposed to these direct water applications within a single year, resulting in a decrease in population numbers significant enough to affect the entire ESU. Based on risk from all use patterns, we rate the likelihood of 2,4-D affecting the ESU as medium.

Triclopyr BEE: In the preceding a.i. summary, we determined that Triclopyr BEE poses a medium risk to fish. As discussed in the a.i. summary, the medium likelihood is based on our concerns about occasional high exposures resulting in a fish mortality event. If a high concentration of the a.i. was to co-occur with peak fish presence, this could cause a significant drop in the size of population. Because the risk of this exposure is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Upper Willamette River Chinook as medium. In order to decrease lambda of the species, multiple mortality events would have to occur, either within the same population or across several populations within the ESU. Given the uses, fate, and toxicity of the chemical, we do not expect mortality to be a common occurance. Thus, we believe that this compound has a medium likelihood of causing an effect to the species as a whole.

Diuron: Diuron was determined to pose a medium risk to salmon. As discussed in the a.i. summary, use in rights-of-way is our largest concern, as it has the highest application rate and runoff potential. Because the risk of this exposure is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Upper Willamette River Chinook as medium. We expect exposure from other uses to lead to declines in the primary productivity of the system, but not to wide-scale fish-health issues. Given the use rates, pattern of uses for this chemical, and relatively widespread agriculture, we believe that there is a medium likelihood of diuron causing a negative effect to multiple populations, or repeated negative effect to the same population. Therefore, we conclude that there is a medium likelihood of an effect to the species as a whole.

Linuron: Given the limited uses of linuron, we expect exposure to be relatively infrequent. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Upper Willamette River Chinook as low. As discussed in the a.i. summary, linuron exposure in agricultural areas may have some effect on the primary productivity of a system. We anticipate that the effects of linuron exposure would have a minimal effect on populations within this ESU. Thus, there would be a low likelihood of an effect to the ESU as a whole.

Captan: In the a.i. summary, we concluded that captan has a low likelihood of affecting salmon. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Upper Willamette

River Chinook as low. Due to the fate properties of captan, we stated that water concentrations will rarely be high enough to cause a physiological response. Therefore, there is a low likelihood of captan causing population level, or species level effects.

Chlorothalonil: In the a.i. summary, we determined that there was a medium likelihood of chlorothalonil having a negative effect on salmon. Higher or repeated exposures, particularly those associated with turf uses, may cause fish mortality. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Upper Willamette River Chinook as medium. The distribution of populations and use sites across the range of the species indicate that it is possible that multiple populations could experience substantial negative effects. Therefore, we believe that there is a medium potential for effects at the ESU level.

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 percentage of cultivated croplands and developed lands that overlap with CC Chinook salmon habitat are 1% and 5.4%, respectively. The most abundant populations are in the Eel River and tributaries, and in the Russian River watershed. While there is little overlap of use sites with the habitat of the Eel River populations, there is substantial overlap in the Russian River watershed. Due to the importance of this population to the ESU, likelihood of negative effects ratings were based primarily on the overlap in this watershed. Juveniles rear in freshwater streams for a few months, and may reside in the estuary for an extended period before entering the ocean.

In general, we expect the populations to have limited exposure to the a.i.s, except for direct water applications of 2,4-D BEE. There is a low amount of development, agriculture, and rights-of-way uses within the range of the ESU. We also expect that the population in the Russian River will have a much higher degree of exposure due to the distribution of land uses. This is particularly important for fungicide exposure, as grapes are an important crop in the Russian River valley. We expect that some individuals may experience either low level chronic effects, or even direct lethality.

2,4-D: 2,4-D was determined to pose a medium risk to fish. In the a.i. summary, we discuss three separate use patterns: direct water application of esters, direct water application of amines, terrestrial application of both esters and amines. While we expect exposure from most uses to be minimal, we are not able to predict where 2,4-D direct water applications may occur so we cannot rule out this type of exposure. Because the risk of this exposure is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of affecting populations of CC Chinook as medium. Although effects from terrestrial applications are expected to reduce primary productivity in-stream, affect fish health, and degrade riparian vegetation in some locations, our greatest concern is the potential for salmonid lethality due to direct water applications of 2,4-D BEE. NMFS believes that multiple populations may be exposed to these direct water applications within a single year, resulting in a decrease in population numbers significant enough to affect the entire ESU. Based on risk from all use patterns, we rate the likelihood of 2,4-D affecting the ESU as medium.

Triclopyr BEE: In the preceding a.i. summary, we determined that Triclopyr BEE poses a medium risk to fish. As discussed in the a.i. summary, the medium likelihood is based on our concerns about occasional high exposures resulting in a fish mortality event. The Russian River system is the most developed, but NMFS does not expect it will receive significantly more inputs from rights-of-way uses as it is not a major transportation or utility route. Because the risk of this exposure is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of affecting populations of CC Chinook as low. In order to decrease lambda of the species, multiple mortality events would have to occur, either within the same population or across several populations within the ESU. Given the uses, fate, and toxicity of the chemical, we do not expect mortality to be a common occurrence. Thus, we believe that this compound has a low likelihood of causing an effect to the species as a whole.

Diuron: Diuron was determined to pose a medium risk to salmon. As discussed in the a.i. summary, use in rights-of-way is our largest concern, as it has the highest application rate and runoff potential. Because the risk of this type of exposure is greater for the population in the Russian River due to development, we rate the likelihood for this population as medium and the likelihood for the remaining populations as low. Therefore we rate the likelihood to populations of CC Chinook as low to medium. We expect exposure to lead to declines in the primary productivity of the system, but not to wide-scale fish-health issues. Given the use rates and pattern of uses for this chemical, we believe that there is a low likelihood of diuron causing a negative effect to multiple populations, or repeated negative effect to the same population. Therefore, we conclude that there is a low likelihood of an effect to the species as a whole.

Linuron: Given the limited uses of linuron, we expect exposure to be relatively infrequent, even in the Russian River watershed. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of CC Chinook as low. As discussed in the a.i. summary, linuron exposure in agricultural areas may have some effect on the primary productivity of a system. We anticipate that the effects of linuron exposure would have a minimal effect on populations within this ESU. Thus, there would be a low likelihood of an effect to the ESU as a whole.

Captan: In the a.i. summary, we concluded that captan has a low likelihood of affecting salmon. Due to cropping patterns, we expect that captan will be used more frequently in Russian River watershed. However, NMFS believes that risk to this population is still low based on captan's fate properties. Therefore, we do not find a reason to differentiate risk among populations, and so we rate the likelihood to populations of CC Chinook as low. As discussed earlier, NMFS believes that water concentrations will rarely be high enough to cause a physiological response. Therefore, there is a low likelihood of captan causing population level, or species level effects.

Chlorothalonil: In the a.i. summary, we determined that there was a medium likelihood of chlorothalonil having a negative effect on salmon. Higher or repeated exposures may cause fish mortality. We expect that these uses will occur more frequently in the Russian River watershed, due to a higher concentration of turf uses and the prevalence of higher-use agricultural crops. Historically, the Russian River population was a potential source population that helped to sustain other populations. As such, it weighs heavily in our population level analysis, so we rate the likelihood of adverse effects to populations of CC Chinook as medium. This consideration also weighs heavily on the ESU analysis, as negative effects to this population can result in lower numbers for multiple populations. Therefore, we believe that there is a medium potential for effects at the ESU level.

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, Harrell, & Nobriga, 2005; Sommer, et al., 2001). 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.

We expect that individuals within this population will be exposed to the six a.i.s. Their range is heavily developed, for both agricultural and urban purposes. The percentage of cultivated croplands and developed lands that overlap with CV Chinook salmon habitat are 21.3% and

10.8%, respectively. The valley also has a high concentration of power and transportation lines, indicating that rights-of-way applications will also occur. Most spawning occurs in the upper waters of three Northern watersheds which are largely undeveloped, thus lowering the likelihood of exposure to some life stages. Much of the rearing and migration of these populations occurs along the Sacramento River, where exposure to the a.i.s is likely to occur. As this area is highly developed, we expect that fish will be exposed to a variety of environmental mixtures. They are also likely to experience pesticide inputs from multiple sources, increasing the likelihood of exposure to each a.i. at intervals shorter than the labeled application interval. We expect that all populations may be exposed to the a.i.s during the rearing period, and may experience adverse, chronic effects from this exposure. There is the possibility that some lethality will occur as well. However, the likelihood of such high concentrations being coincident in time and space with a significant portion of the population is low.

2,4-D: 2,4-D was determined to pose a medium risk to fish. In the a.i. summary, we discuss three separate use patterns: direct water application of esters, direct water application of amines, terrestrial application of both esters and amines. Because the risk of these exposures is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of affecting populations of CV Spring - run Chinook as medium. Although effects from terrestrial applications are expected to reduce primary productivity in-stream, affect fish health, and degrade riparian vegetation in some locations, our greatest concern is the potential for salmonid lethality due to direct water applications of 2,4-D BEE. NMFS believes that multiple populations may be exposed to these direct water applications within a single year, resulting in a decrease in population numbers significant enough to affect the entire ESU. Based on risk from all use patterns, we rate the likelihood of 2,4-D affecting the ESU as medium.

Triclopyr BEE: In the preceding a.i. summary, we determined that Triclopyr BEE poses a medium risk to fish. As discussed in the a.i. summary, the medium likelihood is based on our concerns about occasional high exposures resulting in a fish mortality event. If a high concentration of the a.i. was to co-occur with peak fish presence, this could cause a significant drop in the size of population. Because the risk of this exposure is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of CV Spring - run Chinook as medium. In order to decrease lambda of the species, multiple mortality events would have to occur, either within the same population or across several populations within the ESU. Given the uses, fate, and toxicity of the chemical, we do not expect mortality to be a common occurrence. However, because of the level of development, we expect use to be much more frequent, thereby increasing the likelihood of mortality. Thus, we believe that this compound has a medium likelihood of causing an effect to the species as a whole.

Diuron: Diuron was determined to pose a medium risk to salmon. As discussed in the a.i. summary, use in rights-of-way is our largest concern, as it has the highest application rate and runoff potential. Because the risk of this exposure is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of CV Spring - run Chinook as medium. We expect exposure from other uses to lead to declines in the primary productivity of the system, but not to wide-scale fish-health issues. Given the use rates, pattern of uses for this chemical, and relatively widespread development, we believe that there is a medium likelihood of diuron causing a negative effect to multiple populations, or repeated negative effect to the same population. Therefore, we conclude that there is a medium likelihood of an effect to the species as a whole.

Linuron: Given the limited uses of linuron, we expect exposure to be relatively infrequent. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of CV Spring - run Chinook as low. As discussed in the a.i. summary, linuron exposure in agricultural areas may have some effect on the primary productivity of a system. We anticipate that the effects of linuron exposure would have a minimal effect on populations within this ESU. Thus, there would be a low likelihood of an effect to the ESU as a whole.

Captan: In the a.i. summary, we concluded that captan has a low likelihood of affecting salmon. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of CV Spring - run

Chinook as low. Due to the fate properties of captan, we stated that water concentrations will rarely be high enough to cause a physiological response. Therefore, there is a low likelihood of captan causing population level, or species level effects.

Chlorothalonil: In the a.i. summary, we determined that there was a medium likelihood of chlorothalonil having a negative effect on salmon. Higher or repeated exposures, particularly those associated with turf uses, may cause fish mortality. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of CV Spring - run Chinook as medium. The distribution of populations and use sites across the range of the species indicate that it is possible that multiple populations could experience substantial negative effects. Therefore, we believe that there is a medium potential for effects at the ESU level.

Sacramento River Winter-run Chinook Salmon (Endangered Species)

The Sacramento River Winter-run Chinook salmon ESU is now comprised of a single population. This population rears in the mainstem of the Sacramento River below Keswick Dam. Abundance and productivity have fluctuated greatly over the past two decades. The genetic diversity of this population has been reduced through small population sizes and the influence of hatchery fish. The large fluctuations in productivity and abundance indicate that the species is highly vulnerable to extinction.

We expect that the one population in this ESU may be exposed to the six a.i.s, as its range is restricted to the mainstem Sacramento River. The Central Valley has significant agricultural and urban development, and is a main corridor for many utilities that may use the a.i.s on rights-of-way. The percentage of cultivated croplands and developed lands that overlap with Sacramento River Winter-run Chinook salmon are 25% and 10%, respectively. As this area is highly developed, we expect that fish will be exposed to a variety of environmental mixtures. They are also likely to experience pesticide inputs from multiple sources, increasing the likelihood of exposure to each a.i. at intervals shorter than the labeled application interval. 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. We expect that some individuals from this ESU will experience adverse chronic effects from exposure to the a.i.s. As there is only one population, we do not make separate population and species level calls in the following a.i. summaries.

2,4-D: 2,4-D was determined to pose a medium risk to fish. In the a.i. summary, we discuss three separate use patterns: direct water application of esters, direct water application of amines, terrestrial application of both esters and amines. Although effects from terrestrial applications are expected to reduce primary productivity in-stream, affect fish health, and degrade riparian vegetation in some locations, our greatest concern is the potential for salmonid lethality due to direct water applications of 2,4-D BEE. Based on risk from all use patterns, we rate the likelihood of 2,4-D affecting the ESU as medium.

Triclopyr BEE: In the preceding a.i. summary, we determined that Triclopyr BEE poses a medium risk to fish. As discussed in the a.i. summary, the medium likelihood is based on our concerns about occasional high exposures resulting in a fish mortality event. If a high concentration of the a.i. was to co-occur with peak fish presence, this could cause a significant drop in the size of population. Given the uses, fate, and toxicity of the chemical, we do not expect mortality to be a common occurrence. However, because of the level of development, we expect use to be much more frequent, thereby increasing the likelihood of mortality. Thus, we believe that this compound has a medium likelihood of causing an effect to the remaining population and Sacramento River Winter-run species as a whole.

Diuron: Diuron was determined to pose a medium risk to salmon. As discussed in the a.i. summary, use in rights-of-way is our largest concern, as it has the highest application rate and runoff potential. We expect exposure from other uses to lead to declines in the primary productivity of the system, but not to wide-scale fish-health issues. Given the use rates, pattern of uses for this chemical, and relatively widespread development, we believe that there is a medium likelihood of diuron causing a negative effect to the Sacramento River Winter-run population and species as a whole.

Linuron: Given the limited uses of linuron, we expect exposure to be relatively infrequent. As discussed in the a.i. summary, linuron exposure in agricultural areas may have some effect on the primary productivity of a system. We anticipate that the effects of linuron exposure would have a minimal effect on the population within the Sacramento River Winter-run Chinook. Thus, there would be a low likelihood of an effect to the ESU as a whole.

Captan: In the a.i. summary, we concluded that captan has a low likelihood of affecting salmon. Due to the fate properties of captan, we stated that water concentrations will rarely be high enough to cause a physiological response. Therefore, there is a low likelihood of captan causing population level, or species level effects to the Sacramento River Winter-run Chinook.

Chlorothalonil: In the a.i. summary, we determined that there was a medium likelihood of chlorothalonil having a negative effect on salmon. Higher or repeated exposures, particularly those associated with turf uses, may cause fish mortality. Given number and distribution of use sites throughout the ESU, we believe that there is a medium potential for effects to the remaining Sacramento River Winter-run Chinook.

Hood Canal Summer-run Chum Salmon (Threatened Species)

This ESU has two remaining independent populations made up of multiple spawning aggregations. Much of the historical spatial structure has been lost; with the exception of the Union River, populations on the eastern side of the canal are extirpated. Despite being low, the genetic diversity of the ESU has increased from the low values seen in the 1990s. The two populations have long-term trends above replacement, and while they have increased since the time of listing, abundance is still considered low. The life history of this ESU strongly influences the potential for exposure. Following emergence, fish typically migrate quickly to nearshore marine areas in Hood Canal and Discovery Bay to rear and grow. Average rearing time for juveniles is around 23 days before emigration to the outer Strait of Juan de Fuca and Pacific Ocean.

The area occupied by this ESU is largely undeveloped; roughly 50% of the land is federally owned within the Olympic National Forest. The Forest Service has already consulted on the use of herbicides for invasive plant control within the Olympic Forest, so we are not concerned about forestry use in those areas. Exposure from urban and agricultural lands is likely to be low, as there is a small amount of development. Correspondingly, we expect a low amount of rights-of-way uses. 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.

2,4-D: 2,4-D was determined to pose a medium risk to fish. In the a.i. summary, we discuss three separate use patterns: direct water application of esters, direct water application of amines, terrestrial application of both esters and amines. While we expect exposure from most uses to be minimal, we are not able to predict where 2,4-D direct water applications may occur so we cannot rule out this type of exposure. Because the risk of these exposures is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of affecting populations of Hood Canal Chum as medium. Although effects from terrestrial applications are expected to reduce primary productivity in-stream, affect fish health, and degrade riparian vegetation in some locations, our greatest concern is the potential for salmonid lethality due to direct water applications of 2,4-D BEE. In cases of aquatic weed infestation, 2,4-D can be applied directly to the water. This use can occur anywhere, including areas with low routine pesticide use. NMFS believes that if either of these populations were exposed to these direct water applications, the resulting decrease in population numbers could be significant enough to affect the entire ESU. Based on risk from all use patterns, we rate the likelihood of 2,4-D affecting the ESU as medium.

Triclopyr BEE: In the preceding a.i. summary, we determined that Triclopyr BEE poses a medium risk to fish. As discussed in the a.i. summary, the medium likelihood is based on our concerns about occasional high exposures resulting in a fish mortality event. However, as stated above, we expect exposure to this a.i. to be minimal, thereby decreasing the likelihood of a negative effect. Because the risk of this exposure is fairly consistent between the two populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of potential adverse effects to populations of Hood Canal Chum as low.

Given the uses, fate, and toxicity of the chemical, we do not expect mortality to be a common occurrence. Existing federal and state limitations, described in the a.i. summary in this section, further decrease the likelihood of high exposures. Thus, we believe that this compound has a low likelihood of causing an effect to the species as a whole.

Diuron: Diuron was determined to pose a medium risk to salmon. As discussed in the a.i. summary, use in rights-of-way is our largest concern, as it has the highest application rate and runoff potential. However, as stated above, we expect exposure to this a.i. to be minimal, thereby decreasing the likelihood of a negative effect. Because the risk of this exposure is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of potential adverse effects to the populations of Hood Canal Chum as low. In this ESU, diuron use in rights-of-way is strictly limited by state legislation. Given the use rates, pattern of uses for this chemical, and restrictive state regulations, we believe that there is a low likelihood of diuron causing a negative effect to multiple populations, or repeated negative effect to the same population. Therefore, we conclude that there is a low likelihood of an effect to the species as a whole.

Linuron: Given the limited uses of linuron, we expect exposure to be relatively infrequent. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Hood Canal Chum as low. Due to the very low amount of agricultural land within the ESU, we anticipate that the effects of linuron exposure would have a minimal effect on populations within this ESU. Thus, there would be a low likelihood of an effect to the ESU as a whole.

Captan: In the a.i. summary, we concluded that captan has a low likelihood of affecting salmon. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Hood Canal Chum as low. Due to the fate properties of captan, we stated that water concentrations will rarely be high enough to cause a physiological response. Therefore, there is a low likelihood of captan causing population level, or species level effects.

Chlorothalonil: In the a.i. summary, we determined that there was a medium likelihood of chlorothalonil having a negative effect on salmon. Higher or repeated exposures, particularly those associated with turf uses, may cause fish mortality. While we expect exposure from most uses to be minimal, turf uses may occur throughout the ESU so we cannot rule out this type of exposure. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Hood Canal Chum as medium. NMFS believes that the distribution of populations and use sites across the range of the species indicate that it is possible that populations could experience substantial negative effects. Therefore, we believe that there is a medium potential for effects at the ESU level.

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 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. The Grays River population is largely in undeveloped areas, thus lowering the likelihood of exposure to the a.i.s. The Upper Gorge population is more likely to be exposed, as individuals must migrate past the Portland area, which includes the upstream contributions from the Willamette basin.

Adult chum salmon spawning and occurs in the late fall, from mid-October to December. 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. This relatively short residence period in fresh water results in chum having a lower likelihood of exposure than other salmonids.

2,4-D: 2,4-D was determined to pose a medium risk to fish. In the a.i. summary, we discuss three separate use patterns: direct water application of esters, direct water application of amines, terrestrial application of both esters and amines. While we expect exposure from most uses to be minimal, we are not able to predict where 2,4-D direct water applications may occur so we cannot rule out this type of exposure. Because the risk of these exposures is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of affecting populations of Columbia River Chum as medium. Although effects from terrestrial applications are expected to reduce primary productivity in-stream, affect fish health, and degrade riparian vegetation in some locations, our greatest concern is the potential for salmonid lethality due to direct water applications of 2,4-D BEE. NMFS believes that if either of these populations were exposed to these direct water applications, the resulting decrease in population numbers could be significant enough to affect the entire ESU. Based on risk from all use patterns, we rate the likelihood of 2,4-D affecting the ESU as medium.

Triclopyr BEE: In the preceding a.i. summary, we determined that Triclopyr BEE poses a medium risk to fish. As discussed in the a.i. summary, the medium likelihood is based on our concerns about occasional high exposures resulting in a fish mortality event. Based on land use data within the watersheds and low residency periods, NMFS believes that there is a low likelihood of either population experiencing this type of exposure. Due to the properties of the a.i. and the high volume – high flow conditions in the mainstem Columbia River, we do not believe there will be much exposure during the migration of Lower Gorges individuals. Therefore, the risk of this type of exposure to triclopyr BEE is similar for the two populations. As such, we rate the likelihood of affecting populations of Lower Columbia River Chum as low. Given the uses, fate, and toxicity of the chemical, we do not expect mortality to be a common occurrence. Existing federal and state limitations, described in the a.i. summary in this section, further decrease the likelihood of high exposures. Thus, we believe that this compound has a low likelihood of causing an effect to the species as a whole.

Diuron: Diuron was determined to pose a medium risk to salmon. As discussed in the a.i. summary, use in rights-of-way is our largest concern, as it has the highest application rate and runoff potential. Based on land use data within the watersheds and low residency periods, NMFS believes that there is a low likelihood of either population experiencing this type of exposure. Due to the high volume – high flow conditions in the mainstem Columbia River, we do not believe there will be much exposure during the migration of Lower Gorges individuals. Therefore, the risk of this type of exposure is similar for the two populations. Therefore we rate the likelihood to populations of Lower Columbia River chum as low. In this ESU, diuron use in rights-of-way is strictly limited by state legislation. We expect exposure from other uses to lead to declines in the primary productivity of the system, but not to wide-scale fish-health issues. Given the use rates, pattern of uses for this chemical, and restrictive state regulations, we believe that there is a low likelihood of diuron causing a negative effect to either population. Therefore, we conclude that there is a low likelihood of an effect to the species as a whole.

Linuron: Given the limited uses of linuron, we expect exposure to be relatively infrequent. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Lower Columbia chum as low. As discussed in the a.i. summary, linuron exposure in agricultural areas may have some effect on the primary productivity of a system. We anticipate that the effects of linuron exposure would have a minimal effect on the populations within this ESU. Thus, there would be a low likelihood of an effect to the ESU as a whole.

Captan: In the a.i. summary, we concluded that captan has a low likelihood of affecting salmon. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Lower Columbia Chinook as low. Due to the fate properties of captan, we stated that water concentrations will rarely be high enough to cause a physiological response. Therefore, there is a low likelihood of captan causing population level, or species level effects.

Chlorothalonil: In the a.i. summary, we determined that there was a medium likelihood of chlorothalonil having a negative effect on salmon. Higher or repeated exposures, particularly those associated with turf uses, may cause fish mortality. Because turf uses are distributed throughout the ESU, the risk is fairly consistent among populations. Therefore, we do not find a reason to differentiate risk among populations, and we rate the likelihood to populations of Lower Columbia River Chinook as medium. NMFS believes that if either of these populations were exposed to high applications, the resulting decrease in population numbers could be significant enough to affect the entire ESU. Therefore, we believe that there is a medium potential for effects at the ESU level.

Lower Columbia River Coho Salmon (Threatened Species)

The LCR coho salmon ESU now consists of two populations found in the Sandy and Clackamas Rivers. Both populations have low levels of abundance. The diversity of populations has been eroded by large hatchery influences and low effective population sizes. The spatial structure for this ESU has also been drastically reduced compared to historical levels. Additionally, coho have the most sensitive life history of the salmonids, as they have three distinct cohorts.

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%) make up a fairly substantial portion of this ESU. The percentage of cultivated croplands and developed lands that overlap with LCR chum salmon habitat are 2% and 11.7%, respectively.

The forested areas are largely private, rather than federally controlled. While the spawning areas are in tributaries located in lower-use areas, we expect that these individuals will be exposed to the a.i.s during rearing and migration. The two populations in this ESU must both navigate the waters around Portland, where there is an abundance of rights-of-way in addition to urban and agricultural development. We expect that these populations will have significant exposure from rights-of-way uses. This area has a high concentration of rights-of-way from rail, road, and utilities. These uses are of greater concern as they tend to be higher use rates with greater probability of runoff. This concern is mediated somewhat by the wide distribution of populations throughout the basin, and the fact that exposure will likely occur in higher volume, higher flow habitats, such as the Columbia River. Given the higher likelihood of exposure based on geographic distribution and the higher sensitivity of the species, there is a greater likelihood that the populations, and the ESU as a whole, will be negatively affected by the use of these a.i.s. The likelihood of negative effects is further influenced by the properties of the chemicals themselves.

2,4-D: 2,4-D was determined to pose a medium risk to fish. In the a.i. summary, we discuss three separate use patterns: direct water application of esters, direct water application of amines, terrestrial application of both esters and amines. Because the risk of these exposures is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of affecting populations of Lower Columbia River coho as medium. Although effects from terrestrial applications are expected to reduce primary productivity in-stream, affect fish health, and degrade riparian vegetation in some locations, our greatest concern is the potential for salmonid lethality due to direct water applications of 2,4-D BEE. NMFS believes that if either of these populations were exposed to these direct water applications, the resulting decrease in population numbers could be significant enough to affect the entire ESU. Based on risk from all use patterns, we rate the likelihood of 2,4-D affecting the ESU as medium.

Triclopyr BEE: In the preceding a.i. summary, we determined that Triclopyr BEE poses a medium risk to fish. As discussed in the a.i. summary, the medium likelihood is based on our concerns about occasional high exposures resulting in a fish mortality event. If a high concentration of the a.i. was to co-occur with peak fish presence, this could cause a significant drop in the size of population. Because the risk of these exposures is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of affecting populations of Lower Columbia River coho as medium. Given the uses, fate, and toxicity of the chemical, we do not expect mortality to be a common occurrence. However, given the location of the remaining populations, the proximity of potential high-use areas, and the more sensitive life history, we believe that this compound has a medium likelihood of causing an effect to the species as a whole.

Diuron: Diuron was determined to pose a medium risk to salmon. As discussed in the a.i. summary, use in rights-of-way is our largest concern, as it has the highest application rate and runoff potential. Because the risk of these exposures is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of affecting populations of Lower Columbia River coho as medium. We expect exposure from other uses to lead to declines in the primary productivity of the system, but not to wide-scale fish-health issues. However, given the location of the remaining populations, the proximity of potential high-use areas, and the more sensitive life history, we believe that this compound has a medium likelihood of causing an effect to the species as a whole.

Linuron: Given the limited uses of linuron, we expect exposure to be relatively infrequent. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Lower Columbia chum as low. As discussed in the a.i. summary, linuron exposure in agricultural areas may have some effect on the primary productivity of a system. Again, as we expect there is a low likelihood of exposure, we anticipate that the effects of linuron exposure would have a minimal effect on populations within this ESU. Thus, there would be a low likelihood of an effect to the ESU as a whole.

Captan: In the a.i. summary, we concluded that captan has a low likelihood of affecting salmon. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Lower Columbia chum as low. Due to the fate properties of captan, we stated that water concentrations will rarely be high enough to cause a physiological response. Therefore, there is a low likelihood of captan causing a species level effect.

Chlorothalonil: In the a.i. summary, we determined that there was a medium likelihood of chlorothalonil having a negative effect on salmon. Higher or repeated exposures, particularly those associated with turf uses, may cause fish mortality. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Lower Columbia River Chinook as medium. As the Portland area has a high proportion of rights-of-way uses, we expect that the remaining populations may experience high exposures, resulting in substantial negative effects. NMFS believes that if either of these populations were exposed to high applications, the resulting decrease in population numbers could be significant enough to affect the entire ESU. Therefore, we believe that there is a medium potential for effects at the ESU level.

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 strong however, populations within the ESU experience recruitment failure and long-term negative growth (Good, Waples et al. 2005). Spatial distribution is relatively intact. 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 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. Juvenile coho salmon are often found in small streams less than five feet wide and rear in fresh water for 18 months.

A large portion of this ESU's range is Forest Service land. As any pesticide applications would undergo a separate consultation, we are less concerned with uses within these areas. The low amounts of urban and agricultural lands also indicate a lower likelihood of exposure. While there is the possibility of exposure and subsequent negative effects to individuals, we believe that the potential for negative population level effects is low. The spatial distribution of the populations combined with the distribution of use sites, and relatively low expected use, we do not believe that most a.i.s will have a large enough impact to negatively affect the ESU.

2,4-D: 2,4-D was determined to pose a medium risk to fish. In the a.i. summary, we discuss three separate use patterns: direct water application of esters, direct water application of amines, terrestrial application of both esters and amines. While we expect exposure from most uses to be minimal, we are not able to predict where 2,4-D direct water applications may occur so we cannot rule out this type of exposure. Because the risk of these exposures is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of affecting populations of Oregon Coast coho as medium. Although effects from terrestrial applications are expected to reduce primary productivity in-stream, affect fish health, and degrade riparian vegetation in some locations, our greatest concern is the potential for salmonid lethality due to direct water applications of 2,4-D BEE. In cases of aquatic weed infestation, 2,4-D can be applied directly to the water. This use can occur anywhere, including areas with low routine pesticide use. NMFS believes that multiple populations may be exposed to these direct water applications within a single year, resulting in a decrease in population numbers significant enough to affect the entire ESU. Based on risk from all use patterns, we rate the likelihood of 2,4-D affecting the ESU as medium.

Triclopyr BEE: In the preceding a.i. summary, we determined that Triclopyr BEE poses a medium risk to fish. As discussed in the a.i. summary, the medium likelihood is based on our concerns about occasional high exposures resulting in a fish mortality event. However, as stated above, we expect exposure to this a.i. to be minimal, thereby decreasing the likelihood of a negative effect. Because the risk of this exposure is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of potential adverse effects to populations of Oregon Coast coho as low. Given the uses, fate, and toxicity of the chemical, we do not expect mortality to be a common occurrence. Existing federal limitations, described in the a.i. summary in this section, further decrease the likelihood of high exposures. Thus, we believe that this compound has a low likelihood of causing an effect to the species as a whole.

Diuron: Diuron was determined to pose a medium risk to salmon. As discussed in the a.i. summary, use in rights-of-way is our largest concern, as it has the highest application rate and runoff potential. However, as stated above, we expect exposure to this a.i. to be minimal, thereby decreasing the likelihood of a negative effect. Because the risk of this exposure is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of potential adverse effects to populations of Oregon Coast coho as low. Given the use rates and pattern of uses for this chemical, we believe that there is a low likelihood of diuron causing a negative effect to multiple populations, or repeated negative effect to the same population. Therefore, we conclude that there is a low likelihood of an effect to the species as a whole.

Linuron: Given the limited uses of linuron, we expect exposure to be relatively infrequent. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Oregon Coast coho as low. As discussed in the a.i. summary, linuron exposure in agricultural areas may have some effect on the primary productivity of a system. We anticipate that the effects of linuron exposure would have a minimal effect on populations within this ESU. Thus, there would be a low likelihood of an effect to the ESU as a whole.

Captan: In the a.i. summary, we concluded that captan has a low likelihood of affecting salmon. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Oregon Coast coho as low. Due to the fate properties of captan, we stated that water concentrations will rarely be high enough to cause a physiological response. Therefore, there is a low likelihood of captan causing population level, or species level effects.

Chlorothalonil: In the a.i. summary, we determined that there was a medium likelihood of chlorothalonil having a negative effect on salmon. However, as stated above, we expect exposure to this a.i. to be minimal, thereby decreasing the likelihood of a negative effect. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of adverse effects to populations of

Oregon Coast coho as low. Exposure to chlorothalonil is further restricted by federal and state legislation. As such, there is a low likelihood of multiple populations experiencing adverse effects. Therefore, we believe that there is a low potential for effects at the ESU level.

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 spatial structure and genetic diversity of this ESU. Distribution within individual watersheds has been reduced throughout the entire range. 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 percentage of cultivated croplands and developed lands that overlap with SONCC coho salmon habitat are 2.5% and 4.3%, respectively. As little population data were available for this ESU, we were not able to determine if agricultural and developed areas, which cluster in certain watersheds, co-occur with important populations. Areas with more cropland include the Scott and Shasta watersheds in the Klamath basin, and the Upper and Middle Rough River[13] watersheds. Of the development in this ESU, much is in the Rough River basin, with most of the rest distributed along the coastline and estuaries. The fry rear in backwater, side channels, and shallow channel edges for up to 18 months.

We expect that this ESU will have fairly low exposure to the a.i.s, due to the low agricultural and urban development within its range. Rights-of-way uses are also expected to be low. Roughly 36% of the land is federally owned, including parts of the Redwood forest. While the spatial structure of the population is not well understood, salmon are present throughout the range. Individuals may be exposed to the a.i.s and experience adverse effects. However, given the

[13] The Rough River is also be referred to as the Rouge or Rouge River in other publications, maps, or websites

distribution of land uses, it is unlikely that a large portion of the ESU would experience a high exposure event for any chemical except 2,4-D BEE when applied to water. Therefore, we believe that there is a low likelihood of an ESU-level effect for most a.i.s.

2,4-D: 2,4-D was determined to pose a medium risk to fish. In the a.i. summary, we discuss three separate use patterns: direct water application of esters, direct water application of amines, terrestrial application of both esters and amines. While we expect exposure from most uses to be minimal, we are not able to predict where 2,4-D direct water applications may occur so we cannot rule out this type of exposure. Because the risk of these exposures is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of affecting populations of SONCC coho as medium. Although effects from terrestrial applications are expected to reduce primary productivity in-stream, affect fish health, and degrade riparian vegetation in some locations, our greatest concern is the potential for salmonid lethality due to direct water applications of 2,4-D BEE. In cases of aquatic weed infestation, 2,4-D can be applied directly to the water. This use can occur anywhere, including areas with low routine pesticide use. NMFS believes that multiple populations may be exposed to these direct water applications within a single year, resulting in a decrease in population numbers significant enough to affect the entire ESU. Based on risk from all use patterns, we rate the likelihood of 2,4-D affecting the ESU as medium.

Triclopyr BEE: In the preceding a.i. summary, we determined that Triclopyr BEE poses a medium risk to fish. As discussed in the a.i. summary, the medium likelihood is based on our concerns about occasional high exposures resulting in a fish mortality event. However, as stated above, we expect exposure to this a.i. to be minimal, thereby decreasing the likelihood of a negative effect. Because the risk of this exposure is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of potential adverse effects to populations of SONCC coho as low. Given the uses, fate, and toxicity of the chemical, we do not expect mortality to be a common occurrence. Existing federal limitations, described in the a.i. summary in this section, further decrease the likelihood of high exposures. Thus, we believe that this compound has a low likelihood of causing an effect to the species as a whole.

Diuron: Diuron was determined to pose a medium risk to salmon. As discussed in the a.i. summary, use in rights-of-way is our largest concern, as it has the highest application rate and runoff potential. However, as stated above, we expect exposure to this a.i. to be minimal, thereby decreasing the likelihood of a negative effect. Because the risk of this exposure is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of potential adverse effects to populations of SONCC coho as low. Given the use rates and pattern of uses for this chemical, we believe that there is a low likelihood of diuron causing a negative effect to multiple populations, or repeated negative effect to the same population. Therefore, we conclude that there is a low likelihood of an effect to the species as a whole.

Linuron: Given the limited uses of linuron, we expect exposure to be relatively infrequent. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of SONCC coho as low. As discussed in the a.i. summary, linuron exposure in agricultural areas may have some effect on the primary productivity of a system. We anticipate that the effects of linuron exposure would have a minimal effect on populations within this ESU. Thus, there would be a low likelihood of an effect to the ESU as a whole.

Captan: In the a.i. summary, we concluded that captan has a low likelihood of affecting salmon. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of SONCC coho as low. Due to the fate properties of captan, we stated that water concentrations will rarely be high enough to cause a physiological response. Therefore, there is a low likelihood of captan causing population level, or species level effects.

Chlorothalonil: In the a.i. summary, we determined that there was a medium likelihood of chlorothalonil having a negative effect on salmon. However, as stated above, we expect exposure to this a.i. to be minimal, thereby decreasing the likelihood of a negative effect. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of adverse effects to populations of SONCC coho as low. Exposure to chlorothalonil is further restricted by federal and state legislation. As such, there is a low likelihood of multiple populations experiencing adverse effects. Therefore, we believe that there is a low potential for effects at the ESU level.

Central California Coast (CCC) Coho Salmon (Endangered Species)

The CCC coho salmon ESU includes 11 independent populations. The spatial structure for CCC coho salmon has been substantially modified due to lack of viable source populations and loss of dependent populations. All populations have very low abundances making it difficult to determine long-term population trends. Returns suggest 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. The life histories of this ESU strongly influence the potential for exposure to the 6 a.i.s. Juveniles rear for 18 months, spending two winters in fresh water.

The percentage of cultivated croplands and developed lands that overlap with CCC coho salmon habitat are 2.3% and 9.4%, respectively. Much of the development is centered on San Francisco Bay, and there are also developed areas and agriculture in the Russian River watershed. 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 (Brian C. Spence, et al., 2008). 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 majority of the salmon remaining is in the northern, undeveloped watersheds around the Navarro and Big Rivers.

The populations within this ESU have very different potential for exposure to the a.i.s. We expect that the populations in the Russian River and southern areas will have a higher likelihood of exposure than the more Northern populations. There is some development in the Northern watersheds, as well as potential rights-of-way uses on electric transmission lines. Therefore we expect that all populations may have some degree of exposure. The likelihood of species-level effects is strongly tied to the Russian River population, as it is one of the more important populations. This basin is known for vineyards, so we expect that captan and chlorothalonil will be used on a large portion of agricultural land.

2,4-D: 2,4-D was determined to pose a medium risk to fish. In the a.i. summary, we discuss three separate use patterns: direct water application of esters, direct water application of amines, terrestrial application of both esters and amines. While we expect exposure from most uses to be minimal, we are not able to predict where 2,4-D direct water applications may occur so we cannot rule out this type of exposure. Because the risk of this exposure is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of affecting populations of CCC Coho as medium. Although effects from terrestrial applications are expected to reduce primary productivity in-stream, affect fish health, and degrade riparian vegetation in some locations, our greatest concern is the potential for salmonid lethality due to direct water applications of 2,4-D BEE. In cases of aquatic weed infestation, 2,4-D can be applied directly to the water. This use can occur anywhere, including areas with low routine pesticide use. Historically, the Russian River population was a source population that helped to sustain other populations. This consideration weighs heavily on the ESU analysis, as negative effects to this population can result in lower numbers for multiple populations. Based on risk from all use patterns, we rate the likelihood of 2,4-D affecting the ESU as medium.

Triclopyr BEE: In the preceding a.i. summary, we determined that Triclopyr BEE poses a medium risk to fish. As discussed in the a.i. summary, the medium likelihood is based on our concerns about occasional high exposures resulting in a fish mortality event. The Russian River system is the most developed, but NMFS does not expect it will receive significantly more inputs from rights-of-way uses as it is not a major transportation or utility route. Because the risk of this exposure is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of affecting populations of CCC coho as low. In order to decrease lambda of the species, multiple mortality events would have to occur, either within the same population or across several populations within the ESU. Given the uses, fate, and toxicity of the chemical, we do not expect mortality to be a common occurrence. Thus, we believe that this compound has a low likelihood of causing an effect to the species as a whole.

Diuron: Diuron was determined to pose a medium risk to salmon. As discussed in the a.i. summary, use in rights-of-way is our largest concern, as it has the highest application rate and runoff potential. While the Russian River is not a major transportation or utility route, we do expect some exposure to result from a combination of rights-of way and agricultural uses. Therefore, we rate the likelihood for the Russian River and other southern populations as medium and the likelihood for the more Northern populations as low. Therefore we rate the likelihood to populations of CCC coho as low to medium. We expect exposure to lead to declines in the primary productivity of the system, but not to wide-scale fish-health issues. Given the relative importance of the Russian River population, pattern of uses for this chemical, and land use patterns within the range of the ESU, we believe that there is a medium likelihood of diuron causing a negative effect to the species as a whole.

Linuron: Given the limited uses of linuron, we expect exposure to be relatively infrequent, even in the Russian River watershed. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of CCC coho as low. As discussed in the a.i. summary, linuron exposure in agricultural areas may have some effect on the primary productivity of a system. We anticipate that the effects of linuron exposure would have a minimal effect on populations within this ESU. Thus, there would be a low likelihood of an effect to the ESU as a whole.

Captan: In the a.i. summary, we concluded that captan has a low likelihood of affecting salmon. Due to cropping patterns, we expect that captan will be used more frequently in Russian River watershed. However, NMFS believes that risk to this population is still low based on captan's fate properties. Therefore, we do not find a reason to differentiate risk among populations, and so we rate the likelihood to populations of CCC coho as low. As discussed earlier, NMFS believes that water concentrations will rarely be high enough to cause a physiological response. Therefore, there is a low likelihood of captan causing population level, or species level effects.

Chlorothalonil: In the a.i. summary, we determined that there was a medium likelihood of chlorothalonil having a negative effect on salmon. Higher or repeated exposures may cause fish mortality. We expect that these uses will occur more frequently in the Russian River watershed, due to a higher concentration of turf uses and the prevalence of higher-use agricultural crops. Historically, the Russian River population was a potential source population that helped to sustain other populations. As such, it weighs heavily in our population level analysis, so we rate the likelihood of adverse effects to populations of CC Chinook as medium. This consideration also weighs heavily on the ESU analysis, as negative effects to this population can result in lower numbers for multiple populations. Therefore, we believe that there is a medium potential for effects at the ESU level.

Ozette Lake Sockeye Salmon (Threatened Species)

The Ozette Lake sockeye salmon ESU consists of a single population made up of five spawning aggregations. Uncertainty remains on the growth rate and productivity of the natural component of the ESU. While genetic differences occur between age cohorts and different age groups do not spawn with each other, genetic diversity within the ESU is low. Spatial structure of the population has been altered, as only two beaches are known to be used for spawning (Haggerty, Ritchie, Shellberg, Crewson, & Jolonen, 2007). Overall abundance is also significantly depressed.

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. Roughly 77% of the land in Ozette Basin is managed for timber production (Jacobs, Larson, Meyer, Currence, & Hinton, 1996). Land use of this ESU is primarily forest with private, state, and federal ownership (86% forested, 13% open water, 1% developed land, 0% agriculture). The entire circumference of the lake is within Olympic National Park.

The life histories of this ESU strongly influence the potential for exposure to the 6 a.i.s. Adult spawners enter Ozette River from April to early August and may remain in Ozette Lake for extended periods before spawning (October- February). Spawning occurs along the lakeshore and historically in some of the lakes' tributaries. Fry migrate immediately to the lake where they rear for a year or so before entering the ocean. The predominant pesticide use sites (*i.e.*, urban/residential and forestry uses) overlap with the Lake's freshwater tributaries. As such, the greatest risk of exposure is to those sockeye that utilize freshwater tributary habitats. Direct effects to fish remain a concern within tributaries.

2,4-D: 2,4-D was determined to pose a medium risk to fish. In the a.i. summary, we discuss three separate use patterns: direct water application of esters, direct water application of amines, terrestrial application of both esters and amines. While we expect exposure from most uses to be minimal, we are not able to predict where 2,4-D direct water applications may occur so we cannot rule out this type of exposure. Although effects from terrestrial applications are expected to reduce primary productivity in-stream, affect fish health, and degrade riparian vegetation in some locations, our greatest concern is the potential for salmonid lethality due to direct water applications of 2,4-D BEE. In cases of aquatic weed infestation, 2,4-D can be applied directly to the water. This use can occur anywhere, including areas with low routine pesticide use. Based on risk from all use patterns, we rate the likelihood of 2,4-D affecting the single population that makes up this ESU as medium.

Triclopyr BEE: In the a.i. summary, we determined that Triclopyr BEE poses a medium risk to fish. As discussed in the a.i. summary, the medium likelihood is based on our concerns about occasional high exposures resulting in a fish mortality event. However, as stated above, we expect exposure to this a.i. to be minimal, thereby decreasing the likelihood of a negative effect. Given the uses, fate, and toxicity of the chemical, we do not expect mortality to be a common occurrence. Existing federal and state limitations, described in the a.i. summary in this section, further decrease the likelihood of high exposures. Thus, we believe that this compound has a low likelihood of causing an effect to the species as a whole.

Diuron: Diuron was determined to pose a medium risk to salmon. As discussed in the a.i. summary, use in rights-of-way is our largest concern, as it has the highest application rate and runoff potential. However, given the land use within the ESU, we expect exposure to this a.i. to be minimal, thereby decreasing the likelihood of a negative effect. Given the use rates and pattern of uses for this chemical, we believe that there is a low likelihood of diuron causing a negative effect to the Ozette Lake sockeye.

Linuron: Given the limited uses of linuron, we do not expect that this ESU will be exposed to linuron. Thus, there would be a low likelihood of an effect to the Ozette Lake sockeye.

Captan: In the a.i. summary, we concluded that captan has a low likelihood of affecting salmon. Due to the fate properties of captan, we stated that water concentrations will rarely be high enough to cause a physiological response. Therefore, there is a low likelihood of captan causing negative effects to the ESU.

Chlorothalonil: In the a.i. summary, we determined that there was a medium likelihood of chlorothalonil having a negative effect on salmon. However, given landuse within this ESU, we expect exposure to this a.i. to be minimal, thereby decreasing the likelihood of a negative effect. Exposure to chlorothalonil is further restricted by state legislation. Therefore, we believe that there is a low potential for effects at the ESU level.

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 (NMFS, 2008d). However, this figure has increased to adults numbering in the hundreds over the last two years. The ESU's genetic diversity has been reduced based on low population abundance and a high proportion of hatchery-origin fish.

About 1% of the land surrounding Redfish Lake has been developed, and another 1% is used for agriculture, primarily hay and pasture. More than 50% of the ESUs is composed of evergreen forests. Consequently, forestry uses are the major source of pesticide exposure during spawning and rearing activities. However, Redfish Lake is located in a watershed that is 92% federal land. Therefore, any forestry uses of the chemicals would fall under a separate section 7 consultation. We expect that exposure to the a.i.s will occur during migration to and from Redfish Lake.

Juvenile sockeye remain in the lake for one to three years before migrating through the Snake and Columbia Rivers for several hundred miles to the ocean.

2,4-D: 2,4-D was determined to pose a medium risk to fish. In the a.i. summary, we discuss three separate use patterns: direct water application of esters, direct water application of amines, terrestrial application of both esters and amines. While we expect exposure from most uses to be minimal, we are not able to predict where 2,4-D direct water applications may occur so we cannot rule out this type of exposure. Although effects from terrestrial applications are expected to reduce primary productivity in-stream, affect fish health, and degrade riparian vegetation in some locations, our greatest concern is the potential for salmonid lethality due to direct water applications of 2,4-D BEE. In cases of aquatic weed infestation, 2,4-D can be applied directly to the water. This use can occur anywhere, including areas with low routine pesticide use. Based on risk from all use patterns, we rate the likelihood of 2,4-D affecting the single population that makes up this ESU as medium.

Triclopyr BEE: In the a.i. summary, we determined that Triclopyr BEE poses a medium risk to fish. The medium likelihood is based on our concerns about occasional high exposures resulting in a fish mortality event. However, given landuse within the ESU, we expect exposure to this a.i. to be minimal, thereby decreasing the likelihood of a negative effect. Given the uses, fate, and toxicity of the chemical, we do not expect mortality to be a common occurrence. Existing federal and state limitations, described in the a.i. summary in this section, further decrease the likelihood of high exposures. Thus, we believe that this compound has a low likelihood of causing an effect to the species as a whole.

Diuron: Diuron was determined to pose a medium risk to salmon. As discussed in the a.i. summary, use in rights-of-way is our largest concern, as it has the highest application rate and runoff potential. However, given landuse within the ESU, we expect exposure to this a.i. to be minimal, thereby decreasing the likelihood of a negative effect. Given the use rates and pattern of uses for this chemical, we believe that there is a low likelihood of diuron causing a negative effect to the Snake River sockeye.

Linuron: Given the limited uses of linuron, we do not expect that this ESU will be exposed to linuron. Thus, there would be a low likelihood of an effect to the Snake River sockeye.

Captan: In the a.i. summary, we concluded that captan has a low likelihood of affecting salmon. Due to the fate properties of captan, we stated that water concentrations will rarely be high enough to cause a physiological response. Therefore, there is a low likelihood of captan causing negative effects to the ESU.

Chlorothalonil: In the a.i. summary, we determined that there was a medium likelihood of chlorothalonil having a negative effect on salmon. However, given landuse within the ESU, we expect exposure to this a.i. to be minimal, thereby decreasing the likelihood of a negative effect. Exposure to chlorothalonil is further restricted by federal legislation. Therefore, we believe that there is a low potential for effects at the ESU level.

Puget Sound Steelhead (Threatened Species)

The Puget Sound steelhead is comprised of 53 populations (37 winter-run and 16 summer-run). Summer-run populations are concentrated in northern Puget Sound and Hood Canal. The WDFW 2002 stock assessment categorized 5 populations as healthy, 19 as depressed, 1 as critical, and 27 of unknown status (Washington Department of Fish and Wildlife (WDFW), 2002). 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.

More than 50 percent of the DPS is composed of evergreen, deciduous, or mixed forests. Other pesticide use areas include urban/residential development (15%) and agricultural uses (4%). Cultivated crops (1%) and hay crops and pastures (3%) are primarily distributed on the floodplain and other lowland habitats. The majority of urban/residential also occurs within river and stream valleys in lowland areas, and much of the nearshore marine area also consists of urban/residential development. These areas serve as rearing and migration areas for juveniles. Spawning generally occurs in the forested upper portions of the watersheds. 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 marine waters as one or three-year olds.

2,4-D: 2,4-D was determined to pose a medium risk to fish. In the a.i. summary, we discuss three separate use patterns: direct water application of esters, direct water application of amines, terrestrial application of both esters and amines. Because the risk of these exposures is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of affecting populations of Puget Sound Steelhead as medium. Although effects from terrestrial applications are expected to reduce primary productivity in-stream, affect fish health, and degrade riparian vegetation in some locations, our greatest concern is the potential for salmonid lethality due to direct water applications of 2,4-D BEE. NMFS believes that multiple populations may be exposed to these direct water applications within a single year, resulting in a decrease in population numbers significant enough to affect the entire ESU. Based on risk from all use patterns, we rate the likelihood of 2,4-D affecting the ESU as medium.

Triclopyr BEE: In the preceding a.i. summary, we determined that Triclopyr BEE poses a medium risk to fish. As discussed in the a.i. summary, the medium likelihood is based on our concerns about occasional high exposures resulting in a fish mortality event. If a high concentration of the a.i. was to co-occur with peak fish presence, this could cause a significant drop in the size of population. Because the risk of this exposure is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Puget Sound Steelhead as medium. However, in order to decrease lambda of the species, multiple mortality events would have to occur, either within the same population or across several populations within the DPS. Given the uses, fate, and toxicity of the chemical, we do not expect mortality to be a common occurrence. Existing federal and state limitations, described in the a.i. summary in this section, further decrease the likelihood of high exposures. Thus, we believe that this compound has a low likelihood of causing an effect to the species as a whole.

Diuron: Diuron was determined to pose a medium risk to salmon. As discussed in the a.i. summary, use in rights-of-way is our largest concern, as it has the highest application rate and runoff potential. Because the risk of this exposure is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Puget Sound Steelhead as medium. In this DPS, diuron use in rights-of-way is strictly limited by state legislation. We expect exposure from other uses to lead to declines in the primary productivity of the system, but not to wide-scale fish-health issues. However, given the use rates, pattern of uses for this chemical, and restrictive state regulations, we believe that there is a low likelihood of diuron causing a negative effect to multiple populations, or repeated negative effect to the same population. Therefore, we conclude that there is a low likelihood of an effect to the species as a whole.

Linuron: Given the limited uses of linuron, we expect exposure to be relatively infrequent. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Puget Sound Steelhead as low. As discussed in the a.i. summary, linuron exposure in agricultural areas may have some effect on the primary productivity of a system. We anticipate that the effects of linuron exposure would have a minimal effect on populations within this DPS. Thus, there would be a low likelihood of an effect to the DPS as a whole.

Captan: In the a.i. summary, we concluded that captan has a low likelihood of affecting salmon. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Puget Sound Steelhead as low. Due to the fate properties of captan, we stated that water concentrations will rarely be high enough to cause a physiological response. Therefore, there is a low likelihood of captan causing species level effects.

Chlorothalonil: In the a.i. summary, we determined that there was a medium likelihood of chlorothalonil having a negative effect on salmon. Higher or repeated exposures, particularly those associated with turf uses, may cause fish mortality. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and

therefore we rate the likelihood to populations of Puget Sound Steelhead as medium. The distribution of populations and use sites across the range of the species indicate that it is possible that multiple populations could experience substantial negative effects. Therefore, we believe that there is a medium potential for effects at the DPS level.

Lower Columbia River Steelhead (Threatened Species)

The LCR steelhead DPS includes 23 extant populations. Spatial structure within the DPS, especially in Washington, has been substantially reduced by the loss of access to the upper portions of some basins from tributary hydropower development. Many of the populations in this DPS are small, and the long- and short-term trends in abundance of all individual populations are negative. The genetic diversity of this DPS has also been substantially reduced.

The percentage of cultivated crop lands overlap with LCR Steelhead DPS is about 7%, 4.5 % as hay/pasture land and 2.5% as cultivated crop land. More than 61% of the DPS is composed of evergreen, deciduous forest, and mixed forests. Urban/residential development lands (12%) were a fairly substantial portion of this DPS. 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.

Populations located near the Portland area are expected to have increased exposure to urban uses, while the more Northern populations experience inputs from agricultural and forestry uses. Turf uses, including use on golf courses, are spread throughout the ESU with a higher concentration near Portland and along the mainstem Columbia. We expect that salmonids near Portland will have significant exposure from rights-of-way uses. This area has a high concentration of rights-of-way from rail, road, and utilities. These uses are of greater concern as they tend to be higher use rates with greater probability of runoff. This concern is mediated somewhat by the wide distribution of populations throughout the basin, and the fact that exposure will likely occur in higher volume, higher flow habitats, such as the Columbia River. While agricultural uses should not be discounted, urban uses are of greater concern – particularly for professional lawn applications. 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.

2,4-D: 2,4-D was determined to pose a medium risk to fish. In the a.i. summary, we discuss three separate use patterns: direct water application of esters, direct water application of amines, terrestrial application of both esters and amines. Because the risk of these exposures is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of affecting populations of Lower Columbia River Steelhead as medium. Although effects from terrestrial applications are expected to reduce primary productivity in-stream, affect fish health, and degrade riparian vegetation in some locations, our greatest concern is the potential for salmonid lethality due to direct water applications of 2,4-D BEE. NMFS believes that multiple populations may be exposed to these direct water applications within a single year, resulting in a decrease in population numbers significant enough to affect the entire DPS. Based on risk from all use patterns, we rate the likelihood of 2,4-D affecting the DPS as medium.

Triclopyr BEE: In the preceding a.i. summary, we determined that Triclopyr BEE poses a medium risk to fish. As discussed in the a.i. summary, the medium likelihood is based on our concerns about occasional high exposures resulting in a fish mortality event. If a high concentration of the a.i. was to co-occur with peak fish presence, this could cause a significant drop in the size of population. Because the risk of this type of exposure is greater for populations near the Portland area due to development, we rate the likelihood for these populations as medium and the likelihood for the remaining populations as low. Therefore we rate the likelihood to populations of Lower Columbia River Steelhead as low to medium. However, in order to decrease lambda of the species, multiple mortality events would have to occur, either within the same population or across several populations within the DPS. Given the uses, fate, and toxicity of the chemical, we do not expect mortality to be a common occurrence. Existing federal and state limitations, described in the a.i. summary in this section, further decrease the likelihood of high exposures. Thus, we believe that this compound has a low likelihood of causing an effect to the species as a whole.

Diuron: Diuron was determined to pose a medium risk to salmon. As discussed in the a.i. summary, use in rights-of-way is our largest concern, as it has the highest application rate and runoff potential. The risk of this type of exposure is greater for these populations, as they are located near the Portland area. Therefore we rate the likelihood to populations of Lower Columbia River Steelhead as low to medium. In this DPS, diuron use in rights-of-way is strictly limited by state legislation. We expect exposure from other uses to lead to declines in the primary productivity of the system, but not to wide-scale fish-health issues. Given the use rates, pattern of uses for this chemical, and restrictive state regulations, we believe that there is a low likelihood of diuron causing a negative effect to multiple populations, or repeated negative effect to the same population. Therefore, we conclude that there is a low likelihood of an effect to the species as a whole.

Linuron: Given the limited uses of linuron, we expect exposure to be relatively infrequent. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Lower Columbia Steelhead as low. As discussed in the a.i. summary, linuron exposure in agricultural areas may have some effect on the primary productivity of a system. We anticipate that the effects of linuron exposure would have a minimal effect on populations within this DPS. Thus, there would be a low likelihood of an effect to the DPS as a whole.

Captan: In the a.i. summary, we concluded that captan has a low likelihood of affecting salmon. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Lower Columbia Steelhead as low. Due to the fate properties of captan, we stated that water concentrations will rarely be high enough to cause a physiological response. Therefore, there is a low likelihood of captan causing population level, or species level effects.

Chlorothalonil: In the a.i. summary, we determined that there was a medium likelihood of chlorothalonil having a negative effect on salmon. Higher or repeated exposures, particularly those associated with turf uses, may cause fish mortality. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and

therefore we rate the likelihood to populations of Lower Columbia River Steelhead as medium. The distribution of populations and use sites across the range of the species indicate that it is possible that multiple populations could experience substantial negative effects. Therefore, we believe that there is a medium potential for effects at the DPS level.

Upper Willamette River Steelhead (Threatened Species)

The UWR steelhead DPS is comprised of four extant populations that occupy tributaries draining the east side of the UWR basin. Populations within this DPS have been declining and have exhibited large fluctuations in abundance. Abundance is moderately depressed for the entire DPS. The DPS's spatial distribution and genetic diversity are moderately intact.

The major threats to the survival and recovery of this DPS include habitat loss due to blockages, lost or degraded floodplain connectivity, and degraded water quality within the Willamette mainstem and the lower reaches of its tributaries. Fifty pesticides were detected in streams that drain both agricultural and urban areas. Forty-nine pesticides were detected in streams draining agricultural land, while 25 pesticides were detected in streams draining urban areas. Ten of these pesticides, including azinphos methyl, exceeded EPA criteria for the protection of freshwater aquatic life.

The percentage of cultivated crop lands and developed lands overlapping with this DPS are 14.5% and 10%, respectively. After emergence, steelhead fry typically rear in floodplain habitats associated with their natal rivers and streams for two years.

We expect that populations within this ESU will be exposed to the six a.i.s due to the high amount of agricultural and developed land. Specialty crops are likely to be grown in this area, increasing the likelihood of linuron exposure. Further, while some of the spawning and rearing streams are in forested areas, they are not necessarily in Federal lands. As such, we cannot assume any additional protections from other ESA consultations. The valley is also heavily used by railroads, roads, and electrical transmission lines, increasing the likelihood of rights-of-way applications. We also expect that environmental mixtures will compound the effects of these chemicals. Use of the a.i.s is likely to cause a decrease in primary production, chronic effects, and may result in fish mortality. 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.

2,4-D: 2,4-D was determined to pose a medium risk to fish. In the a.i. summary, we discuss three separate use patterns: direct water application of esters, direct water application of amines, terrestrial application of both esters and amines. Because the risk of these exposures is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of affecting populations of Upper Willamette River Steelhead as medium. Although effects from terrestrial applications are expected to reduce primary productivity in-stream, affect fish health, and degrade riparian vegetation in some locations, our greatest concern is the potential for salmonid lethality due to direct water applications of 2,4-D BEE. NMFS believes that multiple populations may be exposed to these direct water applications within a single year, resulting in a decrease in population numbers significant enough to affect the entire DPS. Based on risk from all use patterns, we rate the likelihood of 2,4-D affecting the DPS as medium.

Triclopyr BEE: In the preceding a.i. summary, we determined that Triclopyr BEE poses a medium risk to fish. As discussed in the a.i. summary, the medium likelihood is based on our concerns about occasional high exposures resulting in a fish mortality event. If a high concentration of the a.i. was to co-occur with peak fish presence, this could cause a significant drop in the size of population. Because the risk of this exposure is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Upper Willamette River Steelhead as medium. In order to decrease lambda of the species, multiple mortality events would have to occur, either within the same population or across several populations within the DPS. Given the uses, fate, and toxicity of the chemical, we do not expect mortality to be a common occurrence. However, because of the level of development, we expect use to be much more frequent, thereby increasing the likelihood of mortality. Thus, we believe that this compound has a medium likelihood of causing an effect to the species as a whole.

Diuron: Diuron was determined to pose a medium risk to salmon. As discussed in the a.i. summary, use in rights-of-way is our largest concern, as it has the highest application rate and runoff potential. Because the risk of this exposure is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Upper Willamette River Steelhead as medium. We expect exposure from other uses to lead to declines in the primary productivity of the system, but not to wide-scale fish-health issues. Given the use rates and pattern of uses for this chemical, we believe that there is a medium likelihood of diuron causing a negative effect to multiple populations, or repeated negative effect to the same population. Therefore, we conclude that there is a medium likelihood of an effect to the species as a whole.

Linuron: Given the limited uses of linuron, we expect exposure to be relatively infrequent. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Upper Willamette River Steelhead as low. As discussed in the a.i. summary, linuron exposure in agricultural areas may have some effect on the primary productivity of a system. We anticipate that the effects of linuron exposure would have a minimal effect on populations within this DPS. Thus, there would be a low likelihood of an effect to the DPS as a whole.

Captan: In the a.i. summary, we concluded that captan has a low likelihood of affecting salmon. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Upper Willamette River Steelhead as low. Due to the fate properties of captan, we stated that water concentrations will rarely be high enough to cause a physiological response. Therefore, there is a low likelihood of captan causing population level, or species level effects.

Chlorothalonil: In the a.i. summary, we determined that there was a medium likelihood of chlorothalonil having a negative effect on salmon. Higher or repeated exposures, particularly those associated with turf uses, may cause fish mortality. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and

therefore we rate the likelihood to populations of Upper Willamette River Chinook as Steelhead. The distribution of populations and use sites across the range of the species indicate that it is possible that multiple populations could experience substantial negative effects. Therefore, we believe that there is a medium potential for effects at the ESU level.

Middle Columbia River Steelhead (Threatened Status)

The MCR steelhead DPS includes 16 extant populations in Oregon and Washington. The spatial structure of this population is relatively intact. The genetic diversity has been compromised by interbreeding with resident and hatchery fish. Population growth rates are near replacement, though abundances are depressed in relation to historic levels.

The percentage of cultivated crop lands and developed lands within the range of this DPS are 17% and 3%, respectively. Orchards are common in this area, and often located in close proximity to rivers. There are few urban centers, but low levels of development are distributed throughout the range. Due to the relatively low levels of development, we do not expect that rights-of-way uses will be a major exposure route, aside from areas directly along the Columbia River. Swim–up fry usually inhabit shallow water along banks of streams or aquatic habitats on stream margins. Juveniles rear in a variety of freshwater habitat for two years.

2,4-D: 2,4-D was determined to pose a medium risk to fish. In the a.i. summary, we discuss three separate use patterns: direct water application of esters, direct water application of amines, terrestrial application of both esters and amines. Because the risk of these exposures is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of affecting populations of Middle Columbia River Steelhead as medium. Although effects from terrestrial applications are expected to reduce primary productivity in-stream, affect fish health, and degrade riparian vegetation in some locations, our greatest concern is the potential for salmonid lethality due to direct water applications of 2,4-D BEE. NMFS believes that multiple populations may be exposed to these direct water applications within a single year, resulting in a decrease in population numbers significant enough to affect the entire DPS. Based on risk from all use patterns, we rate the likelihood of 2,4-D affecting the DPS as medium.

Triclopyr BEE: In the preceding a.i. summary, we determined that Triclopyr BEE poses a medium risk to fish. As discussed in the a.i. summary, the medium likelihood is based on our concerns about occasional high exposures resulting in a fish mortality event. Due to the low concentrations of developed areas and rights-of-way use sites, we think that exposure will be infrequent. Because the risk of this exposure is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Middle Columbia River Steelhead as low. In order to decrease lambda of the species, multiple mortality events would have to occur, either within the same population or across several populations within the DPS. Given the uses, fate, and toxicity of the chemical, we do not expect mortality to be a common occurrence. Thus, we believe that this compound has a low likelihood of causing an effect to the species as a whole.

Diuron: Diuron was determined to pose a medium risk to salmon. As discussed in the a.i. summary, use in rights-of-way is our largest concern, as it has the highest application rate and runoff potential. Due to the low concentration of rights-of-way use sites, we think that exposure will be infrequent. Because the risk of this exposure is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Middle Columbia River Steelhead as low. We expect exposure from other uses to lead to declines in the primary productivity of the system, but not to wide-scale fish-health issues. Given the potential use sites, pattern of uses, and additional state limitations we believe that there is a low likelihood of diuron causing a negative effect to multiple populations, or repeated negative effect to the same population. Therefore, we conclude that there is a low likelihood of an effect to the species as a whole.

Linuron: Given the limited uses of linuron, we expect exposure to be relatively infrequent. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Middle Columbia River Steelhead as low. As discussed in the a.i. summary, linuron exposure in agricultural areas may have some effect on the primary productivity of a system. We anticipate that the effects of linuron exposure would have a minimal effect on populations within this DPS. Thus, there would be a low likelihood of an effect to the DPS as a whole.

Captan: In the a.i. summary, we concluded that captan has a low likelihood of affecting salmon. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Middle Columbia River Steelhead as low. As orchards are common and fruit trees have some of the highest use-rates and permitted number of application per year, this ESU is more likely to be exposed to captan. However, due to the fate properties of captan, we stated that water concentrations will rarely be high enough to cause a physiological response. Therefore, there is a low likelihood of captan causing population level, or species level effects.

Chlorothalonil: In the a.i. summary, we determined that there was a medium likelihood of chlorothalonil having a negative effect on salmon. Higher or repeated exposures, particularly those associated with turf uses, may cause fish mortality. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Middle Columbia River Steelhead as medium. The distribution of populations and use sites across the range of the species indicate that it is possible that multiple populations could experience substantial negative effects. Therefore, we believe that there is a medium potential for effects at the ESU level.

Upper Columbia River Steelhead (Threatened Species)

The UCR steelhead DPS consists of four extant populations in Washington State. Abundance data indicate that these populations are below the minimum threshold for recovery and have negative growth rates. Adult returns are dominated by hatchery fish and experience reduced genetic diversity from homogenization of populations. The spatial structure of this DPS has been severely altered, with 50% of its habitat cutoff by the Grand Coulee Dam.

Newly emerged fry move about considerably and seek suitable rearing habitat, such as stream margins or cascades. The majority of juveniles smolt as two-year olds, though some individuals may rear for as long as seven years in these fresh water systems.

While this ESU has very few populations left, we do not expect that there will be much exposure to any of the a.i.s with the exception of direct water applications of 2, 4-D BEE. There is very little agriculture and urban development within the ESU, and correspondingly less right-of-way. The percentage of cultivated crop lands and developed lands within the range of the ESU are 13% and 4%, respectively. There is some agriculture in the spawning and rearing areas in the Wenatchee, Methow, and Okenogan watersheds. In the Entiat, there is intense agriculture the Upper Columbia Irrigation District. However, the water is heavily used and re-used in irrigation. Forested lands make up about 45% of the ESU. Much of the forested land is federally owned; any program involving the use of pesticides would be covered under its own ESA consultation. Therefore, we are considering exposure to be minimal in these areas as well. Most exposure will occur during migration along the Columbia River. This exposure is of less concern as it is a high volume, high flow system.

2,4-D: 2,4-D was determined to pose a medium risk to fish. In the a.i. summary, we discuss three separate use patterns: direct water application of esters, direct water application of amines, terrestrial application of both esters and amines. While we expect exposure from most uses to be minimal, we are not able to predict where 2,4-D direct water applications may occur so we cannot rule out this type of exposure. Because the risk of these exposures is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of affecting populations of Upper Columbia Steelhead as medium. Although effects from terrestrial applications are expected to reduce primary productivity in-stream, affect fish health, and degrade riparian vegetation in some locations, our greatest concern is the potential for salmonid lethality due to direct water applications of 2,4-D BEE. In cases of aquatic weed infestation, 2,4-D can be applied directly to the water. This use can occur anywhere, including areas with low routine pesticide use. NMFS believes that if any one of the populations was exposed to these direct water applications, the resulting decrease in population numbers could be significant enough to affect the entire DPS. Based on risk from all use patterns, we rate the likelihood of 2,4-D affecting the DPS as medium.

Triclopyr BEE: In the preceding a.i. summary, we determined that Triclopyr BEE poses a medium risk to fish. As discussed in the a.i. summary, the medium likelihood is based on our concerns about occasional high exposures resulting in a fish mortality event. However, as stated above, we expect exposure to this a.i. to be minimal, thereby decreasing the likelihood of a negative effect. Because the risk of this exposure is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of potential adverse effects to populations of Upper Columbia Steelhead as low. Given the uses, fate, and toxicity of the chemical, we do not expect mortality to be a common occurrence. Existing federal and state limitations, described in the a.i. summary in this section, further decrease the likelihood of high exposures. Thus, we believe that this compound has a low likelihood of causing an effect to the species as a whole.

Diuron: Diuron was determined to pose a medium risk to salmon. As discussed in the a.i. summary, use in rights-of-way is our largest concern, as it has the highest application rate and runoff potential. However, as stated above, we expect exposure to this a.i. to be minimal, thereby decreasing the likelihood of a negative effect. Because the risk of this exposure is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of potential adverse effects to populations of Upper Columbia River Steelhead as low. In this DPS, diuron use in rights-of-way is strictly limited by state legislation. Given the use rates, pattern of uses for this chemical, and restrictive state regulations, we believe that there is a low likelihood of diuron causing a negative effect to multiple populations, or repeated negative effect to the same population. Therefore, we conclude that there is a low likelihood of an effect to the species as a whole.

Linuron: Given the limited uses of linuron, we expect exposure to be relatively infrequent. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Upper Columbia Steelhead as low. As discussed in the a.i. summary, linuron exposure in agricultural areas may have some effect on the primary productivity of a system. We anticipate that the effects of linuron exposure would have a minimal effect on populations within this DPS. Thus, there would be a low likelihood of an effect to the DPS as a whole.

Captan: In the a.i. summary, we concluded that captan has a low likelihood of affecting salmon. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Upper Columbia Steelhead as low. Due to the fate properties of captan, we stated that water concentrations will rarely be high enough to cause a physiological response. Therefore, there is a low likelihood of captan causing population level, or species level effects.

Chlorothalonil: In the a.i. summary, we determined that there was a medium likelihood of chlorothalonil having a negative effect on salmon. Higher or repeated exposures, particularly those associated with turf uses, may cause fish mortality. While we expect exposure from most uses to be minimal, turf uses may occur throughout the DPS so we cannot rule out this type of exposure. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Upper Columbia Steelhead as medium. NMFS believes that if any of the four populations was exposed to chlorothalonil as a result of turf uses, the resulting decrease in population numbers could be significant enough to affect the entire DPS. The distribution of populations and use sites across the range of the species indicate that it is possible that populations could experience substantial negative effects. Therefore, we believe that there is a medium potential for effects at the DPS level.

Snake River Basin Steelhead (Threatened Species)

The SR basin steelhead DPS includes 23 populations that are spatially distributed in each of the six major geographic areas in the Snake River basin (T. P. Good, et al., 2005). The historic spatial structure is relatively unaltered. While population growth rates show mixed long- and short-term trends in productivity, overall abundances remain well below their interim recovery criteria. Genetic diversity has been reduced, particularly for the B-run steelhead, those whose life history pattern includes spending two or more years in freshwater, and two or more years in the ocean before their upriver migration. A-run steelhead are smaller, have a shorter freshwater and ocean residence. Juveniles typically rear in floodplain habitats associated with their natal rivers and streams for more than a year. SR basin steelhead typically smolt after two or three years.

Exposure of the a.i.s to the Snake River Steelhead populations is likely to be fairly low. Potential exposure from use within the DPS includes use on evergreen forests (52%), agricultural lands including use on cultivated crops (8%) and hay/pasture (1%), and use in urban/residential or other developed areas (2%). As many spawn and rear in U.S. Forest Service lands, any pesticide use would be authorized under additional ESA consultations. Existing Opinions on weed control programs in these areas aim to minimize pesticide runoff and strictly limit the amount of land that can be treated. Given these conditions, we do not believe that populations in these areas will experience adverse effects from any of the a.i.s with the exception of adverse effects resulting from direct water applications of 2, 4-D BEE. Some populations may experience exposure from agricultural or urban uses, particularly during migration. Since these exposures will occur in a high volume high flow system, we expect population effects to be minimal, with the exception noted above.

2,4-D: 2,4-D was determined to pose a medium risk to fish. In the a.i. summary, we discuss three separate use patterns: direct water application of esters, direct water application of amines, terrestrial application of both esters and amines. While we expect exposure from most uses to be minimal, we are not able to predict where 2,4-D direct water applications may occur so we cannot rule out this type of exposure. Because the risk of these exposures is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of affecting populations of Snake River Steelhead as medium. Although effects from terrestrial applications are expected to reduce primary productivity in-stream, affect fish health, and degrade riparian vegetation in some locations, our greatest concern is the potential for salmonid lethality due to direct water applications of 2,4-D BEE. In cases of aquatic weed infestation, 2,4-D can be applied directly to the water. This use can occur anywhere, including areas with low routine pesticide use. NMFS believes that multiple populations may be exposed to these direct water applications within a single year, resulting in a decrease in population numbers significant enough to affect the entire DPS. Based on risk from all use patterns, we rate the likelihood of 2,4-D affecting the DPS as medium.

Triclopyr BEE: In the preceding a.i. summary, we determined that Triclopyr BEE poses a medium risk to fish. As discussed in the a.i. summary, the medium likelihood is based on our concerns about occasional high exposures resulting in a fish mortality event. However, as stated above, we expect exposure to this a.i. to be minimal, thereby decreasing the likelihood of a negative effect. Because the risk of this exposure is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of potential adverse effects to populations of Snake River Steelhead as low. Given the uses, fate, and toxicity of the chemical, we do not expect mortality to be a common occurrence. Existing federal limitations, described in the a.i. summary in this section, further decrease the likelihood of high exposures. Thus, we believe that this compound has a low likelihood of causing an effect to the species as a whole.

Diuron: Diuron was determined to pose a medium risk to salmon. As discussed in the a.i. summary, use in rights-of-way is our largest concern, as it has the highest application rate and runoff potential. However, as stated above, we expect exposure to this a.i. to be minimal, thereby decreasing the likelihood of a negative effect. Because the risk of this exposure is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of potential adverse effects to populations of Snake River Steelhead as low. Given the use rates and pattern of uses for this chemical, we believe that there is a low likelihood of diuron causing a negative effect to multiple populations, or repeated negative effect to the same population. Therefore, we conclude that there is a low likelihood of an effect to the species as a whole.

Linuron: Given the limited uses of linuron, we expect exposure to be relatively infrequent. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Snake River Steelhead as low. As discussed in the a.i. summary, linuron exposure in agricultural areas may have some effect on the primary productivity of a system. We anticipate that the effects of linuron exposure would have a minimal effect on populations within this DPS. Thus, there would be a low likelihood of an effect to the DPS as a whole.

Captan: In the a.i. summary, we concluded that captan has a low likelihood of affecting salmon. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Snake River Steelhead as low. Due to the fate properties of captan, we stated that water concentrations will rarely be high enough to cause a physiological response. Therefore, there is a low likelihood of captan causing population level, or species level effects.

Chlorothalonil: In the a.i. summary, we determined that there was a medium likelihood of chlorothalonil having a negative effect on salmon. However, as stated above, we expect exposure to this a.i. to be minimal, thereby decreasing the likelihood of a negative effect. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of adverse effects to populations of Snake River Steelhead as low. Exposure to chlorothalonil is further restricted by state legislation. As such, there is a low likelihood of multiple populations experiencing adverse effects. Therefore, we believe that there is a low potential for effects at the DPS level.

Northern California Steelhead (Threatened Species)

The NC steelhead DPS includes 15 historically independent populations of winter steelhead and 4 extant populations of summer steelhead. The loss of summer-run steelhead populations has significantly reduced the genetic diversity. Most populations are in decline and have low abundances and production. Although the DPS spatial structure is relatively intact, the distribution within most watersheds has been restricted by physical and temperature barriers. Juvenile steelhead remain in fresh water for two or more years, rearing in streams and lagoons.

In general, we expect the populations to have limited exposure to the a.i.s, with the exception of exposure resulting from 2, 4-D BEE direct water applications. There is a low amount of development, agriculture, and rights-of-way uses within the range of the ESU. The percentage of cultivated crop lands and developed lands overlapping with NC steelhead habitat are less than 1% and 19%, and there are few areas of concentrated agriculture. Most appears to hay/pasture, concentrated in the Lower Eel watershed and some of the other coastal valleys. Development is concentrated primarily near Eureka, on the coast in the Mad River and Redwood Creek watersheds. Much of the land area in this DPS is heavily forested, and there are a number of state and national parks.

2,4-D: 2,4-D was determined to pose a medium risk to fish. In the a.i. summary, we discuss three separate use patterns: direct water application of esters, direct water application of amines, terrestrial application of both esters and amines. While we expect exposure from most uses to be minimal, we are not able to predict where 2,4-D direct water applications may occur so we cannot rule out this type of exposure. Because the risk of these exposures is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of affecting populations of Northern California Steelhead as medium. Although effects from terrestrial applications are expected to reduce primary productivity in-stream, affect fish health, and degrade riparian vegetation in some locations, our greatest concern is the potential for salmonid lethality due to direct water applications of 2,4-D BEE. In cases of aquatic weed infestation, 2,4-D can be applied directly to the water. This use can occur anywhere, including areas with low routine pesticide use. NMFS believes that multiple populations may be exposed to these direct water applications within a single year, resulting in a decrease in population numbers significant enough to affect the entire DPS. Based on risk from all use patterns, we rate the likelihood of 2,4-D affecting the DPS as medium.

Triclopyr BEE: In the preceding a.i. summary, we determined that Triclopyr BEE poses a medium risk to fish. As discussed in the a.i. summary, the medium likelihood is based on our concerns about occasional high exposures resulting in a fish mortality event. However, as stated above, we expect exposure to this a.i. to be minimal, thereby decreasing the likelihood of a negative effect. Because the risk of this exposure is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of potential adverse effects to populations of Northern California Steelhead as low. Given the uses, fate, and toxicity of the chemical, we do not expect mortality to be a common occurrence. Thus, we believe that this compound has a low likelihood of causing an effect to the species as a whole.

Diuron: Diuron was determined to pose a medium risk to salmon. As discussed in the a.i. summary, use in rights-of-way is our largest concern, as it has the highest application rate and runoff potential. However, as stated above, we expect exposure to this a.i. to be minimal, thereby decreasing the likelihood of a negative effect. Because the risk of this exposure is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of potential adverse effects to populations of Northern California Steelhead as low. Given the use rates and pattern of uses for this chemical, we believe that there is a low likelihood of diuron causing a negative effect to multiple populations, or repeated negative effect to the same population. Therefore, we conclude that there is a low likelihood of an effect to the species as a whole.

Linuron: Given the limited uses of linuron, we expect exposure to be relatively infrequent. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Northern California Steelhead as low. As discussed in the a.i. summary, linuron exposure in agricultural areas may have some effect on the primary productivity of a system. We anticipate that the effects of linuron exposure would have a minimal effect on populations within this DPS. Thus, there would be a low likelihood of an effect to the DPS as a whole.

Captan: In the a.i. summary, we concluded that captan has a low likelihood of affecting salmon. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of Northern California Steelhead as low. Due to the fate properties of captan, we stated that water concentrations will rarely be high enough to cause a physiological response. Therefore, there is a low likelihood of captan causing population level, or species level effects.

Chlorothalonil: In the a.i. summary, we determined that there was a medium likelihood of chlorothalonil having a negative effect on salmon. However, as stated above, we expect exposure to this a.i. to be minimal, thereby decreasing the likelihood of a negative effect. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of adverse effects to populations of Northern California Steelhead as low. As such, there is a low likelihood of multiple populations experiencing adverse effects. Therefore, we believe that there is a low potential for effects at the DPS level.

Central California Coast (CCC) Steelhead (Threatened Species)

The CCC steelhead DPS includes nine historic independent populations, all of which are nearly extirpated. Data on abundance and population growth rates are scarce, but available information strongly suggests that no population is viable. The loss of spatial structure and hatchery influences have likely reduced the genetic diversity for this DPS. Juvenile steelhead remain in fresh water for one or more years rearing in small tributaries and floodplain habitats. Age to smoltificaton for this DPS is typically 1 to 4 years. Steelhead have a more adaptive life history than some of the other salmon species, including overlapping generations and iteropary.

High densities of crop farming occur throughout the San Joaquin Basin, the Delta, and along the lower Sacramento River. There is also agriculture in the Russian River valley. The Russian River population is one of the largest runs. Southern portions of DPS include the heavily developed areas around San Francisco Bay. The percentage of cultivated croplands and developed lands that overlap with CCV steelhead habitat are 27% and 10%, respectively. 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.

We expect that this population is exposed to multiple stressors as a result of these registrations. This area is highly developed, so we expect exposure to uses in urban, residential, and industrial areas. There is also a high concentration of roads, railroads and power lines resulting in multiple pathways for exposure to rights-of-way uses. The large bays within the DPS dilute exposure concentrations in the major waterways. Given these factors and the long residency period of steelhead, we expect that the populations will be exposed to all 6 a.i.s to some extent.

2,4-D: 2,4-D was determined to pose a medium risk to fish. In the a.i. summary, we discuss three separate use patterns: direct water application of esters, direct water application of amines, terrestrial application of both esters and amines. Because the risk of this exposure is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of affecting populations of CCC Steelhead as medium. Although effects from terrestrial applications are expected to reduce primary productivity in-stream, affect fish health, and degrade riparian vegetation in some locations, our greatest concern is the potential for salmonid lethality due to direct water applications of 2,4-D BEE. Based on risk from all use patterns, we rate the likelihood of 2,4-D affecting the DPS as medium.

Triclopyr BEE: In the preceding a.i. summary, we determined that Triclopyr BEE poses a medium risk to fish. As discussed in the a.i. summary, the medium likelihood is based on our concerns about occasional high exposures resulting in a fish mortality event. Because the risk of this exposure is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of affecting populations of CCC Steelhead as medium. If a high concentration of the a.i. was to co-occur with peak fish presence, this could cause a significant drop in the size of population. In order to decrease lambda of the species, multiple mortality events would have to occur, either within the same population or across several populations within the ESU. Given the fate and toxicity of the chemical, we do not expect mortality to be a common occurrence. However, given the degree of development and rights-of-ways within the range of the ESU, we expect use to be much more frequent, thereby increasing the likelihood of mortality. Thus, we believe that this compound has a medium likelihood of causing an effect to the species as a whole.

Diuron: Diuron was determined to pose a medium risk to salmon. As discussed in the a.i. summary, use in rights-of-way is our largest concern, as it has the highest application rate and runoff potential. As stated earlier, we expect that there will be a high degree of rights-of-way uses within this DPS. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations. Therefore we rate the likelihood to populations of CCC Steelhead as medium. We expect exposure to lead to declines in the primary productivity of the system, but not to wide-scale fish-health issues. Given the pattern of uses for this chemical and land use patterns within the range of the DPS, we believe that there is a medium likelihood of diuron causing a negative effect to the species as a whole.

Linuron: Given the limited uses of linuron, we expect exposure to be relatively infrequent. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of CCC Steelhead as low. As discussed in the a.i. summary, linuron exposure in agricultural areas may have some effect on the primary productivity of a system. We anticipate that the effects of linuron exposure would have a minimal effect on populations within this DPS. Thus, there would be a low likelihood of an effect to the DPS as a whole.

Captan: In the a.i. summary, we concluded that captan has a low likelihood of affecting salmon. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and so we rate the likelihood to populations of CCC Steelhead as low. As discussed earlier, NMFS believes that water concentrations will rarely be high enough to cause a physiological response. Therefore, there is a low likelihood of captan causing population level, or species level effects.

Chlorothalonil: In the a.i. summary, we determined that there was a medium likelihood of chlorothalonil having a negative effect on salmon. Higher or repeated exposures, particularly those associated with turf uses, may cause fish mortality. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of adverse effects to populations of CCC Steelhead as medium. The distribution of populations and use sites across the range of the species indicate that it is possible that multiple populations could experience substantial negative effects. Therefore, we believe that there is a medium potential for effects at the DPS level.

California Central Valley (CCV) Steelhead (Threatened Species)

The CCV steelhead DPS consisted of 81 historical and independent populations. The spatial structure of the CCV steelhead has been greatly reduced by loss of habitat diversity and tributary access from dams. Available information shows a significant long-term downward trend in abundance for this DPS (NMFS, 2009a). Population losses and reduction in abundance have reduced the genetic diversity that existed within the DPS.

We expect that individuals within this population will be exposed to the six a.i.s. Their range is heavily developed, for both agricultural and urban purposes. The percentage of agriculture, developed, and forested lands that overlap with CCV steelhead habitat are 32%, 10%, and 58%, respectively. Heavy use of agricultural pesticides and the high probability of mixtures increase likelihood of negative effects for this species. They are also likely to experience pesticide inputs from multiple sources, increasing the likelihood of exposure to each a.i. at intervals shorter than the labeled application interval. The valley also has a high concentration of power and transportation lines, indicating that rights-of-way applications will also occur. Juveniles typically rear for multiple years in fresh water. Juveniles also feed and rear in a variety of habitats, including the Sacramento River, the Delta, non-natal intermittent tributaries, tidal marshes, non-tidal freshwater marshes, and other shallow areas in the Delta as rearing areas for short periods during out-migration to the sea.

2,4-D: 2,4-D was determined to pose a medium risk to fish. In the a.i. summary, we discuss three separate use patterns: direct water application of esters, direct water application of amines, terrestrial application of both esters and amines. Because the risk of these exposures is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of affecting populations of CCV Steelhead as medium. Although effects from terrestrial applications are expected to reduce primary productivity in-stream, affect fish health, and degrade riparian vegetation in some locations, our greatest concern is the potential for salmonid lethality due to direct water applications of 2,4-D BEE. NMFS believes that multiple populations may be exposed to these direct water applications within a single year, resulting in a decrease in population numbers significant enough to affect the entire DPS. Based on risk from all use patterns, we rate the likelihood of 2,4-D affecting the DPS as medium.

Triclopyr BEE: In the preceding a.i. summary, we determined that Triclopyr BEE poses a medium risk to fish. As discussed in the a.i. summary, the medium likelihood is based on our concerns about occasional high exposures resulting in a fish mortality event. If a high concentration of the a.i. was to co-occur with peak fish presence, this could cause a significant drop in the size of population. Because the risk of this exposure is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of CCV Steelhead as medium. Given the uses, fate, and toxicity of the chemical, we do not expect mortality to be a common occurrence. However, given the degree of development and rights-of-ways within the range of the DPS, we expect use to be much more frequent, thereby increasing the likelihood of mortality. Thus, we believe that this compound has a medium likelihood of causing an effect to the species as a whole.

Diuron: Diuron was determined to pose a medium risk to salmon. As discussed in the a.i. summary, use in rights-of-way is our largest concern, as it has the highest application rate and runoff potential. Because the risk of this exposure is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of CCV Steelhead as medium. We expect exposure from other uses to lead to declines in the primary productivity of the system, but not to wide-scale fish-health issues. Given the use rates, pattern of uses for this chemical, and relatively widespread development, we believe that there is a medium likelihood of diuron causing a negative effect to multiple populations, or repeated negative effect to the same population. Therefore, we conclude that there is a medium likelihood of an effect to the species as a whole.

Linuron: Given the limited uses of linuron, we expect exposure to be relatively infrequent. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of CCV Steelhead as low. As discussed in the a.i. summary, linuron exposure in agricultural areas may have some effect on the primary productivity of a system. We anticipate that the effects of linuron exposure would have a minimal effect on populations within this DPS. Thus, there would be a low likelihood of an effect to the DPS as a whole.

Captan: In the a.i. summary, we concluded that captan has a low likelihood of affecting salmon. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of CCV Steelhead as low. Due to the fate properties of captan, we stated that water concentrations will rarely be high enough to cause a physiological response. Therefore, there is a low likelihood of captan causing population level, or species level effects.

Chlorothalonil: In the a.i. summary, we determined that there was a medium likelihood of chlorothalonil having a negative effect on salmon. Higher or repeated exposures, particularly those associated with turf uses, may cause fish mortality. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of CCV Steelhead as medium. The distribution of populations and use sites across the range of the species indicate that it is possible that multiple populations could experience substantial negative effects. Therefore, we believe that there is a medium potential for effects at the DPS level.

South-Central California Coast (S-CCC) Steelhead (Threatened Species)

The S-CCC steelhead DPS includes all naturally spawned steelhead in streams from the Pajaro River to the Santa Maria River. Population growth rates are unknown, though abundances are very depressed. Generally, juvenile steelhead remain in fresh water for one or more years before migrating downstream to smolt. Steelhead have a more adaptive life history than some of the other species, including overlapping generations, and iteropary. Following emergence, fry rear in smaller tributaries and floodplain habitats

Little information is available on the spatial structure or genetic diversity of this DPS. Because of the lack of information as to which populations are more important to the DPS, we have given the benefit of doubt to the species, and assumed that the populations in the mainstem of the Salinas and Pajaro Rivers, both of which have areas of intensive agriculture and development, are important.

The percentage of cultivated crop lands and developed lands that overlap with this DPS' range are 7% and 10%, respectively. Because of the degree of development in the system, we also expect that there will be a moderate to high amount of land which may have right-of-way applications. Agriculturally, the area is known for lettuces, strawberries, cut flowers, and vineyards. The volume of berries and grapes grown in the area makes it more likely that fungicides will be used within the basin. Agriculture is the dominant land use in the Salinas River valley, and there are areas of intense agriculture in the Pajaro watershed as well. Areas higher in the Salinas and Pajaro watersheds and along some of the coastal areas are much less developed, so are less affected.

2,4-D: 2,4-D was determined to pose a medium risk to fish. In the a.i. summary, we discuss three separate use patterns: direct water application of esters, direct water application of amines, terrestrial application of both esters and amines. Because the risk of these exposures is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of affecting populations of S-CCC steelhead as medium. Although effects from terrestrial applications are expected to reduce primary productivity in-stream, affect fish health, and degrade riparian vegetation in some locations, our greatest concern is the potential for salmonid lethality due to direct water applications of 2,4-D BEE. NMFS believes that multiple populations may be exposed to these direct water applications within a single year, resulting in a decrease in population numbers significant enough to affect the entire DPS. Based on risk from all use patterns, we rate the likelihood of 2,4-D affecting the DPS as medium.

Triclopyr BEE: In the preceding a.i. summary, we determined that Triclopyr BEE poses a medium risk to fish. As discussed in the a.i. summary, the medium likelihood is based on our concerns about occasional high exposures resulting in a fish mortality event. If a high concentration of the a.i. was to co-occur with peak fish presence, this could cause a significant drop in the size of population. Because the risk of this exposure is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of S-CCC steelhead as medium. Given the uses, fate, and toxicity of the chemical, we do not expect mortality to be a common occurrence. However, given the degree of development and rights-of-ways within the range of the DPS, we expect use to be much more frequent, thereby increasing the likelihood of mortality. Thus, we believe that this compound has a medium likelihood of causing an effect to the species as a whole.

Diuron: Diuron was determined to pose a medium risk to salmon. As discussed in the a.i. summary, use in rights-of-way is our largest concern, as it has the highest application rate and runoff potential. Because the risk of this exposure is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of S-CCC steelhead as medium. We expect exposure from other uses to lead to declines in the primary productivity of the system, but not to wide-scale fish-health issues. Given the use rates, pattern of uses for this chemical, and amount of development, we believe that there is a medium likelihood of diuron causing a negative effect to multiple populations, or repeated negative effect to the same population. Therefore, we conclude that there is a medium likelihood of an effect to the species as a whole.

Linuron: Given the limited uses of linuron, we expect exposure to be relatively infrequent. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of S-CCC steelhead as low. As discussed in the a.i. summary, linuron exposure in agricultural areas may have some effect on the primary productivity of a system. We anticipate that the effects of linuron exposure would have a minimal effect on populations within this DPS. Thus, there would be a low likelihood of an effect to the DPS as a whole.

Captan: In the a.i. summary, we concluded that captan has a low likelihood of affecting salmon. Due to cropping patterns, we expect that captan will be used more frequently in the Salinas River valley. However, NMFS believes that risk to this population is still low based on captan's fate properties. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of S-CCC steelhead as low. Due to the fate properties of captan, we stated that water concentrations will rarely be high enough to cause a physiological response. Therefore, there is a low likelihood of captan causing population level, or species level effects.

Chlorothalonil: In the a.i. summary, we determined that there was a medium likelihood of chlorothalonil having a negative effect on salmon. Higher or repeated exposures, particularly those associated with turf uses, may cause fish mortality. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of S-CCC steelhead as medium. The widespread distribution of use sites across the range of the species indicates that it is possible that multiple populations could experience substantial negative effects. Therefore, we believe that there is a medium potential for effects at the DPS level.

Southern California (SC) Steelhead (Endangered Species)

The SC steelhead DPS includes populations in five major and several small coastal river basins in California from the Santa Maria River to the U.S.–Mexican border. Long-term estimates and population trends are lacking for the streams within the DPS. The DPS experiences reduced and fragmented distribution, and large variations in annual spawner runs. Abundance is extremely low. SC steelhead juveniles may rear in fresh water or at the upper end of coastal lagoons for the first or second summer before migrating downstream to smolt.

This area is highly developed, so we expect exposure to uses in urban, residential, and industrial areas. There is also a high concentration of roads, railroads and power lines resulting in multiple pathways for exposure to rights-of-way uses. The percentage of cultivated crop lands and developed lands within SC steelhead habitat are about 5% and 34%, respectively. The agricultural areas are mostly along the coast of the more northern portion of the DPS. Some of the spawning and rearing areas are in the upper portions of these watersheds, away from the areas heavy development. Additionally, some populations overlap with portions of the Los Padres National Forest.

2,4-D: 2,4-D was determined to pose a medium risk to fish. In the a.i. summary, we discuss three separate use patterns: direct water application of esters, direct water application of amines, terrestrial application of both esters and amines. Because the risk of these exposures is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood of affecting populations of SC steelhead as medium. Although effects from terrestrial applications are expected to reduce primary productivity in-stream, affect fish health, and degrade riparian vegetation in some locations, our greatest concern is the potential for salmonid lethality due to direct water applications of 2,4-D BEE. NMFS believes that multiple populations may be exposed to these direct water applications within a single year, resulting in a decrease in population numbers significant enough to affect the entire DPS. Based on risk from all use patterns, we rate the likelihood of 2,4-D affecting the DPS as medium.

Triclopyr BEE: In the preceding a.i. summary, we determined that Triclopyr BEE poses a medium risk to fish. As discussed in the a.i. summary, the medium likelihood is based on our concerns about occasional high exposures resulting in a fish mortality event. If a high concentration of the a.i. was to co-occur with peak fish presence, this could cause a significant drop in the size of population. Because the risk of this exposure is fairly consistent among populations, we do not find a reason to differentiate risk among populations. However, given the uses, fate, and toxicity of the chemical, along with the distribution of populations, we do not expect mortality to be a common occurrence. Thus, we believe that this compound has a low likelihood of causing an effect to the species as a whole.

Diuron: Diuron was determined to pose a medium risk to salmon. As discussed in the a.i. summary, use in rights-of-way is our largest concern, as it has the highest application rate and runoff potential. Because the risk of this exposure is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of SC steelhead as medium. We expect exposure from other uses to lead to declines in the primary productivity of the system, but not to wide-scale fish-health issues. Given the use rates, pattern of uses for this chemical, and amount of development, we believe that there is a medium likelihood of diuron causing a negative effect to multiple populations, or repeated negative effect to the same population. Therefore, we conclude that there is a medium likelihood of an effect to the species as a whole.

Linuron: Given the limited uses of linuron, we expect exposure to be relatively infrequent. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of SC steelhead as low. As discussed in the a.i. summary, linuron exposure in agricultural areas may have some effect on the primary productivity of a system. We anticipate that the effects of linuron exposure would have a minimal effect on populations within this DPS. Thus, there would be a low likelihood of an effect to the DPS as a whole.

Captan: In the a.i. summary, we concluded that captan has a low likelihood of affecting salmon. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of SC steelhead as low. Due to the fate properties of captan, we stated that water concentrations will rarely be high enough to cause a physiological response. Therefore, there is a low likelihood of captan causing population level, or species level effects.

Chlorothalonil: In the a.i. summary, we determined that there was a medium likelihood of chlorothalonil having a negative effect on salmon. Higher or repeated exposures, particularly those associated with turf uses, may cause fish mortality. Because the risk is fairly consistent among populations, we do not find a reason to differentiate risk among populations, and therefore we rate the likelihood to populations of SC steelhead as medium. The widespread distribution of use sites across the range of the species indicates that it is possible that multiple populations could experience substantial negative effects. Therefore, we believe that there is a medium potential for effects at the DPS level.

Table 146. Potential for reduction in reproduction, abundance, or distribution.

Species	ESU/DPS	Herbicides				Fungicides	
		2,4-D	Triclopyr BEE	Diuron	Linuron	Captan	Chlorothalonil
Chinook	Puget Sound	Medium	Low	Low	Low	Low	Medium
	Lower Columbia River	Medium	Low	Low	Low	Low	Medium
	Upper Columbia River Spring - Run	Medium	Low	Low	Low	Low	Medium
	Snake River Fall - Run	Medium	Low	Low	Low	Low	Low
	Snake River Spring/Summer - Run	Medium	Low	Low	Low	Low	Low
	Upper Willamette River	Medium	Medium	Medium	Low	Low	Medium
	California Coastal	Medium	Low	Low	Low	Low	Medium
	Central Valley Spring - Run	Medium	Medium	Medium	Low	Low	Medium
	Sacramento River Winter - Run	Medium	Medium	Medium	Low	Low	Medium
Chum	Hood Canal Summer - Run	Medium	Low	Low	Low	Low	Medium
	Columbia River	Medium	Low	Low	Low	Low	Medium
Coho	Lower Columbia River	Medium	Medium	Medium	Low	Low	Medium
	Oregon Coast	Medium	Low	Low	Low	Low	Low
	Southern Oregon and Northern California Coast	Medium	Low	Low	Low	Low	Low
	Central California Coast	Medium	Low	Medium	Low	Low	Medium
Sockeye	Ozette Lake	Medium	Low	Low	Low	Low	Low
	Snake River	Medium	Low	Low	Low	Low	Low
Steelhead	Puget Sound	Medium	Low	Low	Low	Low	Medium
	Lower Columbia River	Medium	Low	Low	Low	Low	Medium
	Upper Willamette River	Medium	Medium	Medium	Low	Low	Medium
	Middle Columbia River	Medium	Low	Low	Low	Low	Medium
	Upper Columbia River	Medium	Low	Low	Low	Low	Medium
	Snake River	Medium	Low	Low	Low	Low	Low
	Northern California	Medium	Low	Low	Low	Low	Low
	Central California Coast	Medium	Medium	Medium	Low	Low	Medium
	California Central Valley	Medium	Medium	Medium	Low	Low	Medium
	South-Central California Coast	Medium	Medium	Medium	Low	Low	Medium
	Southern California	Medium	Low	Medium	Low	Low	Medium

Designated Critical Habitat Specific Evaluations for Each a.i.

Below, we summarize the current status of high and medium conservation value watersheds for each species, including baseline stressors. As exposure to 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 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

Of 61 assessed watersheds (HUC 5), 40 and 9 are of high and medium conservation value, respectively. Nineteen nearshore marine areas are also of high conservation value. Of the high value conservation watersheds, 32 and 40 are exposed to pesticides from agriculture and urban land uses, respectively. Among the medium value watersheds, six and nine are exposed to pesticides from agriculture and urban land uses, respectively. 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, reduced substrate conditions for spawning and incubation, and degraded water quality. Estuary PCEs in the northwest Puget Sound are also degraded from impaired water quality (*e.g.*, contaminants), altered salinity conditions, lack of natural cover, and modification of and lack of access to tidal marshes and their channels. 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 (1%) and hay crops and pastures (3%) are primarily distributed on the floodplain and other lowland habitats. The majority of urban/residential land use also occurs within river and stream valleys in lowland areas, much of the nearshore marine area also consists of urban/residential.

2,4-D: In the a.i. summary at the beginning of the Integration and Synthesis section, we determined that 2,4-D has a medium likelihood of degrading PCEs based on effects to water quality and riparian vegetation. We expect that habitat in heavily agricultural and/or urban areas are more likely to experience frequent low-level inputs. While there is moderate overlap between developed land and rearing and migratory PCEs, we do not believe that the level of effect from this degree of overlap is sufficient to result in an appreciable decline in the value of these PCEs and in the designated critical habitat of the Puget Sound Chinook.

Triclopyr BEE: As stated in the a.i. summary, triclopyr BEE is unlikely to result in reductions to PCEs. While applications to riparian zones could alter the community structure, that change could result in either a beneficial or detrimental effect to the habitat. As triclopyr BEE is often used in restoration projects, it is difficult to draw conclusions on all uses of this chemical. NMFS also believes that reductions in prey availability or instream primary productivity are unlikely. Therefore, there is a low likelihood of triclopyr BEE causing an appreciable decline in the conservation value of PCEs or critical habitat of the Puget Sound Chinook.

Diuron: In the a.i. summary, we determined that diuron has a medium likelihood of reducing the conservation value of PCEs and critical habitat due to effects on water quality, primary productivity instream, and riparian vegetation. We expect that habitat in heavily agricultural areas are more likely to experience frequent low-level inputs from multiple sources. We are also concerned about exposure resulting from applications to rights-of-way. Additionally, diuron’s persistence in the water column indicates that areas downstream from the application site could be affected. The moderate degree of overlap between developed land and rearing and migratory PCEs along with the toxicity and persistence of diuron may lead to widespread effects to critical habitat. As such, we believe that there may be an appreciable decline in the value of these PCEs and in the designated critical habitat of the Puget Sound Chinook.

Linuron: As it has very limited uses, we expect that exposure to linuron will be minimal. As such, we determined in the a.i. summary that linuron has a low likelihood of affecting PCEs. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Captan: As stated in the a.i. summary, we believe that captan has a low likelihood of causing a reduction in PCEs. Captan degrades very rapidly, so we do not anticipate much exposure. It is unlikely to reduce prey abundance or modify primary productivity or riparian vegetation. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Chlorothalonil: In the a.i. summary, we determined that chlorothalonil has a medium likelihood of degrading PCEs based on degradation of water quality. We anticipate frequent, low-level inputs to occur in highly agricultural or developed areas. As with the species analysis, we have strong concerns regarding use on turf grass, particularly on golf courses. There is a high degree of overlap between these land use categories and rearing and migratory PCEs. Based on the prevalence of this co-occurrence, we believe that there may be an appreciable decline in the value of these PCEs and in the designated critical habitat of the Puget Sound Chinook.

Lower Columbia River (LCR) Chinook Salmon

Thirty-one and 13 watersheds are of high and medium conservation value, respectively. Four additional unoccupied watersheds received a “possibly high” rating for species conservation as well. Our GIS analysis indicates 26 of 31 high conservation value watersheds are exposed to pesticide applications from agriculture and urban land uses, respectively. All 13 medium and 4 low conservation watersheds are also exposed to pesticide applications from both land uses.

Spawning and rearing PCEs for LCR Chinook salmon have been degraded by timber harvests, agriculture, and urbanization. These land uses have reduced floodplain connectivity and water quality, and removed natural cover in several rivers. Hydropower development projects have also reduced the timing and magnitude of water flows, thereby altering required water quantity to form and maintain physical habitat conditions for juvenile fish growth and mobility. Migration PCEs are also affected by several dams along the migration route used by adult and juvenile fish. The survival of yearlings in the ocean is also affected by habitat conditions in the estuary, such as changes in food availability and the presence of contaminants.

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.

2,4-D: In the a.i. summary at the beginning of the Integration and Synthesis section, we determined that 2,4-D has a medium likelihood of degrading PCEs based on effects to water quality and riparian vegetation. We expect that habitat in heavily agricultural and/or urban areas are more likely to experience frequent low-level inputs. While there is some overlap between developed land and rearing and migratory PCEs, we do not believe that the level of effect from this degree of overlap is sufficient to result in an appreciable decline in the value of these PCEs and in the designated critical habitat of the Lower Columbia River Chinook.

Triclopyr BEE: As stated in the a.i. summary, triclopyr BEE is unlikely to result in reductions to PCEs. While applications to riparian zones could alter the community structure, that change could result in either a beneficial or detrimental effect to the habitat. As triclopyr BEE is often used in restoration projects, it is difficult to draw conclusions on all uses of this chemical. NMFS also believes that reductions in prey availability or instream primary productivity are unlikely. Therefore, there is a low likelihood of triclopyr BEE causing an appreciable decline in the conservation value of PCEs or critical habitat of the Lower Columbia River Chinook.

Diuron: In the a.i. summary, we determined that diuron has a medium likelihood of reducing the conservation value of PCEs and critical habitat due to effects on water quality, primary productivity instream, and riparian vegetation. We expect that habitat in heavily agricultural areas is more likely to experience frequent low-level inputs from multiple sources. We are also concerned about exposure resulting from applications to rights-of-way. Additionally, diuron's persistence in the water column indicates that areas downstream from the application site could be affected. In the case of this ESU, we expect that exposure may result from applications in the heavily agricultural Upper Willamette basin. The moderate degree of overlap between developed land and rearing and migratory PCEs along with the toxicity and persistence of diuron may lead to widespread effects to critical habitat. As such, we believe that there may be an appreciable decline in the value of these PCEs and in the designated critical habitat of the Lower Columbia River Chinook.

Linuron: As it has very limited uses, we expect that exposure to linuron will be minimal. As such, we determined in the a.i. summary that linuron has a low likelihood of affecting PCEs. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Captan: As stated in the a.i. summary, we believe that captan has a low likelihood of causing a reduction in PCEs. Captan degrades very rapidly, so we do not anticipate much exposure. It is unlikely to reduce prey abundance or modify primary productivity or riparian vegetation. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Chlorothalonil: In the a.i. summary, we determined that chlorothalonil has a medium likelihood of degrading PCEs based on degradation of water quality. We anticipate frequent, low-level inputs to occur in highly agricultural or developed areas. As with the species analysis, we have strong concerns regarding use on turf grass, particularly on golf courses. There is a moderate degree of overlap between these land use categories and rearing and migratory PCEs. Based on the level of co-occurrence, we believe that there will not be an appreciable decline in the value of these PCEs and in the designated critical habitat of the Lower Columbia River Chinook.

Upper Columbia River (UCR) Spring-run Chinook Salmon

Twenty-six and five watersheds are of high and medium conservation value, respectively. Our GIS analysis indicates 23 and 26 high conservation watersheds are exposed to pesticide applications from agriculture and urban land uses, respectively. All medium conservation value watersheds are also exposed to pesticides from both land uses.

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.

2,4-D: In the a.i. summary at the beginning of the Integration and Synthesis section, we determined that 2,4-D has a medium likelihood of degrading PCEs based on effects to water quality and riparian vegetation. We expect that habitat in heavily agricultural and/or urban areas are more likely to experience frequent low-level inputs. As there is limited agriculture and development in this ESU, there is a low likelihood of 2,4-D causing an appreciable decline in the value of these PCEs and in the designated critical habitat of the Upper Columbia River Chinook.

Triclopyr BEE: As stated in the a.i. summary, triclopyr BEE is unlikely to result in reductions to PCEs. While applications to riparian zones could alter the community structure, that change could result in either a beneficial or detrimental effect to the habitat. As triclopyr BEE is often used in restoration projects, it is difficult to draw conclusions on all uses of this chemical. NMFS also believes that reductions in prey availability or instream primary productivity are unlikely. Therefore, there is a low likelihood of triclopyr BEE causing an appreciable decline in the conservation value of PCEs or critical habitat of the Upper Columbia River Chinook.

Diuron: In the a.i. summary, we determined that diuron has a medium likelihood of reducing the conservation value of PCEs and critical habitat due to effects on water quality, primary productivity instream, and riparian vegetation. We expect that habitat in heavily agricultural areas is more likely to experience frequent low-level inputs from multiple sources. We are also concerned about exposure resulting from applications to rights-of-way. However, given the land use within this ESU, we do not believe much exposure will occur. As such, we believe that there is a low likelihood of diuron causing an appreciable decline in the value of these PCEs and in the designated critical habitat of the Upper Columbia River Chinook.

Linuron: As it has very limited uses, we expect that exposure to linuron will be minimal. As such, we determined in the a.i. summary that linuron has a low likelihood of affecting PCEs. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Captan: As stated in the a.i. summary, we believe that captan has a low likelihood of causing a reduction in PCEs. Captan degrades very rapidly, so we do not anticipate much exposure. It is unlikely to reduce prey abundance or modify primary productivity or riparian vegetation. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Chlorothalonil: In the a.i. summary, we determined that chlorothalonil has a medium likelihood of degrading PCEs based on degradation of water quality. We anticipate frequent, low-level inputs will occur in highly agricultural or developed areas. There is a low degree of overlap between these land use categories and all PCEs. Based on the level of co-occurrence, we believe that there will not be an appreciable decline in the value of these PCEs and in the designated critical habitat of the Upper Columbia River Chinook.

Snake River (SR) Fall-run Chinook Salmon

Individual watersheds within the range of SR Fall-run Chinook salmon have not been evaluated by the CHART team for their conservation value. However, the Lower Columbia River corridor is of high conservation value as it connects several populations with the ocean and is used by rearing/migrating juveniles and migrating adults. The Columbia River estuary is also a unique and essential area for juveniles and adults making the physiological transition between life in freshwater and marine habitats. 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.

2,4-D: In the a.i. summary at the beginning of the Integration and Synthesis section, we determined that 2,4-D has a medium likelihood of degrading PCEs based on effects to water quality and riparian vegetation. We expect that habitat in heavily agricultural and/or urban areas are more likely to experience frequent low-level inputs. As there is limited agriculture and development in this ESU, there is a low likelihood of 2,4-D causing an appreciable decline in the value of these PCEs and in the designated critical habitat of the SR Fall-run Chinook.

Triclopyr BEE: As stated in the a.i. summary, triclopyr BEE is unlikely to result in reductions to PCEs. While applications to riparian zones could alter the community structure, that change could result in either a beneficial or detrimental effect to the habitat. As triclopyr BEE is often used in restoration projects, it is difficult to draw conclusions on all uses of this chemical. NMFS also believes that reductions in prey availability or instream primary productivity are unlikely. Therefore, there is a low likelihood of triclopyr BEE causing an appreciable decline in the conservation value of PCEs or critical habitat of the SR Fall-run Chinook.

Diuron: In the a.i. summary, we determined that diuron has a medium likelihood of reducing the conservation value of PCEs and critical habitat due to effects on water quality, primary productivity instream, and riparian vegetation. We expect that habitat in heavily agricultural areas is more likely to experience frequent low-level inputs from multiple sources. We are also concerned about exposure resulting from applications to rights-of-way. However, given the land use within this ESU, we do not believe much exposure will occur. As such, we believe that there is a low likelihood of diuron causing an appreciable decline in the value of these PCEs and in the designated critical habitat of the SR Fall-run Chinook.

Linuron: As it has very limited uses, we expect that exposure to linuron will be minimal. As such, we determined in the a.i. summary that linuron has a low likelihood of affecting PCEs. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Captan: As stated in the a.i. summary, we believe that captan has a low likelihood of causing a reduction in PCEs. Captan degrades very rapidly, so we do not anticipate much exposure. It is unlikely to reduce prey abundance or modify primary productivity or riparian vegetation. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Chlorothalonil: In the a.i. summary, we determined that chlorothalonil has a medium likelihood of degrading PCEs based on degradation of water quality. We anticipate frequent, low-level inputs will occur in highly agricultural or developed areas. There is a low degree of overlap between these land use categories and all PCEs. Based on the level of co-occurrence, we believe that there will not be an appreciable decline in the value of these PCEs and in the designated critical habitat of the SR Fall-run Chinook.

Snake River (SR) Spring/Summer-run Chinook Salmon

Watersheds within the range of SR Spring/Summer-run Chinook salmon were not evaluated by the CHART team for their conservation value. However, the Lower Columbia River is of high conservation value as it connects every population with the ocean and is used by rearing/migrating juveniles and migrating adults. Juveniles of this ESU rely on adequate fresh water quality and prey abundance for migrating and rearing in freshwater habitats including migratory routes from natal reaches leading to alternative summer-rearing or overwintering areas.

Spawning and juvenile 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 ESU, and those that are present are clustered mostly around the mainstem Snake and Columbia Rivers. The Snake River is a high-volume, high-flow system, and salmon use it primarily as a migratory corridor.

2,4-D: In the a.i. summary at the beginning of the Integration and Synthesis section, we determined that 2,4-D has a medium likelihood of degrading PCEs based on effects to water quality and riparian vegetation. We expect that habitat in heavily agricultural and/or urban areas are more likely to experience frequent low-level inputs. As there is limited agriculture and development in this ESU, there is a low likelihood of 2,4-D causing an appreciable decline in the value of these PCEs and in the designated critical habitat of the SR Spring/Summer-run Chinook.

Triclopyr BEE: As stated in the a.i. summary, triclopyr BEE is unlikely to result in reductions to PCEs. While applications to riparian zones could alter the community structure, that change could result in either a beneficial or detrimental effect to the habitat. As triclopyr BEE is often used in restoration projects, it is difficult to draw conclusions on all uses of this chemical. NMFS also believes that reductions in prey availability or instream primary productivity are unlikely. Therefore, there is a low likelihood of triclopyr BEE causing an appreciable decline in the conservation value of PCEs or critical habitat of the SR Spring/Summer-run Chinook.

Diuron: In the a.i. summary, we determined that diuron has a medium likelihood of reducing the conservation value of PCEs and critical habitat due to effects on water quality, primary productivity instream, and riparian vegetation. We expect that habitat in heavily agricultural areas is more likely to experience frequent low-level inputs from multiple sources. We are also concerned about exposure resulting from applications to rights-of-way. However, given the land use within this ESU, we do not believe much exposure will occur. As such, we believe that there is a low likelihood of diuron causing an appreciable decline in the value of these PCEs and in the designated critical habitat of the SR Spring/Summer-run Chinook.

Linuron: As it has very limited uses, we expect that exposure to linuron will be minimal. As such, we determined in the a.i. summary that linuron has a low likelihood of affecting PCEs. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Captan: As stated in the a.i. summary, we believe that captan has a low likelihood of causing a reduction in PCEs. Captan degrades very rapidly, so we do not anticipate much exposure. It is unlikely to reduce prey abundance or modify primary productivity or riparian vegetation. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Chlorothalonil: In the a.i. summary, we determined that chlorothalonil has a medium likelihood of degrading PCEs based on degradation of water quality. We anticipate frequent, low-level inputs will occur in highly agricultural or developed areas. There is a low degree of overlap between these land use categories and all PCEs. Based on the level of co-occurrence, we believe that there will not be an appreciable decline in the value of these PCEs and in the designated critical habitat of the SR Spring/Summer-run Chinook.

Upper Willamette River (UWR) Chinook Salmon

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. 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 10.5% and 9%, respectively. Spawning, rearing, and migration freshwater PCEs in these exposed watersheds (including mainstem and floodplain wetlands) likely experience reductions in water quality and prey abundance.

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.

2,4-D: In the a.i. summary at the beginning of the Integration and Synthesis section, we determined that 2,4-D has a medium likelihood of degrading PCEs based on effects to water quality and riparian vegetation. We expect that habitat in heavily agricultural and/or urban areas are more likely to experience frequent low-level inputs. There is a high degree of overlap between developed land and rearing and migratory PCEs. Based on the prevalence of this co-occurrence, we believe that there may be an appreciable decline in the value of these PCEs and in the designated critical habitat of the Upper Willamette River Chinook.

Triclopyr BEE: As stated in the a.i. summary, triclopyr BEE is unlikely to result in reductions to PCEs. While applications to riparian zones could alter the community structure, that change could result in either a beneficial or detrimental effect to the habitat. As triclopyr BEE is often used in restoration projects, it is difficult to draw conclusions on all uses of this chemical. Those modifications would be site specific, and given the scope of this analysis we cannot draw a conclusion either way. NMFS also believes that reductions in prey availability or instream primary productivity are unlikely. Therefore, there is a low likelihood of triclopyr BEE causing an appreciable decline in the conservation value of PCEs or critical habitat of the Upper Willamette River Chinook.

Diuron: In the a.i. summary, we determined that diuron has a medium likelihood of reducing the conservation value of PCEs and critical habitat due to effects on water quality, primary productivity instream, and riparian vegetation. We expect that habitat in heavily agricultural areas are more likely to experience frequent low-level inputs from multiple sources. There is a high degree of overlap between developed land and rearing and migratory PCEs. Based on the prevalence of this co-occurrence, we believe that there may be an appreciable decline in the value of these PCEs and in the designated critical habitat of the Upper Willamette River Chinook.

Linuron: As it has very limited uses, we expect that exposure to linuron will be minimal. As such, we determined in the a.i. summary that linuron has a low likelihood of affecting PCEs. Therefore, we do not expect that the conservation value of PCES and critical habitat will be appreciably reduced.

Captan: As stated in the a.i. summary, we believe that captan has a low likelihood of causing a reduction in PCEs. Captan degrades very rapidly, so we do not anticipate much exposure. It is unlikely to reduce prey abundance or modify primary productivity or riparian vegetation.

Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Chlorothalonil: In the a.i. summary, we determined that chlorothalonil has a medium likelihood of degrading PCEs based on degradation of water quality. We anticipate frequent, low-level inputs to occur in highly agricultural or developed areas. There is a high degree of overlap between these land use categories and rearing and migratory PCEs. Based on the prevalence of this co-occurrence, we believe that there may be an appreciable decline in the value of these PCEs and in the designated critical habitat of the Upper Willamette River Chinook.

California Coastal (CC) Chinook Salmon

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. 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 a low population abundance across the ESU.

2,4-D: In the a.i. summary at the beginning of the Integration and Synthesis section, we determined that 2,4-D has a medium likelihood of degrading PCEs based on effects to water quality and riparian vegetation. We expect that habitat in heavily agricultural and/or urban areas are more likely to experience frequent low-level inputs. As there is limited agriculture and development in this ESU, there is a low likelihood of 2,4-D causing an appreciable decline in the value of these PCEs and in the designated critical habitat of the CC Chinook.

Triclopyr BEE: As stated in the a.i. summary, triclopyr BEE is unlikely to result in reductions to PCEs. While applications to riparian zones could alter the community structure, that change could result in either a beneficial or detrimental effect to the habitat. As triclopyr BEE is often used in restoration projects, it is difficult to draw conclusions on all uses of this chemical. NMFS also believes that reductions in prey availability or instream primary productivity are unlikely. Therefore, there is a low likelihood of triclopyr BEE causing an appreciable decline in the conservation value of PCEs or critical habitat of the CC Chinook.

Diuron: In the a.i. summary, we determined that diuron has a medium likelihood of reducing the conservation value of PCEs and critical habitat due to effects on water quality, primary productivity instream, and riparian vegetation. We expect that habitat in heavily agricultural areas is more likely to experience frequent low-level inputs from multiple sources. We are also concerned about exposure resulting from applications to rights-of-way. However, given the land use within this ESU, we do not believe much exposure will occur. As such, we believe that there is a low likelihood of diuron causing an appreciable decline in the value of these PCEs and in the designated critical habitat of the CC Chinook.

Linuron: As it has very limited uses, we expect that exposure to linuron will be minimal. As such, we determined in the a.i. summary that linuron has a low likelihood of affecting PCEs. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Captan: As stated in the a.i. summary, we believe that captan has a low likelihood of causing a reduction in PCEs. Captan degrades very rapidly, so we do not anticipate much exposure. It is unlikely to reduce prey abundance or modify primary productivity or riparian vegetation.

Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Chlorothalonil: In the a.i. summary, we determined that chlorothalonil has a medium likelihood of degrading PCEs based on degradation of water quality. We anticipate frequent, low-level inputs will occur in highly agricultural or developed areas. There is a low degree of overlap between these land use categories and all PCEs. Based on the level of co-occurrence, we believe that there will not be an appreciable decline in the value of these PCEs and in the designated critical habitat of the CC Chinook.

Central Valley (CV) Spring-run Chinook Salmon

Of 38 occupied watersheds, 28 and 3 are of high and medium conservation value, respectively. Four of these watersheds comprise portions of the San Francisco-San Pablo-Suisun Bay estuarine complex which provides rearing and migratory habitat for CV Spring-run Chinook salmon. Our GIS analysis indicates 17 and 28 high conservation value watersheds are exposed to pesticides from agriculture and urban land uses, respectively. Of the medium conservation watersheds, two and three watersheds are exposed to from the above land uses as well. All low value watersheds are exposed to pesticide applications from urban land uses, 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 from the mainstem of larger rivers through the Sacramento River watershed, thereby reducing effective foraging. The migration PCE is degraded by lack of natural cover along the migration corridors. Juvenile migration is further obstructed by water diversions along the Sacramento River and by two large state and federal water-export facilities in the Sacramento-San Joaquin Delta. Agriculture and urban runoff containing a suite of pollutants further impair water quality of receiving systems used by this species.

Intensive agricultural development occurs in the California Central Valley and may impact waters draining into the Sacramento River. We further expect rearing and migration PCEs in non-natal tributaries, intermittent streams, and floodplain habitats may also experience likely reductions in water quality and prey abundance. 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.

2,4-D: In the a.i. summary at the beginning of the Integration and Synthesis section, we determined that 2,4-D has a medium likelihood of degrading PCEs based on effects to water quality and riparian vegetation. We expect that habitat in heavily agricultural and/or urban areas are more likely to experience frequent low-level inputs. There is a high degree of overlap between developed land and rearing and migratory PCEs. Based on the prevalence of this co-occurrence, we believe that there may be an appreciable decline in the value of these PCEs and in the designated critical habitat of the CV Spring-run Chinook.

Triclopyr BEE: As stated in the a.i. summary, triclopyr BEE is unlikely to result in reductions to PCEs. While applications to riparian zones could alter the community structure, that change could result in either a beneficial or detrimental effect to the habitat. As triclopyr BEE is often used in restoration projects, it is difficult to draw conclusions on all uses of this chemical. Those modifications would be site specific, and given the scope of this analysis we cannot draw a conclusion either way. NMFS also believes that reductions in prey availability or instream primary productivity are unlikely. Therefore, there is a low likelihood of triclopyr BEE causing an appreciable decline in the conservation value of PCEs or critical habitat of the CV Spring-run Chinook.

Diuron: In the a.i. summary, we determined that diuron has a medium likelihood of reducing the conservation value of PCEs and critical habitat due to effects on water quality, primary productivity instream, and riparian vegetation. We expect that habitat in heavily agricultural areas are more likely to experience frequent low-level inputs from multiple sources. There is a high degree of overlap between developed land and rearing and migratory PCEs. Based on the prevalence of this co-occurrence, we believe that there may be an appreciable decline in the value of these PCEs and in the designated critical habitat of the CV Spring-run Chinook.

Linuron: As it has very limited uses, we expect that exposure to linuron will be minimal. As such, we determined in the a.i. summary that linuron has a low likelihood of affecting PCEs. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Captan: As stated in the a.i. summary, we believe that captan has a low likelihood of causing a reduction in PCEs. Captan degrades very rapidly, so we do not anticipate much exposure. It is unlikely to reduce prey abundance or modify primary productivity or riparian vegetation. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Chlorothalonil: In the a.i. summary, we determined that chlorothalonil has a medium likelihood of degrading PCEs based on degradation of water quality. We anticipate frequent, low-level inputs to occur in highly agricultural or developed areas. There is a high degree of overlap between these land use categories and rearing and migratory PCEs. Based on the prevalence of this co-occurrence, we believe that there may be an appreciable decline in the value of these PCEs and in the designated critical habitat of the CV Spring-run Chinook.

Sacramento River Winter-run Chinook Salmon

Individual subbasins or river sections were not evaluated for their conservation value. However, the entire Sacramento River and the Delta are considered of high conservation value for spawning, rearing, and migration.

Spawning and rearing PCEs are currently degraded by elevated water temperature and lost access to historic spawning areas in upper watersheds with cool and clean water throughout the summer. The rearing PCE is degraded and is affected by loss of floodplain habitat connection from the mainstem of larger rivers through the Sacramento River watershed, thereby reducing effective foraging. The migration PCE is degraded by lack of natural cover along the migration corridors. Juvenile migration is further obstructed by water diversions along the Sacramento River and by two large state and federal water-export facilities in the Sacramento-San Joaquin Delta. 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.

2,4-D: In the a.i. summary at the beginning of the Integration and Synthesis section, we determined that 2,4-D has a medium likelihood of degrading PCEs based on effects to water quality and riparian vegetation. We expect that habitat in heavily agricultural and/or urban areas are more likely to experience frequent low-level inputs. There is a high degree of overlap between developed land and rearing and migratory PCEs. Based on the prevalence of this co-occurrence, we believe that there may be an appreciable decline in the value of these PCEs and in the designated critical habitat of the Sacramento winter-run Chinook.

Triclopyr BEE: As stated in the a.i. summary, triclopyr BEE is unlikely to result in reductions to PCEs. While applications to riparian zones could alter the community structure, that change could result in either a beneficial or detrimental effect to the habitat. As triclopyr BEE is often used in restoration projects, it is difficult to draw conclusions on all uses of this chemical. Those modifications would be site specific, and given the scope of this analysis we cannot draw a conclusion either way. NMFS also believes that reductions in prey availability or instream primary productivity are unlikely. Therefore, there is a low likelihood of triclopyr BEE causing an appreciable decline in the conservation value of PCEs or critical habitat of the Sacramento winter-run Chinook.

Diuron: In the a.i. summary, we determined that diuron has a medium likelihood of reducing the conservation value of PCEs and critical habitat due to effects on water quality, primary productivity instream, and riparian vegetation. We expect that habitat in heavily agricultural areas are more likely to experience frequent low-level inputs from multiple sources. There is a high degree of overlap between developed land and rearing and migratory PCEs. Based on the prevalence of this co-occurrence, we believe that there may be an appreciable decline in the value of these PCEs and in the designated critical habitat of the Sacramento winter-run Chinook.

Linuron: As it has very limited uses, we expect that exposure to linuron will be minimal. As such, we determined in the a.i. summary that linuron has a low likelihood of affecting PCEs. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Captan: As stated in the a.i. summary, we believe that captan has a low likelihood of causing a reduction in PCEs. Captan degrades very rapidly, so we do not anticipate much exposure. It is unlikely to reduce prey abundance or modify primary productivity or riparian vegetation. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Chlorothalonil: In the a.i. summary, we determined that chlorothalonil has a medium likelihood of degrading PCEs based on degradation of water quality. We anticipate frequent, low-level inputs to occur in highly agricultural or developed areas. There is a high degree of overlap between these land use categories and rearing and migratory PCEs. Based on the prevalence of this co-occurrence, we believe that there may be an appreciable decline in the value of these PCEs and in the designated critical habitat of the Sacramento winter-run Chinook.

Hood Canal Summer-run Chum Salmon

Of 12 assessed watersheds, nine and three are of high and medium conservation value, respectively. Five nearshore marine areas were also rated as high conservation value. Many of the watersheds have less than four miles of spawning habitat and none are greater than 8.5 miles in length. Our GIS analysis indicates seven and nine high conservation value watersheds are exposed to pesticides from agriculture and urban land uses, respectively. All three medium conservation watersheds are exposed to both land uses as well.

The spawning PCE is degraded by excessive fine sediment in gravel. The rearing PCE is degraded by loss of access to sloughs in the estuary and nearshore areas and excessive predation. Migration and rearing PCEs in estuaries are impaired by the loss of functional floodplain areas. These degraded conditions likely maintain low population abundance across the ESU.

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.

2,4-D: In the a.i. summary at the beginning of the Integration and Synthesis section, we determined that 2,4-D has a medium likelihood of degrading PCEs based on effects to water quality and riparian vegetation. We expect that habitat in heavily agricultural and/or urban areas are more likely to experience frequent low-level inputs. As there is limited agriculture and development in this ESU, there is a low likelihood of 2,4-D causing an appreciable decline in the value of these PCEs and in the designated critical habitat of the Hood Canal chum.

Triclopyr BEE: As stated in the a.i. summary, triclopyr BEE is unlikely to result in reductions to PCEs. While applications to riparian zones could alter the community structure, that change could result in either a beneficial or detrimental effect to the habitat. As triclopyr BEE is often used in restoration projects, it is difficult to draw conclusions on all uses of this chemical. NMFS also believes that reductions in prey availability or instream primary productivity are unlikely. Therefore, there is a low likelihood of triclopyr BEE causing an appreciable decline in the conservation value of PCEs or critical habitat of the Hood Canal chum.

Diuron: In the a.i. summary, we determined that diuron has a medium likelihood of reducing the conservation value of PCEs and critical habitat due to effects on water quality, primary productivity instream, and riparian vegetation. We expect that habitat in heavily agricultural areas is more likely to experience frequent low-level inputs from multiple sources. We are also concerned about exposure resulting from applications to rights-of-way. However, given the land use within this ESU, we do not believe much exposure will occur. As such, we believe that there is a low likelihood of diuron causing an appreciable decline in the value of these PCEs and in the designated critical habitat of the Hood Canal chum.

Linuron: As it has very limited uses, we expect that exposure to linuron will be minimal. As such, we determined in the a.i. summary that linuron has a low likelihood of affecting PCEs. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Captan: As stated in the a.i. summary, we believe that captan has a low likelihood of causing a reduction in PCEs. Captan degrades very rapidly, so we do not anticipate much exposure. It is unlikely to reduce prey abundance or modify primary productivity or riparian vegetation. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Chlorothalonil: In the a.i. summary, we determined that chlorothalonil has a medium likelihood of degrading PCEs based on degradation of water quality. We anticipate frequent, low-level inputs will occur in highly agricultural or developed areas. There is a low degree of overlap between these land use categories and all PCEs. Based on the level of co-occurrence, we believe that there will not be an appreciable decline in the value of these PCEs and in the designated critical habitat of the Hood Canal chum.

Columbia River (CR) Chum Salmon

Of 19 assessed watersheds, 16 and 3 are of high and medium conservation value, respectively. Our GIS analysis indicates all high and medium conservation value watersheds are exposed to pesticide applications from agriculture, developed areas, and forestry adjacent to CR chum salmon habitat.

The migration PCE for this species has been significantly impacted by dams obstructing adult migration and access to historic spawning sites. Water quality and cover for estuary and rearing PCEs have decreased and are not likely to maintain their intended function to conserve the species. Elevated water temperature further prevents CR chum salmon from inhabiting 272.8 km of waters within its range.

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.

2,4-D: In the a.i. summary at the beginning of the Integration and Synthesis section, we determined that 2,4-D has a medium likelihood of degrading PCEs based on effects to water quality and riparian vegetation. We expect that habitat in heavily agricultural and/or urban areas are more likely to experience frequent low-level inputs. While there is some overlap between developed land and rearing and migratory PCEs, we do not believe that the level of effect from this degree of overlap is sufficient to result in an appreciable decline in the value of these PCEs and in the designated critical habitat of the Columbia River chum.

Triclopyr BEE: As stated in the a.i. summary, triclopyr BEE is unlikely to result in reductions to PCEs. While applications to riparian zones could alter the community structure, that change could result in either a beneficial or detrimental effect to the habitat. As triclopyr BEE is often used in restoration projects, it is difficult to draw conclusions on all uses of this chemical. NMFS also believes that reductions in prey availability or instream primary productivity are unlikely. Therefore, there is a low likelihood of triclopyr BEE causing an appreciable decline in the conservation value of PCEs or critical habitat of the Columbia River chum.

Diuron: In the a.i. summary, we determined that diuron has a medium likelihood of reducing the conservation value of PCEs and critical habitat due to effects on water quality, primary productivity instream, and riparian vegetation. We expect that habitat in heavily agricultural areas is more likely to experience frequent low-level inputs from multiple sources. We are also concerned about exposure resulting from applications to rights-of-way. Additionally, diuron's persistence in the water column indicates that areas downstream from the application site could be affected. In the case of this ESU, the locations of spawning PCEs make exposure less likely. Additionally, the short residency periods decreases the reliance on areas that would be affected by applications upstream in the heavily agricultural Upper Willamette basin. As such, we believe that there is a low likelihood of an appreciable decline in the value of these PCEs and in the designated critical habitat of the Columbia River chum.

Linuron: As it has very limited uses, we expect that exposure to linuron will be minimal. As such, we determined in the a.i. summary that linuron has a low likelihood of affecting PCEs. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Captan: As stated in the a.i. summary, we believe that captan has a low likelihood of causing a reduction in PCEs. Captan degrades very rapidly, so we do not anticipate much exposure. It is unlikely to reduce prey abundance or modify primary productivity or riparian vegetation. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Chlorothalonil: In the a.i. summary, we determined that chlorothalonil has a medium likelihood of degrading PCEs based on degradation of water quality. We anticipate frequent, low-level inputs to occur in highly agricultural or developed areas. As with the species analysis, we have strong concerns regarding use on turf grass, particularly on golf courses. There is a moderate degree of overlap between these land use categories and rearing and migratory PCEs. Based on the level of co-occurrence, we believe that there will not be an appreciable decline in the value of these PCEs and in the designated critical habitat of the Columbia River chum.

Oregon Coast (OC) Coho Salmon

Of 80 watersheds, 45 and 27 are of high and medium conservation value, respectively. Our GIS analysis indicates 39 and 44 high conservation watersheds are exposed to pesticides from agriculture and urban areas, respectively. Of the medium conservation watersheds, 18 and 23 are exposed to pesticide applications from the above respective land uses. 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.

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.

2,4-D: In the a.i. summary at the beginning of the Integration and Synthesis section, we determined that 2,4-D has a medium likelihood of degrading PCEs based on effects to water quality and riparian vegetation. We expect that habitat in heavily agricultural and/or urban areas are more likely to experience frequent low-level inputs. As there is limited agriculture and development in this ESU, there is a low likelihood of 2,4-D causing an appreciable decline in the value of these PCEs and in the designated critical habitat of the OC coho.

Triclopyr BEE: As stated in the a.i. summary, triclopyr BEE is unlikely to result in reductions to PCEs. While applications to riparian zones could alter the community structure, that change could result in either a beneficial or detrimental effect to the habitat. As triclopyr BEE is often used in restoration projects, it is difficult to draw conclusions on all uses of this chemical. NMFS also believes that reductions in prey availability or instream primary productivity are unlikely. Therefore, there is a low likelihood of triclopyr BEE causing an appreciable decline in the conservation value of PCEs or critical habitat of the OC coho.

Diuron: In the a.i. summary, we determined that diuron has a medium likelihood of reducing the conservation value of PCEs and critical habitat due to effects on water quality, primary productivity instream, and riparian vegetation. We expect that habitat in heavily agricultural areas is more likely to experience frequent low-level inputs from multiple sources. We are also concerned about exposure resulting from applications to rights-of-way. However, given the land use within this ESU, we do not believe much exposure will occur. As such, we believe that there is a low likelihood of diuron causing an appreciable decline in the value of these PCEs and in the designated critical habitat of the OC coho.

Linuron: As it has very limited uses, we expect that exposure to linuron will be minimal. As such, we determined in the a.i. summary that linuron has a low likelihood of affecting PCEs. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Captan: As stated in the a.i. summary, we believe that captan has a low likelihood of causing a reduction in PCEs. Captan degrades very rapidly, so we do not anticipate much exposure. It is unlikely to reduce prey abundance or modify primary productivity or riparian vegetation. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Chlorothalonil: In the a.i. summary, we determined that chlorothalonil has a medium likelihood of degrading PCEs based on degradation of water quality. We anticipate frequent, low-level inputs will occur in highly agricultural or developed areas. There is a low degree of overlap between these land use categories and all PCEs. Based on the level of co-occurrence, we believe that there will not be an appreciable decline in the value of these PCEs and in the designated critical habitat of the OC coho.

Southern Oregon/Northern California Coast (SONCC) Coho Salmon

Although watersheds within this ESU were not evaluated for their conservation value, the northern coastal streams that are designated as critical habitat are of good quality. Throughout this ESU's range, the spawning PCE has been degraded by fines in spawning gravel from logging. The rearing PCE has been considerably degraded in many inland watersheds by the loss of riparian vegetation, resulting in unsuitable high temperatures. Rearing and migration PCEs have been reduced by the disconnection of floodplain and off-channel habitats in low gradient reaches of streams. Elevated water temperature further prevents SONCC coho salmon from inhabiting 3,249.2 km of waters within its range.

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.

2,4-D: In the a.i. summary at the beginning of the Integration and Synthesis section, we determined that 2,4-D has a medium likelihood of degrading PCEs based on effects to water quality and riparian vegetation. We expect that habitat in heavily agricultural and/or urban areas are more likely to experience frequent low-level inputs. As there is limited agriculture and development in this ESU, there is a low likelihood of 2,4-D causing an appreciable decline in the value of these PCEs and in the designated critical habitat of the SONCC coho.

Triclopyr BEE: As stated in the a.i. summary, triclopyr BEE is unlikely to result in reductions to PCEs. While applications to riparian zones could alter the community structure, that change could result in either a beneficial or detrimental effect to the habitat. As triclopyr BEE is often used in restoration projects, it is difficult to draw conclusions on all uses of this chemical. NMFS also believes that reductions in prey availability or instream primary productivity are unlikely. Therefore, there is a low likelihood of triclopyr BEE causing an appreciable decline in the conservation value of PCEs or critical habitat of the SONCC coho.

Diuron: In the a.i. summary, we determined that diuron has a medium likelihood of reducing the conservation value of PCEs and critical habitat due to effects on water quality, primary productivity instream, and riparian vegetation. We expect that habitat in heavily agricultural areas is more likely to experience frequent low-level inputs from multiple sources. We are also concerned about exposure resulting from applications to rights-of-way. However, given the land use within this ESU, we do not believe much exposure will occur. As such, we believe that there is a low likelihood of diuron causing an appreciable decline in the value of these PCEs and in the designated critical habitat of the SONCC coho.

Linuron: As it has very limited uses, we expect that exposure to linuron will be minimal. As such, we determined in the a.i. summary that linuron has a low likelihood of affecting PCEs. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Captan: As stated in the a.i. summary, we believe that captan has a low likelihood of causing a reduction in PCEs. Captan degrades very rapidly, so we do not anticipate much exposure. It is unlikely to reduce prey abundance or modify primary productivity or riparian vegetation. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Chlorothalonil: In the a.i. summary, we determined that chlorothalonil has a medium likelihood of degrading PCEs based on degradation of water quality. We anticipate frequent, low-level inputs will occur in highly agricultural or developed areas. There is a low degree of overlap between these land use categories and all PCEs. Based on the level of co-occurrence, we believe that there will not be an appreciable decline in the value of these PCEs and in the designated critical habitat of the SONCC coho.

Central California Coast (CCC) Coho Salmon

Individual watersheds have not been evaluated for their conservation value. Nevertheless, there is a distinct trend of increasing degradation in quality and quantity of all PCEs as the habitat progresses south through the species range along the Lost Coast to Navarro Point and the Santa Cruz Mountains. Spawning and incubation substrate and juvenile rearing habitat are generally degraded.

Much of the development is centered around San Francisco Bay, and developed and agricultural areas also occur in the Russian River watershed. The northern, undeveloped watersheds around the Navarro and Big Rivers are used by the majority of this species. Given these land uses, we expect the freshwater rearing PCE may experience reductions in water quality and prey abundance during allowable pesticide applications adjacent to freshwater systems.

2,4-D: In the a.i. summary at the beginning of the Integration and Synthesis section, we determined that 2,4-D has a medium likelihood of degrading PCEs based on effects to water quality and riparian vegetation. We expect that habitat in heavily agricultural and/or urban areas are more likely to experience frequent low-level inputs. While there is some overlap between developed land and rearing and migratory PCEs, we do not believe that the level of effect from this degree of overlap is sufficient to result in an appreciable decline in the value of these PCEs and in the designated critical habitat of the CCC coho.

Triclopyr BEE: As stated in the a.i. summary, triclopyr BEE is unlikely to result in reductions to PCEs. While applications to riparian zones could alter the community structure, that change could result in either a beneficial or detrimental effect to the habitat. As triclopyr BEE is often used in restoration projects, it is difficult to draw conclusions on all uses of this chemical. NMFS also believes that reductions in prey availability or instream primary productivity are unlikely. Therefore, there is a low likelihood of triclopyr BEE causing an appreciable decline in the conservation value of PCEs or critical habitat of the CCC coho.

Diuron: In the a.i. summary, we determined that diuron has a medium likelihood of reducing the conservation value of PCEs and critical habitat due to effects on water quality, primary productivity instream, and riparian vegetation. We expect that habitat in heavily agricultural areas is more likely to experience frequent low-level inputs from multiple sources. We are also concerned about exposure resulting from applications to rights-of-way. While there is some agriculture in the Russian River, we do not believe that the level of effect from this degree of overlap is sufficient to result in an appreciable decline. As such, we believe that there is a low likelihood of diuron causing an appreciable decline in the value of PCEs and in the designated critical habitat of the CCC coho.

Linuron: As it has very limited uses, we expect that exposure to linuron will be minimal. As such, we determined in the a.i. summary that linuron has a low likelihood of affecting PCEs. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Captan: As stated in the a.i. summary, we believe that captan has a low likelihood of causing a reduction in PCEs. Captan degrades very rapidly, so we do not anticipate much exposure. It is unlikely to reduce prey abundance or modify primary productivity or riparian vegetation. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Chlorothalonil: In the a.i. summary, we determined that chlorothalonil has a medium likelihood of degrading PCEs based on degradation of water quality. We anticipate frequent, low-level inputs will occur in highly agricultural or developed areas. We expect that these inputs will occur in the Russian River watershed, due to a higher concentration of turf uses and the prevalence of higher-use agricultural crops. Based on this level of co-occurrence and importance of this watershed, we believe that there may be an appreciable decline in the value of these PCEs and in the designated critical habitat of the CCC coho.

Ozette Lake Sockeye Salmon

The Ozette Lake watershed is of high conservation value. The entire circumference of the lake is within Olympic National Park. Ozette Lake and portions of three tributaries support spawning and rearing PCEs. Ozette River supports rearing and migration PCEs; its river mouth also provides estuarine habitat. Migration habitat is also affected by low water flow in summer and elevated water temperature which pose as a thermal barrier for migration.

Spawning habitat has been affected by the loss of tributary spawning areas, low water levels in summer, and vegetation and sediment that have reduced the quantity and suitability of beaches for spawning. The rearing PCE is degraded by excessive predation, competition with non-native species, and loss of rearing habitat. Migration habitat is affected by high water temperatures and low water flows in summer.

Ozette Lake is in a sparsely populated area, with less than 1% of land developed within the range of this ESU. Similarly, there is no cultivated cropland. However, salmonid habitat may be at risk of exposure form forestry-related uses. Land use is primarily forest with private, state, and federal ownership (86% forested, 13% open water, 1% developed land, 0% agriculture). The predominant pesticide use sites (*i.e.*, urban/residential and forestry) overlap with the Lake's freshwater tributaries. Thus, the greatest risk of exposure to freshwater PCEs are in tributary habitats. However, we do not expect a reduction in prey abundance within these tributaries. Although private residences along tributaries may have small, non-commercial crops for pesticide applications, it is unlikely that restricted use pesticides would be applied.

2,4-D: In the a.i. summary at the beginning of the Integration and Synthesis section, we determined that 2,4-D has a medium likelihood of degrading PCEs based on effects to water quality and riparian vegetation. We expect that habitat in heavily agricultural and/or urban areas are more likely to experience frequent low-level inputs. As there is limited agriculture and development in this ESU, there is a low likelihood of 2,4-D causing an appreciable decline in the value of these PCEs and in the designated critical habitat of the Ozette Lake sockeye.

Triclopyr BEE: As stated in the a.i. summary, triclopyr BEE is unlikely to result in reductions to PCEs. While applications to riparian zones could alter the community structure, that change could result in either a beneficial or detrimental effect to the habitat. As triclopyr BEE is often used in restoration projects, it is difficult to draw conclusions on all uses of this chemical. NMFS also believes that reductions in prey availability or instream primary productivity are unlikely. Therefore, there is a low likelihood of triclopyr BEE causing an appreciable decline in the conservation value of PCEs or critical habitat of the Ozette Lake sockeye.

Diuron: In the a.i. summary, we determined that diuron has a medium likelihood of reducing the conservation value of PCEs and critical habitat due to effects on water quality, primary productivity instream, and riparian vegetation. We expect that habitat in heavily agricultural areas is more likely to experience frequent low-level inputs from multiple sources. We are also concerned about exposure resulting from applications to rights-of-way. However, given the land use within this ESU, we do not believe much exposure will occur. As such, we believe that there is a low likelihood of diuron causing an appreciable decline in the value of these PCEs and in the designated critical habitat of the Ozette Lake sockeye.

Linuron: As it has very limited uses, we expect that exposure to linuron will be minimal. As such, we determined in the a.i. summary that linuron has a low likelihood of affecting PCEs. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Captan: As stated in the a.i. summary, we believe that captan has a low likelihood of causing a reduction in PCEs. Captan degrades very rapidly, so we do not anticipate much exposure. It is unlikely to reduce prey abundance or modify primary productivity or riparian vegetation. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Chlorothalonil: In the a.i. summary, we determined that chlorothalonil has a medium likelihood of degrading PCEs based on degradation of water quality. We anticipate frequent, low-level inputs will occur in highly agricultural or developed areas. There is a low degree of overlap between these land use categories and all PCEs. Based on the level of co-occurrence, we believe that there will not be an appreciable decline in the value of these PCEs and in the designated critical habitat of the Ozette Lake sockeye.

Snake River Sockeye Salmon

Conservation values of individual watersheds have not been reported. Nevertheless, all areas occupied and used by migrating SR sockeye are considered of high conservation value as this species is limited to a single lake within the SR basin.

The quality and quantity of rearing and migration PCEs have been reduced by land uses that disrupt access to foraging areas, increase the amount of fines in the stream substrate, and reduce instream cover. Water quality is impaired by a suite of anthropogenic pollutants which enter surface waters and riverine sediments from the headwaters of the Salmon River to the Columbia River estuary. The migration PCE is also affected by four dams in the SR basins that obstructs migration and increases mortality of downstream migrating juveniles. Given the migration distance traveled by this species, adequate passage conditions (water quality and quantity available at specific times) is critical.

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.

2,4-D: In the a.i. summary at the beginning of the Integration and Synthesis section, we determined that 2,4-D has a medium likelihood of degrading PCEs based on effects to water quality and riparian vegetation. We expect that habitat in heavily agricultural and/or urban areas are more likely to experience frequent low-level inputs. As there is limited agriculture and development in this ESU, there is a low likelihood of 2,4-D causing an appreciable decline in the value of these PCEs and in the designated critical habitat of the Snake River sockeye.

Triclopyr BEE: As stated in the a.i. summary, triclopyr BEE is unlikely to result in reductions to PCEs. While applications to riparian zones could alter the community structure, that change could result in either a beneficial or detrimental effect to the habitat. As triclopyr BEE is often used in restoration projects, it is difficult to draw conclusions on all uses of this chemical.

NMFS also believes that reductions in prey availability or instream primary productivity are unlikely. Therefore, there is a low likelihood of triclopyr BEE causing an appreciable decline in the conservation value of PCEs or critical habitat of the Snake River sockeye.

Diuron: In the a.i. summary, we determined that diuron has a medium likelihood of reducing the conservation value of PCEs and critical habitat due to effects on water quality, primary productivity instream, and riparian vegetation. We expect that habitat in heavily agricultural areas is more likely to experience frequent low-level inputs from multiple sources. We are also concerned about exposure resulting from applications to rights-of-way. However, given the land use within this ESU, we do not believe much exposure will occur. As such, we believe that there is a low likelihood of diuron causing an appreciable decline in the value of these PCEs and in the designated critical habitat of the Snake River sockeye.

Linuron: As it has very limited uses, we expect that exposure to linuron will be minimal. As such, we determined in the a.i. summary that linuron has a low likelihood of affecting PCEs. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Captan: As stated in the a.i. summary, we believe that captan has a low likelihood of causing a reduction in PCEs. Captan degrades very rapidly, so we do not anticipate much exposure. It is unlikely to reduce prey abundance or modify primary productivity or riparian vegetation. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Chlorothalonil: In the a.i. summary, we determined that chlorothalonil has a medium likelihood of degrading PCEs based on degradation of water quality. We anticipate frequent, low-level inputs will occur in highly agricultural or developed areas. There is a low degree of overlap between these land use categories and all PCEs. Based on the level of co-occurrence, we believe that there will not be an appreciable decline in the value of these PCEs and in the designated critical habitat of the Snake River sockeye.

Lower Columbia River Steelhead

Of 41 watersheds listed as critical habitat for LCR steelhead, 28 and 11 are of high and medium conservation value, respectively. Our GIS analysis indicates 21 and 26 high conservation watersheds are exposed to pesticides from agriculture and urban/residential land uses, respectively. Of the medium conservation watersheds, 11 and 10 are also exposed to pesticide applications from the above respective land uses. The two low conservation value watersheds are exposed to pesticides applied in both agricultural and urban settings.

The water quality of the rearing PCE within the lower portion and alluvial valleys of many watersheds has been degraded by agricultural runoff into tributaries reaches and the mainstem Columbia River. Consequently, invertebrate production in these aquatic systems is also affected. Elevated water temperature further prevents LCR steelhead from inhabiting 341.5 km of waters within its range.

2,4-D: In the a.i. summary at the beginning of the Integration and Synthesis section, we determined that 2,4-D has a medium likelihood of degrading PCEs based on effects to water quality and riparian vegetation. We expect that habitat in heavily agricultural and/or urban areas are more likely to experience frequent low-level inputs. While there is some overlap between developed land and rearing and migratory PCEs, we do not believe that the level of effect from this degree of overlap is sufficient to result in an appreciable decline in the value of these PCEs and in the designated critical habitat of the Lower Columbia River steelhead.

Triclopyr BEE: As stated in the a.i. summary, triclopyr BEE is unlikely to result in reductions to PCEs. While applications to riparian zones could alter the community structure, that change could result in either a beneficial or detrimental effect to the habitat. As triclopyr BEE is often used in restoration projects, it is difficult to draw conclusions on all uses of this chemical. NMFS also believes that reductions in prey availability or instream primary productivity are unlikely. Therefore, there is a low likelihood of triclopyr BEE causing an appreciable decline in the conservation value of PCES or critical habitat of the Lower Columbia River steelhead.

Diuron: In the a.i. summary, we determined that diuron has a medium likelihood of reducing the conservation value of PCEs and critical habitat due to effects on water quality, primary productivity instream, and riparian vegetation. We expect that habitat in heavily agricultural areas is more likely to experience frequent low-level inputs from multiple sources. We are also concerned about exposure resulting from applications to rights-of-way. Additionally, diuron's persistence in the water column indicates that areas downstream from the application site could be affected. In the case of this ESU, we expect that exposure may result from applications in the heavily agricultural Upper Willamette basin. The moderate degree of overlap between developed land and rearing and migratory PCEs along with the toxicity and persistence of diuron may lead to widespread effects to critical habitat. As such, we believe that there may be an appreciable decline in the value of these PCEs and in the designated critical habitat of the Lower Columbia River steelhead.

Linuron: As it has very limited uses, we expect that exposure to linuron will be minimal. As such, we determined in the a.i. summary that linuron has a low likelihood of affecting PCEs. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Captan: As stated in the a.i. summary, we believe that captan has a low likelihood of causing a reduction in PCEs. Captan degrades very rapidly, so we do not anticipate much exposure. It is unlikely to reduce prey abundance or modify primary productivity or riparian vegetation. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Chlorothalonil: In the a.i. summary, we determined that chlorothalonil has a medium likelihood of degrading PCEs based on degradation of water quality. We anticipate frequent, low-level inputs to occur in highly agricultural or developed areas. As with the species analysis, we have strong concerns regarding use on turf grass, particularly on golf courses. There is a moderate degree of overlap between these land use categories and rearing and migratory PCEs. Based on the level of co-occurrence, we believe that there will not be an appreciable decline in the value of these PCEs and in the designated critical habitat of the Lower Columbia River steelhead.

Upper Willamette River Steelhead

Of the watersheds assessed, 14 and 6 are of high and medium conservation value, respectively. Our GIS analysis indicates all high and medium conservation value watersheds are exposed to pesticide applications from agriculture and urban areas adjacent to UWR steelhead critical habitat. All 17 of the low conservation value watersheds are at risk of exposure to pesticides applied in agricultural and urban areas.

Existing water quality necessary for juvenile rearing within many watersheds have been impaired by pollutants in agricultural runoff. Consequently, invertebrate production for salmonids in several watersheds and in the mainstem Columbia River is affected. As several dams obstruct migrating fish along the migratory corridor, the migration PCE is also reduced by these features. Elevated water temperature further prevents UWR steelhead from inhabiting 1,668 km of waters within its range.

Given these uses, we expect the freshwater rearing PCE in floodplain habitats, rivers, and streams may experience reductions in water quality and prey abundance during allowable pesticide applications adjacent to these systems.

2,4-D: In the a.i. summary at the beginning of the Integration and Synthesis section, we determined that 2,4-D has a medium likelihood of degrading PCEs based on effects to water quality and riparian vegetation. We expect that habitat in heavily agricultural and/or urban areas are more likely to experience frequent low-level inputs. There is a high degree of overlap between developed land and rearing and migratory PCEs. Based on the prevalence of this co-occurrence, we believe that there may be an appreciable decline in the value of these PCEs and in the designated critical habitat of the Upper Willamette River steelhead.

Triclopyr BEE: As stated in the a.i. summary, triclopyr BEE is unlikely to result in reductions to PCEs. While applications to riparian zones could alter the community structure, that change could result in either a beneficial or detrimental effect to the habitat. As triclopyr BEE is often used in restoration projects, it is difficult to draw conclusions on all uses of this chemical. Those modifications would be site specific, and given the scope of this analysis we cannot draw a conclusion either way. NMFS also believes that reductions in prey availability or instream primary productivity are unlikely. Therefore, there is a low likelihood of triclopyr BEE causing an appreciable decline in the conservation value of PCES or critical habitat of the Upper Willamette River steelhead.

Diuron: In the a.i. summary, we determined that diuron has a medium likelihood of reducing the conservation value of PCEs and critical habitat due to effects on water quality, primary productivity instream, and riparian vegetation. We expect that habitat in heavily agricultural areas are more likely to experience frequent low-level inputs from multiple sources. There is a high degree of overlap between developed land and rearing and migratory PCEs. Based on the prevalence of this co-occurrence, we believe that there may be an appreciable decline in the value of these PCEs and in the designated critical habitat of the Upper Willamette River steelhead.

Linuron: As it has very limited uses, we expect that exposure to linuron will be minimal. As such, we determined in the a.i. summary that linuron has a low likelihood of affecting PCEs. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Captan: As stated in the a.i. summary, we believe that captan has a low likelihood of causing a reduction in PCEs. Captan degrades very rapidly, so we do not anticipate much exposure. It is unlikely to reduce prey abundance or modify primary productivity or riparian vegetation. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Chlorothalonil: In the a.i. summary, we determined that chlorothalonil has a medium likelihood of degrading PCEs based on degradation of water quality. We anticipate frequent, low-level inputs to occur in highly agricultural or developed areas. There is a high degree of overlap between these land use categories and rearing and migratory PCEs. Based on the prevalence of this co-occurrence, we believe that there may be an appreciable decline in the value of these PCEs and in the designated critical habitat of the Upper Willamette River steelhead.

Middle Columbia River Steelhead

Of the 106 assessed watersheds, 73 and 24 are of high and medium conservation value, respectively. The lower Columbia River rearing/migration corridor downstream of the spawning range is also of high conservation value. Our GIS analysis indicates 67 and 68 high conservation watersheds are exposed to pesticides from agriculture and urban areas, respectively. Of the medium conservation watersheds, 23 and 24 watersheds are also exposed to pesticide applications from the above respective land uses. All 9 of the low conservation value watersheds are at risk of exposure to pesticides applied in agricultural and urban areas.

The current condition of critical habitat for MCR steelhead is moderately degraded. The water quality attribute for the rearing PCE within many watersheds is reduced. Consequently, invertebrate production in these watersheds and in the mainstem Columbia River is also reduced. Loss of riparian vegetation to grazing has resulted in elevated water temperature in the John Day Basin. Elevated water temperature prevents MCR steelhead from inhabiting 3,727.9 km of waters within its range. In the Yakima River, 72 streams and river segments are also listed as impaired waters and 83% exceed temperature standards. As several dams obstruct fish along their migratory corridor, these features further degrade the migration PCE.

2,4-D: In the a.i. summary at the beginning of the Integration and Synthesis section, we determined that 2,4-D has a medium likelihood of degrading PCEs based on effects to water quality and riparian vegetation. We expect that habitat in heavily agricultural and/or urban areas are more likely to experience frequent low-level inputs. As there is limited agriculture and development in this ESU, there is a low likelihood of 2,4-D causing an appreciable decline in the value of these PCEs and in the designated critical habitat of the MCR steelhead.

Triclopyr BEE: As stated in the a.i. summary, triclopyr BEE is unlikely to result in reductions to PCEs. While applications to riparian zones could alter the community structure, that change could result in either a beneficial or detrimental effect to the habitat. As triclopyr BEE is often used in restoration projects, it is difficult to draw conclusions on all uses of this chemical. NMFS also believes that reductions in prey availability or instream primary productivity are unlikely. Therefore, there is a low likelihood of triclopyr BEE causing an appreciable decline in the conservation value of PCEs or critical habitat of the MCR steelhead.

Diuron: In the a.i. summary, we determined that diuron has a medium likelihood of reducing the conservation value of PCEs and critical habitat due to effects on water quality, primary productivity instream, and riparian vegetation. We expect that habitat in heavily agricultural areas is more likely to experience frequent low-level inputs from multiple sources. We are also concerned about exposure resulting from applications to rights-of-way. However, given the land use within this ESU, we do not believe much exposure will occur. As such, we believe that there is a low likelihood of diuron causing an appreciable decline in the value of these PCEs and in the designated critical habitat of the MCR steelhead.

Linuron: As it has very limited uses, we expect that exposure to linuron will be minimal. As such, we determined in the a.i. summary that linuron has a low likelihood of affecting PCEs. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Captan: As stated in the a.i. summary, we believe that captan has a low likelihood of causing a reduction in PCEs. Captan degrades very rapidly, so we do not anticipate much exposure. It is unlikely to reduce prey abundance or modify primary productivity or riparian vegetation. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Chlorothalonil: In the a.i. summary, we determined that chlorothalonil has a medium likelihood of degrading PCEs based on degradation of water quality. We anticipate frequent, low-level inputs to occur in highly agricultural or developed areas. As with the species analysis, we have strong concerns regarding use on turf grass, particularly on golf courses. There is a moderate degree of overlap between these land use categories and rearing and migratory PCEs. Based on the level of co-occurrence, we believe that there will not be an appreciable decline in the value of these PCEs and in the designated critical habitat of the MCR steelhead.

Upper Columbia River Steelhead

Of the 41 watersheds occupied by UCR steelhead, 31 and 7 are of high and medium conservation value, respectively. The lower Columbia River rearing/migration corridor downstream of the species' spawning range is also of high conservation value. Our GIS analysis indicates 28 and 31 high conservation watersheds are exposed to pesticides from agriculture and urban areas, respectively. All seven medium and all three low conservation value watersheds are exposed to pesticide applications from the above land uses.

The current condition of UCR steelhead critical habitat is moderately degraded. Habitat quality in tributary streams range from excellent to poor. Water quality for the rearing PCEs within many watersheds has been reduced from agriculture runoff. Consequently, invertebrate production in several watersheds and in the mainstem Columbia River is also reduced. Several dams obstruct fish migrating through the migratory corridor and further impact the migration PCEs. There is some agriculture in the spawning and rearing areas in the Wenatchee, Methow, and Okenogan watersheds. Intense agriculture occurs in the Upper Columbia Irrigation District within the Entiat watershed. The water is heavily used and re-used for irrigation.

2,4-D: In the a.i. summary at the beginning of the Integration and Synthesis section, we determined that 2,4-D has a medium likelihood of degrading PCEs based on effects to water quality and riparian vegetation. We expect that habitat in heavily agricultural and/or urban areas are more likely to experience frequent low-level inputs. As there is limited agriculture and development in this ESU, there is a low likelihood of 2,4-D causing an appreciable decline in the value of these PCEs and in the designated critical habitat of the Upper Columbia River steelhead.

Triclopyr BEE: As stated in the a.i. summary, triclopyr BEE is unlikely to result in reductions to PCEs. While applications to riparian zones could alter the community structure, that change could result in either a beneficial or detrimental effect to the habitat. As triclopyr BEE is often used in restoration projects, it is difficult to draw conclusions on all uses of this chemical. NMFS also believes that reductions in prey availability or instream primary productivity are unlikely. Therefore, there is a low likelihood of triclopyr BEE causing an appreciable decline in the conservation value of PCEs or critical habitat of the Upper Columbia River steelhead.

Diuron: In the a.i. summary, we determined that diuron has a medium likelihood of reducing the conservation value of PCEs and critical habitat due to effects on water quality, primary productivity instream, and riparian vegetation. We expect that habitat in heavily agricultural areas is more likely to experience frequent low-level inputs from multiple sources. We are also concerned about exposure resulting from applications to rights-of-way. However, given the land use within this ESU, we do not believe much exposure will occur. As such, we believe that there is a low likelihood of diuron causing an appreciable decline in the value of these PCEs and in the designated critical habitat of the Upper Columbia River steelhead.

Linuron: As it has very limited uses, we expect that exposure to linuron will be minimal. As such, we determined in the a.i. summary that linuron has a low likelihood of affecting PCEs. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Captan: As stated in the a.i. summary, we believe that captan has a low likelihood of causing a reduction in PCEs. Captan degrades very rapidly, so we do not anticipate much exposure. It is unlikely to reduce prey abundance or modify primary productivity or riparian vegetation. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Chlorothalonil: In the a.i. summary, we determined that chlorothalonil has a medium likelihood of degrading PCEs based on degradation of water quality. We anticipate frequent, low-level inputs will occur in highly agricultural or developed areas. There is a low degree of overlap between these land use categories and all PCEs. Based on the level of co-occurrence, we believe that there will not be an appreciable decline in the value of these PCEs and in the designated critical habitat of the Upper Columbia River steelhead.

Snake River Basin Steelhead

Of the watersheds assessed, 229 and 41 are of high and medium conservation value, respectively. The Columbia River migration corridor is also of high conservation value. Our GIS analysis indicates 163 and 99 high conservation watersheds are exposed to pesticides from agriculture and urban areas, respectively. Of the medium conservation watersheds, 34 and 28 are also exposed to pesticide applications from the above land uses. Of the low conservation value watersheds, 12 are exposed to pesticides applied in agricultural areas, while 9 are exposed to those applied in urban areas.

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.

2,4-D: In the a.i. summary at the beginning of the Integration and Synthesis section, we determined that 2,4-D has a medium likelihood of degrading PCEs based on effects to water quality and riparian vegetation. We expect that habitat in heavily agricultural and/or urban areas are more likely to experience frequent low-level inputs. As there is limited agriculture and development in this ESU, there is a low likelihood of 2,4-D causing an appreciable decline in the value of these PCEs and in the designated critical habitat of the SR steelhead.

Triclopyr BEE: As stated in the a.i. summary, triclopyr BEE is unlikely to result in reductions to PCEs. While applications to riparian zones could alter the community structure, that change could result in either a beneficial or detrimental effect to the habitat. As triclopyr BEE is often used in restoration projects, it is difficult to draw conclusions on all uses of this chemical. NMFS also believes that reductions in prey availability or instream primary productivity are unlikely. Therefore, there is a low likelihood of triclopyr BEE causing an appreciable decline in the conservation value of PCEs or critical habitat of the SR steelhead.

Diuron: In the a.i. summary, we determined that diuron has a medium likelihood of reducing the conservation value of PCEs and critical habitat due to effects on water quality, primary productivity instream, and riparian vegetation. We expect that habitat in heavily agricultural areas is more likely to experience frequent low-level inputs from multiple sources. We are also concerned about exposure resulting from applications to rights-of-way. However, given the land use within this ESU, we do not believe much exposure will occur. As such, we believe that there is a low likelihood of diuron causing an appreciable decline in the value of these PCEs and in the designated critical habitat of the SR steelhead.

Linuron: As it has very limited uses, we expect that exposure to linuron will be minimal. As such, we determined in the a.i. summary that linuron has a low likelihood of affecting PCEs. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Captan: As stated in the a.i. summary, we believe that captan has a low likelihood of causing a reduction in PCEs. Captan degrades very rapidly, so we do not anticipate much exposure. It is unlikely to reduce prey abundance or modify primary productivity or riparian vegetation. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Chlorothalonil: In the a.i. summary, we determined that chlorothalonil has a medium likelihood of degrading PCEs based on degradation of water quality. We anticipate frequent, low-level inputs will occur in highly agricultural or developed areas. There is a low degree of overlap between these land use categories and all PCEs. Based on the level of co-occurrence, we believe that there will not be an appreciable decline in the value of these PCEs and in the designated critical habitat of the SR steelhead.

Northern California Steelhead

Of the 50 assessed watersheds, 27 and 14 are of high and medium conservation value, respectively. Two estuarine habitat areas used for rearing and migration (Humboldt Bay and the Eel River Estuary) are also of high conservation value. Our GIS analysis indicates 10 and 27 high conservation watersheds are exposed to agriculture and urban areas, respectively. Of the medium conservation watersheds, 2 and 14 are also exposed to pesticide applications from the same above land uses, respectively. Of the low watersheds, all nine may be exposed to pesticides applied in urban areas, while only one is at risk of exposure to pesticides applied in agricultural areas.

The current condition of critical habitat for NC steelhead is moderately degraded. Removal of riparian vegetation within portions of its range promotes elevated water temperature and consequently affects the rearing PCE in freshwater and estuaries. Spawning PCE attributes such as the quality of substrate supporting spawning, incubation, and larval development are degraded by silt and sediment fines in the spawning gravel. Access to tributaries in many watersheds is affected by bridges, culverts, and forest road construction. Consequently, these uses reduce the function of the migration PCE for adults.

There are few areas of concentrated agriculture and most appear to be hay/pasture and are concentrated in the Lower Eel watershed and some of the other coastal valleys. Development is concentrated primarily near Eureka, on the coast in the Mad River and Redwood Creek watersheds. Much of the land area in this DPS is heavily forested, and there is a number of state and national parks.

2,4-D: In the a.i. summary at the beginning of the Integration and Synthesis section, we determined that 2,4-D has a medium likelihood of degrading PCEs based on effects to water quality and riparian vegetation. We expect that habitat in heavily agricultural and/or urban areas are more likely to experience frequent low-level inputs. As there is limited agriculture and development in this ESU, there is a low likelihood of 2,4-D causing an appreciable decline in the value of these PCEs and in the designated critical habitat of the NC steelhead.

Triclopyr BEE: As stated in the a.i. summary, triclopyr BEE is unlikely to result in reductions to PCEs. While applications to riparian zones could alter the community structure, that change could result in either a beneficial or detrimental effect to the habitat. As triclopyr BEE is often used in restoration projects, it is difficult to draw conclusions on all uses of this chemical. NMFS also believes that reductions in prey availability or instream primary productivity are unlikely. Therefore, there is a low likelihood of triclopyr BEE causing an appreciable decline in the conservation value of PCEs or critical habitat of the NC steelhead.

Diuron: In the a.i. summary, we determined that diuron has a medium likelihood of reducing the conservation value of PCEs and critical habitat due to effects on water quality, primary productivity instream, and riparian vegetation. We expect that habitat in heavily agricultural areas is more likely to experience frequent low-level inputs from multiple sources. We are also concerned about exposure resulting from applications to rights-of-way. However, given the land use within this ESU, we do not believe much exposure will occur. As such, we believe that there is a low likelihood of diuron causing an appreciable decline in the value of these PCEs and in the designated critical habitat of the NC steelhead.

Linuron: As it has very limited uses, we expect that exposure to linuron will be minimal. As such, we determined in the a.i. summary that linuron has a low likelihood of affecting PCEs. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Captan: As stated in the a.i. summary, we believe that captan has a low likelihood of causing a reduction in PCEs. Captan degrades very rapidly, so we do not anticipate much exposure. It is unlikely to reduce prey abundance or modify primary productivity or riparian vegetation. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Chlorothalonil: In the a.i. summary, we determined that chlorothalonil has a medium likelihood of degrading PCEs based on degradation of water quality. We anticipate frequent, low-level inputs will occur in highly agricultural or developed areas. There is a low degree of overlap between these land use categories and all PCEs. Based on the level of co-occurrence, we believe that there will not be an appreciable decline in the value of these PCEs and in the designated critical habitat of the NC steelhead.

Central California Coast (CCC) Steelhead

Of 47 occupied watersheds, 19 and 15 are of high and medium conservation value, respectively. Our GIS analysis indicates 12 and 15 high conservation watersheds are exposed to pesticide applications from agriculture and urban areas, respectively. Of the medium conservation watersheds, 8 and 13 are also exposed to the above land uses areas, respectively. Of the low conservation watersheds, 9 are exposed to agricultural applications, while 15 are exposed to applications in urban areas. Throughout the species' range, habitat conditions and quality have been degraded by a lack of channel complexity, eroded banks, turbid and contaminated water, low summer flow and high water temperatures, multiple contaminants found at toxic levels, and restricted access to cooler head waters from migration barriers.

The current condition of designated critical habitat for CCC steelhead is poor. The spawning PCE is impacted by sediment fines in the spawning gravel, which limits the production of aquatic stream insects adapted to running water. Elevated water temperature and impaired water quality have further reduced the quality, quantity, and function of the rearing PCE within most streams.

High densities of crop farming occur throughout the San Joaquin Basin, the Delta, and along the lower Sacramento River. Agriculture also occurs in the Russian River valley. Most of the watersheds in this DPS are heavily developed, and/or have intensive agriculture in the river valley. Given these land uses, rearing and migration PCEs in small freshwater tributaries and floodplains and the San Francisco-San Pablo-Suisan Bay estuarine complex may experience reductions in water quality and prey abundance during allowable pesticide applications adjacent to these systems.

2,4-D: In the a.i. summary at the beginning of the Integration and Synthesis section, we determined that 2,4-D has a medium likelihood of degrading PCEs based on effects to water quality and riparian vegetation. We expect that habitat in heavily agricultural and/or urban areas are more likely to experience frequent low-level inputs. There is a high degree of overlap between developed land and rearing and migratory PCEs. Based on the prevalence of this co-occurrence, we believe that there may be an appreciable decline in the value of these PCEs and in the designated critical habitat of the CCC steelhead.

Triclopyr BEE: As stated in the a.i. summary, triclopyr BEE is unlikely to result in reductions to PCEs. While applications to riparian zones could alter the community structure, that change could result in either a beneficial or detrimental effect to the habitat. As triclopyr BEE is often used in restoration projects, it is difficult to draw conclusions on all uses of this chemical. Those modifications would be site specific, and given the scope of this analysis we cannot draw a conclusion either way. NMFS also believes that reductions in prey availability or instream primary productivity are unlikely. Therefore, there is a low likelihood of triclopyr BEE causing an appreciable decline in the conservation value of PCES or critical habitat of the CCC steelhead.

Diuron: In the a.i. summary, we determined that diuron has a medium likelihood of reducing the conservation value of PCEs and critical habitat due to effects on water quality, primary productivity instream, and riparian vegetation. We expect that habitat in heavily agricultural areas are more likely to experience frequent low-level inputs from multiple sources. There is a high degree of overlap between developed land and rearing and migratory PCEs. Due to diuron’s persistence, we are also concerned about the effects applications upstream in the central valley. Based on the prevalence of this co-occurrence, we believe that there may be an appreciable decline in the value of these PCEs and in the designated critical habitat of the CCC steelhead.

Linuron: As it has very limited uses, we expect that exposure to linuron will be minimal. As such, we determined in the a.i. summary that linuron has a low likelihood of affecting PCEs. Therefore, we do not expect that the conservation value of PCES and critical habitat will be appreciably reduced.

Captan: As stated in the a.i. summary, we believe that captan has a low likelihood of causing a reduction in PCEs. Captan degrades very rapidly, so we do not anticipate much exposure. It is unlikely to reduce prey abundance or modify primary productivity or riparian vegetation. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Chlorothalonil: In the a.i. summary, we determined that chlorothalonil has a medium likelihood of degrading PCEs based on degradation of water quality. We anticipate frequent, low-level inputs to occur in highly agricultural or developed areas. There is a high degree of overlap between these land use categories and rearing and migratory PCEs. Based on the prevalence of this co-occurrence, we believe that there may be an appreciable decline in the value of these PCEs and in the designated critical habitat of the CCC steelhead.

California Central Valley (CCV) Steelhead

Of 67 occupied watersheds, 37 and 18 are of high and medium conservation value, respectively. Our GIS analysis indicates 24 and 37 high conservation watersheds are exposed to pesticide applications from agriculture and urban areas, respectively. Of the medium conservation watersheds, 14 and 17 watersheds are exposed to pesticide applications from the above land uses, respectively. Of the low conservation watersheds, 12 are exposed to applications in urban areas, while 5 are exposed to urban applications.

The current condition of CCV steelhead critical habitat is degraded and does not function well for ensuring species recovery. The Sacramento-San Joaquin River Delta serves little function for juvenile CCV steelhead rearing and their physiological transition to salt water. Water flow and temperature, especially during the summer months affect the condition of the spawning PCE in floodplains and flood bypasses. The rearing PCE is degraded by channelized, leveed, and riprapped river reaches and sloughs in the Sacramento-San Joaquin system. Stream channels commonly have elevated water temperature. The current condition of migration corridors is poor. Both migration and rearing PCEs are affected by dense urbanization and agriculture along the mainstems and in the Delta which contribute to reduced water quality from contaminants in runoff. The RBDD gates obstruct migrating juveniles and adults. State and federal government pumps and associated fish facilities alter flow in the Delta and consequently obstruct migrations along the migratory corridor.

Heavy uses of agricultural pesticides and the high probably of mixtures increase the likelihood of negative effects on PCEs and critical habitat. As there is a continuous run of steelhead throughout the year, the conditions of the rearing PCE in a variety of habitat are important for this DPS. Given these land uses, freshwater rearing and migration PCEs in the Sacramento River, the Delta, tributaries, tidal and non-tidal marshes, and other shallow areas in the Delta may experience reductions in water quality and prey abundance during allowable pesticide applications adjacent to these systems.

2,4-D: In the a.i. summary at the beginning of the Integration and Synthesis section, we determined that 2,4-D has a medium likelihood of degrading PCEs based on effects to water quality and riparian vegetation. We expect that habitat in heavily agricultural and/or urban areas are more likely to experience frequent low-level inputs. There is a high degree of overlap between developed land and rearing and migratory PCEs. Based on the prevalence of this co-occurrence, we believe that there may be an appreciable decline in the value of these PCEs and in the designated critical habitat of the CCV steelhead.

Triclopyr BEE: As stated in the a.i. summary, triclopyr BEE is unlikely to result in reductions to PCEs. While applications to riparian zones could alter the community structure, that change could result in either a beneficial or detrimental effect to the habitat. As triclopyr BEE is often used in restoration projects, it is difficult to draw conclusions on all uses of this chemical. Those modifications would be site specific, and given the scope of this analysis we cannot draw a conclusion either way. NMFS also believes that reductions in prey availability or instream primary productivity are unlikely. Therefore, there is a low likelihood of triclopyr BEE causing an appreciable decline in the conservation value of PCEs or critical habitat of the CCV steelhead.

Diuron: In the a.i. summary, we determined that diuron has a medium likelihood of reducing the conservation value of PCEs and critical habitat due to effects on water quality, primary productivity instream, and riparian vegetation. We expect that habitat in heavily agricultural areas are more likely to experience frequent low-level inputs from multiple sources. There is a high degree of overlap between developed land and rearing and migratory PCEs. Based on the prevalence of this co-occurrence, we believe that there may be an appreciable decline in the value of these PCEs and in the designated critical habitat of the CCV steelhead.

Linuron: As it has very limited uses, we expect that exposure to linuron will be minimal. As such, we determined in the a.i. summary that linuron has a low likelihood of affecting PCEs. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Captan: As stated in the a.i. summary, we believe that captan has a low likelihood of causing a reduction in PCEs. Captan degrades very rapidly, so we do not anticipate much exposure. It is unlikely to reduce prey abundance or modify primary productivity or riparian vegetation. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Chlorothalonil: In the a.i. summary, we determined that chlorothalonil has a medium likelihood of degrading PCEs based on degradation of water quality. We anticipate frequent, low-level inputs to occur in highly agricultural or developed areas. There is a high degree of overlap between these land use categories and rearing and migratory PCEs. Based on the prevalence of this co-occurrence, we believe that there may be an appreciable decline in the value of these PCEs and in the designated critical habitat of the CCV steelhead.

South-Central California Coast (S-CCC) Steelhead

Of 29 occupied watersheds, 12 and 11 are of high and medium conservation value, respectively. Our GIS analysis indicates all high conservation watersheds are exposed to pesticide applications from agriculture and urban areas. Of the medium conservation watersheds, 9 and 11 watersheds are exposed to pesticide applications from agriculture and urban areas, respectively. All 6 of the low conservation value watersheds are at risk of exposure to pesticides applied in agricultural and urban areas.

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 there are areas of intense agriculture in the Pajaro watershed as well. Areas higher in the Salinas and Pajaro watersheds and along some of the coastal areas are less affected.

2,4-D: In the a.i. summary at the beginning of the Integration and Synthesis section, we determined that 2,4-D has a medium likelihood of degrading PCEs based on effects to water quality and riparian vegetation. We expect that habitat in heavily agricultural and/or urban areas are more likely to experience frequent low-level inputs. While there is some overlap between developed land and rearing and migratory PCEs, we do not believe that the level of effect from this degree of overlap is sufficient to result in an appreciable decline in the value of these PCEs and in the designated critical habitat of the S-CCC steelhead.

Triclopyr BEE: As stated in the a.i. summary, triclopyr BEE is unlikely to result in reductions to PCEs. While applications to riparian zones could alter the community structure, that change could result in either a beneficial or detrimental effect to the habitat. As triclopyr BEE is often used in restoration projects, it is difficult to draw conclusions on all uses of this chemical. NMFS also believes that reductions in prey availability or instream primary productivity are unlikely. Therefore, there is a low likelihood of triclopyr BEE causing an appreciable decline in the conservation value of PCEs or critical habitat of the S-CCC steelhead.

Diuron: In the a.i. summary, we determined that diuron has a medium likelihood of reducing the conservation value of PCEs and critical habitat due to effects on water quality, primary productivity instream, and riparian vegetation. We expect that habitat in heavily agricultural areas is more likely to experience frequent low-level inputs from multiple sources. We are also concerned about exposure resulting from applications to rights-of-way. While there is a moderate amount of development within this DPS, we do not believe that the level of effect from this degree of overlap is sufficient to result in an appreciable decline. As such, we believe that there is a low likelihood of diuron causing an appreciable decline in the value of PCEs and in the designated critical habitat of the S-CCC steelhead.

Linuron: As it has very limited uses, we expect that exposure to linuron will be minimal. As such, we determined in the a.i. summary that linuron has a low likelihood of affecting PCEs. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Captan: As stated in the a.i. summary, we believe that captan has a low likelihood of causing a reduction in PCEs. Captan degrades very rapidly, so we do not anticipate much exposure. It is unlikely to reduce prey abundance or modify primary productivity or riparian vegetation. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Chlorothalonil: In the a.i. summary, we determined that chlorothalonil has a medium likelihood of degrading PCEs based on degradation of water quality. We anticipate frequent, low-level inputs will occur in highly agricultural or developed areas. Agriculturally, the area is known for lettuces, strawberries, cut flowers, and vineyards. Due to a greater concentration of these higher-use agricultural crops, we expect that frequent, low-level inputs will occur in the Salinas watershed. Based on this co-occurrence, we believe that there may be an appreciable decline in the value of these PCEs and in the designated critical habitat of the S-CCC steelhead.

Southern California (SC) Steelhead

Of 29 freshwater and estuarine watersheds, 21 and 5 are of high and medium conservation value, respectively. Our GIS analysis indicates 15 and 21 high conservation watersheds are exposed to pesticide applications from agriculture and urban areas, respectively. Of the medium conservation watersheds, all five watersheds are exposed to pesticide applications from the same above land uses. All three low conservation value watersheds are exposed to pesticides used in urban areas, and two are exposed to those applied in agricultural areas.

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.

2,4-D: In the a.i. summary at the beginning of the Integration and Synthesis section, we determined that 2,4-D has a medium likelihood of degrading PCEs based on effects to water quality and riparian vegetation. We expect that habitat in heavily agricultural and/or urban areas are more likely to experience frequent low-level inputs. While there is some overlap between developed land and rearing and migratory PCEs, we do not believe that the level of effect from this degree of overlap is sufficient to result in an appreciable decline in the value of these PCEs and in the designated critical habitat of the SC steelhead.

Triclopyr BEE: As stated in the a.i. summary, triclopyr BEE is unlikely to result in reductions to PCEs. While applications to riparian zones could alter the community structure, that change could result in either a beneficial or detrimental effect to the habitat. As triclopyr BEE is often used in restoration projects, it is difficult to draw conclusions on all uses of this chemical. NMFS also believes that reductions in prey availability or instream primary productivity are unlikely. Therefore, there is a low likelihood of triclopyr BEE causing an appreciable decline in the conservation value of PCEs or critical habitat of the SC steelhead.

Diuron: In the a.i. summary, we determined that diuron has a medium likelihood of reducing the conservation value of PCEs and critical habitat due to effects on water quality, primary productivity instream, and riparian vegetation. We expect that habitat in heavily agricultural areas is more likely to experience frequent low-level inputs from multiple sources. We are also concerned about exposure resulting from applications to rights-of-way. While the Los Angeles area is heavily developed, there is little overlap between the developed areas and designated critical habitat. Therefore, we do not believe this degree of overlap is sufficient to result in an appreciable decline. As such, we believe that there is a low likelihood of diuron causing an appreciable decline in the value of PCEs and in the designated critical habitat of the SC steelhead.

Linuron: As it has very limited uses, we expect that exposure to linuron will be minimal. As such, we determined in the a.i. summary that linuron has a low likelihood of affecting PCEs. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Captan: As stated in the a.i. summary, we believe that captan has a low likelihood of causing a reduction in PCEs. Captan degrades very rapidly, so we do not anticipate much exposure. It is unlikely to reduce prey abundance or modify primary productivity or riparian vegetation. Therefore, we do not expect that the conservation value of PCEs and critical habitat will be appreciably reduced.

Chlorothalonil: In the a.i. summary, we determined that chlorothalonil has a medium likelihood of degrading PCEs based on degradation of water quality. We anticipate frequent, low-level inputs will occur in highly agricultural or developed areas. Due to a greater concentration of turf uses and moderate amount of agriculture, we expect that frequent, low-level inputs will occur. Based on this co-occurrence, we believe that there may be an appreciable decline in the value of these PCEs and in the designated critical habitat of the SC steelhead.

Table 147. Appreciable reduction in conservation value of critical habitat.

Species	ESU/DPS	Herbicides				Fungicides	
		2,4-D	Triclopyr BEE	Diuron	Linuron	Captan	Chlorothalonil
Chinook	Puget Sound	No	No	Yes	No	No	Yes
	Lower Columbia River	No	No	Yes	No	No	No
	Upper Columbia River Spring - Run	No	No	No	No	No	No
	Snake River Fall - Run	No	No	No	No	No	No
	Snake River Spring/Summer - Run	No	No	No	No	No	No
	Upper Willamette River	Yes	No	Yes	No	No	Yes
	California Coastal	No	No	No	No	No	No
	Central Valley Spring - Run	Yes	No	Yes	No	No	Yes
	Sacramento River Winter - Run	Yes	No	Yes	No	No	Yes
Chum	Hood Canal Summer - Run	No	No	No	No	No	No
	Columbia River	No	No	No	No	No	No
Coho	Lower Columbia River	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*
	Oregon Coast	No	No	No	No	No	No
	Southern Oregon and Northern California Coast	No	No	No	No	No	No
	Central California Coast	No	No	No	No	No	Yes
Sockeye	Ozette Lake	No	No	No	No	No	No
	Snake River	No	No	No	No	No	No
Steelhead	Puget Sound	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*
	Lower Columbia River	No	No	Yes	No	No	No
	Upper Willamette River	Yes	No	Yes	No	No	Yes
	Middle Columbia River	No	No	No	No	No	No
	Upper Columbia River	No	No	No	No	No	No
	Snake River	No	No	No	No	No	No
	Northern California	No	No	No	No	No	No
	Central California Coast	Yes	No	Yes	No	No	Yes
	California Central Valley	Yes	No	Yes	No	No	Yes
	South-Central California Coast	No	No	No	No	No	Yes
	Southern California	No	No	No	No	No	Yes

Conclusion

In the *Integration and Synthesis of Effects to Listed Species* section, we described NMFS' assessment of the likelihood of negative effects posed to the survival and recovery of listed Pacific salmonids as a result of EPA's registration of 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil. Conclusions in this final opinion include jeopardy and adverse modification determinations for chlorothalonil and adverse modification determinations for 2,4-D that were revised after the March 1, 2011 draft opinion . Changes in the chlorothalonil determinations were based on further analysis of turf use data NMFS received on February 14, 2011, as well as the revised environmental fate parameters from the Drinking Water Assessment received on April 26, 2011(EPA 2011). Revisions to the 2,4-D adverse modification determinations were based on additional analysis of use patterns of aquatic applications and uses related to restoration activities.

The likelihood of effects assigned to each ESU/DPS for each a.i. reflects NMFS' evaluation of the likelihood that a compound will cause reductions in species' viability. Direct water applications of the 2,4-D BEE are anticipated to kill salmonids if applied when salmonids are present. Because of the lack of restrictions on where and when direct water applications can occur, we believe the likelihood for an appreciable reduction in reproduction, numbers, or distribution at the population level is medium for all populations in all ESUs/DPSs for 2,4-D BEE. The BEE form is the only ester currently registered for direct water application. Given differences in the fate and toxicity of the 2,4-D acids and amines, we do not expect their use to kill salmonids. Based on the 2,4-D BEE authorization, we conclude that the registration of 2,4-D is likely to jeopardize all listed salmonid ESUs/DPSs.

We expect that triclopyr BEE, diuron and chlorothalonil will have a negative impact on most listed salmonids. However, any mortality events or sublethal effects caused by these a.i.s are not expected to affect a large enough portion of the populations to reduce the species' viability. Therefore, we conclude triclopyr BEE, diuron and chlorothalonil are not likely to jeopardize any listed ESU/DPS (Table 148). Linuron and captan were not likely to reduce the reproduction, numbers or distribution of any listed salmonid ESUs or DPSs, we conclude they are not likely to jeopardize any listed salmonids.

The Integration and Synthesis of Effects to Designated Critical Habitat section described NMFS' determination as to whether the proposed action will likely result in an appreciable decline or reduction in the conservation value of critical habitat the PCES that comprise that critical habitat for ESA-listed species. This biological opinion evaluated the PCE attributes to determine the likelihood of reducing the quality of spawning, rearing, migration, and estuarine habitat. We expect the stressors of the action will primarily affect water quality, prey and plant-based components of the critical habitat. These determinations translate directly to whether or not we expect the a.i. is likely to destroy or adversely modify the critical habitat of each ESU or DPS. We do not expect the registration of triclopyr BEE, linuron, or captan is likely to result in the destruction or adverse modification critical habitat of any listed salmonids. We do expect that registered uses of 2,4-D, diuron, and chlorothalonil is likely to adversely modify the critical habitat of some listed Pacific salmonids. Final determinations for the adverse modification of critical habitat are given below in Table 149.

Table 148. Jeopardy determinations for a.i.s.

Species	ESU	Herbicides				Fungicides	
		2,4-D[14]	Triclopyr BEE	Diuron	Linuron	Captan	Chlorothalonil
Chinook	Puget Sound	**Jeopardy**	No	No	No	No	No
	Lower Columbia River	**Jeopardy**	No	No	No	No	No
	Upper Columbia River Spring - Run	**Jeopardy**	No	No	No	No	No
	Snake River Fall - Run	**Jeopardy**	No	No	No	No	No
	Snake River Spring/Summer - Run	**Jeopardy**	No	No	No	No	No
	Upper Willamette River	**Jeopardy**	No	No	No	No	No
	California Coastal	**Jeopardy**	No	No	No	No	No
	Central Valley Spring - Run	**Jeopardy**	No	No	No	No	No
	Sacramento River Winter - Run	**Jeopardy**	No	No	No	No	No
Chum	Hood Canal Summer - Run	**Jeopardy**	No	No	No	No	No
	Columbia River	**Jeopardy**	No	No	No	No	No
Coho	Lower Columbia River	**Jeopardy**	No	No	No	No	No
	Oregon Coast	**Jeopardy**	No	No	No	No	No
	Southern Oregon and Northern California Coast	**Jeopardy**	No	No	No	No	No
	Central California Coast	**Jeopardy**	No	No	No	No	No
Sockeye	Ozette Lake	**Jeopardy**	No	No	No	No	No
	Snake River	**Jeopardy**	No	No	No	No	No
Steelhead	Puget Sound	**Jeopardy**	No	No	No	No	No
	Lower Columbia River	**Jeopardy**	No	No	No	No	No
	Upper Willamette River	**Jeopardy**	No	No	No	No	No
	Middle Columbia River	**Jeopardy**	No	No	No	No	No
	Upper Columbia River	**Jeopardy**	No	No	No	No	No
	Snake River	**Jeopardy**	No	No	No	No	No
	Northern California	**Jeopardy**	No	No	No	No	No
	Central California Coast	**Jeopardy**	No	No	No	No	No
	California Central Valley	**Jeopardy**	No	No	No	No	No
	South-Central California Coast	**Jeopardy**	No	No	No	No	No
	Southern California	**Jeopardy**	No	No	No	No	No

[14] Aquatic applications of 2,4-D BEE for the control of aquatic weeds weighed heavily in NMFS' determination of Jeopardy for this a.i.

Table 149. Adverse modifiction determinations.

Species	ESU	Herbicides				Fungicides	
		2,4-D	Triclopyr BEE	Diuron	Linuron	Captan	Chlorothalonil
Chinook	Puget Sound	No	No	**Ad Mod**	No	No	**Ad Mod**
	Lower Columbia River	No	No	**Ad Mod**	No	No	No
	Upper Columbia River Spring - Run	No	No	No	No	No	No
	Snake River Fall - Run	No	No	No	No	No	No
	Snake River Spring/Summer - Run	No	No	No	No	No	No
	Upper Willamette River	**Ad Mod**	No	**Ad Mod**	No	No	**Ad Mod**
	California Coastal	No	No	No	No	No	No
	Central Valley Spring - Run	**Ad Mod**	No	**Ad Mod**	No	No	**Ad Mod**
	Sacramento River Winter - Run	**Ad Mod**	No	**Ad Mod**	No	No	**Ad Mod**
Chum	Hood Canal Summer - Run	No	No	No	No	No	No
	Columbia River	No	No	No	No	No	No
Coho	Lower Columbia River	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*
	Oregon Coast	No	No	No	No	No	No
	Southern Oregon and Northern California Coast	No	No	No	No	No	No
	Central California Coast	No	No	No	No	No	**Ad Mod**
Sockeye	Ozette Lake	No	No	No	No	No	No
	Snake River	No	No	No	No	No	No
Steelhead	Puget Sound	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*
	Lower Columbia River	No	No	**Ad Mod**	No	No	No
	Upper Willamette River	**Ad Mod**	No	**Ad Mod**	No	No	**Ad Mod**
	Middle Columbia River	No	No	No	No	No	No
	Upper Columbia River	No	No	No	No	No	No
	Snake River	No	No	No	No	No	No
	Northern California	No	No	No	No	No	No
	Central California Coast	**Ad Mod**	No	**Ad Mod**	No	No	**Ad Mod**
	California Central Valley	**Ad Mod**	No	**Ad Mod**	No	No	**Ad Mod**
	South-Central California Coast	No	No	No	No	No	**Ad Mod**
	Southern California	No	No	No	No	No	**Ad Mod**

Reasonable and Prudent Alternatives

Regulations (50 CFR §402.02) implementing section 7 of the ESA define reasonable and prudent alternatives as alternative actions, identified during formal consultation, that: (1) can be implemented in a manner consistent with the intended purpose of the action; (2) can be implemented consistent with the scope of the action agency's legal authority and jurisdiction; (3) are economically and technologically feasible; and (4) NMFS believes would avoid the likelihood of jeopardizing the continued existence of listed species or resulting in the destruction or adverse modification of critical habitat.

This Opinion has concluded that EPA's proposed registration of certain uses of 2,4-D, including aquatic uses of 2,4-D BEE are likely to jeopardize the continued existence of the 28 endangered and threatened Pacific salmonids. This Opinion has also concluded that 2,4-D, diuron, and chlorothalonil are likely to adversely modify or destroy designated critical habitat for one or more of the 28 threatened and endangered salmonids. NMFS reached these conclusions because predicted concentrations of these a.i.s in salmonid habitats are likely to cause adverse effects to Pacific salmonids, water quality, salmonid prey, natural cover, and/or substrate in freshwater rearing, spawning, and foraging areas.

The Reasonable and Prudent Alternative (RPA) accounts for the following issues: (1) the action will result in exposure to other chemical stressors in addition to the a.i. that may increase the risk of the action to listed species, including unspecified inert ingredients, adjuvants, and tank mixes; (2) exposure to chemical mixtures containing the a.i.s; and (3) exposure to other chemicals and physical stressors 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 adverse modification by reducing the concentrations of each of these a.i.s and their associated stressors of the action within the designated critical habitat. In the proposed RPA, NMFS does not attempt to ensure there is no take of listed species. NMFS believes take will occur, and has provided an incidental take statement exempting that take from the take prohibitions, so long as the action is conducted according to the RPA and reasonable and prudent measures (RPM). Avoiding take altogether would most likely entail canceling registration, or prohibiting use in watersheds inhabited by salmonids. The goal of the RPA is to reduce exposure to ensure that the action is not likely to jeopardize listed species, destroy or adversely modify critical habitat.

The RPA is comprised of seven required elements that must be implemented in its entirety within one year of the EPA's receipt of this Opinion to ensure the registration of these pesticides is not likely to jeopardize endangered or threatened Pacific salmonids under the jurisdiction of NMFS or destroy or adversely modify critical habitat designated for these species. For each active ingredient, the elements of the RPA apply only to those ESUs/DPSs where NMFS has determined that registration of that a.i. is likely to jeopardize listed species and/or destroy or adversely modify designated critical habitat (Table 148and Table 149). These elements rely upon recognized practices for reducing the loading of pesticide products into aquatic habitats.

Specific Elements of the Reasonable and Prudent Alternative

Elements 1-7 shall be specified on FIFRA labels of all pesticide products containing 2,4-D, diuron, and chlorothalonil when used within ESUs or DPSs where jeopardy or adverse modification of designated critical habitat has been determined. Alternatively, the label could direct pesticide users to the EPA's Endangered Species Protection Program (ESPP) bulletins that specify elements 1-7 in the applicable counties. The derivation of concentration limits used in the elements below is described in Appendix 1.

*Element 1***.** The following applies to broadcast spray applications of pesticide products containing 2,4-D, diuron, and chlorothalonil in applicable ESUs or DPSs. These pesticides shall only be broadcast applied when there is minimal potential for drift to listed salmonid-bearing waters. Do not apply when wind speeds are below 2 mph or exceed 10 mph, except when winds in excess of 10 mph will carry drift away from salmonid-bearing waters.

Element 2. Do not apply pesticide products containing 2,4-D, diuron, or chlorothalonil when soil is saturated, or when a precipitation event , likely to produce direct runoff to salmonid-bearing waters from the treated area, is forecasted by NOAA/NWS (National Weather Service) or other similar forecasting service within 48 h following application.

Element 3. 2,4-D BEE specific requirements:
Do not apply pesticide products containing 2,4-D butoxyethyl ester directly to any surface waters accessible to listed salmonids.

Element 4. 2,4-D specific requirements designed to protect native riparian vegetation and designated critical habitat.

1. Do not apply 2,4-D directly to native riparian vegetation except as part of a native riparian vegetation restoration project. Control of invasive plants within the riparian habitat shall be by individual plant treatments for woody species, and spot treatment of less than 1/10 acre for herbaceous species.
2. EPA will implement NMFS approved risk reduction measures to ensure maximum concentrations of terrestrially applied 2,4-D do not exceed a peak of 100 µg/L in salmonid-bearing waters.[15]

Element 5. Diuron-specific requirements within areas designated critical as habitat for the specified ESU/DPSs in Table 149. This element is designed to protect native riparian vegetation and reduce direct exposure to listed fish:

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[15] Within ESUs or DPSs where jeopardy or adverse modification of designated critical habitat has been determined, EPA will implement NMFS approved risk reduction measures to ensure maximum concentrations of the a.i.s predicted in salmonid bearing waters or associated native riparian vegetation will not exceed the specified value. NMFS encourages EPA to take into account existing state programs that reduce exposure potential to salmonid-bearing waters when developing protocols. These values represent the highest concentrations that may be achieved in salmonid habitats, rather than time-weighted average concentrations, and consider the range in potential droplet size spectrum, release heights, wind speeds, and wind directions that may be associated with all labeled application methods (*e.g.,* agricultural applications, vector control in public health programs, *etc*.). The maximum predicted concentrations shall account for potential contributions from both runoff and drift to salmonid habitats, as appropriate. Risk reduction measures shall account for the predicted maximum concentrations in all salmonid-bearing water, including a modeled floodplain habitat of 1-2 m wide and 0.1 m deep. They shall also account for potential increases in aquatic concentrations associated with the maximum application rate and the maximum number of times an a.i. may be applied per season according to label restrictions.

1. Do not apply diuron directly to native riparian vegetation.
2. Do not apply diuron to intermittently flooded low lying sites, marshes, swamps, and bogs that may be seasonally connected to habitats that contain listed salmonids.
3. EPA will implement NMFS approved risk reduction measures to ensure diuron drift to native riparian vegetation does not exceed 0.10 lbs/A[13].
4. When native riparian vegetation is not present, EPA will implement NMFS approved risk reduction measures to ensure maximum concentrations of diuron do not exceed 5.0 μg/L in salmonid-bearing waters.

Element 6. Chlorothalonil-specific requirements within areas designated as critical habitat for the specified ESU/DPSs in Table 149.

1. EPA will implement NMFS approved risk reduction measures to ensure maximum concentrations of chlorothalonil do not exceed a peak concentration of 1.05 μg/L, or a 21 d time-weighted-average concentration of 0.18 μg/L in salmonid-bearing waters.[13,16] Reduction measures may include reduced single and annual application rates.
2. Application to conifers will be limited to the following uses: (i) conifer nursery beds; (ii) Christmas tree and bough production plantations; (iii) tree seed orchards; and (iv) landscape situations (ornamental or specimen trees in a residential or commercial landscape).

Element 7. Report all incidents of fish mortality that occur within the vicinity of the treatment area, including areas downstream and downwind, and in the four days following application 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). EPA shall submit an annual report to NMFS OPR that identifies the total number of fish affected and incident locations.

[16] Calculation of 21 d time-weighted-average shall be for a static floodplain habitat (2 m wide and 0.1 m deep) and include the evaluation of maximum annual application rates according to label restrictions.

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)). ESU/DPS applicable reasonable and prudent alternatives have been summarized in Table 150.

Table 150. RPA elements (1 - 7) applicable to each ESU/DPS and to each a.i. combination.

Species	ESU	Herbicides: RPA Elements that apply				Fungicides: RPA Elements that apply	
		2,4-D	Triclopyr BEE	Diuron	Linuron	Captan	Chlorothalonil
Chinook	Puget Sound	1,2,3,7		1,2,5,7			1,2,6,7
	Lower Columbia River	1,2,3,7		1,2,5,7			
	Upper Columbia River Spring - Run	1,2,3,7					
	Snake River Fall - Run	1,2,3,7					
	Snake River Spring/Summer - Run	1,2,3,7					
	Upper Willamette River	1,2,3,4,7		1,2,5,7			1,2,6,7
	California Coastal	1,2,3,7					
	Central Valley Spring - Run	1,2,3,4,7		1,2,5,7			1,2,6,7
	Sacramento River Winter - Run	1,2,3,4,7		1,2,5,7			1,2,6,7
Chum	Hood Canal Summer - Run	1,2,3,7					
	Columbia River	1,2,3,7					
Coho	Lower Columbia River	1,2,3,7					
	Oregon Coast	1,2,3,7					
	Southern Oregon and Northern California Coast	1,2,3,7					
	Central California Coast	1,2,3,7					1,2,6,7
Sockeye	Ozette Lake	1,2,3,7					
	Snake River	1,2,3,7					
Steelhead	Puget Sound	1,2,3,7					
	Lower Columbia River	1,2,3,7		1,2,5,7			
	Upper Willamette River	1,2,3,4,7		1,2,5,7			1,2,6,7
	Middle Columbia River	1,2,3,7					
	Upper Columbia River	1,2,3,7					
	Snake River	1,2,3,7					
	Northern California	1,2,3,7					
	Central California Coast	1,2,3,4,7		1,2,5,7			1,2,6,7
	California Central Valley	1,2,3,4,7		1,2,5,7			1,2,6,7
	South-Central California Coast	1,2,3,7					1,2,6,7
	Southern California	1,2,3,7					1,2,6,7

Incidental Take Statement

Section 9(a)(1) of the ESA prohibits the taking of endangered species without a specific permit or exemption. Protective regulations adopted pursuant to section 4(d) of the ESA extend the prohibition to threatened species. Take is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt to engage in any such conduct (50 CFR 222.102). Harm is further defined by NMFS to include significant habitat modification or degradation that results in death or injury to listed species by significantly impairing essential behavioral patterns, including breeding, feeding, or sheltering. Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity conducted by the Federal agency or applicant (50 CFR 402.02). Under the terms of section 7(b)(4) and section 7(o)(2), taking that is incidental to and not intended as part of the agency action, whether implemented as proposed or as modified by reasonable and prudent alternatives, is not considered to be prohibited taking under the ESA provided that such taking is in compliance with the terms and conditions of this Incidental Take Statement.

Amount or Extent of Take

As described earlier in this Opinion, this is a consultation on the EPA's registration of pesticide products containing 2,4-D, triclopyr BEE, diuron, linuron, captan, chlorothalonil, and their formulations as they are used in the Pacific Northwest and California and the effects 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 as described in the *Description of the Proposed Action* and elsewhere in the document. The goal of this Opinion is to evaluate the impacts to NMFS' listed resources from the EPA's broad authorization of applied pesticide products. This Opinion is a partial consultation because pursuant to the court's order, EPA sought consultation on only 26 listed Pacific salmonids under NMFS' jurisdiction. However, even though the court's order did not address the two more recently listed ESUs and DPSs, NMFS analyzed the impacts of EPA's actions to them because they belong to the same taxon and the analysis requires consideration of the same information. Consultation with NMFS will be completed when EPA makes effect determinations on all remaining species under NMFS' jurisdiction and consults with NMFS as necessary.

For this Opinion, NMFS anticipates the general direct and indirect effects that would occur from EPA's registration of pesticide products across the states of California, Idaho, Oregon, and Washington to 28 listed Pacific salmonids under NMFS' jurisdiction during the 15-year duration of the proposed action. Recent and historical surveys indicate that listed salmonids occur in the action area, in places where they will be exposed to the stressors of the action. The RPA above and RPMs below provided in this Opinion are designed to reduce this exposure but not eliminate it. Pesticide runoff and drift of 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil 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, perennial, intermittent, and ephemeral streams. These inputs into aquatic habitats are especially high when rainfall immediately follows applications. The effects of pesticides and other contaminants found in right of ways and 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 six a.i.s on salmonids includes direct and indirect toxicological effects. Within this range, effects include impairments of physiological functions to the extent that fish die or are unable to perform necessary life functions (such as predator avoidance, foraging, migration and reductions in reproductive success). Adverse impacts to riparian vegetation could lead to increased water temperature, increased sedimentation from bank instability, reductions in cover, alterations to or decreases in prey production, and reduction in chemical and nutrient filtering from upland sources. Impacts to aquatic vegetation would reduce dissolved oxygen, natural cover, alter or reduce the prey base, affect growth, and lead to an increased susceptibility to predation. These results are not the purpose of the proposed action. Therefore, incidental take of listed salmonids is reasonably certain to occur over the 15-year duration of the proposed action.

Given the variability of real-life conditions, the broad nature and scope of the proposed action, and the migratory nature of salmon, the best scientific and commercial data available are not sufficient to enable NMFS to estimate a specific amount of incidental take associated with the proposed action. As explained in the *Description of the Proposed Action* and the *Effects of the Proposed Action* sections, NMFS identified multiple uncertainties associated with the proposed action. Areas of uncertainty include:

1. Unable to quantify effect of herbicides on salmon habitat due to variability in plant susceptibility to the herbicides and variability in species composition and density in the various locations.
2. Incomplete information on the proposed action (*i.e.*, no master labels summarizing all stressors of the action and all authorized uses of pesticide products). ;
3. Limited use and exposure data on stressors of the action for non-agricultural uses of these pesticides;
4. Minimal information on exposure and toxicity for pesticide formulations, adjuvants, and other/inert ingredients within registered formulations;
5. Little information on permitted tank mixtures and associated exposure estimates;
6. Limited data on toxicity of environmental mixtures;
7. Responses from exposure to combinations of the 6 a.i.s and other stressors in the baseline;
8. Annual variable conditions regarding land use, crop cover, and pest pressure;
9. Variable temporal and spatial conditions within each ESU, especially at the population-level; and
10. Variable conditions of water bodies in which salmonids live.

NMFS therefore identifies, as a surrogate for the allowable extent of take, the ability of this action to proceed without any fish kills attributed to the legal use of 2,4-D, triclopyr BEE, diuron, linuron, captan, or chlorothalonil, or any compounds, degradates, or mixtures in aquatic habitats containing individuals from any ESU/DPS. Because of the difficulty of detecting salmonid deaths, the fishes killed do not have to be listed salmonids. In general, salmonids appear to be more sensitive to these a.i.s than many other species of fish, so that if there are kills of other freshwater fishes attributed to use of these pesticides, it is likely that salmonids have also died, even if no dead salmonids can be located. In addition, if stream conditions due to pesticide use kill less sensitive fishes in certain areas, the potential for lethal and non-lethal takes in downstream areas increases. A fish kill is considered attributable to one of these six ingredients, its metabolites, or degradates, if the a.i is known to have been applied in the vicinity, may reasonably be supposed to have run off or drifted into the affected area, and if surface water samples, or pathology indicate lethal levels of the a.i.(s).

NMFS notes that with increased monitoring and study of the impact of these pesticides on water quality, particularly water quality in off-channel habitats, NMFS will be able to refine this incidental take statement, and future incidental take statements, to allow other measures of the extent of take.

Reasonable and Prudent Measures

The measures described below are non-discretionary measures to avoid or minimize take that must be undertaken by the EPA so that they become binding conditions of any grant or permit issued to the applicant(s), as appropriate, for the exemption in section 7(o)(2) to apply. The EPA has a continuing duty to regulate the activity covered by this incidental take statement. If the EPA (1) fails to assume and implement the terms and conditions or (2) fails to require the applicant(s) to adhere to the terms and conditions of the incidental take statement through enforceable terms that are added to the permit or grant document, the protective coverage of section 7(o)(2) may lapse. In order to monitor the impact of incidental take, the EPA must report the progress of the action and its impact on the species to NMFS OPR as specified in the incidental take statement [50 CFR§402.14(i)(3)].

To satisfy its obligations pursuant to section 7(a)(2) of the ESA, the EPA must monitor (a) the direct, indirect, and cumulative impacts of its long-term registration of pesticide products containing 2,4-D, triclopyr BEE, diuron, linuron, captan, or chlorothalonil; and (b) the consequences of those effects on listed Pacific salmonids under NMFS' jurisdiction. The purpose of the monitoring program is for the EPA to use the results of the monitoring data and modify the registration process in order to reduce exposure and minimize the effect of exposure where pesticides will occur in salmonid habitat. NMFS believes all measures described as part of the proposed action, together with use of the Reasonable and Prudent Measures and Terms and Conditions described below, are necessary and appropriate to minimize the likelihood of incidental take of listed species due to implementation of the proposed action.

The EPA shall:

1. Minimize the amount and extent of incidental take from use of pesticide products containing 2,4-D, triclopyr BEE, diuron, linuron, captan, or chlorothalonil by reducing the potential of these chemicals to reach salmon-bearing waters;
2. Minimize the effects of 2,4-D during direct water applications;
3. Monitor any incidental take or surrogate measure of take that occurs from the action; and
4. Report annually to NMFS OPR on the monitoring results from the previous year.

Terms and Conditions

To be exempt from the prohibitions of section 9 of the ESA, within one year following the date of issuance of this Opinion, the EPA must comply with the following terms and conditions. These terms and conditions implement the reasonable and prudent measures described above. These terms and conditions are non-discretionary. Terms and conditions 1, 3, and 6(a).shall be specified on FIFRA labels of all pesticide products containing 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil. Alternately, the labels could direct pesticide users to the EPA's ESPP bulletins that specify these terms and conditions.

1. This pesticide shall only be broadcast applied when there is minimal potential for drift to listed salmonid-bearing waters. Do not broadcast spray when wind speeds are below 2 mph or exceed 10 mph, except when winds in excess of 10 mph will carry drift away from salmonid-bearing waters.

2. Products containing 2,4-D (except 2,4-D BEE) may be applied to salmon bearing waters providing the following:
    a. Applications are only to control non-native (exotic) invasive plant species;
    b. Applications are only during timing windows provided in Appendix 9 (page 967), and
    c. Applications will minimally affect non-target native vegetation.

3. Do not apply pesticide products containing 2,4-D, triclopyr BEE, diuron, linuron, captan, or chlorothalonil (include only relevant a.i. or pesticide product name on label/bulletin) when soil is saturated , or when a precipitation event likely to produce direct runoff to salmon bearing waters from the treated area is forecasted by NOAA/NWS (National Weather Service) or other similar forecasting service within 48 h following application.

4. Do not apply diuron to intermittently flooded low lying sites, marshes, swamps, and bogs that may be seasonally connected to habitats that contain listed salmonids.

5. Chlorothalonil applications to conifers will be limited to the following uses: (i) conifer nursery beds; (ii) Christmas tree and bough production plantations; (iii) tree seed orchards; and (iv) landscape situations (ornamental or specimen trees in a residential or commercial landscape).

6. Regarding all products containing 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil:

    a. EPA shall include the following instructions requiring reporting of fish kills either on the labels or ESPP Bulletins :

    NOTICE: Incidents where salmon appear injured or killed as a result of pesticide applications shall be reported to NMFS OPR at 301-713-1401 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. The finder may be asked to carry out instructions provided by NMFS OPR to collect specimens or take other measures to ensure that evidence intrinsic to the specimen is preserved.

b. EPA shall report to NMFS OPR any incidences regarding 2,4-D, triclopyr BEE, diuron, linuron, captan, or chlorothalonil effects on aquatic ecosystems added to its incident database that EPA has classified as “probable” or “highly probable.”

7. In addition to the labeling requirements above, EPA shall develop and implement a NMFS-approved effectiveness monitoring plan for floodplain habitats, and produce annual reports of the results. NMFS encourages EPA to work with local, state, and other agencies to assist in plan development and implementation. The plan shall identify representative floodplain habitats prone to drift and runoff of pesticides within agricultural and non-agricultural areas. The representative sampling sites shall include habitats currently used by threatened and endangered Pacific salmonids, as identified by NMFS biologists. Sampling sites include at least two sites for each general species (*i.e.*, coho salmon, chum salmon, steelhead, sockeye salmon, and ocean-type Chinook and stream-type Chinook salmon). Sampling shall consist of daily collection of surface water samples for seven consecutive days during three periods of high application for 2,4-D, triclopyr BEE, diuron, linuron, captan and chlorothalonil. The report shall be submitted to NMFS OPR and will summarize annual monitoring data and provide all raw data.

Conservation Recommendations

Section 7(a) (1) of the ESA directs federal agencies to use their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of endangered and threatened species. Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of a proposed action on listed species or critical habitat, to help implement recovery plans, or to develop information.

The following conservation recommendations would provide information for future consultations involving future authorizations of pesticide a.i.s that may affect listed species:

1. Conduct mixture toxicity analysis in screening-level and endangered species biological evaluations;
2. Develop models to estimate pesticide concentrations in off-channel habitats; and

3. Develop models to estimate pesticide concentrations in aquatic habitats associated with non-agricultural applications, particularly in residential and industrial environments.
4. Develop and implement a program to educate users of pesticide about the potential adverse effects on salmonids and their designated critical habitat. Educational materials should discuss measures and techniques appropriate for reducing input of pesticides to aquatic habitats.

In order for NMFS to be kept informed of actions minimizing or avoiding adverse effects or benefiting listed species or their habitats, the EPA should notify NMFS OPR of any conservation recommendations it implements in the final action.

Reinitiation Notice

This concludes formal consultation on the EPA's proposed registration of pesticide products containing 2,4-D, triclopyr BEE, diuron, linuron, captan, chlorothalonil, and their formulations to ESA-listed Pacific salmonids under the jurisdiction of the NMFS. As provided in 50 CFR 402.16, reinitiation of formal consultation is required where discretionary federal agency involvement or control over the action has been retained (or is authorized by law) and if: (1) the extent of take specified in the *Incidental Take Statement* is exceeded; (2) new information reveals effects of this action that may affect listed species or designated critical habitat in a manner or to an extent not previously considered in this biological opinion; (3) the identified action is subsequently modified in a manner that causes an effect to the listed species or critical habitat that was not considered in this Opinion; or (4) a new species is listed or critical habitat designated that may be affected by the identified action. If reinitiation of consultation appears warranted due to one or more of the above circumstances, EPA must contact NMFS OPR. In the event reinitiation conditions (1), (2), or (3) is met, reinitiation will be only for the a.i.(s) which meet that condition, not for all 6 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.

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Appendix 1 – NMFS' consideration of maximum concentration limits for 2,4-D, diuron, and chlorothalonil

<u>**2,4-D**</u>

2,4-D: March 1, 2011 Draft limit of 10µg/L from Brock literature review (Forsyth et al 1997).

Brock 2000 reviewed ecosystem studies and provided $NOEC_{ecco}$ values for comparison to water quality criteria. The 10 µg/L value comes from a study by Forsyth et al. Summary information for that mesocosm study is provided below.

Mesocosm: test units consisted of 1 meter square enclosures in a 12 hectare prairie wetland pond (12-ha); test units 50-70 cm deep; lost 43% of volume of water in units during 60d course of study.

Treatment: A single application of 2,4-D was applied to test units and concentrations were left to naturally degrade in the wetland post application; 2 test concentrations were evaluated with nominal peak concentrations of 10 and 100 µg/L; response measurements were collected at 30 and 60 days post application.

Results: The 100 µg/L treatment inhibited growth and caused mortality in both submerged aquatic macrophytes evaluated (sago pondweed and a water milfoil). The 10 µg/L responses were not statistically different than controls.

2,4-D Task Force comments:

2,4-D Task Force recommend we use a maximum concentration limit of 500 µg/L – The task force claims it's the 24-hr NOEC from a provided study (Green et al. 1990)

Lab study: 15 gallon test units

Treatment: Short duration exposures (12 hr -72 hr) were evaluated at concentrations starting at 500 µg/L, after short-term exposure plants were placed in clean water free of contaminants and assessed 4 weeks post-treatment.

Results: Injury increased with both exposure time and concentration. Minimal injury was observed after 12 hr exposure at 500 µg/L. Some effects apparent at all treatments, severe effects at 500 µg/L after 72 hrs (biomass about 20% of control, 92% visual injury to plants).

Comparison of 2 studies

Task force argued study conditions in the outdoor mesocosm don't represent suitable habitat for salmonids. It's not a flowing habitat and temperatures were 20 – 26 deg C. They suggest we use a 24 hr "NOEC" from a lab study.

The temperatures in the outdoor mesocosm study were not optimal for salmon habitat. However, the team agreed the effects to plants are meaningful, and in fact, more relevant than the laboratory test recommended by the task force. We found no valid reasons to discount the mesocosm study which showed effects to the two species of plants evaluated. Additionally, the lab study the task force recommended showed severe injury to plants at the lowest test concentration (500 µg/L) with exposure at 72-hrs. There was also evidence of effects at shorter exposure periods (e.g. 12 hr, 24 hr). Decreased biomass and visual injury showed dose-response pattern related to both test concentration and duration. The argument by the task force that NMFS should go with the 24-hr NOEC of 500 µg/L is flawed for several reasons: (1)2,4-D does not degrade quickly in water and exposures in excess of 24 hrs are expected (stable to hydrolysis, photolytic ½ life is 13 days, metabolic ½ life is 15 days); (2) results suggest severe effects could occur at 500 µg/L with relatively short exposures (72 hrs); and (3) the paper does not provide a true NOEC that involved statistical comparison to controls. The authors indicated 500 µg/L had little or no effect for short duration exposures (e.g. 12 and 24 hr exposure), but that was in the context of selecting efficacious application rates to control milfoil, not for assessing ecological responses of salmonid habitat.

Considerations for developing maximum concentration limits for 2,4-D in salmonid habitats

Consideration	Increase likelihood of over-protecting	Increase likelihood of under-protecting
2,4-D in surface water		
Use of NOEC endpoint	X	
Use of peak concentrations for maximum concentration limit	X	
Beneficial habitat responses	X	
Assumption that surrogate species accurately reflect sensitivity of salmonids and their habitat	May result in either over- or under-protecting	
Uncertainty associated with effectiveness of risk reduction measures employed	May result in either over- or under-protecting	
Baseline stressors that may contribute to response (e.g. temperature, dissolved oxygen, other pesticides, other contaminants)		X
Exposure to formulation inerts		X
Exposure to other a.i.s in formulation		X
Exposure to other contaminants in tank mixtures		X

NMFS Decision – for 2nd draft value for 2,4-D in surface water: After weighing the available information (provided above, and considering additional information discussed in draft opinion), the team agreed it was appropriate to use the mesocosm LOEC value of 100 µg/L for the maximum concentration limit (Forsyth et al 1997).

DIURON

March 1, 2011 Draft limit of 0.05 lbs/A for Riparian Habitats

NMFS selected this value after considering the FIFRA guideline studies for nontarget plants. The EC25s with crop species range from 0.002 – 0.021 lbs/A. We also considered that diuron is used to eliminate weeds at labeled rates of 0.8 lbs/A (maximum single application rate in corn) to 12 lbs/A (max rate for non-crop land uses). We felt significant effects to riparian habitats would likely occur somewhere between those two ranges but recognized there is uncertainty about riparian response given lack of toxicity data on riparian species. At the applicant meeting with diuron registrants we asked if they could provide additional information that would help us characterize potential vegetation response in riparian zone.

DuPont Response to Riparian Habitat limit

Diuron is most effective on plants that are actively growing (early during development). Woody species and mature perennial grasses are significantly less sensitive than plants that are tested in lab studies which would represent worst case. DuPont provided three papers to support their position that Diuron has greater affect on young growing plants than on established perennials (Young 1970, Young and Evans 1972, Tworkoski et al. 200). They also provided a paper to support the related position that plant in mature fields show resistance compared to recently established fields (Tompkins and Grant 1977). A fifth paper cited was claimed to show that treatment of an established range pasture at 1 kg/ha (0.89 lbs/A) did not result in decreased grass, broadleaf, or total cover.

Finally, DuPont stated that they believe there is no relevant technical information that would justify a drift target for protection of riparian communities below 0.1 lb ai/A, and that exposure of riparian plant communities to any level of diuron drift is highly unlikely because diuron has a “use pattern with very limited aerial application.”

NMFS review of information provided by DuPont.

The information DuPont provided does support that mature plants are not as sensitive as actively growing young plants. We expect riparian habitat response will be variable depending on the composition of the riparian habitat. We also expect well established habitats that include a variety of species will be less impacted than newly established or restored riparian habitats and those dominated by annual grasses. DuPont did not provide information to evaluate the relative sensitivity of riparian plants. It is unclear why DuPont chose to identify 0.1 lb a.i/A as a threshold below which limits would not be justified. NMFS had asked DuPont to provide a recommendation for a threshold value for riparian habitats, but it was unclear if DuPont was recommending 0.1 lbs a.i/A as that threshold. We do not agree that a target of < 0.1 lb a.i./A could not be justified as the regulatory toxicity studies suggest impacts to plants below < 0.05 lbs a.i./A. However, we recognize that there is a great deal of uncertainty with regard to the likely response of riparian vegetation and we expect established perennials in the riparian zone will be less sensitive to diuron than the species tested. We noted that 0.1 lb a.i./A is approximately 10% of the lower range of maximum application rates for diuron use sites.

Considerations for developing maximum drift limits for diuron in riparian habitats

Consideration	**Increase likelihood of over-protecting**	**Increase likelihood of under-protecting**
Diuron in riparian habitats		
Use of lab EC25 on new plants versus sensitivity of established riparian plants	X	
Assumption that surrogate species accurately reflect sensitivity of riparian plants	May result in either over- or under-protecting	
Uncertainty associated with effectiveness of risk reduction measures employed	May result in either over- or under-protecting	
Exposure to other a.i.s and inerts in formulations		x
Exposure to other contaminants in tank mixtures		x

NMFS Decision for 2nd draft value for diuron drift to riparian vegetation: After weighing the available information (provided above, and considering additional information discussed in draft opinion), the team agreed it was appropriate to use 0.10 lbs a.i./A as the limit for diuron drift to riparian vegetation.

NMFS Proposed limit of 2.9 µg/L of diuron in surface waters

The 2.9 µg/L value was the $NOEC_{ecco}$ from Brock literature review (Flum and Shannon 1986).
Microcosm: Test units were flasks inoculated with zooplankton, amphipods, ostracods, filamentous algae, protozoans, and microbes
Treatment: Single exposure in static test systems (flasks)
Results: The Minimum Effect Level (MEL) = 2.9-28.5 µg/L based on the NOEC of 2.9 µg/L (pH), and LOEC of 28.5 µg/L (pH). An EC20 representing a 20% decline in dissolved oxygen was estimated at 594 µg/L.
Study evaluated treatment related changes in ecosystem level variables by evaluating changes in pH, and dissolved oxygen as indicators of trophic level interactions. These are recognized as indirect measures of primary production (Brock 2000).

<u>DuPont position</u>

DuPont suggested that it's more appropriate to use biological endpoints than the abiotic measurements used in the Flum and Shannon study. Additionally they claim the microcosm was unrealistic for salmon habitats because it was not conducted in running water. They recommend NMFS instead use the median EC_{50} value of 28 µg/L for algae and vascular plants (from NMFS response section, table of assessment endpoints and measures for diuron).
DuPont summarized a mesocosm study by Knauert et al 2008. A stagnant system showed effects to photosynthetic activity within 2 days by 56% at 5 µg/L. Complete recovery of effects on phytoplankton were observed by day 96. DuPont suggests that this is a worst case scenario because it is not in flowing water and shows effects are short-term and transient.
A second mesocosm study (Knauert et al 2010) was also summarized. DuPont indicates phytoplankton pre-exposed to diuron were less sensitive than phytoplankton that were not (by factor

of 3.7). Their conclusion was that phytoplankton pre-exposed to low concentrations of diuron will be less sensitive than subsequent exposure.

NMFS consideration of this information.

The abiotic measure (pH and DO) in the Flum and Shannon 1986 study are relevant not just as indicators of changes to primary production, but of water quality to salmon. We considered the Knauert studies referenced by DuPont. Two of the studies were outdoor mesocosm investigations that showed adverse response at 5 µg/L in phytoplankton (Knauert et al 2008) and aquatic macroophytes (Knauert et al 2010). Both studies showed mixture exposure with other photosystem II-inhibiting herbicides increased the toxicity in a dose-additive manner. These finding suggest the presence of other photosystem-II inhibiting herbicides in the baseline can increase adverse responses of primary producers exposed to diuron.

Considerations for developing maximum concentration limits for diuron in salmonid habitats

Consideration	Increase likelihood of over-protecting	Increase likelihood of under-protecting
Diuron in surface water		
Use of NOEC endpoint	X	
Use of peak concentrations for maximum concentration limit	X	
Beneficial habitat responses	X	
Assumption that surrogate species accurately reflect sensitivity of salmonids and their habitat	May result in either over- or under-protecting	
Uncertainty associated with effectiveness of risk reduction measures employed	May result in either over- or under-protecting	
Baseline stressors that may contribute to response (e.g. temperature, dissolved oxygen, other pesticides, other contaminants)		x
Exposure to formulation inerts		x
Exposure to other a.i.s in formulation		x
Exposure to other contaminants in tank mixtures		x

NMFS Decision for 2nd draft value for diuron in surface waters: After weighing the available information (provided above, and considering additional information discussed in draft opinion), the team agreed it was appropriate to use 5 µg/L for the maximum concentration limit.

CHLOROTHALONIL

March 1, 2011 Draft RPA

NMFS did not provide a maximum concentration limit for chlorothalonil at the issuance of the first draft. Information to evaluate concentrations that cause more complex ecological responses to chlorothalonil were lacking (e.g. ecosystem responses, population level responses). During the meeting with chlorothalonil applicants NMFS requested suggestions for risk reduction. NMFS received no suggestions for maximum concentration limits for chlorothalonil or other risk reduction measures beyond label clarification of chlorothalonil forestry uses.

NMFS consideration of available information

The pesticide team previously discussed deferring to other regulatory criteria. There are no water quality criteria for chlorothalonil in the U.S. Information on Canada's water quality criteria is provided below.

Canadian Water Quality Guideline for Protection of Aquatic Life: 0.18 µg/L.

How derived: By applying a safety factor of 0.1 to the most sensitive aquatic invertebrate LOEC (lowest test concentration with statistically significant effects).

Study information: The endpoint was based on *Daphnia magna* immobilization from exposure to a formulated chlorothalonil product. The reported duration of the study was 22 days.

Other: Supporting material for the Canadian Water Quality guidelines suggest the 0.18 µg/L criteria "is comparable to those proposed by Davies and Cook (1990) for the Australian water quality guidelines of 0.06 and 0.2 µg/L (Level I and Level II protections, respectively) for the maintenance of aquatic ecosystems. "

NMFS considered that the 0.18 µg/L value was derived with chronic exposure. Chronic exposure is certainly a concern given. King and Balogh 2010 found detections of chlorothalonil April – Nov in surface water from applications that primarily occurred in late October. The team felt this was an appropriate threshold for chronic exposure rather than a maximum peak exposure.

Salmonids were among the most sensitive species tested with acute lethality starting at 10.5 µg/L in a 96-hr LC50. The team considered if applying a similar safety factor approach was adequate for peak exposure.

Study information for the 96-hr LC50 Of 10.5 µg/L: The study units included a flow-though system, with O_2 of 5.12 mg/L, and temperature of 16 deg C. The study found a positive relationship between exposure duration and toxicity. Additionally, the study found that lower oxygen levels caused an increased sensitivity to chlorothalonil, even when the oxygen level was not harmful to individuals in the control treatment.

Considerations for developing maximum concentration limits for chlorothalonil in salmonid habitats

Consideration	Increase likelihood of over-protecting	Increase likelihood of under-protecting
Chlorothalonil in surface water		
Use of safety factor applied to chronic LOEC endpoint for concentration limit	x	
Use of safety factor applied to peak concentrations for maximum concentration limit	x	
Assumption that surrogate species accurately reflect sensitivity of salmonids and their habitat	May result in either over- or under-protecting	
Uncertainty associated with effectiveness of risk reduction measures employed	May result in either over- or under-protecting	
Use of single species endpoint to represent complex ecosystem responses	May result in either over- or under-protecting	
Baseline stressors that may contribute to response (e.g. temperature, dissolved oxygen, other pesticides, other contaminants)		X
Exposure to formulation inerts		X
Exposure to other a.i.s in formulation		x
Exposure to other contaminants in tank mixtures		x

NMFS Decision for 2nd draft value for chlorothalonil in surface waters: After weighing the available information (provided above, and considering additional information discussed in draft opinion), the team agreed it was appropriate to use a peak of 1.05 µg/L for the maximum concentration limit of chlorothalonil (1/10 the salmonid LC50), and a chronic (21 d time-weighted-average) concentration of 0.18 µg/L (consistent with Canadian Water Quality value).

Appendix 2. Species and Population Annual Rates of Growth

Chinook Salmon

ESU	Population	λ - H=0	95% CI -lower	95% CI - upper
California Coastal	Eel River	N/A	N/A	N/A
	Redwood Creek	N/A	N/A	N/A
	Mad River	N/A	N/A	N/A
	Humboldt Bay tributaries	N/A	N/A	N/A
	Bear River	N/A	N/A	N/A
	Mattole River	N/A	N/A	N/A
	Tenmile to Gualala	N/A	N/A	N/A
	Russain River	N/A	N/A	N/A
Central Valley Spring - Run (Good et al., 2005 - 90% CI)	Butte Creek - spring run	1.300	1.060	1.600
	Deer Creek - spring run	1.170	1.040	1.350
	Mill Creek - spring run	1.190	1.000	1.470
Lower Columbia River (Good et al., 2005) (# = McElhany et al., 2007)	Youngs Bay	N/A	N/A	N/A
	Grays River - fall run	0.944	0.739	1.204
	Big Creek	N/A	N/A	N/A
	Elochoman River - fall run	1.037	0.813	1.323
	Clatskanie River #	0.990	0.824	1.189
	Mill, Abernathy, Germany Creeks - fall run	0.981	0.769	1.252
	Scappose Creek	N/A	N/A	N/A
	Coweeman River - fall run	1.092	0.855	1.393
	Lower Cowlitz River - fall run	0.998	0.776	1.282
	Upper Cowlitz River - fall run	N/A	N/A	N/A
	Toutle River - fall run	N/A	N/A	N/A
	Kalamaha River - fall run	0.937	0.763	1.242
	Salmon Creek / Lewis River - fall run	0.984	0.771	1.256
	Clackamas River - fall run	N/A	N/A	N/A
	Washougal River - fall run	1.025	0.803	1.308
	Sandy River - fall run	N/A	N/A	N/A
	Lower Gorge tributaries	N/A	N/A	N/A
	Upper Gorge tributaries - fall run	0.959	0.751	1.224
	Hood River - fall run	N/A	N/A	N/A
	Big White Salmon River - fall run	0.963	0.755	1.229
	Sandy River - late fall run	0.943	0.715	1.243
	North Fork Lewis River - late fall run	0.968	0.756	1.204
	Upper Cowlitz River - spring run	N/A	N/A	N/A
	Cispus River	N/A	N/A	N/A
	Tilton River	N/A	N/A	N/A
	Toutle River - spring run	N/A	N/A	N/A
	Kalamaha River - spring run	N/A	N/A	N/A
	Lewis River - spring run	N/A	N/A	N/A
	Sandy River - spring run #	0.961	0.853	1.083
	Big White Salmon River - spring run	N/A	N/A	N/A
	Hood River - spring run	N/A	N/A	N/A

Chinook Salmon (continued)

ESU	Population	λ - H=0	95% CI -lower	95% CI - upper
Upper Columbia River Spring - Run (FCRPS)	Methow River	1.100	N/A	N/A
	Twisp River	N/A	N/A	N/A
	Chewuch River	N/A	N/A	N/A
	Lost / Early River	N/A	N/A	N/A
	Entiat River	0.990	N/A	N/A
	Wenatchee River	1.010	N/A	N/A
	Chiawawa River	N/A	N/A	N/A
	Nason River	N/A	N/A	N/A
	Upper Wenatchee River	N/A	N/A	N/A
	White River	N/A	N/A	N/A
	Little Wenatchee River	N/A	N/A	N/A
Puget Sound (only have λ where hatchery fish = native fish), (Good et al., 2005)	Nooksack - North Fork	0.750	0.680	0.820
	Nooksack - South Fork	0.940	0.880	0.990
	Lower Skagit	1.050	0.960	1.140
	Upper Skagit	1.050	0.990	1.110
	Upper Cascade	1.060	1.010	1.110
	Lower Sauk	1.010	0.890	1.130
	Upper Sauk	0.960	0.900	1.020
	Suiattle	0.990	0.930	1.050
	Stillaguamish - North Fork	0.920	0.880	0.960
	Stillaguamish - South Fork	0.990	0.970	1.010
	Skykomish	0.870	0.840	0.900
	Snoqualmie	1.000	0.960	1.040
	North Lake Washington	1.070	1.000	1.140
	Cedar	0.990	0.920	1.060
	Green	0.670	0.610	0.730
	White	1.160	1.100	1.220
	Puyallup	0.950	0.890	1.010
	Nisqually	1.040	0.970	1.110
	Skokomish	1.040	1.000	1.080
	Dosewallips	1.170	1.070	1.270
	Duckabush	N/A	N/A	N/A
	Hamma Hamma	N/A	N/A	N/A
	Mid Hood Canal	N/A	N/A	N/A
	Dungeness	1.090	0.980	1.200
	Elwha	0.950	0.840	1.060
Sacramento River Winter - Run (Good, 2005 - 90% CI))	Sacramento River - winter run	0.970	0.870	1.090

Chinook Salmon (continued)

ESU	Population	λ - H=0	95% CI -lower	95% CI - upper
Snake River Fall - Run (Good, 2005)	Lower Snake River	1.024	N/A	N/A
Snake River Spring/Summer - Run (FCRPS)	Tucannon River	1.000	N/A	N/A
	Wenaha River	1.100	N/A	N/A
	Wallowa River	N/A	N/A	N/A
	Lostine River	1.050	N/A	N/A
	Minam River	1.050	N/A	N/A
	Catherine Creek	0.970	N/A	N/A
	Upper Grande Ronde River	N/A	N/A	N/A
	South Fork Salmon River	1.110	N/A	N/A
	Secesh River	1.070	N/A	N/A
	Johnson Creek	N/A	N/A	N/A
	Big Creek Spring Run	1.090	N/A	N/A
	Big Creek Summer Run	1.090	N/A	N/A
	Loon Creek	N/A	N/A	N/A
	Marsh Creek	1.080	N/A	N/A
	Bear Valley / Elk Creek	1.100	N/A	N/A
	North Fork Salmon River	N/A	N/A	N/A
	Lemhi River	1.020	N/A	N/A
	Pahsimeroi River	1.080	N/A	N/A
	East Fork Salmon Spring Run	1.040	N/A	N/A
	East Fork Salmon Summer Run	1.040	N/A	N/A
	Yankee Fork Spring Run	N/A	N/A	N/A
	Yankee Fork Summer Run	N/A	N/A	N/A
	Valley Creek Spring Run	N/A	N/A	N/A
	Valley Creek Summer Run	N/A	N/A	N/A
	Upper Salmon Spring Run	1.060	N/A	N/A
	Upper Salmon Summer Run	1.060	N/A	N/A
	Alturas Lake Creek	N/A	N/A	N/A
	Imnaha River	1.050	N/A	N/A
	Big Sheep Creek	N/A	N/A	N/A
	Lick Creek	N/A	N/A	N/A
Upper Williamette River (McElhany et al., 2007)	Clackamas River	0.967	0.849	1.102
	Molalla River	N/A	N/A	N/A
	North Santiam River	N/A	N/A	N/A
	South Santiam River	N/A	N/A	N/A
	Calapooia River	N/A	N/A	N/A
	McKenzie River	0.927	0.761	1.129
	Middle Fork Williamette River	N/A	N/A	N/A
	Upper Fork Williamette River	N/A	N/A	N/A

Chum Salmon

ESU	Population	λ - H=0	95% CI -lower	95% CI - upper
Columbia River	Youngs Bay	N/A	N/A	N/A
	Grays River	0.954	0.855	1.064
	Big Creek	N/A	N/A	N/A
	Elochoman River	N/A	N/A	N/A
	Clatskanie River	N/A	N/A	N/A
	Mill, Abernathy and German Creeks	N/A	N/A	N/A
	Scappose Creek	N/A	N/A	N/A
	Cowlitz River	N/A	N/A	N/A
	Kalama River	N/A	N/A	N/A
	Lewis River	N/A	N/A	N/A
	Salmon Creek	N/A	N/A	N/A
	Clackamus River	N/A	N/A	N/A
	Sandy River	N/A	N/A	N/A
	Washougal River	N/A	N/A	N/A
	Lower Gorge tributaries	0.984	0.883	1.096
	Upper Gorge tributaries	N/A	N/A	N/A
Hood Canal Summer - Run (only have λ where hatchery fish reproductive potential = native fish; Good et. al., 2005)	Jimmycomelately Creek	0.850	0.690	1.010
	Salmon / Snow Creeks	1.230	1.130	1.330
	Big / Little Quilcene rivers	1.390	1.170	1.610
	Lilliwaup Creek	1.190	0.750	1.630
	Hamma Hamma River	1.300	1.110	1.490
	Duckabush River	1.100	0.930	1.270
	Dosewallips River	1.170	0.930	1.410
	Union River	1.150	1.050	1.250
	Chimacum Creek	N/A	N/A	N/A
	Big Beef Creek	N/A	N/A	N/A
	Dewetto Creek	N/A	N/A	N/A

Coho Salmon

ESU	Population	λ - H=0	95% CI -lower	95% CI - upper
Central California Coast	Ten Mile River	N/A	N/A	N/A
	Noyo River	N/A	N/A	N/A
	Big River	N/A	N/A	N/A
	Navarro River	N/A	N/A	N/A
	Garcia River	N/A	N/A	N/A
	Other Mendacino County Rivers	N/A	N/A	N/A
	Gualala River	N/A	N/A	N/A
	Russain River	N/A	N/A	N/A
	Other Sonoma County Rivers	N/A	N/A	N/A
	Martin County	N/A	N/A	N/A
	San Mateo County	N/A	N/A	N/A
	Santa Cruz County	N/A	N/A	N/A
	San Lorenzo River	N/A	N/A	N/A
Lower Columbia River (Good et al., 2005)	Youngs Bay	N/A	N/A	N/A
	Grays River	N/A	N/A	N/A
	Elochoman River	N/A	N/A	N/A
	Clatskanie River	N/A	N/A	N/A
	Mill, Abernathy, Germany Creeks	N/A	N/A	N/A
	Scappose Creek	N/A	N/A	N/A
	Cispus River	N/A	N/A	N/A
	Tilton River	N/A	N/A	N/A
	Upper Cowlitz River	N/A	N/A	N/A
	Lower Cowlitz River	N/A	N/A	N/A
	North Fork Toutle River	N/A	N/A	N/A
	South Fork Toutle River	N/A	N/A	N/A
	Coweeman River	N/A	N/A	N/A
	Kalama River	N/A	N/A	N/A
	North Fork Lewis River	N/A	N/A	N/A
	East Fork Lewis River	N/A	N/A	N/A
	Upper Clackamas River	1.028	0.898	1.177
	Lower Clackamas River	N/A	N/A	N/A
	Salmon Creek	N/A	N/A	N/A
	Upper Sandy River	1.102	0.874	1.172
	Lower Sandy River	N/A	N/A	N/A
	Washougal River	N/A	N/A	N/A
	Lower Columbia River gorge tributaries	N/A	N/A	N/A
	White Salmon	N/A	N/A	N/A
	Upper Columbia River gorge tributaries	N/A	N/A	N/A
	Hood River	N/A	N/A	N/A

Coho Salmon (continued)

ESU	Population	λ - H=0	95% CI -lower	95% CI - upper
Southern Oregon and Northern California Coast	Southern Oregon and Northern California Coast	N/A	N/A	N/A
Oregon Coast	Necanicum	N/A	N/A	N/A
	Nehalem	N/A	N/A	N/A
	Tillamook	N/A	N/A	N/A
	Nestucca	N/A	N/A	N/A
	Siletz	N/A	N/A	N/A
	Yaquima	N/A	N/A	N/A
	Alsea	N/A	N/A	N/A
	Siuslaw	N/A	N/A	N/A
	Umpqua	N/A	N/A	N/A
	Coos	N/A	N/A	N/A
	Coquille	N/A	N/A	N/A

Sockeye Salmon

ESU	Population	λ - H=0	95% CI -lower	95% CI - upper
Ozette Lake	Ozette Lake	N/A	N/A	N/A
Snake River	Snake River	N/A	N/A	N/A

Steelhead

DPS	Population	λ - H=0	95% CI -lower	95% CI - upper
Central California Coast (Good et al., 2005)	Russain River	N/A	N/A	N/A
	Lagunitas	N/A	N/A	N/A
	San Gregorio	N/A	N/A	N/A
	Waddell Creek	N/A	N/A	N/A
	Scott Creek	N/A	N/A	N/A
	San Vincente Creek	N/A	N/A	N/A
	San Lorenzo River	N/A	N/A	N/A
	Soquel Creek	N/A	N/A	N/A
	Aptos Creek	N/A	N/A	N/A
California Central Valley (Good et al., 2005)	Sacramento River	0.950	0.900	1.020
Lower Columbia River (Good et al., 2005)	Cispus River	N/A	N/A	N/A
	Tilton River	N/A	N/A	N/A
	Upper Cowlitz River	N/A	N/A	N/A
	Lower Cowlitz River	N/A	N/A	N/A
	Coweeman River	0.908	0.792	1.041
	South Fork Toutle River	0.938	0.830	1.059
	North Fork Toutle River	1.062	0.915	1.233
	Kalama River - winter run	1.010	9.130	1.117
	Kalama River - summer run	0.981	0.889	1.083
	North Fork Lewis River - winter run	N/A	N/A	N/A
	North Fork Lewis River - summer run	N/A	N/A	N/A
	East Fork Lewis River - winter run	N/A	N/A	N/A
	East Fork Lewis River - summer run	N/A	N/A	N/A
	Salmon Creek	N/A	N/A	N/A
	Washougal River - winter run	N/A	N/A	N/A
	Washougal River - summer run	1.003	0.884	1.138
	Clackamas River	0.971	0.901	1.047
	Sandy River	0.945	0.850	1.051
	Lower Columbia gorge tributaries	N/A	N/A	N/A
	Upper Columbia gorge tributaries	N/A	N/A	N/A

Steelhead (continued)

DPS	Population	λ - H=0	95% CI -lower	95% CI - upper
Middle Columbia River (Good et al., 2005)	Klickitat River	N/A	N/A	N/A
	Yakima River	1.009	N/A	N/A
	Fifteenmile Creek	0.981	N/A	N/A
	Deschutes River	1.022	N/A	N/A
	John Day - upper main stream	0.975	N/A	N/A
	John Day - lower main stream	0.981	N/A	N/A
	John Day - upper north fork	1.011	N/A	N/A
	John Day - lower north fork	1.013	N/A	N/A
	John Day - middle fork	0.966	N/A	N/A
	John Day - south fork	0.967	N/A	N/A
	Umatilla River	1.007	N/A	N/A
	Touchet River	0.961	N/A	N/A
Northern California (Good et al., 2005)	Redwood Creek	N/A	N/A	N/A
	Mad River - winter run	1.000	0.930	1.050
	Eel River - summer run	0.980	0.930	1.040
	Mattole River	N/A	N/A	N/A
	Ten Mile river	N/A	N/A	N/A
	Noyo River	N/A	N/A	N/A
	Big River	N/A	N/A	N/A
	Navarro River	N/A	N/A	N/A
	Garcia River	N/A	N/A	N/A
	Gualala River	N/A	N/A	N/A
	Other Humboldt County streams	N/A	N/A	N/A
	Other Mendocino County streams	N/A	N/A	N/A
Puget Sound*	Puget Sound	N/A	N/A	N/A
Snake River (Good et al., 2005)	Tucannon River	0.886	N/A	N/A
	Lower Granite run	0.994	N/A	N/A
	Snake A run	0.998	N/A	N/A
	Snake B run	0.927	N/A	N/A
	Asotin Creek	N/A	N/A	N/A
	Upper Grande Ronde River	0.967	N/A	N/A
	Joseph Creek	1.069	N/A	N/A
	Imnaha River	1.045	N/A	N/A
	Camp Creek	1.077	N/A	N/A
South-Central California Coast	South-Central California Coast	N/A	N/A	N/A
Southern California	Santa Ynez River	N/A	N/A	N/A
	Ventura River	N/A	N/A	N/A
	Matilija River	N/A	N/A	N/A
	Creek River	N/A	N/A	N/A
	Santa Clara River	N/A	N/A	N/A

Steelhead (continued)

DPS	Population	λ - H=0	95% CI -lower	95% CI - upper
Upper Columbia River (Good et al., 2005)	Wenatchee / Entiat Rivers	1.067	N/A	N/A
	Methow / Okanogan Rivers	1.086	N/A	N/A
Upper Williamette River (McElhany et al., 2007)	Molalla River	0.988	0.790	1.235
	North Santiam River	0.983	0.789	1.231
	South Santiam River	0.976	0.855	1.114
	Calapooia River	1.023	0.743	1.409

Appendix 3: Abbreviations / 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 Divsion
BLM	Bureau of Land Management
BMP	Best Management Practices
BOR	Bureau of Reclaimation
BOR	Bureau of Reclamation
BPA	Bonneville Power Administration
BRT	Biological Review Team (NOAA Fisheries)
BY	Brood Years
CAISMP	California Aquatic Invasive Species Management Plan
CALFED	CALFED Bay-Delta Program (California Resource Agency)
CBFWA	Columbia Basin Fish and Wildlife Authority
CBI	Confidential Business Information
CC	California Coastal
CCC	Central California Coast
CCV	Central California Valley
CDPR	California Department of Pesticide Regulation
CHART	Critical Habitat Assessment Review Team
CIDMP	Comprehensive Irrigation District Management Plan
CFR	Code of Federal Regulations
cfs	cubic feet per second
CDFG	California Department of Fish and Game
Corps	U.S. Department of the Army Corps of Engineers

CSOs	combined sewer/stormwater overflows
CSWP	California State Water Project
CURES	Coalition for Urban/Rural Environmental Stewardship
CVP	Central Valley Projects
CVRWQCB	Central Valley Regional Water Quality Control Board
CWA	Clean Water Act
d	day
DCI	Date Call-Ins
DDD	Dichloro Diphenyl Dichloroethane
DDE	Diphenyl Dichlorethylene
DDT	Dichloro Diphenyl Trichloroethane
DER	Data Evaluation Review
DEQ	Oregon Department of Environmental Quality
DIP	Demographically Independent Population
DOE	Washington State Department of Ecology
DPS	Distinct Population Segment
EC	Emulsifiable Concentrate Pesticide Formulation
EC_{50}	Median Effect Concentration
EEC	Estimated Environmental Concentration
EFED	Environmental Fate and Effects Division
EIM	Environmental Information Management
EPA	U.S. Environmental Protection Agency
ESPP	Endangered Species Protection Program
ESA	Endangered Species Act
ESU	Evolutionarily Significant Unit
EU	European Union
EXAMS	Tier II Surface Water Computer Model
FERC	Federal Energy Regulatory Commission
FCRPS	Federal Columbia River Power System
FFDCA	Federal Food and Drug Cosmetic Act
FIFRA	Federal Insecticide, Fungicide, and Rodenticide Act

FQPA	Food Quality Protection Act
ft	feet
GENEEC	Generic Estimated Exposure Concentration
h	hour
HCP	Habitat Conservation Plan
HSRG	Hatchery Scientific Review Group
HUC	Hydrological Unit Code
IBI	Indices of Biological Integrity
ICTRT	Interior Columbia Technical Recovery Team
ILWP	Irrigated Lands Waiver Program
IPCC	Intergovernmental Panel on Climate Change
IRED	Interim Re-registration Decision
LCFRB	Lower Columbia Fish Recovery Board
ISG	Independent Science Group
ITS	Incidental Take Statement
km	kilometer
Lbs	Pounds
LC_{50}	Median Lethal Concentration.
LCR	Lower Columbia River
LOAEC	Lowest Observed Adverse Effect Concentration.
LOEL	Lowest Observed Adverse Effect level
LOC	Level of Concern
LOEC	Lowest Observed Effect Concentration
LOQ	Limit of Quantification
LRL	Laboratory Reporting Level
LWD	Large Woody Debris
m	meter
MCR	Middle Columbia River
mg/L	milligrams per liter
MOA	Memorandum of Agreement
MPG	Major Population Group

MRID	Master Record Identification Number
MTBE	Methyl tert-butyl ether
NASA	National Aeronautics and Space Administration
NAWQA	U.S. Geological Survey National Water-Quality Assessment
NC	Northern California
NEPA	National Environmental Protection Agency
NLCD	Natural Land Cover Data
NP	Nonylphenol
NPDES	National Pollutant Discharge Elimination System
NPS	National Parks Services
NRCS	Natural Resources Conservation Service
NWS	National Weather Service
NEPA	National Environmental Policy Act
NMA	National Mining Association
NMC	*N*-methyl carbamates
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
NOAEC	No Observed Adverse Effect Concentration
NPDES	National Pollution Discharge Eliminating System
NPIRS	National Pesticide Information Retrieval System
NRC	National Research Council
OC	Oregon Coast
ODFW	Oregon Division of Fish and Wildlife
OP	Organophosphates
Opinion	Biological Opinion
OPP	EPA Office of Pesticide Program
PAH	polyaromatic hydrocarbons
PBDEs	polybrominated diphenyl ethers
PCBs	polychlorinated biphenyls
PCEs	primary constituent elements
POP	Persistent Organic Pollutants

ppb	Parts Per Billion
PPE	Personal Protection Equipment
PSP	Pesticide Stewardship Partnerships
PSAMP	Puget Sound Assessment and Monitoring Program
PSAT	Puget Sound Action Team
PRIA	Pesticide Registration Improvement Act
PRZM	Pesticide Root Zone Model
PUR	Pesticide Use Reporting
QA/QC	Quality Assurance/Quality Control
RED	Re-registration Eligibility Decision
REI	Restricted Entry Interval
RPA	Reasonable and Prudent Alternatives
RPM	reasonable and prudent measures
RQ	Risk Quotient
SAP	Scientific Advisory Panel
SAR	smolt-to-adult return rate
SASSI	Salmon and Steelhead Stock Inventory
SC	Southern California
S-CCC	South-Central California Coast
SONCC	Southern Oregon Northern California Coast
SLN	Special Local Need (Registrations under Section 24(c) of FIFRA)
SR	Snake River
TCE	Trichloroethylene
TCP	3,5,6-trichloro-2-pyridinal
TGAI	Technical Grade Active Ingredient
TIE	Toxicity Identification Evaluation
TMDL	Total Maximum Daily Load
TRT	Technical Recovery Team
UCR	Upper Columbia River
USFS	United States Forest Service
USC	United States Code

USFWS	United States Fish and Wildlife Service
USGS	United States Geological Survey
UWR	Upper Willamette River
VOC	Volatile Organic Compounds
VSP	Viable Salmonid Population
WDFW	Washington Department of Fish and Wildlife
WLCRTRT	Willamette/Lower Columbia River Technical Review Team
WQS	Water Quality Standards
WWTIT	Western Washington Treaty Indian Tribes
WWTP	Wastewater Treatment Plant

Appendix 4: Glossary

303(d) waters Section 303 of the federal Clean Water Act requires states to prepare a list of all surface waters in the state for which beneficial uses – such as drinking, recreation, aquatic habitat, and industrial use - are impaired by pollutants. These are water quality limited estuaries, lakes, and streams that do not meet the state's surface water quality standards and are not expected to improve within the next two years. After water bodies are put on the 303(d) list they enter into a Total Maximum Daily Load Clean Up Plan.

Active ingredient	The component(s) that kills or otherwise affects the pest. A.i.s are always listed on the label (FIFRA 2(a)).
Adulticide	A compound that kills the adult life stage of the pest insect.
Anadromous Fish	Species that are hatched in freshwater migrate to and mature in salt water and return to freshwater to spawn.
Adjuvant	A compound that aides the operation or improves the effectiveness of a pesticide.
Alevin	Life history stage of a salmonid immediately after hatching and before the yolk-sac is absorbed. Alevins usually remain buried in the gravel in or near the egg nest (redd) until their yolk sac is absorbed when they swim up and enter the water column.
Anadromy	The life history pattern that features egg incubation and early juvenile development in freshwater migration to sea water for adult development, and a return to freshwater for spawning.
Assessment Endpoint	Explicit expression of the actual ecological value that is to be protected (*e.g.*, growth of juvenile salmonids).

Bioaccumulation	Accumulation through the food chain (*i.e.*, consumption of food, water/sediment) or direct water and/or sediment exposure.
Bioconcentration	Uptake of a chemical across membranes, generally used in reference to waterborne exposures.
Biomagnification	Transfer of chemicals via the food chain through two or more trophic levels as a result of bioconcentration and bioaccumulation.
Degradates	New compounds formed by the transformation of a pesticide by chemical or biological reactions.
Distinct Population Segment	A listable entity under the ESA that meets tests of discreteness and significance according to USFWS and NMFS policy. A population is considered distinct (and hence a "species" for purposes of conservation under the ESA) if it is discrete from an significant to the remainder of its species based n factors such as physical, behavioral, or genetic characteristics, it occupies an unusual or unique ecological setting, or its loss would represent a significant gap in the species' range.
Escapement	The number of fish that survive to reach the spawning grounds or hatcheries. The escapement plus the number of fish removed by harvest form the total run size.
Evolutionarily Significant Unit	A group of Pacific salmon or steelhead trout that is (1) substantially reproductively isolated from other conspecific units and (2) represent an important component of the evolutionary legacy of the species.
Fall Chinook	This salmon stock returns from the ocean in late summer and early

Salmon	fall to head upriver to its spawning grounds, distinguishing it from other stocks which migrate in different seasons.
Fate	Dispersal of a material in various environmental compartments (sediment, water air, biota) as a result of transport, transformation, and degradation.
Flowable	A pesticide formulation that can be mixed with water to form a suspension in a spray tank.
Fry	Stage in salmonid life history when the juvenile has absorbed its yolk sac and leaves the gravel of the redd to swim up into the water column. The fry stage follows the alevin stage and in most salmonid species is followed by the parr, fingerling, and smolt stages. However, chum salmon juveniles share characteristics of both the fry and smolt stages and can enter sea water almost immediately after becoming fry.
Half-pounder	A life history trait of steelhead exhibited in the Rogue, Klamath, Mad, and Eel Rivers of southern Oregon and northern California. Following smoltification, half-pounders spend only 2-4 months in the ocean, then return to fresh water. They overwinter in fresh water and emigrate to salt water again the following spring. This is often termed a false spawning migration, as few half-pounders are sexually mature.
Hatchery	Salmon hatcheries use artificial procedures to spawn adults and raise the resulting progeny in fresh water for release into the natural environment, either directly from the hatchery or by transfer into another area. In some cases, fertilized eggs are outplanted (usually in "hatch-boxes"), but it is more common to release fry or smolts.
Inert ingredients	"an ingredient which is not active" (FIFRA 2(m)). It may be toxic or enhance the toxicity of the active ingredient.

Iteroparous	Capable of spawning more than once before death
Jacks	Male salmon that return from the ocean to spawn one or more years before full-sized adults return. For coho salmon in California, Oregon, Washington, and southern British Columbia, jacks are 2 years old, having spent only 6 months in the ocean, in contrast to adults, which are 3 years old after spending 1 ½ years in the ocean.
Jills	Female salmon that return from the ocean to spawn one or more years before full-sized adult returns. For sockeye salmon in Oregon, Washington, and southern British Columbia, jills are 3 years old (age 1.1), having spent only one winter in the ocean in contrast to more typical sockeye salmon that are age 1.2, 1.32.2, or 2.3 on return.
Kokanee	The self-perpetuating, non-anadromous form of *O. nerka* that occurs in balanced sex ration populations and whose parents, for several generations back, have spent their whole lives in freshwater.
Lambda	Also known as Population growth rate, or the rate at which the abundance of fish in a population increases or decreases.
LRL	Laboratory Reporting Level (USGS NAWQA data)- Generally equal to twice the yearly determined LT-MDL. The LRL controls false negative error. The probability of falsely reporting a non-detection for a sample that contained an analyte at a concentration equal to or greater that the LRL is predicted to be less than or equal to 1 percent.
Major Population Group (MPG)	A group of salmonid populations that are geographically and genetically cohesive. The MPG is a level of organization between demographically independent populations and the ESU.

Main channel	The stream channel that includes the thalweg (longitudinal continuous deepest portion of the channel.
Metabolite	A transformation product resulting from metabolism.
Mode of Action	A series of key processes that begins with the interaction of a pesticide with a receptor site and proceeds through operational and anatomical changes in an organisms that result in sublethal or lethal effects.
Natural fish	A fish that is produced by parents spawning in a stream or lake bed, as opposed to a controlled environment such as a hatchery.
Nonylphenols	A type of APE and is an example of an adjuvant that may be present as an ingredient of a formulated product or added to a tank mix prior to application.
Off-channel habitat	Water bodies and/or inundated areas that are connected (accessible to salmonid juveniles) seasonally or annually to the main channel of a stream including but not limited to features such as side channels, alcoves, ox bows, ditches, and floodplains.
Parr	The stage in anadromous salmonid development between absorption of the yolk sac and transformation to smolt before migration seaward.
Persistence	The tendency of a compound to remain in its original chemical form in the environment.
Pesticide	Any substance or mixture of substances intended for preventing, destroying, repelling or mitigating any pest.
Reasonable and	Recommended alternative actins identified during formal

Prudent Alternative (RPA)	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 area	Riparian habitats are the transitional zones between terrestrial and aquatic ecosystems and are distinguished by gradients in biological and physical conditions, ecological processes, and biota. They are areas through which surface and subsurface hydrology connect waterbodies with their adjacent uplands. They include those portions of terrestrial ecosystems that significantly influence exchanges of energy and matter with aquatic ecosystems (i.e. zone of influence). Riparian areas are the products of water and material interactions in three dimensions – longitudinal, lateral, and vertical. They include portions of the channel system and associated features (e.g., gravel bars, islands, wood debris); a vegetated zone of varying successional states influenced by floods, sediment deposition, soil-formation processes, and water availability; and a transitional zone to the uplands of the valley wall – all underlain by an alluvial aquifer. Riparian areas are adjacent to rivers, and perennial, intermittent, and ephemeral streams, and lakes, and estuarine-marine shorelines.
Risk	The probability of harm from actual or predicted concentrations of a chemical in the aquatic environment – a scientific judgment.
Salmon bearing Waters	Fresh, brackish and marine waters accessible to salmonids.

Salmonid	Fish of the family *Salmonidae*, including salmon, trout, chars, grayling, and whitefish. In general usage, the term usually refers to salmon, trout, and chars.
SASSI	A cooperative program by WDFW and WWTIT to inventory and evaluate the status of Pacific salmonids in Washington State. The SASSI report is a series of publications from this program.
Semelparous	The condition in an individual organism of reproducing only once in a lifetime.
Smolt	A juvenile salmon or steelhead migrating to the ocean and undergoing physiological changes to adapt from freshwater to a saltwater environment.
Sublethal	Below the concentration that directly causes death. Exposure to sublethal concentrations of a material may produce less obvious effect on behavior, biochemical, and/or physiological function of the organism often leading to indirect death.
Surfactant	A substance that reduces the interfacial or surface tension of a system or a surface-active substance.
Synergism	A phenomenon in which the toxicity of a mixture of chemicals is greater than that which would be expected from a simple summation of the toxicities of the individual chemicals present in the mixture.
Technical Grade Active Ingredient (TGAI)	Pure or almost pure active ingredient. Available to formulators. Most toxicology data are developed with the TGAI. The percent AI is listed on all labels.

Technical Recovery Teams (TRT)	Teams convened by NOAA Fisheries to develop technical products related to recovery planning. TRTs are complemented by planning forums unique to specific states, tribes, or reigns, which use TRT and other technical products to identify recovery actions.
Teratogenic	Effects produced during gestation that evidence themselves as altered structural or functional processes in offspring.
Total Maximum Daily Load (TMDL)	defines how much of a pollutant a water body can tolerate (absorb) daily and remain compliant with applicable water quality standards. All pollutant sources in the watershed combined, including non-point sources, are limited to discharging no more than the TMDL.
Unique Mixture	A specific combination of 2 or more compounds, regardless of the presence of other compounds.
Viable Salmonid Population	An independent population of Pacific salmon or steelhead trout that has a negligible risk of extinction over a 100-year time frame. Viability at the independent population scale is evaluated based on the parameters of abundance, productivity, spatial structure, and diversity.
VSP Parameters	Abundance, productivity, spatial structure, and diversity. These describe characteristics of salmonid populations that are useful in evaluating population viability. See NOAA Technical Memorandum NMFS-NWFSC-, "Viable salmonid populations and the recovery of evolutionarily significant units," McElhany et al., June 2000.
WDFW	Washington Department of Fish and Wildlife is a co-manager of salmonids and salmonid fisheries in Washington State with WWTIT and other fisheries groups. The agency was formed in the early 1990s by the combination of the

Washington Department of Fisheries and the Washington Department of Wildlife.

WWTIT Western Washington Treaty Indian Tribes is an organization of Native American tribes with treaty fishing rights recognized by the U.S. government. WWTIT is a co-manager of salmonids and salmonid fisheries in western Washington in cooperation with the WDFW and other fisheries groups.

WQS "A water quality standard defines the water quality goals of a waterbody, or portion thereof, by designating the use or uses to be made of the water and by setting criteria necessary to protect public health or welfare, enhance the quality of water and serve the purposes of the Clean Water

Appendix 5: Co-occurrence Analysis for Integration and Synthesis

Our species viability assessment considers the spatial, temporal, and biological overlap of ESA-listed species with the stressors of the action. Where there is co-occurrence, salmonids may be exposed to and affected by the a.i. and its associated stressors.

Because pesticides are registered for specific uses, we determine where specific portions of the proposed action may be carried out based on the type of use. National Land Cover Database (NLCD) land use categories were used as a surrogate for use sites: cultivated crops or hay/pasture for a specific crop or crops; developed areas for residential and urban uses, pest control, and disease vector control; and managed forests for forestry applications. While cropping patterns may shift or lands may become fallow over a longer period of time, the NLCD dataset is the most relevant method of estimating exposure. As we cannot determine where a certain crop will be cultivated, we assume that any pesticide registered for use on an agricultural crop could be applied in an area defined as agricultural land use. We did consider differences in state regulations and SLN registrations, as well as general cropping trends for different basins.

However, we cannot determine where rights-of-way uses will occur based on land use information. We assume that rights-of-way will be concentrated in urban areas, but will also be present in rural areas as well. In more remote areas, roads and railroads are often situated along river valleys, sometimes in close proximity to the stream or river.

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.

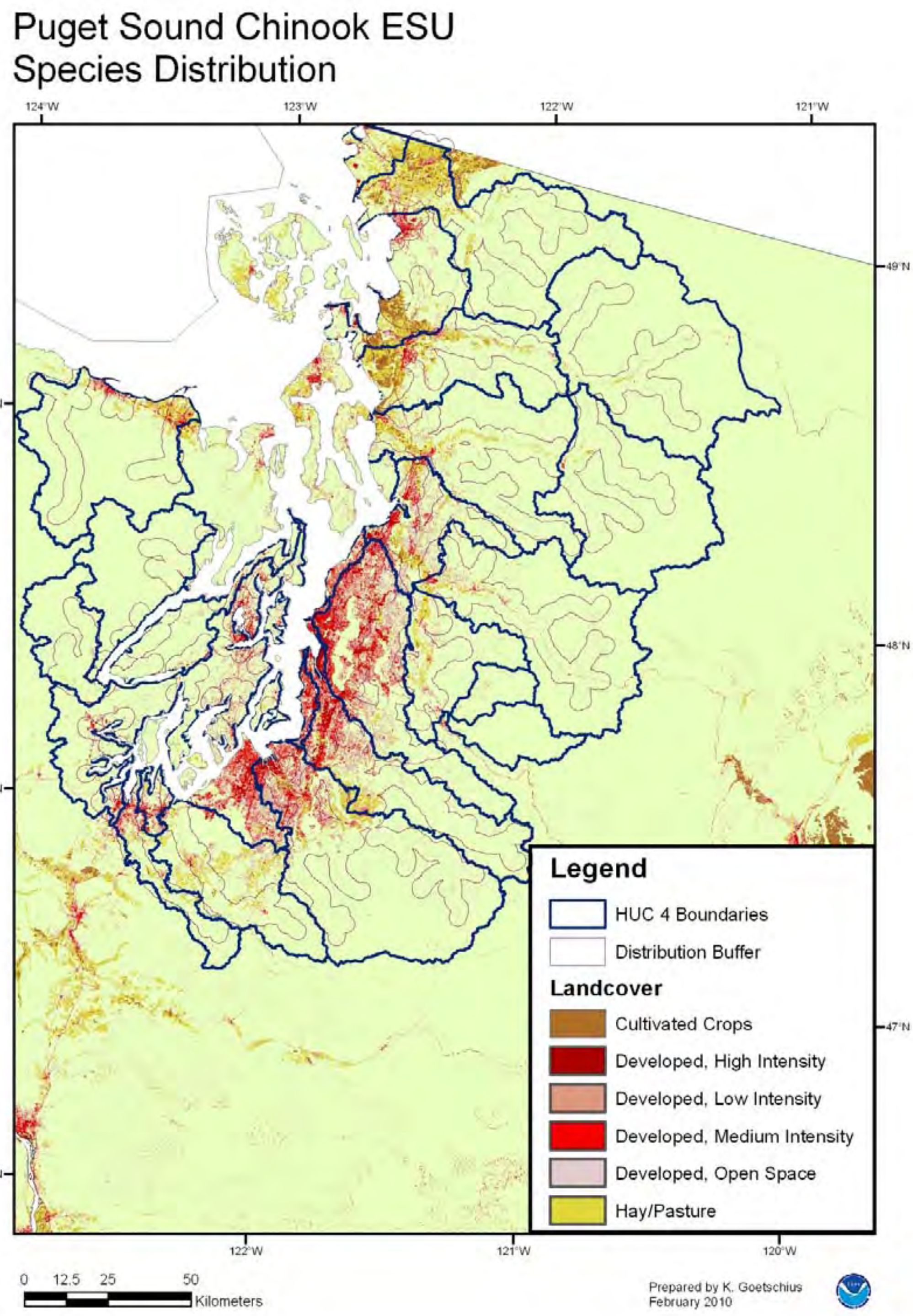
For all ESUs/DPSs, a 2.5 km “buffer” was created on each side of salmonid aquatic habitat. This distance was selected by the team as it is large enough to account for discrepancies between GIS layers due to channel alteration / migration, but not so large that it would encompass the entire range of an 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.

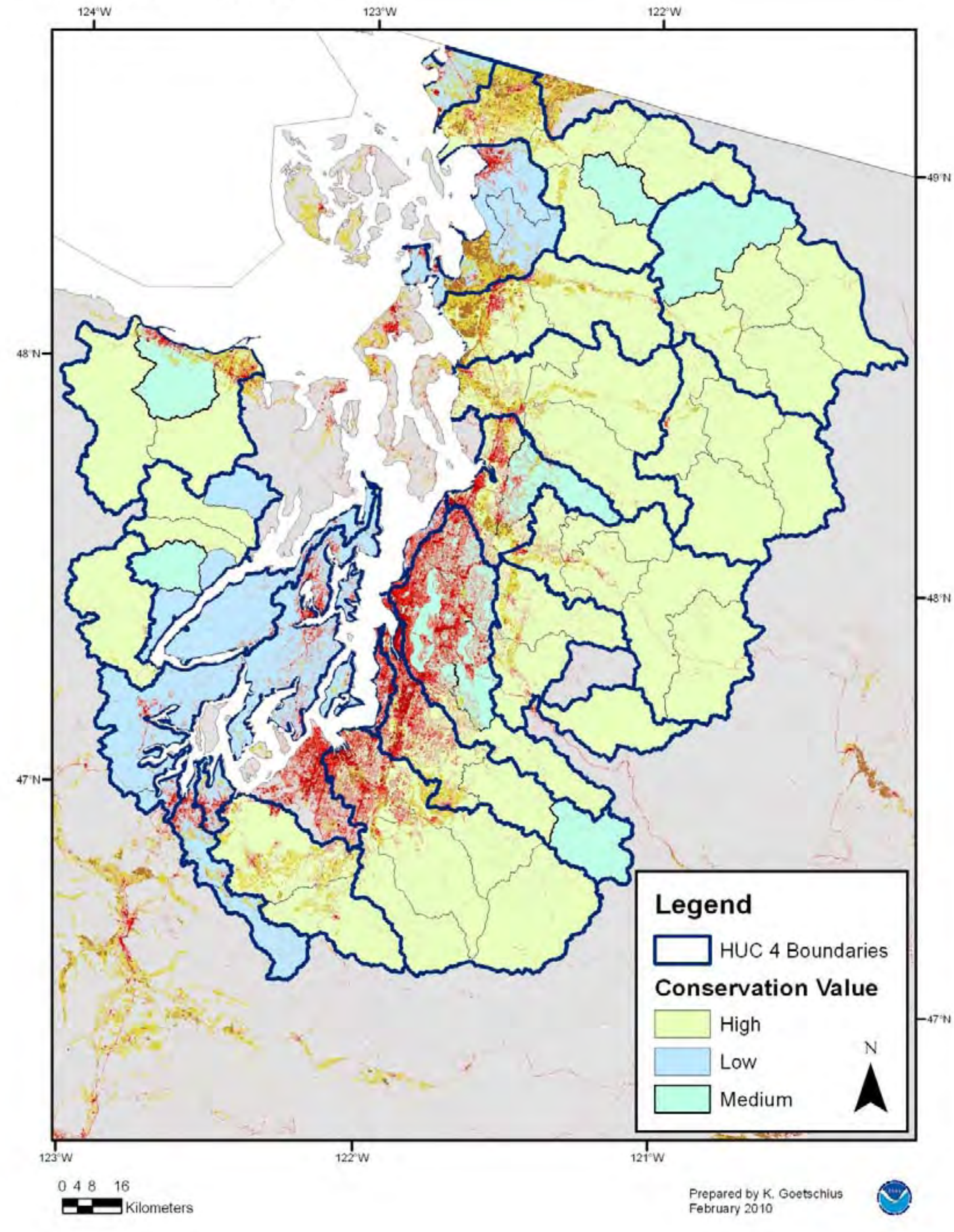
We further considered the existing environmental mixtures, seasonally elevated water temperatures, and other factors which influence the survival of the species, such as loss of habitat features, hydropower and water management conditions, and invasive species or predators. Other important factors that were taken into consideration include location of federal land, railroad lines, and electrical transmission lines.

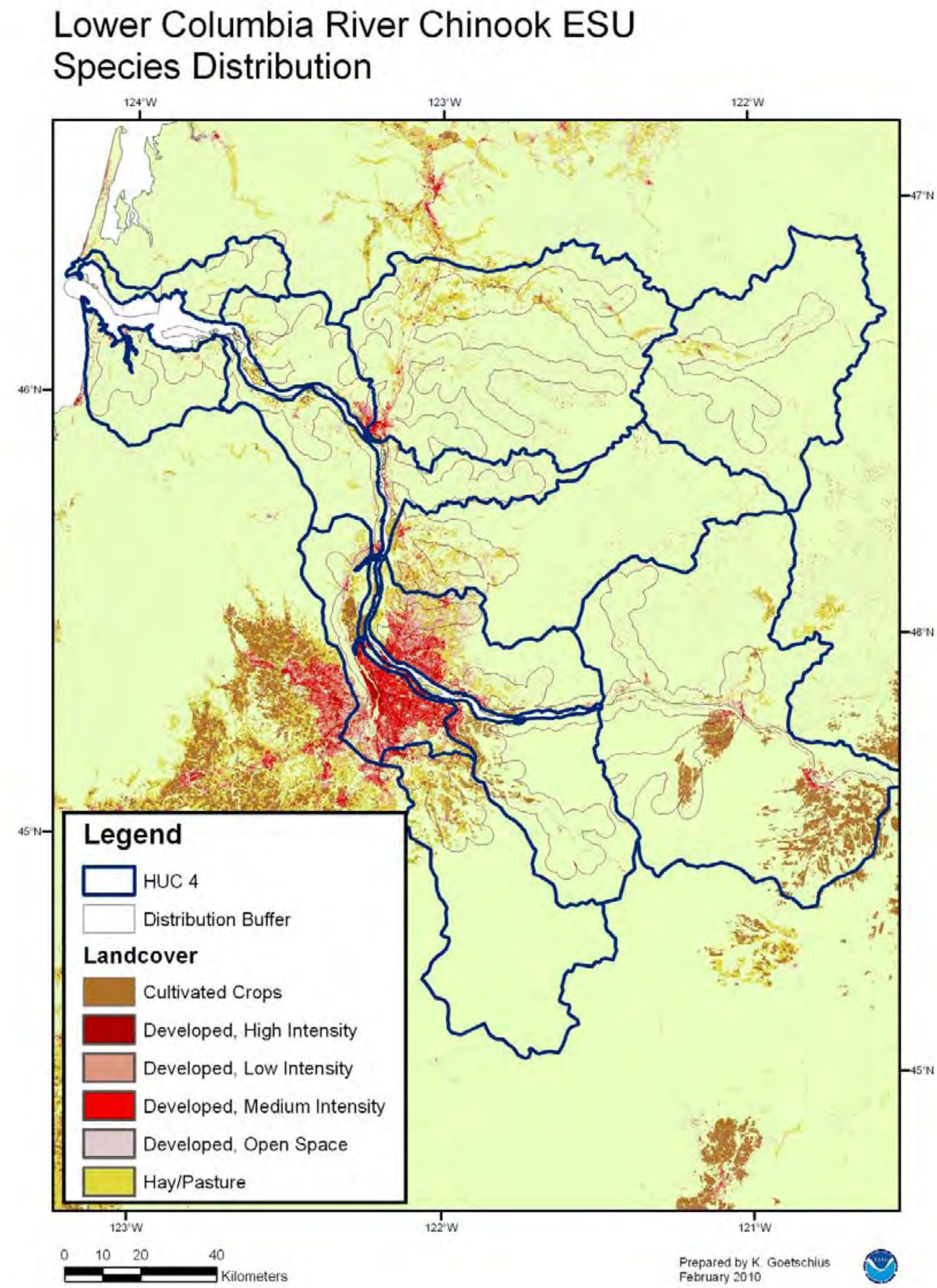
To illustrate the co-occurrence analysis process, this appendix includes two maps for each ESU/DPS. The first map shows the range of the ESU with each HUC 4 outlined in blue, the 2.5 km buffer in burgundy and relevant categories from the NLCD land use layer. This map aided in the Species analyses. The second map was used in the critical habitat analysis. For 19 of the species, conservation values have been assigned to the HUC 5 level units. In Idaho, Oregon, and Washington, these units are referred to as watersheds, while California uses the term “hydrological sub-area” or HSA. The Critical Habitat maps show either, (a) all designated HUC5s and their conservation values, or (b) the species map with the buffer removed. The exceptions to this are Snake River Fall-Run Chinook and Ozette Lake Sockeye, as they cover such small areas, and the two species for which critical habitat has not been designated (Columbia River Coho and Puget Sound Steelhead). These four species each only have one map. The following species have conservation values assigned by HUC5:

1. Puget Sound Chinook
2. Lower Columbia River Chinook
3. Upper Columbia River Spring Run Chinook
4. Upper Willamette River Chinook
5. California Coastal Chinook
6. Central Valley Spring Run Chinook
7. Columbia River Chum
8. Hood Canal Chum
9. Oregon Coast Coho
10. Lower Columbia River Steelhead
11. Middle Columbia River Steelhead
12. Upper Columbia River Steelhead
13. Upper Willamette River Steelhead
14. Snake River Steelhead
15. Northern California Steelhead
16. Central California Coast Steelhead
17. California Central Valley Steelhead
18. South-Central California Coast Steelhead
19. Southern California Steelhead


Puget Sound Chinook ESU
Species Distribution
124°W
123°W
122°W
121°W
49°N
48°N
47°N
46°N
48°N
47°N
122°W
121°W
120°W
Legend
HUC 4 Boundaries
Distribution Buffer
Landcover
Cultivated Crops
Developed, High Intensity
Developed, Low Intensity
Developed, Medium Intensity
Developed, Open Space
Hay/Pasture
0 12.5 25 50
Kilometers
Prepared by K. Goetschius
February 2010

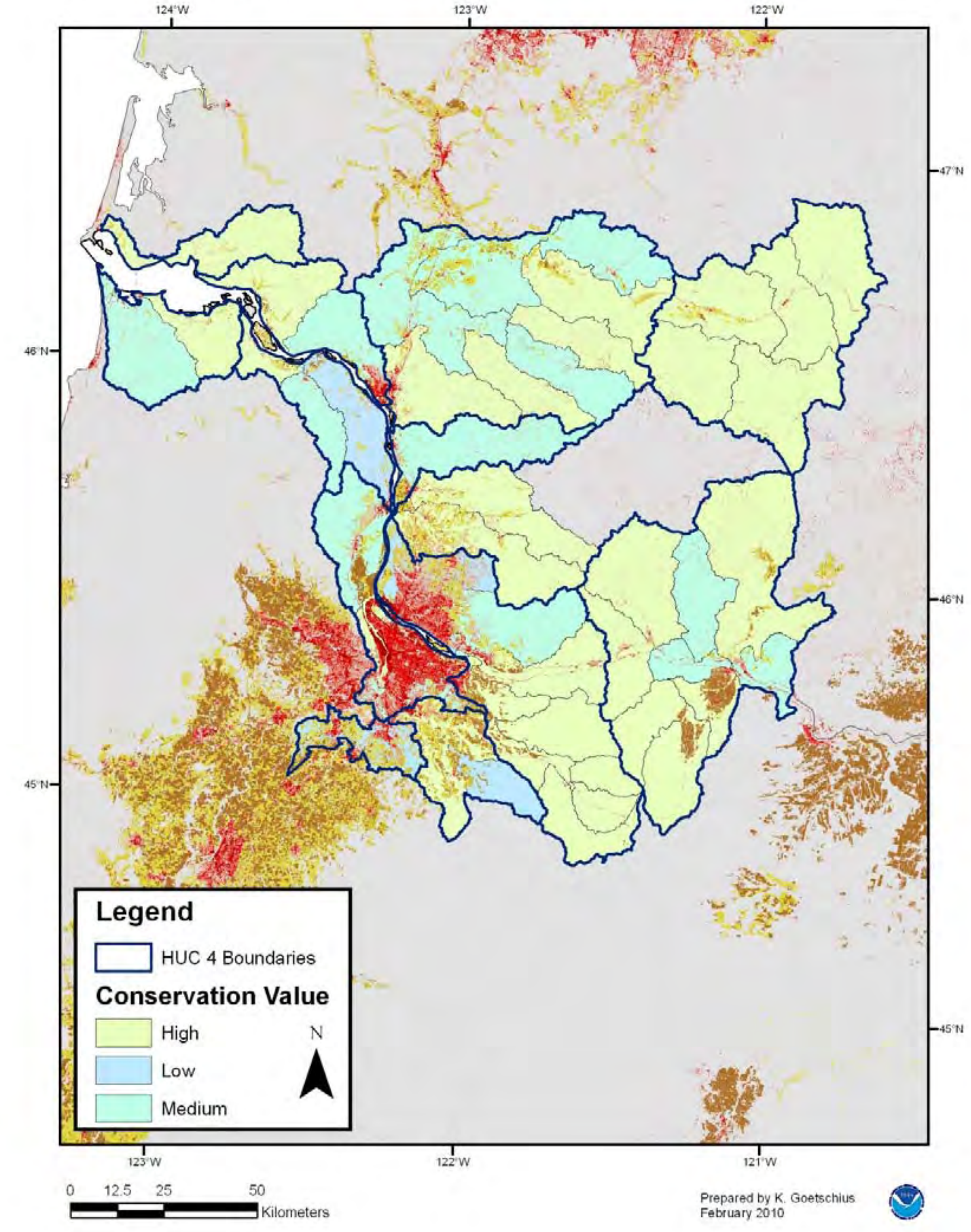
Puget Sound Chinook ESU
Critical Habitat



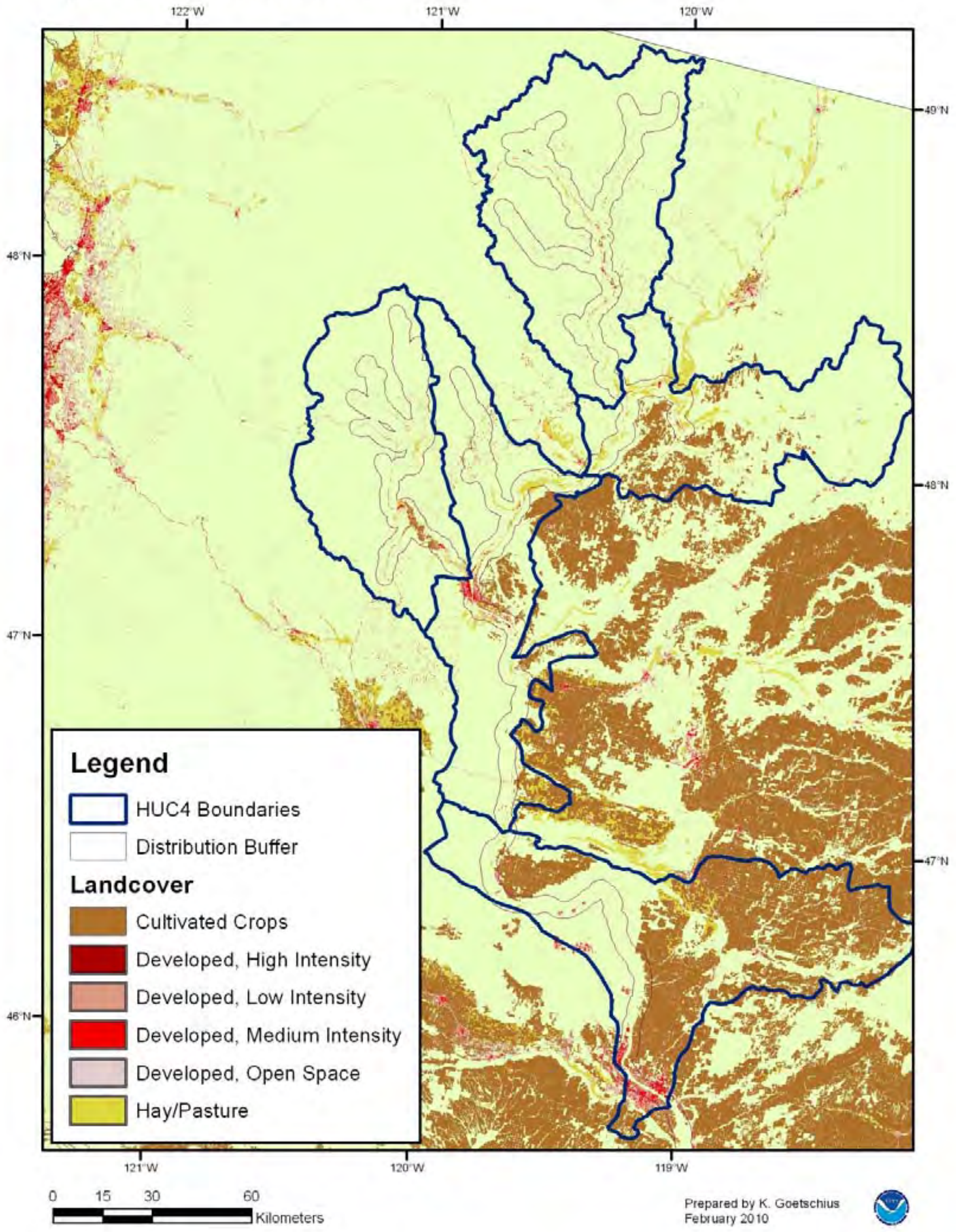


Lower Columbia River Chinook ESU
Species Distribution
124°W
123°W
122°W
47°N
46°N
45°N
121°W
Legend
HUC 4
Distribution Buffer
Landcover
Cultivated Crops
Developed, High Intensity
Developed, Low Intensity
Developed, Medium Intensity
Developed, Open Space
Hay/Pasture
0 10 20 40
Kilometers
Prepared by K. Goetschius
February 2010

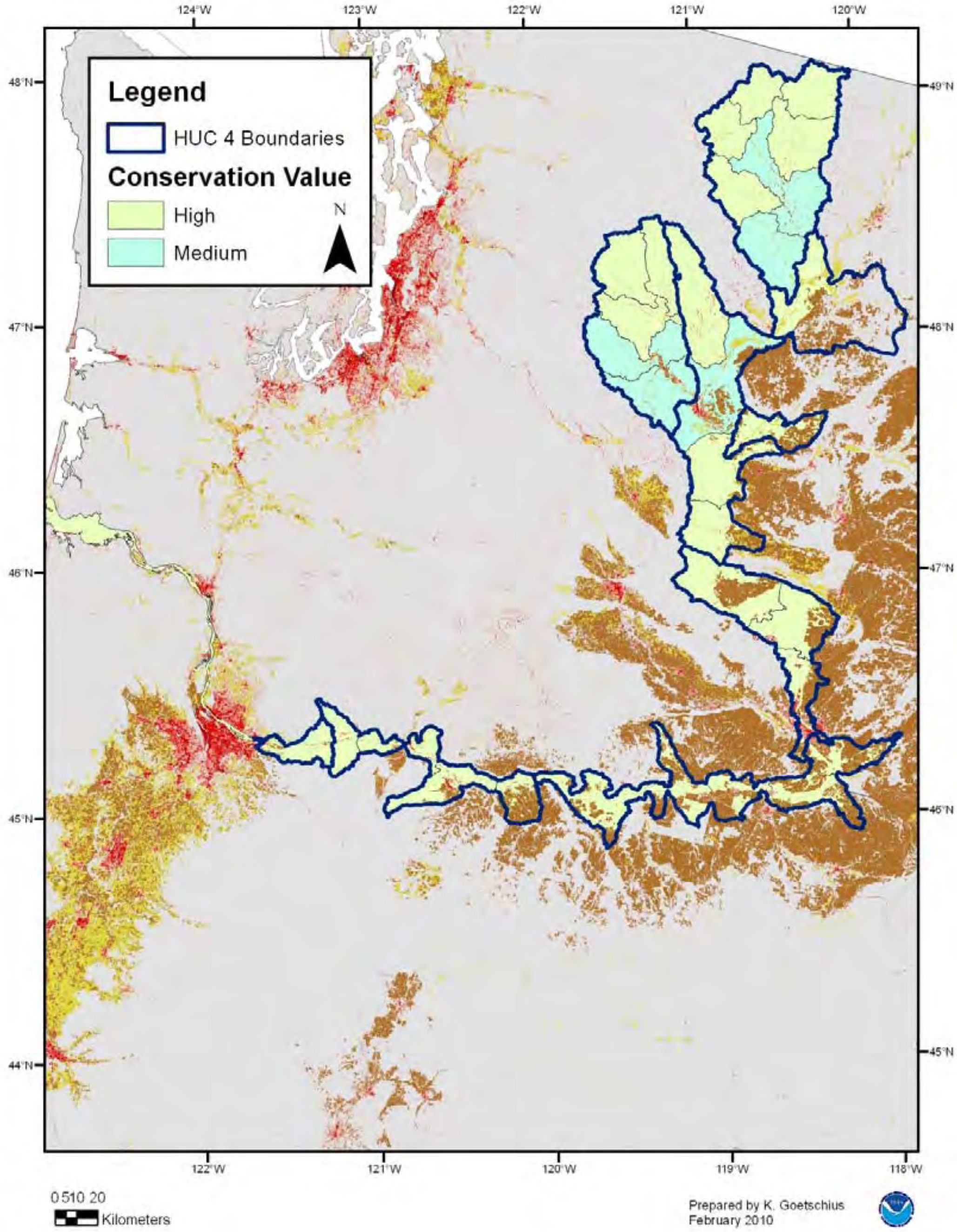
Lower Columbia River Chinook ESU
Critical Habitat

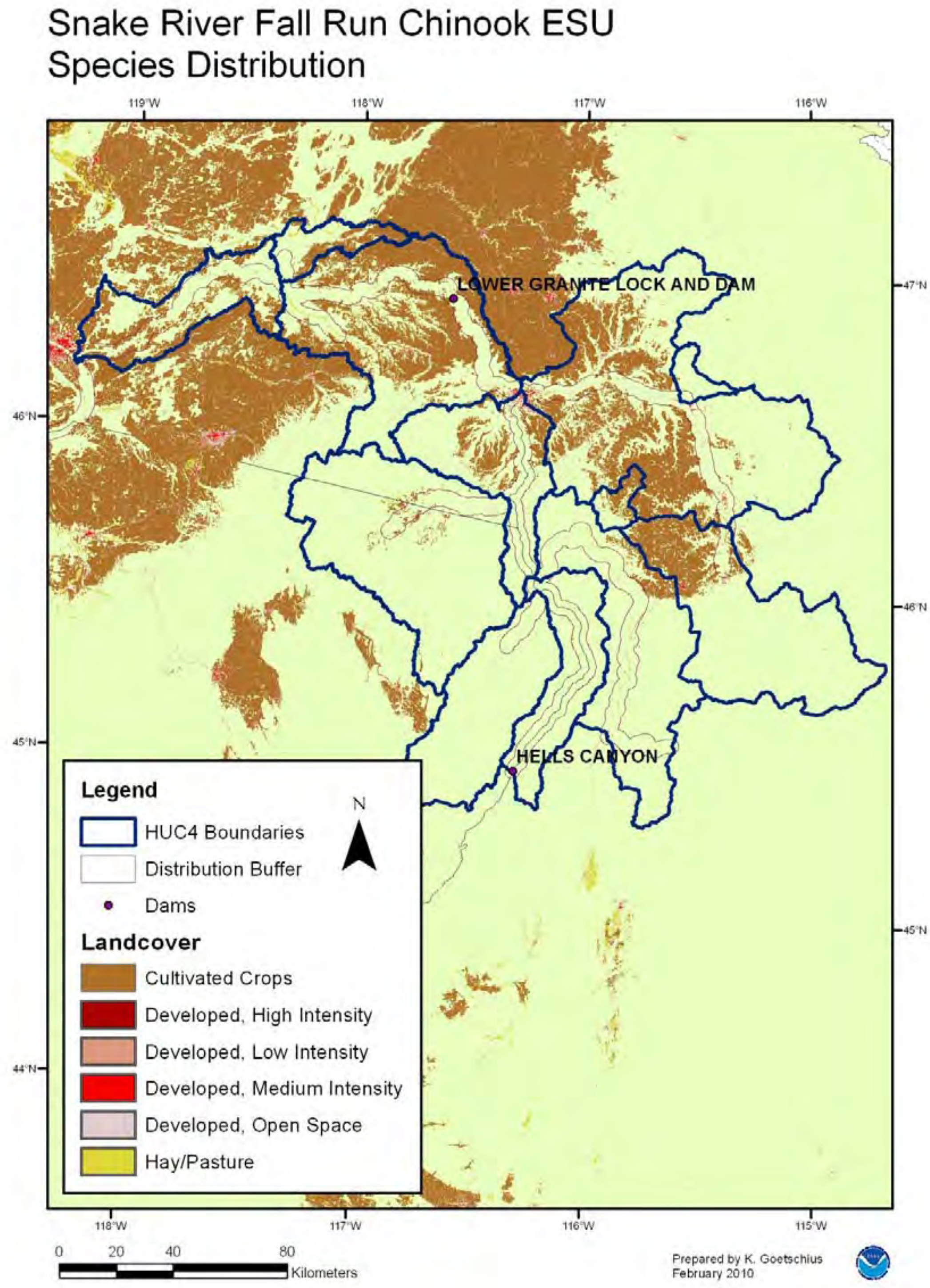


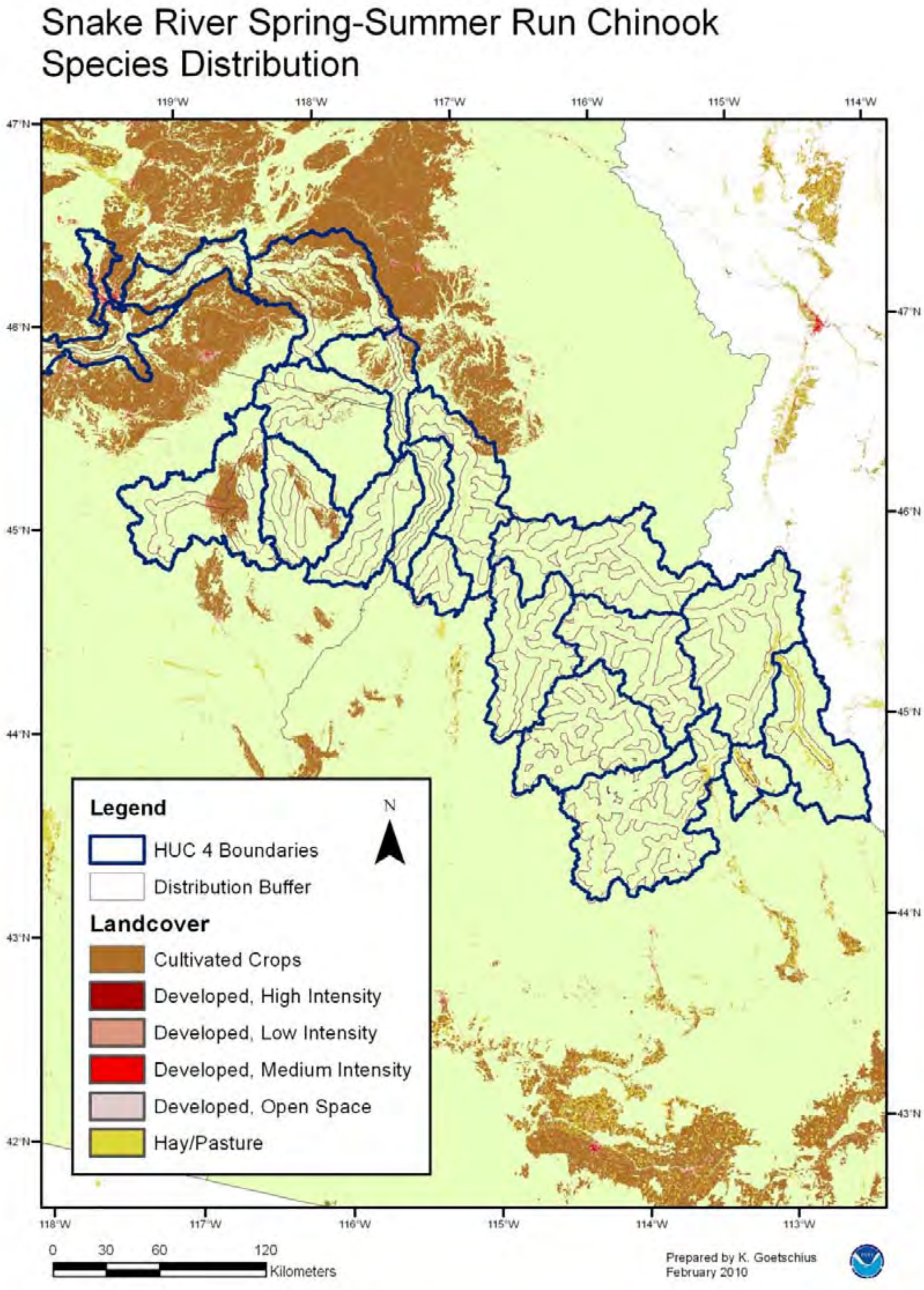
Upper Columbia Spring-Run Chinook ESU Species Distribution



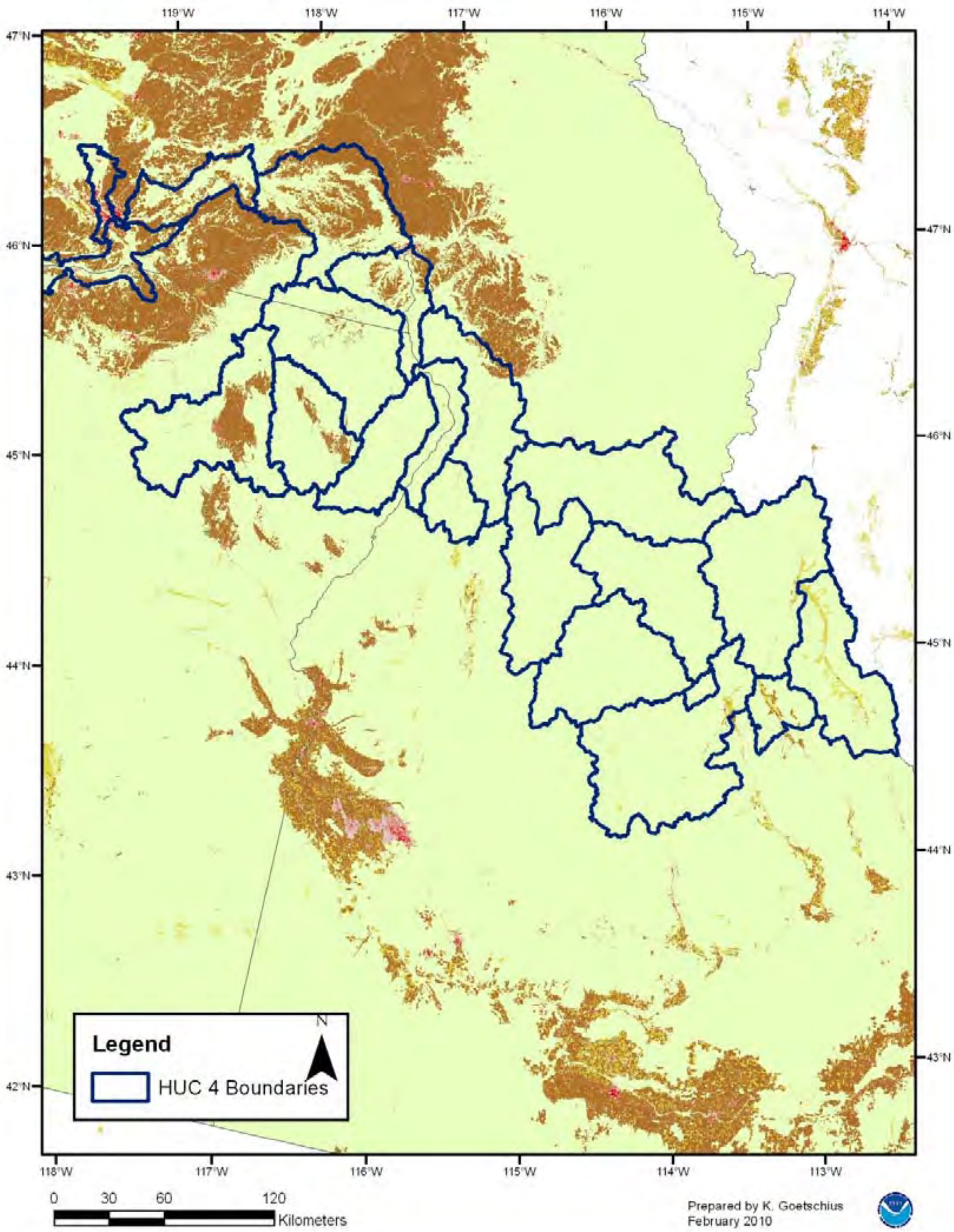
Upper Coulmbia River Chinook ESU
Critical Habitat

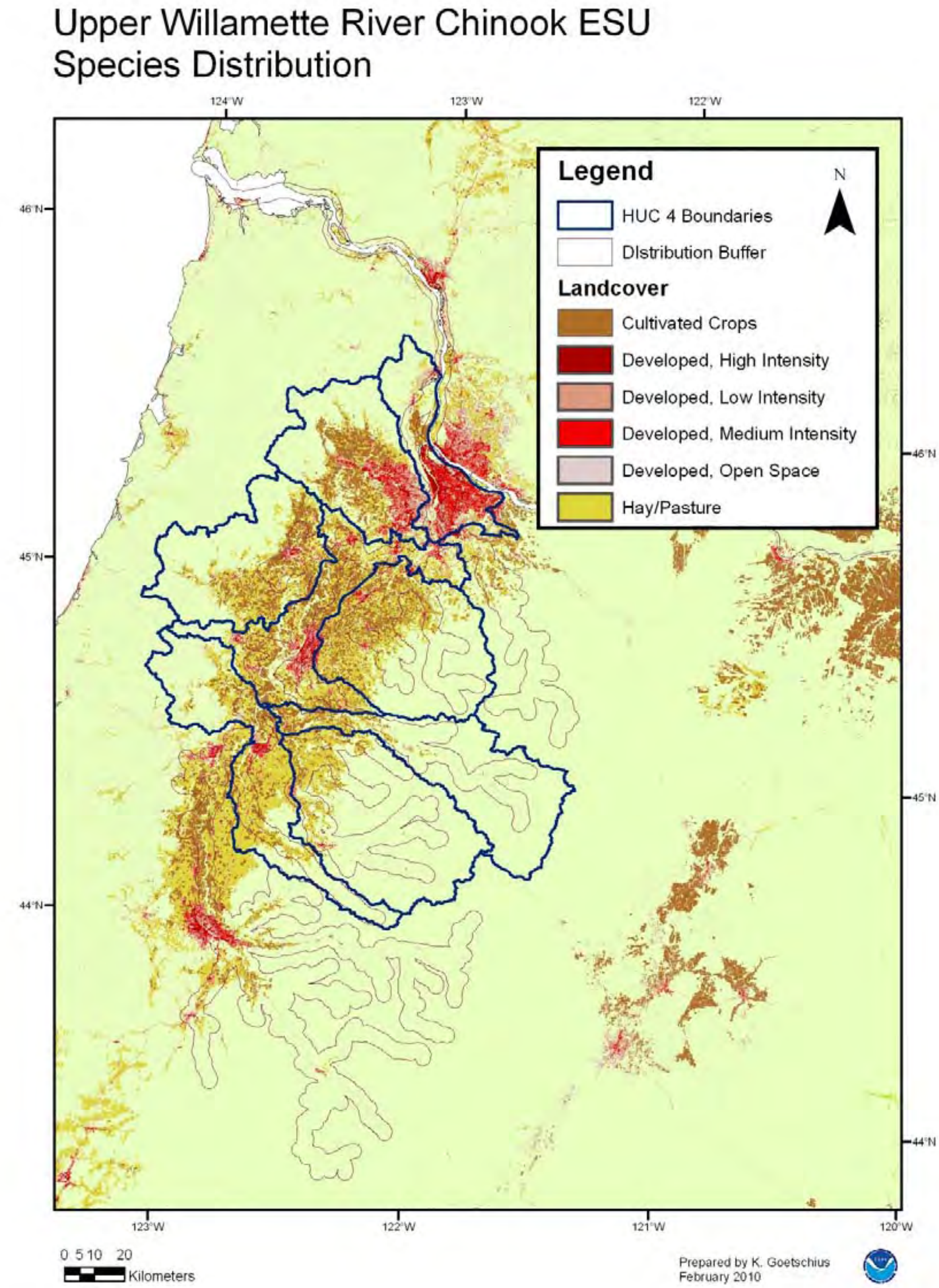



Snake River Fall Run Chinook ESU
Species Distribution
LOWER GRANITE LOCK AND DAM
HELLS CANYON
Legend
HUC4 Boundaries
Distribution Buffer
Dams
Landcover
Cultivated Crops
Developed, High Intensity
Developed, Low Intensity
Developed, Medium Intensity
Developed, Open Space
Hay/Pasture
N
0 20 40 80 Kilometers
Prepared by K. Goetschius
February 2010


Snake River Spring-Summer Run Chinook
Species Distribution
Legend
HUC 4 Boundaries
Distribution Buffer
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
February 2010

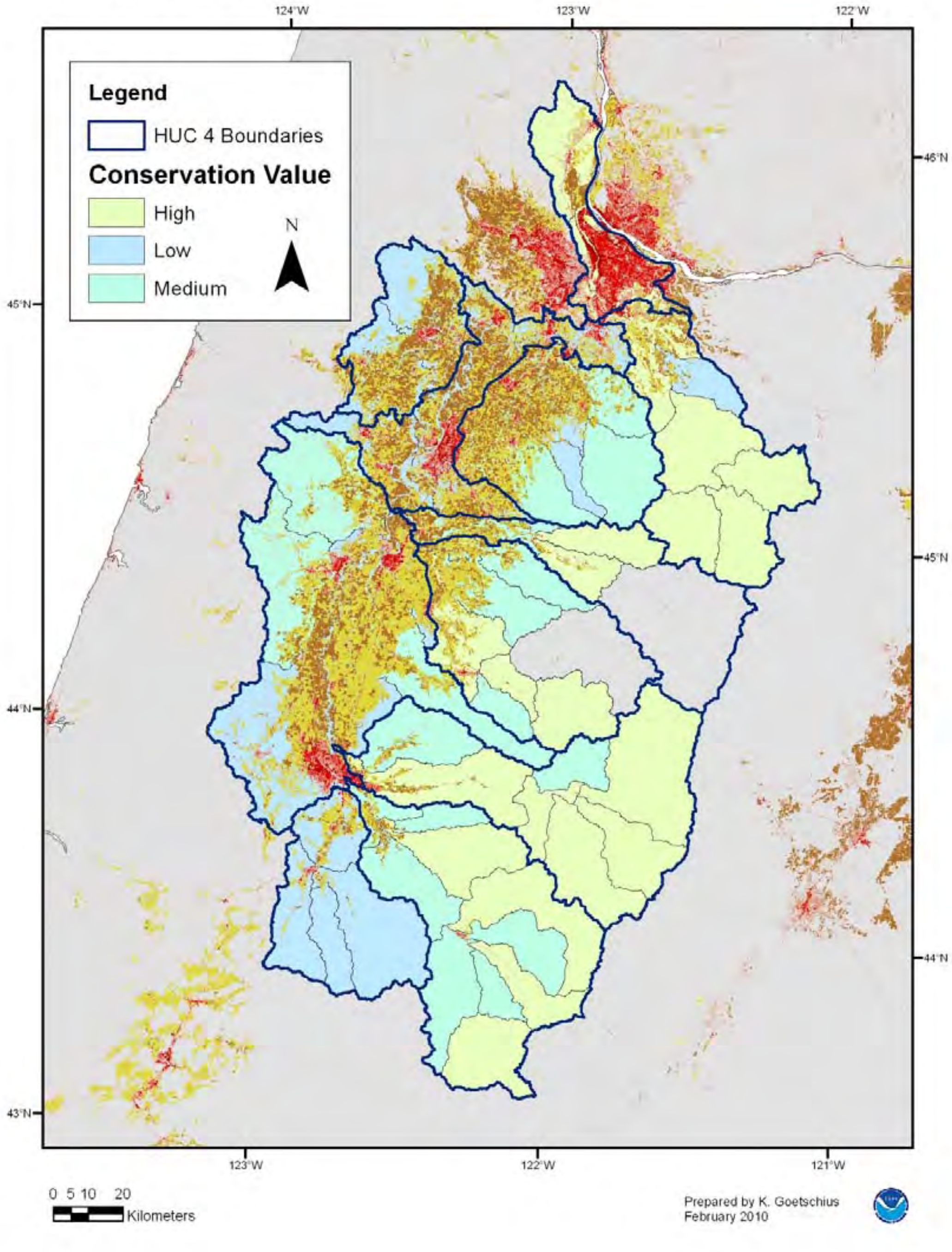
Snake River Spring-Summer Run Chinook Critical Habitat



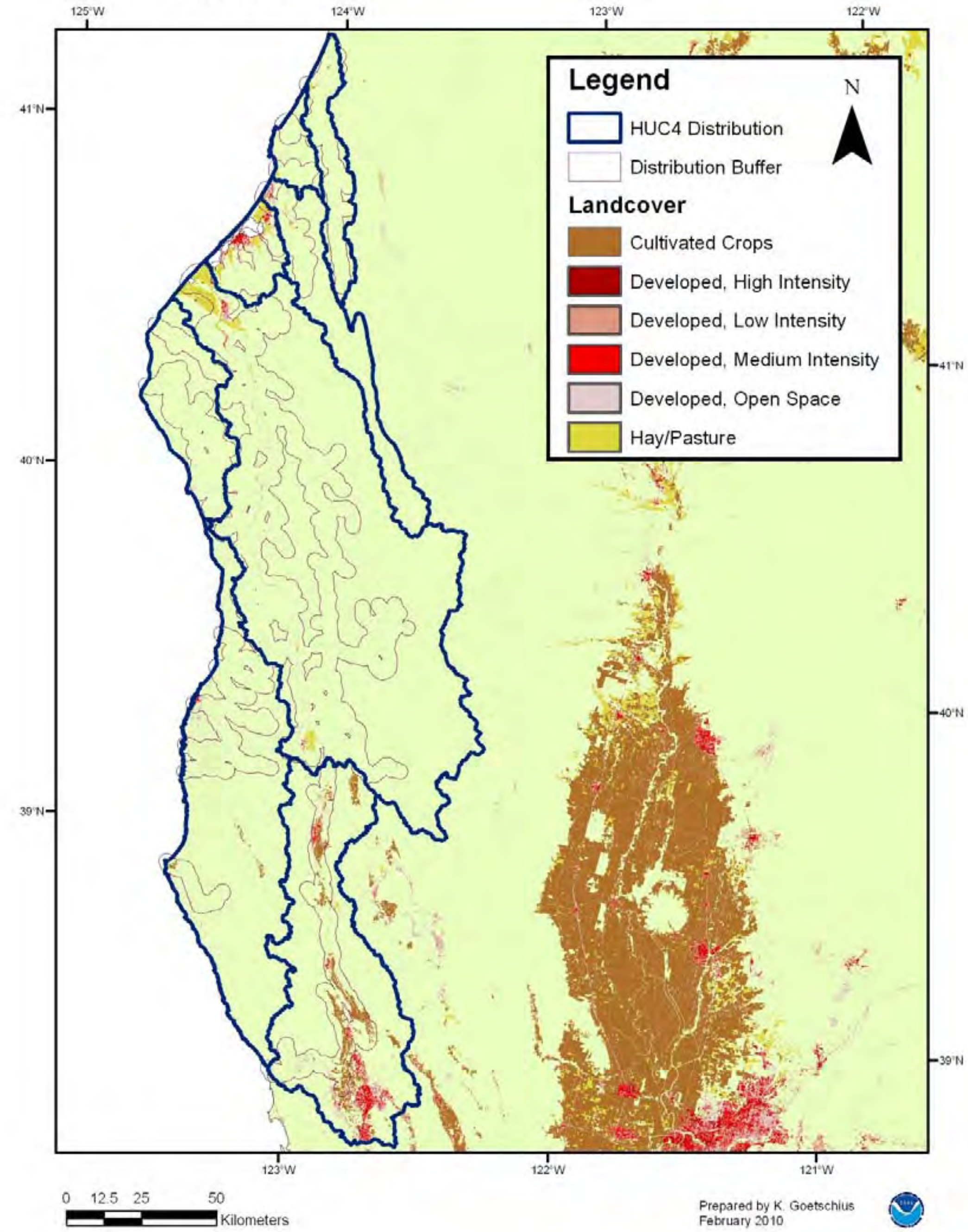


Upper Willamette River Chinook ESU
Species Distribution
Legend
HUC 4 Boundaries
DIstribution Buffer
Landcover
Cultivated Crops
Developed, High Intensity
Developed, Low Intensity
Developed, Medium Intensity
Developed, Open Space
Hay/Pasture
N
124°W
123°W
122°W
121°W
120°W
46°N
45°N
44°N
0 5 10 20
Kilometers
Prepared by K. Goetschius
February 2010

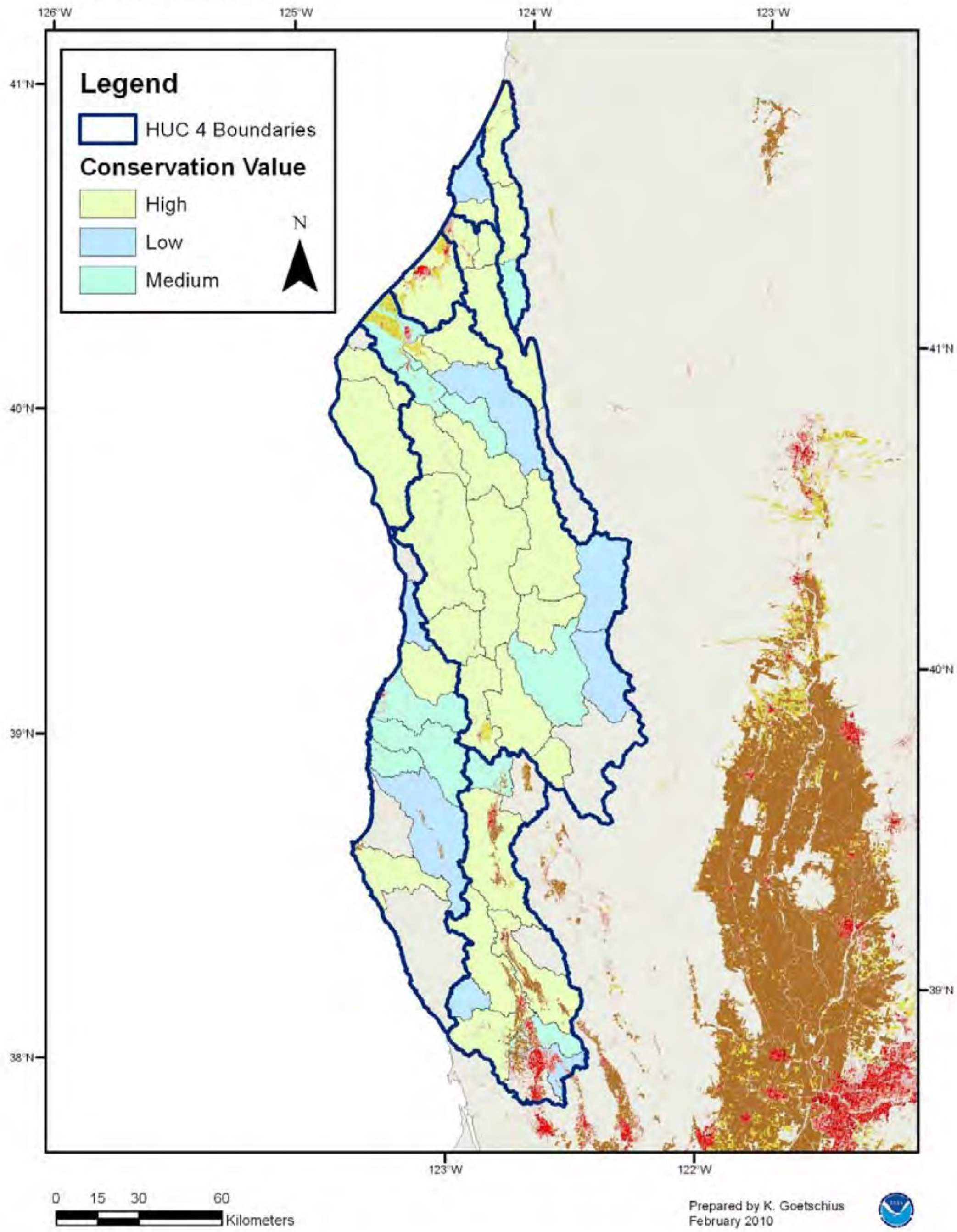
Upper Willamette River Chinook ESU Critical Habitat



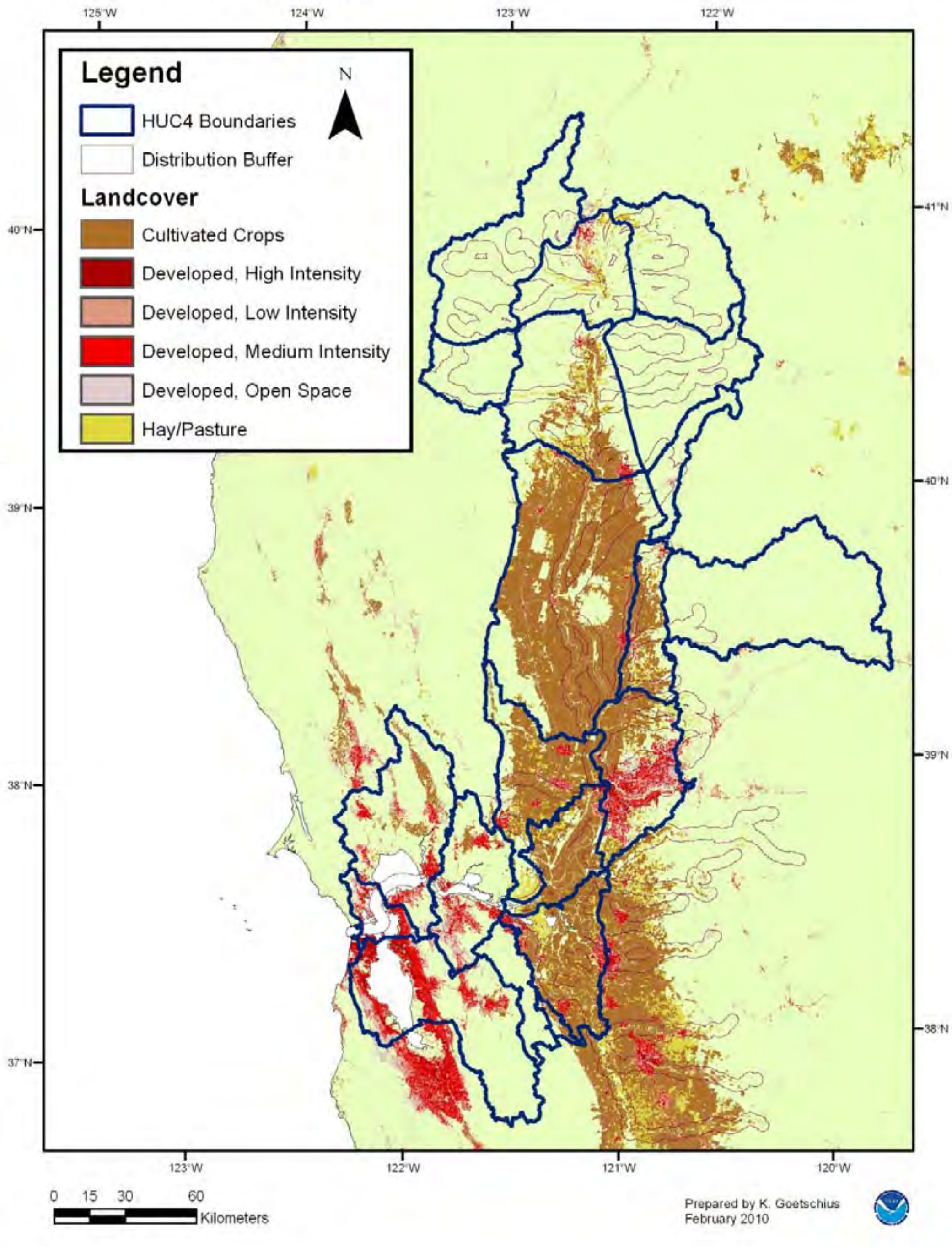
California Coastal Chinook ESU
Species Distribution



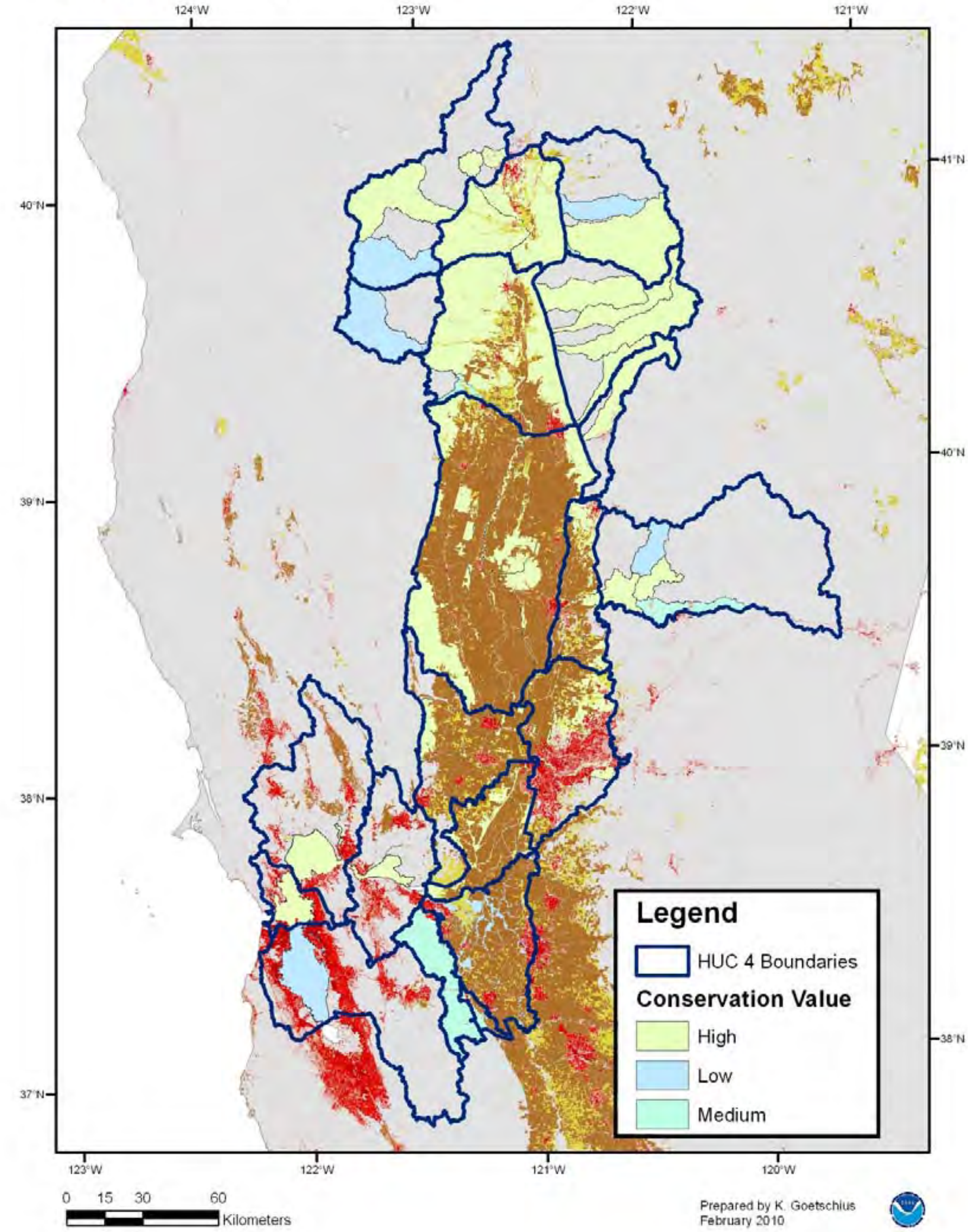
California Coastal Chinook ESU
Critical Habitat



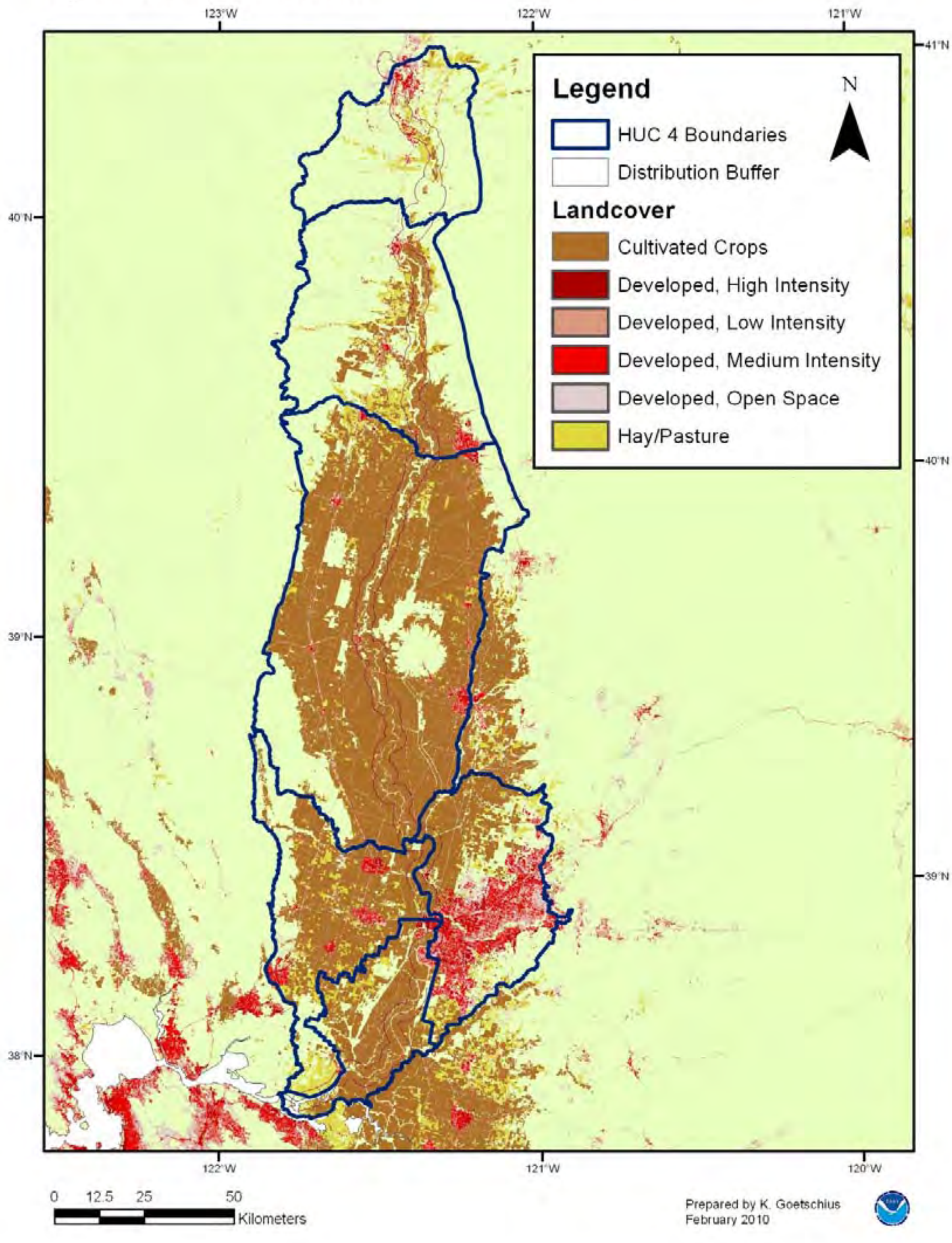
Central Valley Spring-Run Chinook ESU
Species Distribution



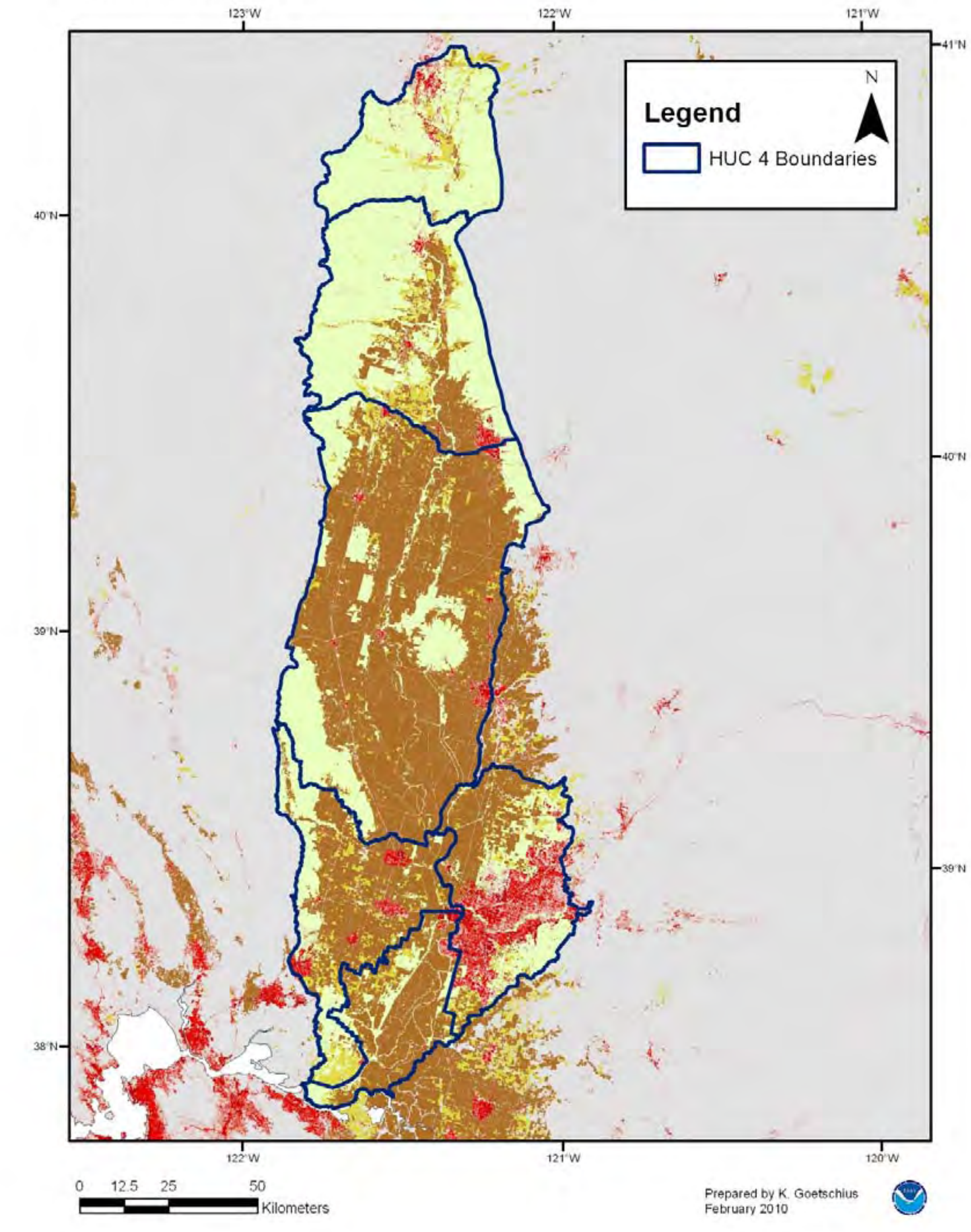
Central Valley Spring-Run Chinook ESU
Critical Habitat



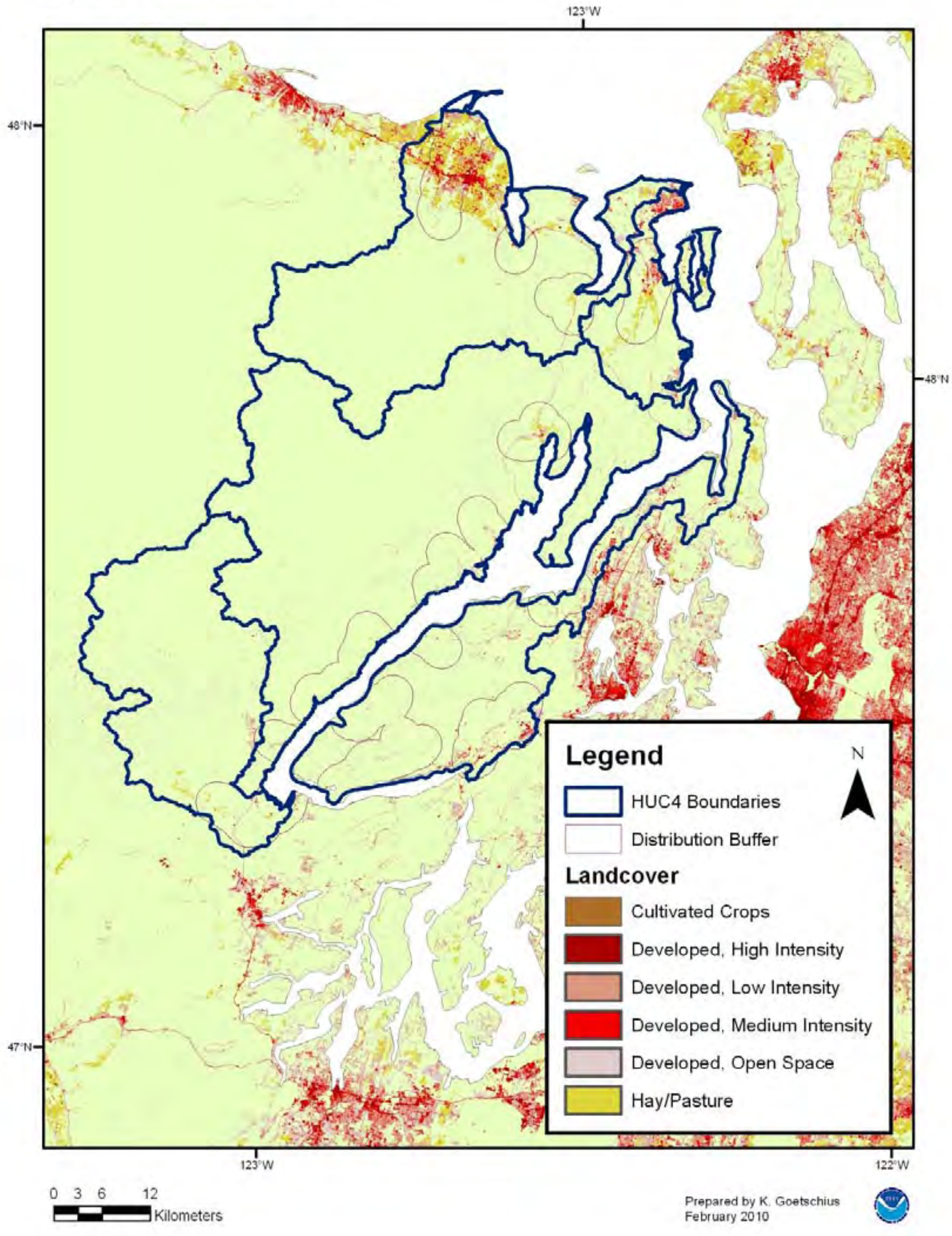
Sacramento River Winter Run Chinook ESU Species Distribution



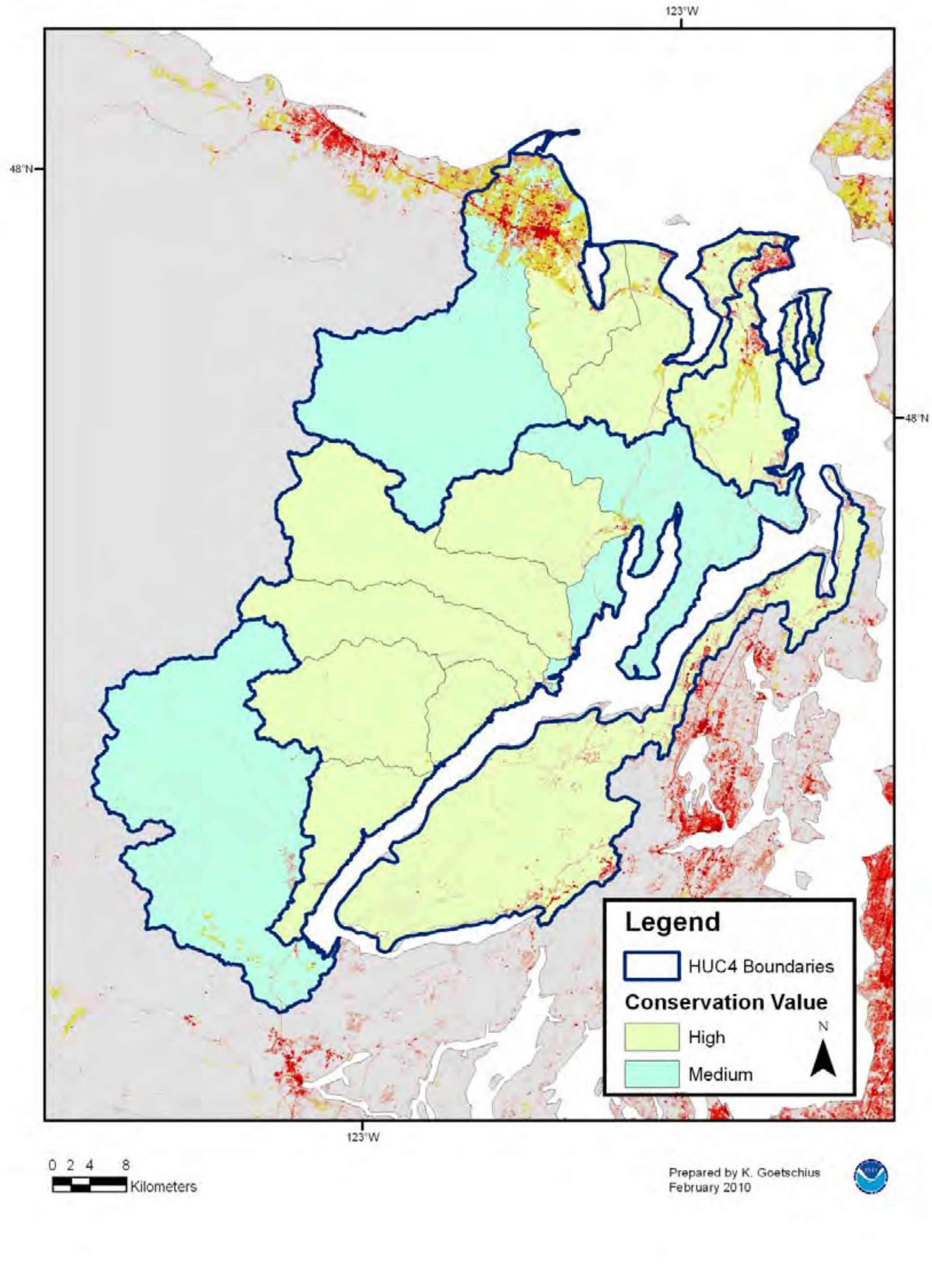
Sacramento River Winter Run Chinook ESU Critical Habitat



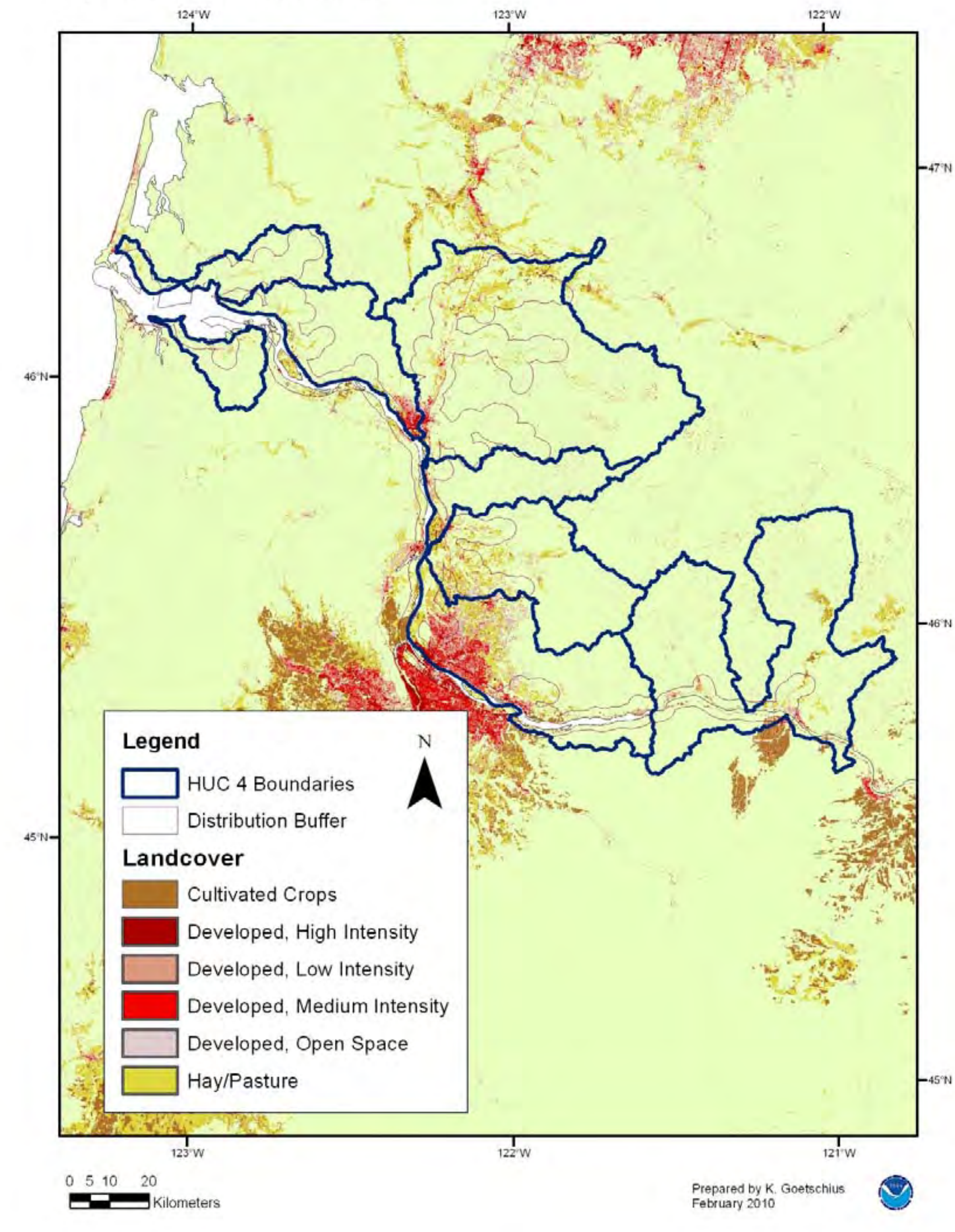
Hood Canal Summer-Run Chum ESU
Species Distribution



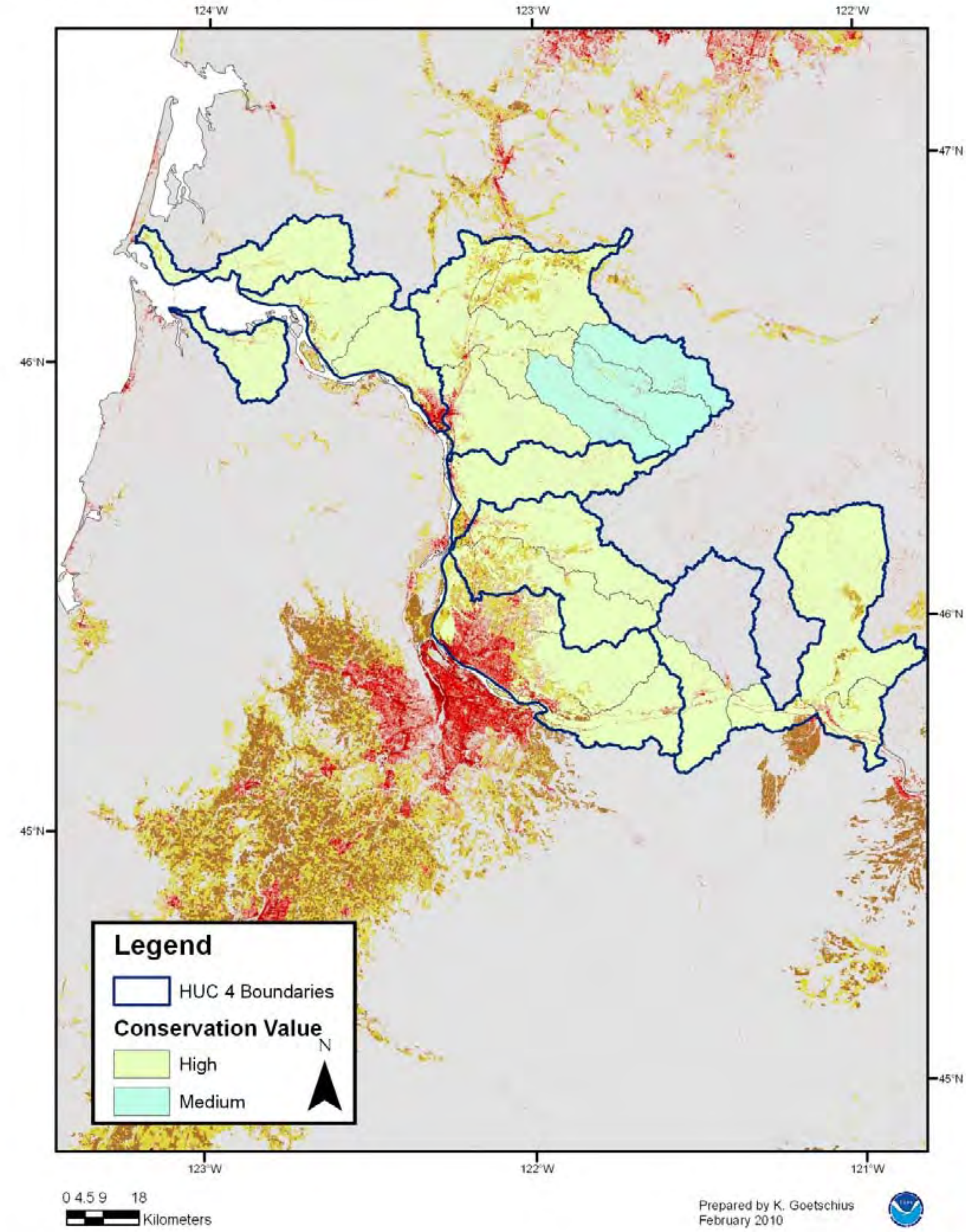
Hood Canal Summer-Run Chum ESU
Critical Habitat

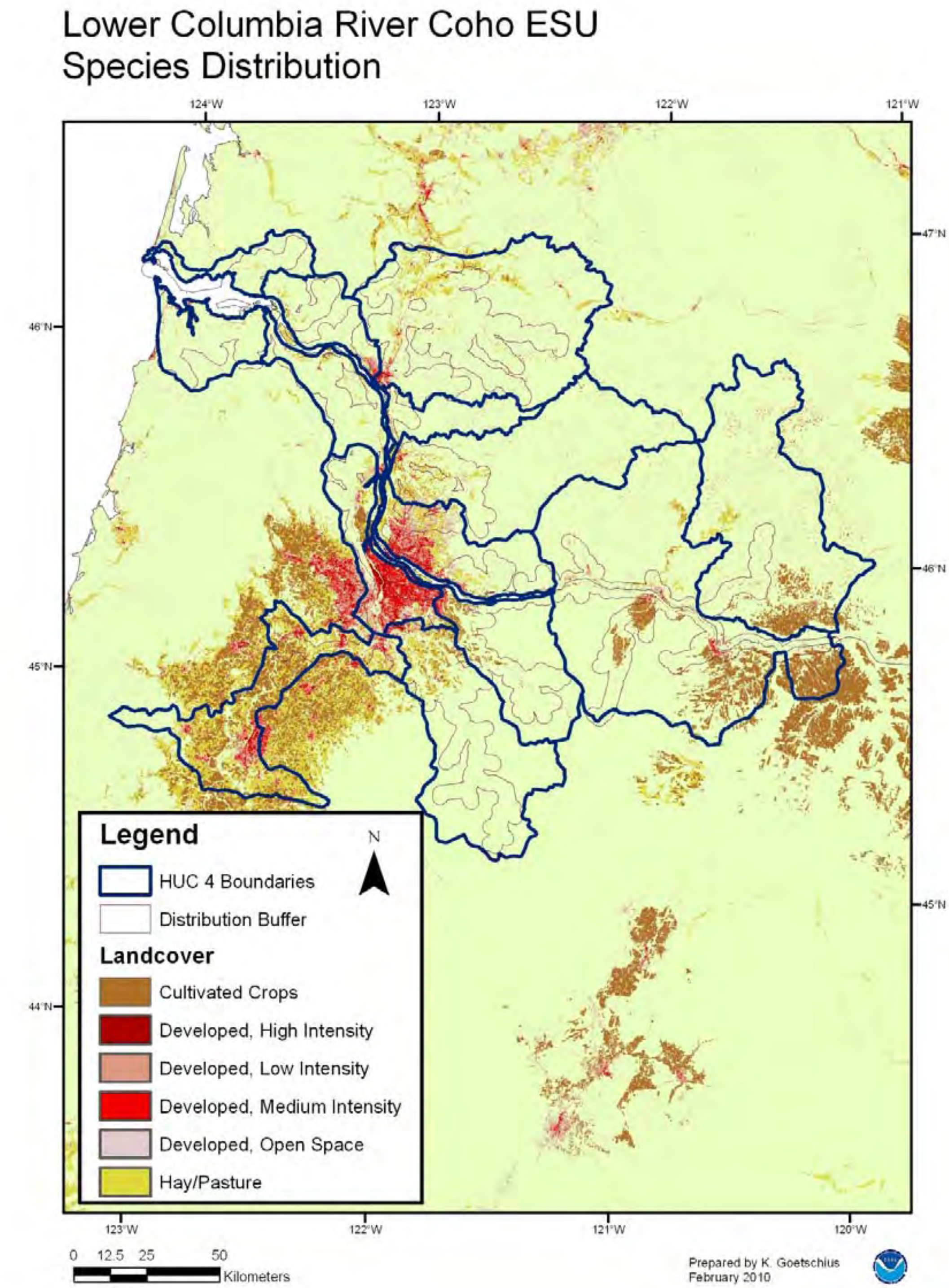


Columbia River Chum ESU
Species Distribution

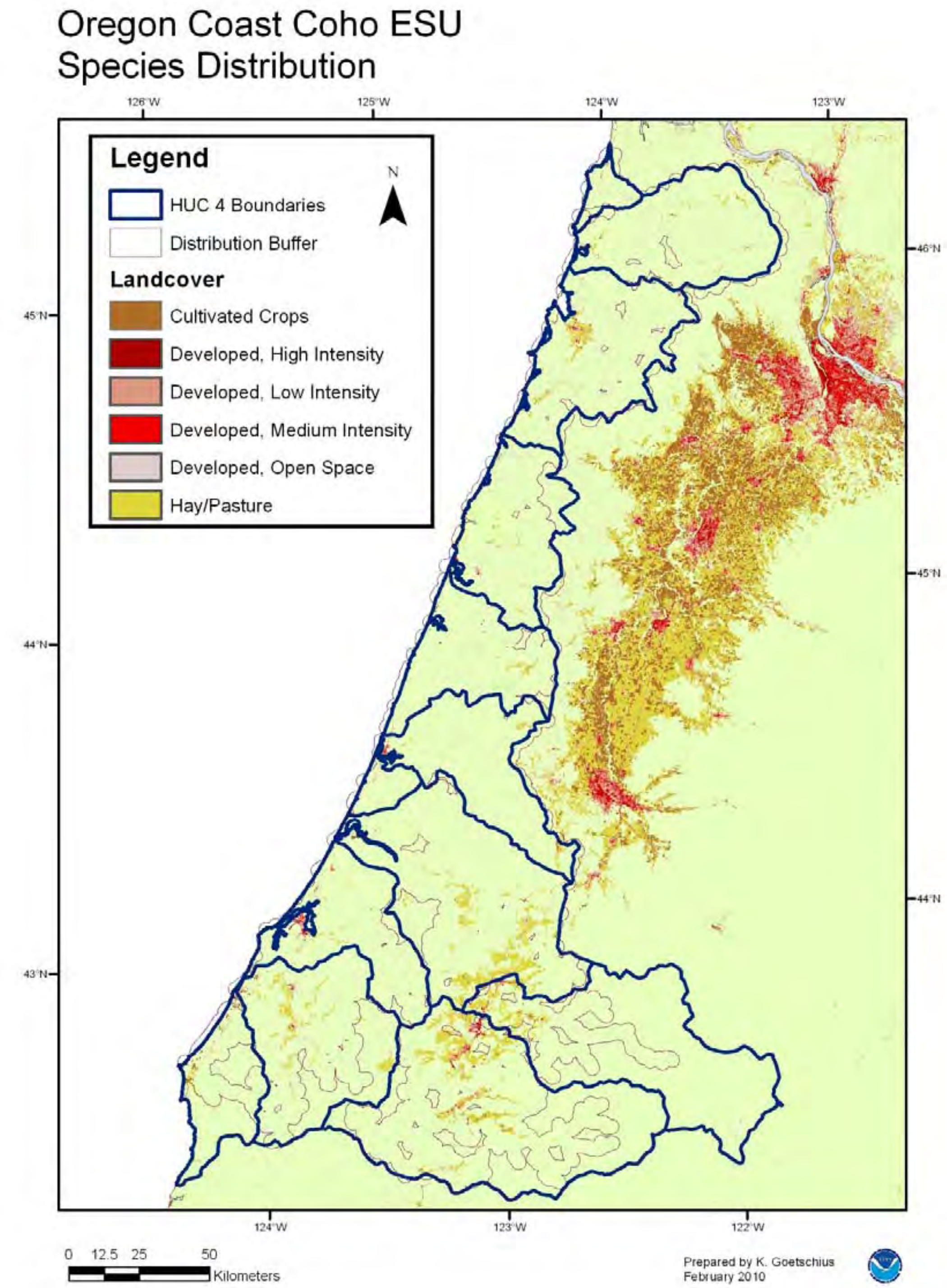


Columbia River Chum ESU
Critical Habitat

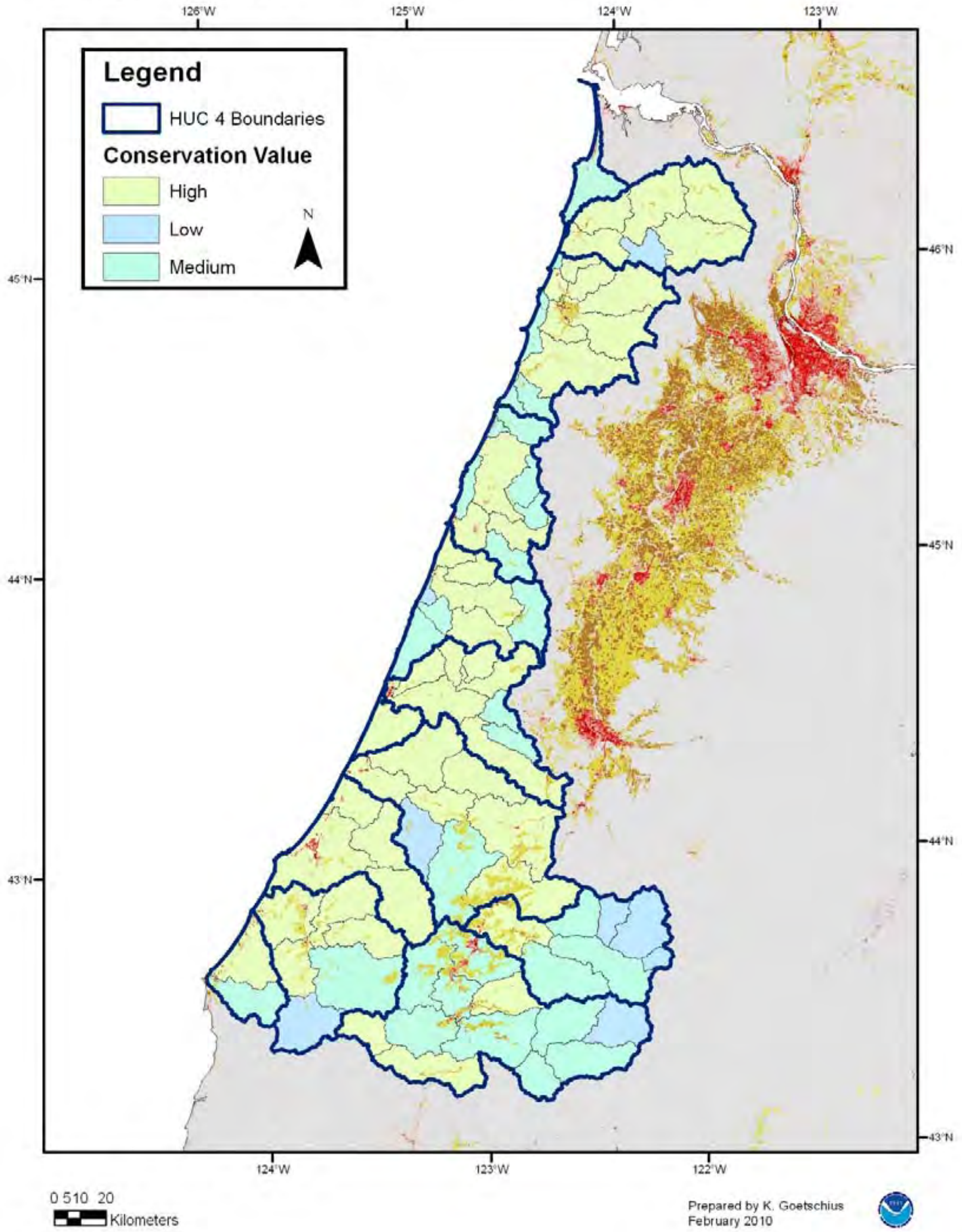




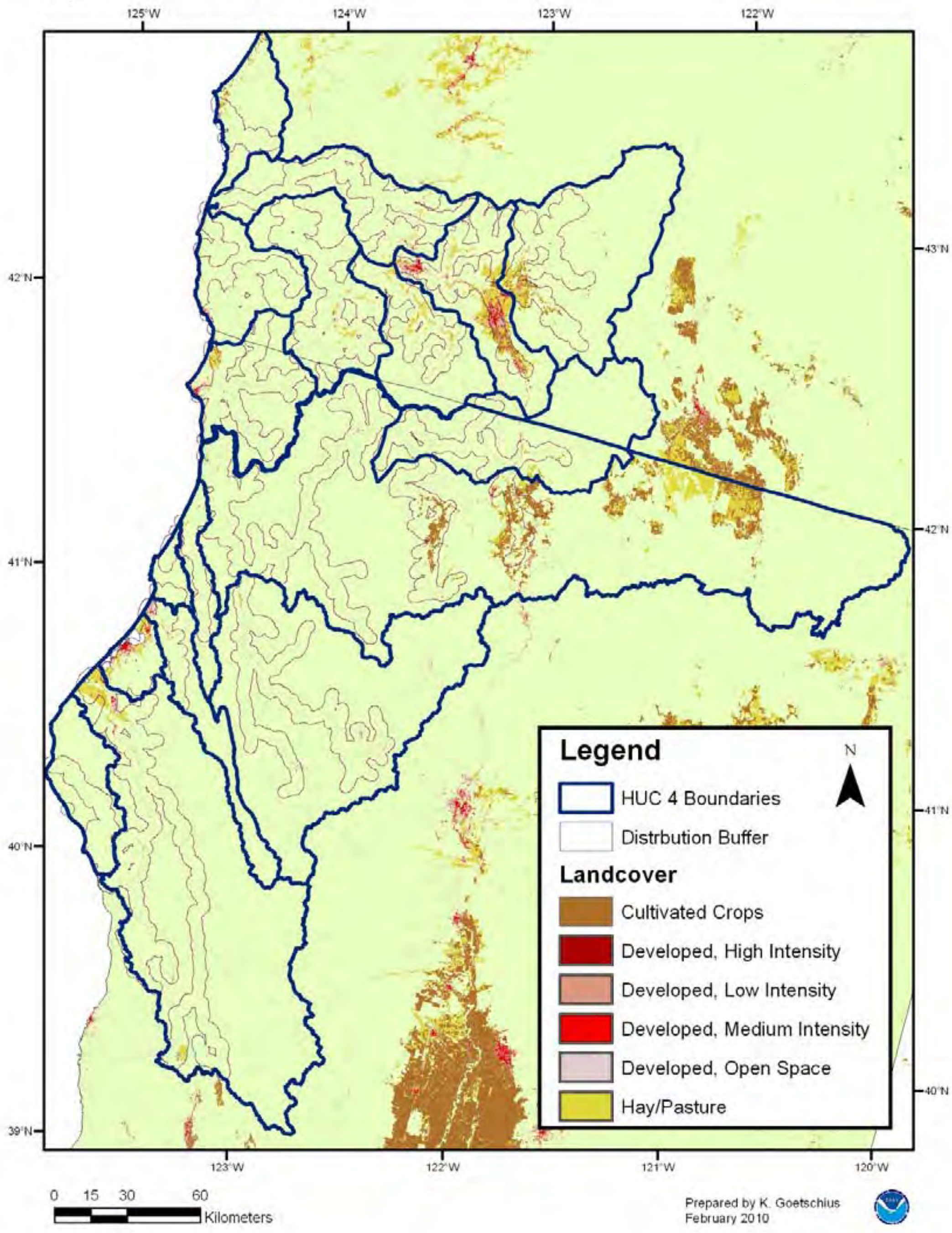
Lower Columbia River Coho ESU
Species Distribution
124°W
123°W
122°W
121°W
47°N
46°N
45°N
44°N
120°W
Legend
HUC 4 Boundaries
Distribution Buffer
Landcover
Cultivated Crops
Developed, High Intensity
Developed, Low Intensity
Developed, Medium Intensity
Developed, Open Space
Hay/Pasture
N
0 12.5 25 50
Kilometers
Prepared by K. Goetschius
February 2010


Oregon Coast Coho ESU
Species Distribution
Legend
HUC 4 Boundaries
Distribution Buffer
Landcover
Cultivated Crops
Developed, High Intensity
Developed, Low Intensity
Developed, Medium Intensity
Developed, Open Space
Hay/Pasture
N
126°W
125°W
124°W
123°W
122°W
46°N
45°N
44°N
43°N
0 12.5 25 50
Kilometers
Prepared by K. Goetschius
February 2010

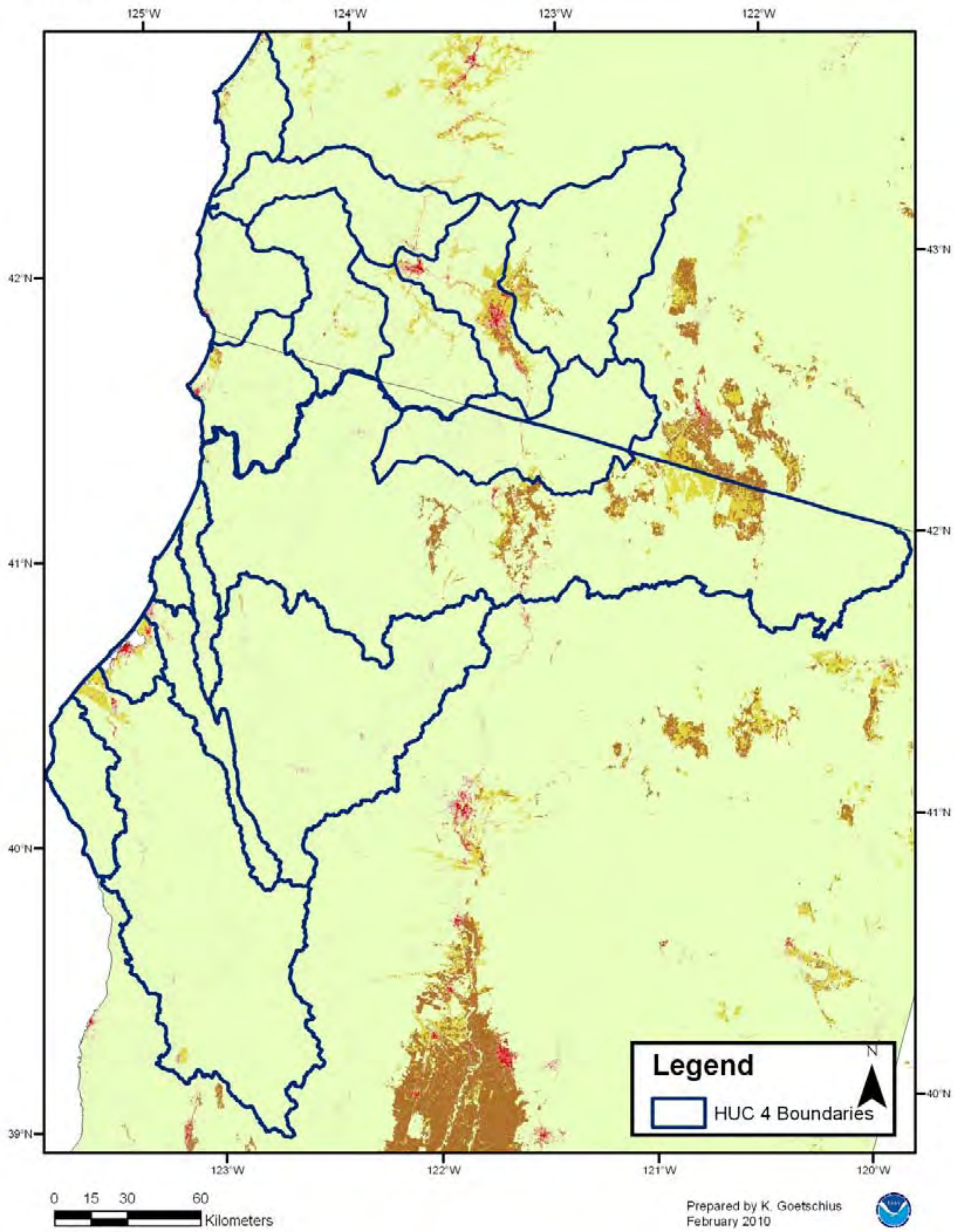
Oregon Coast Coho ESU
Critical Habitat



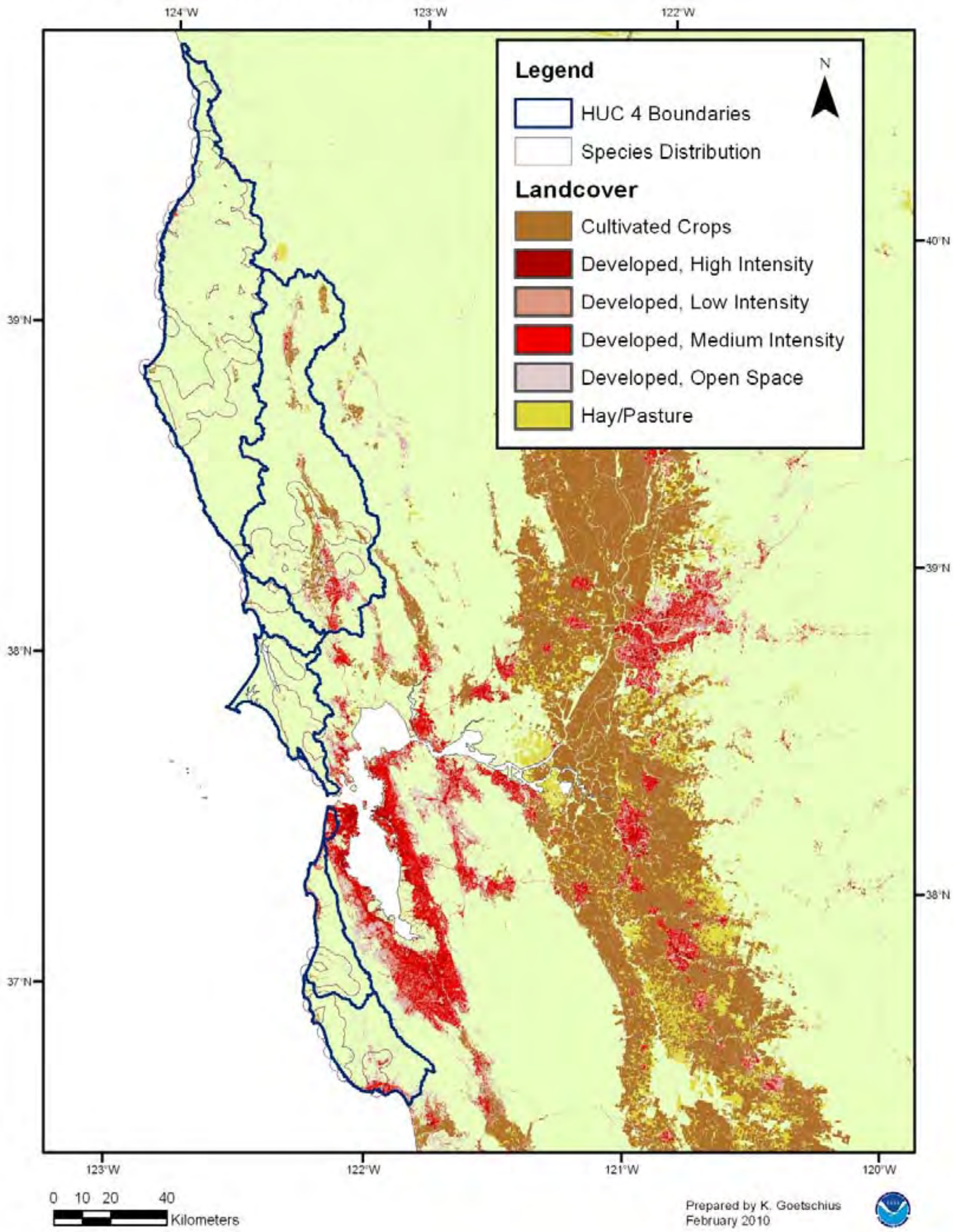
Southern Oregon Northern California Coho ESU Species Distribution



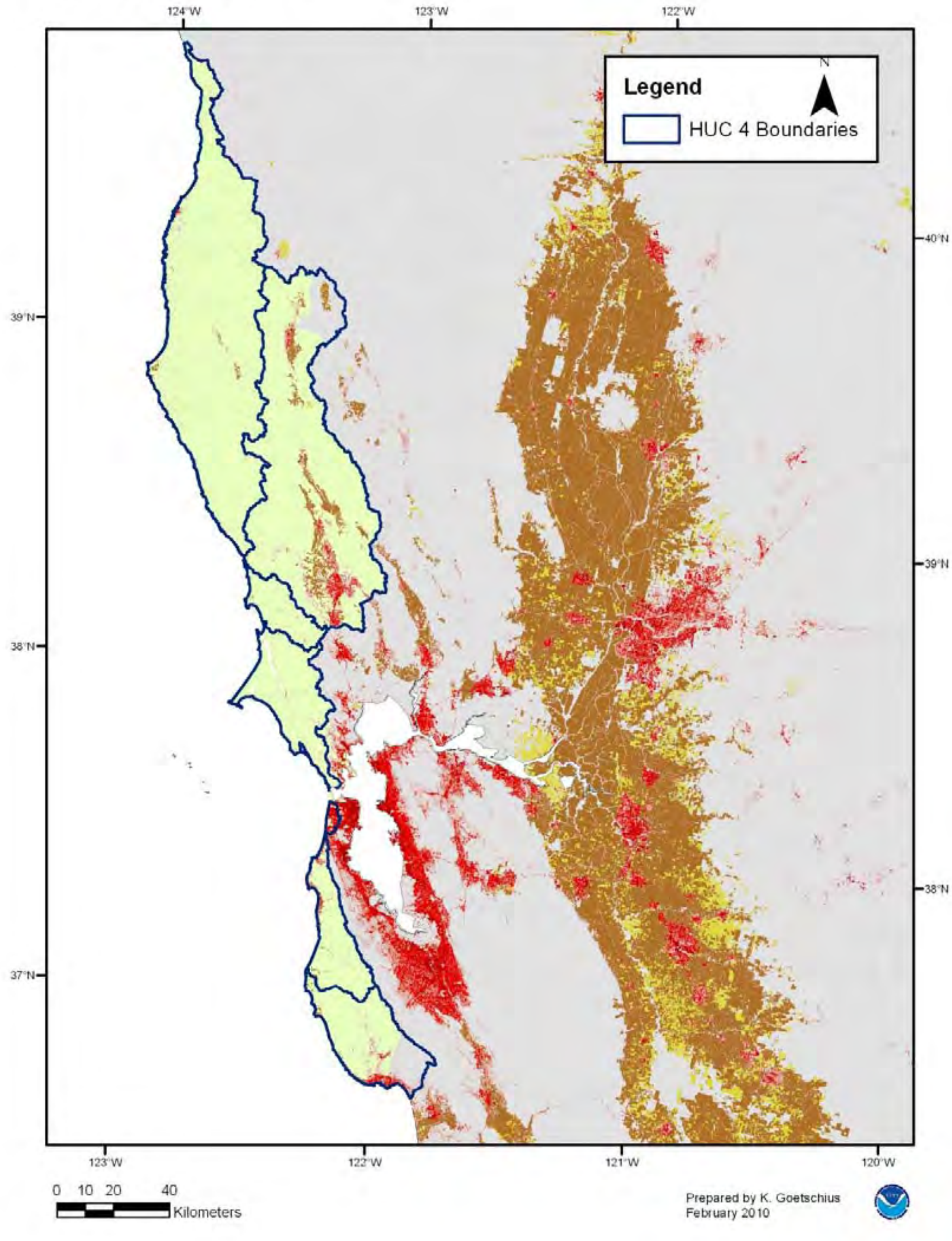
Southern Oregon Northern California Coho ESU Critical Habitat



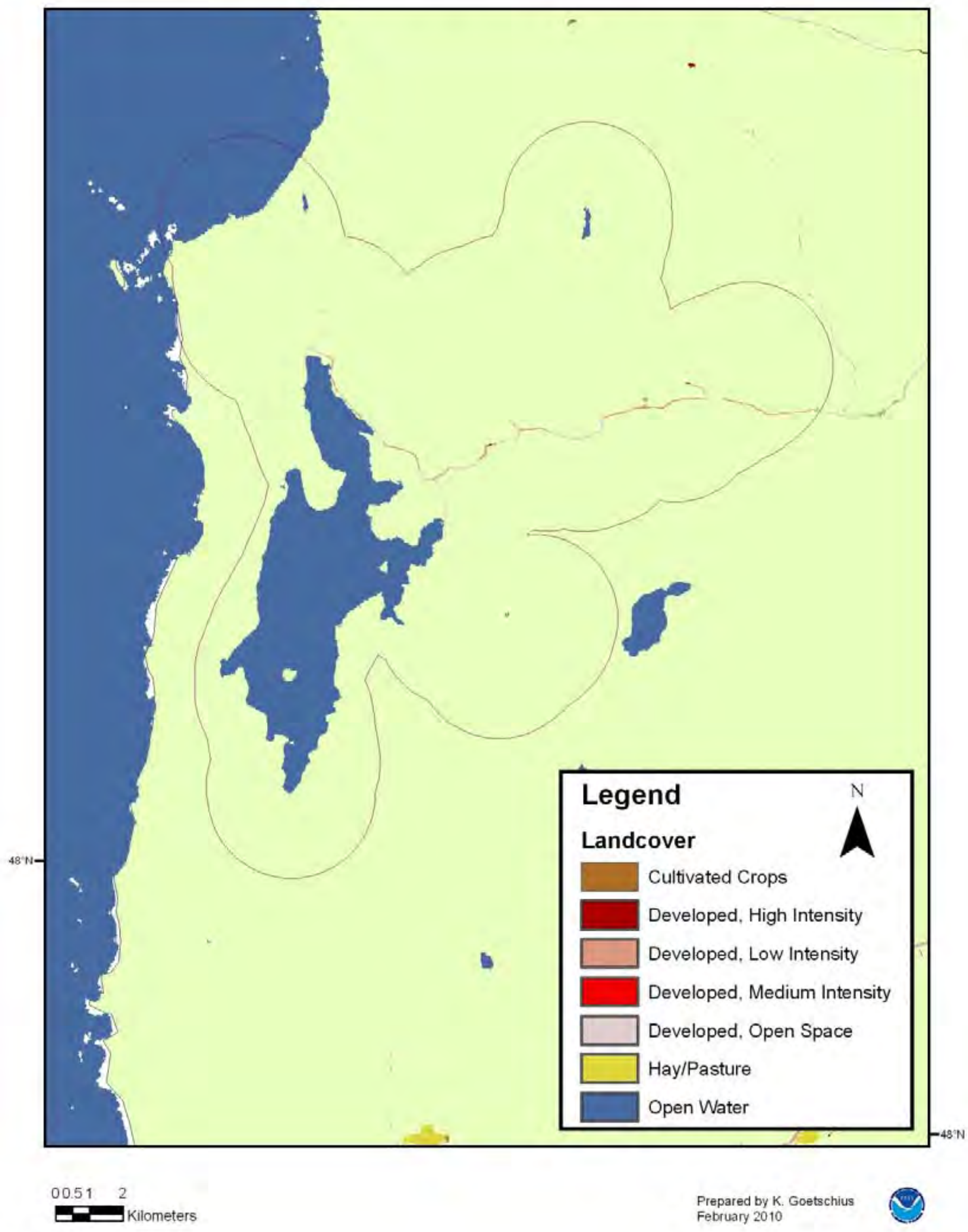
Central California Coastal Coho Species Distribution



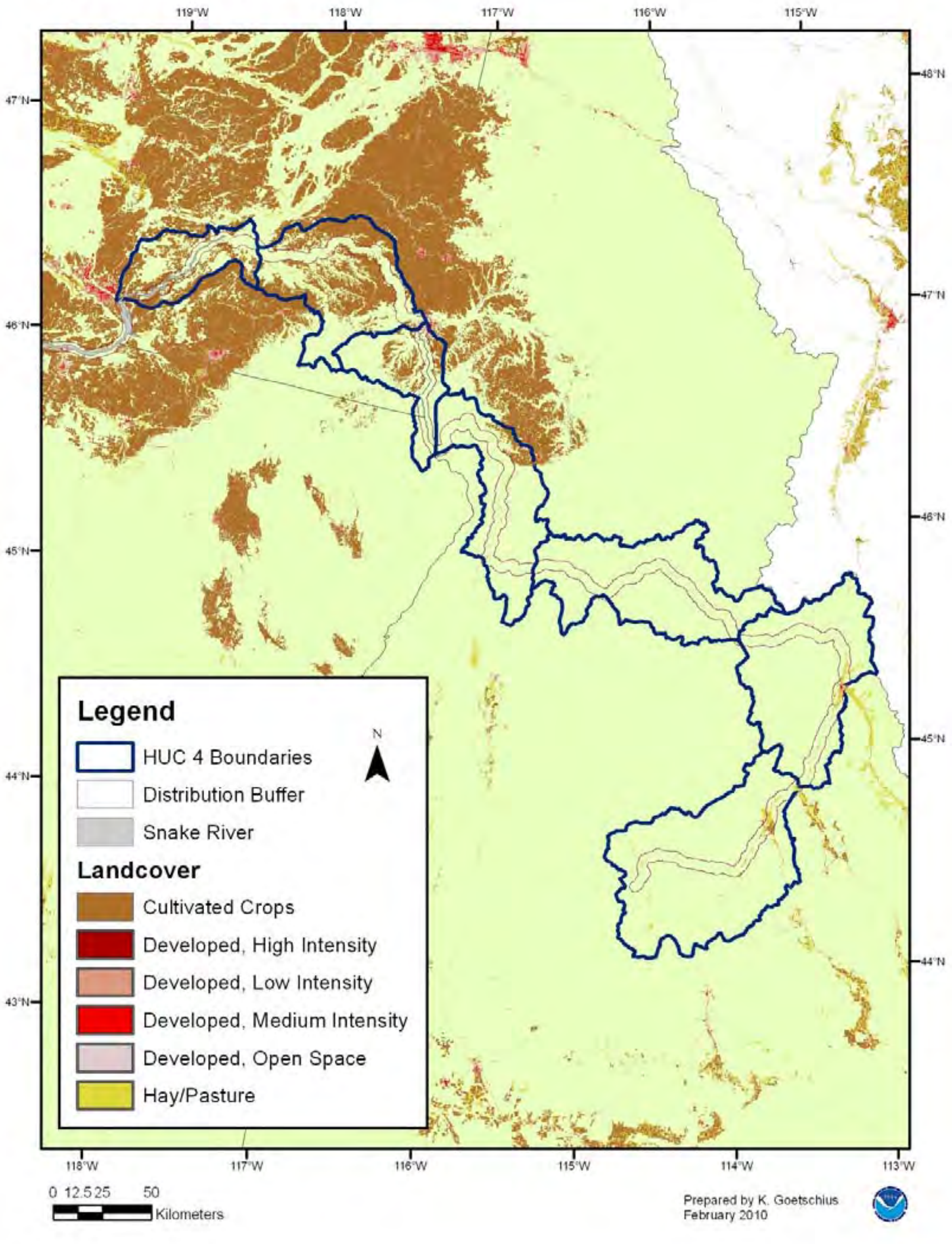
Central California Coastal Coho Critical Habitat



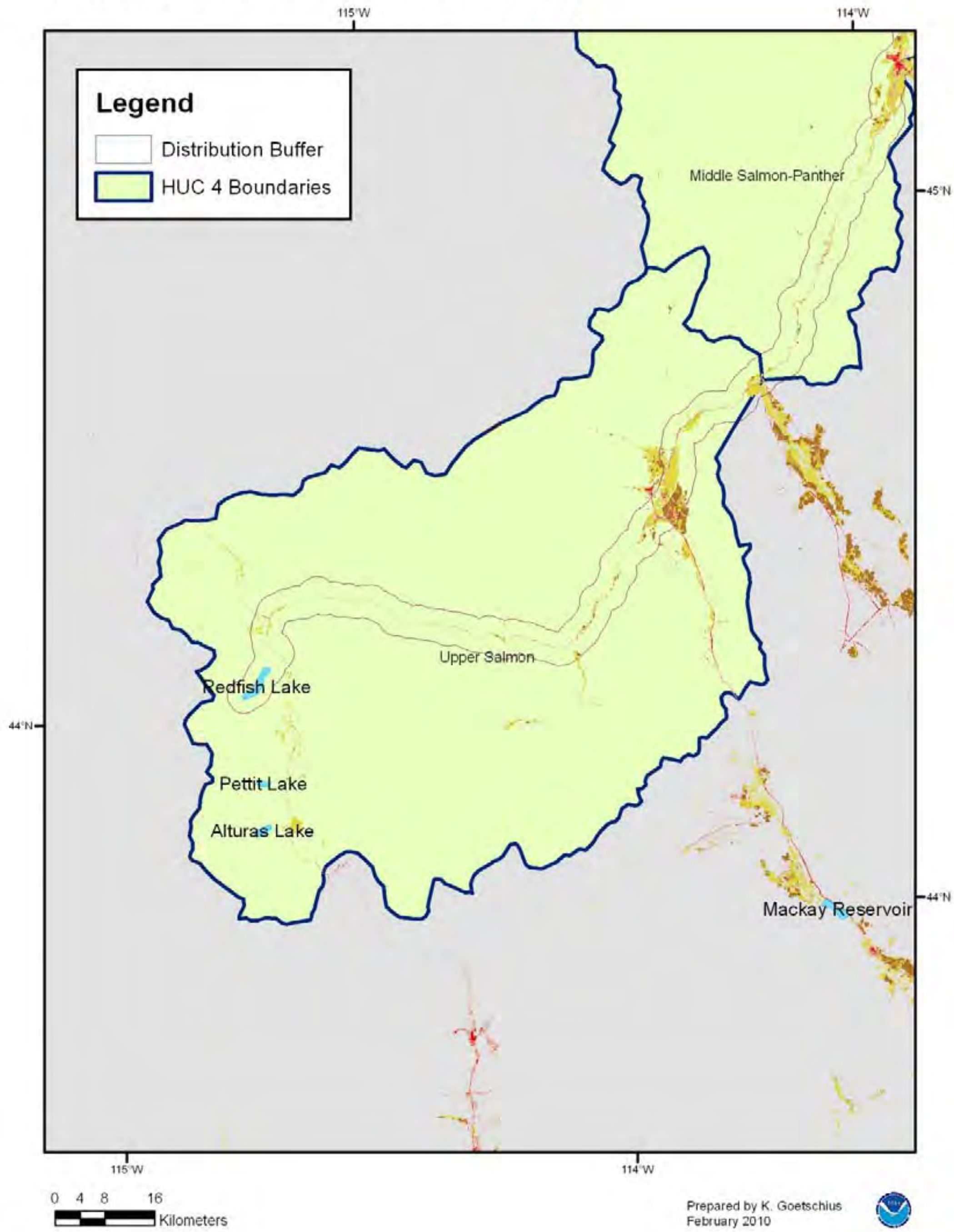
Ozette Lake Sockeye
Species Distribution and Critical Habitat



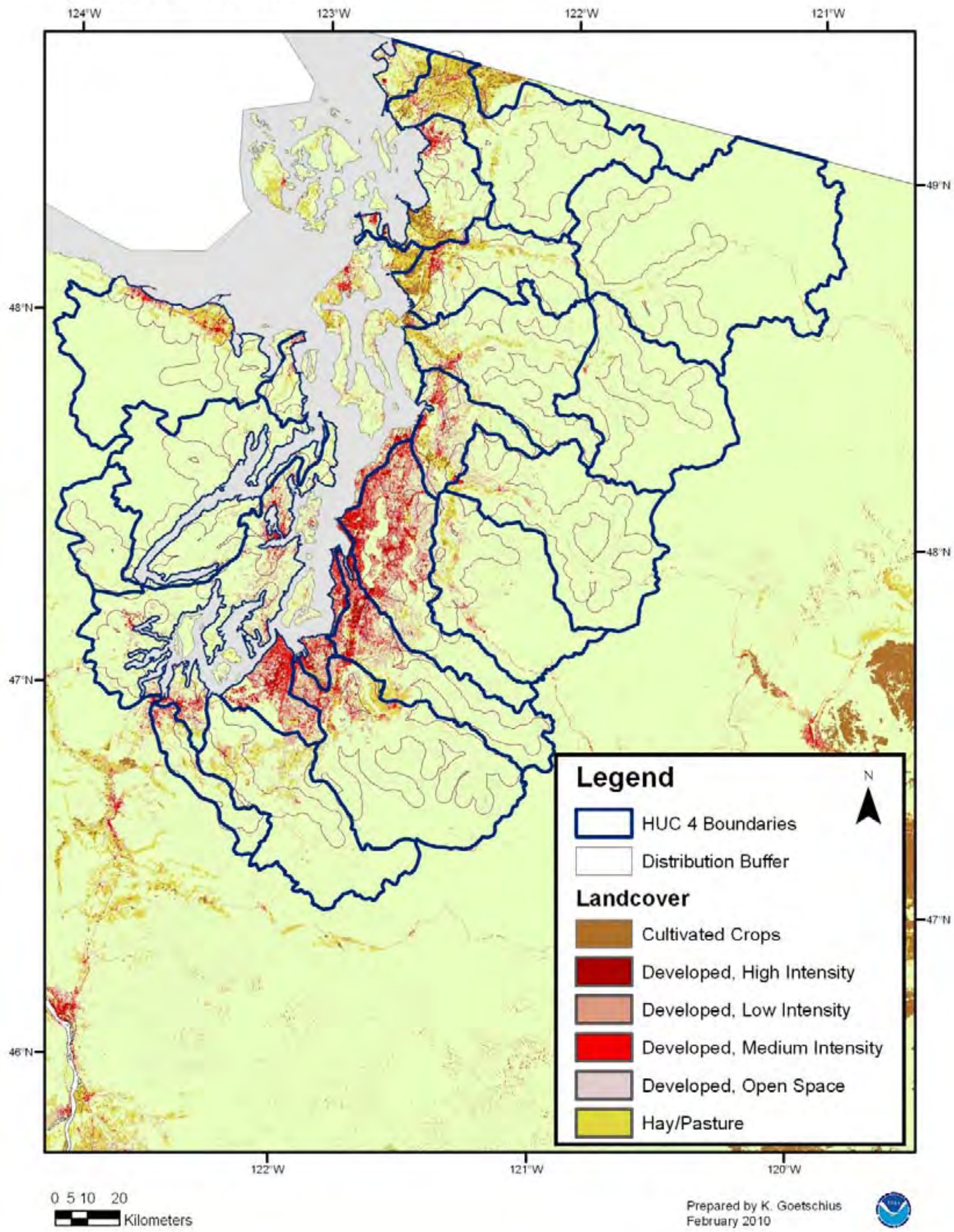
Snake River Sockeye ESU
Species Distribution

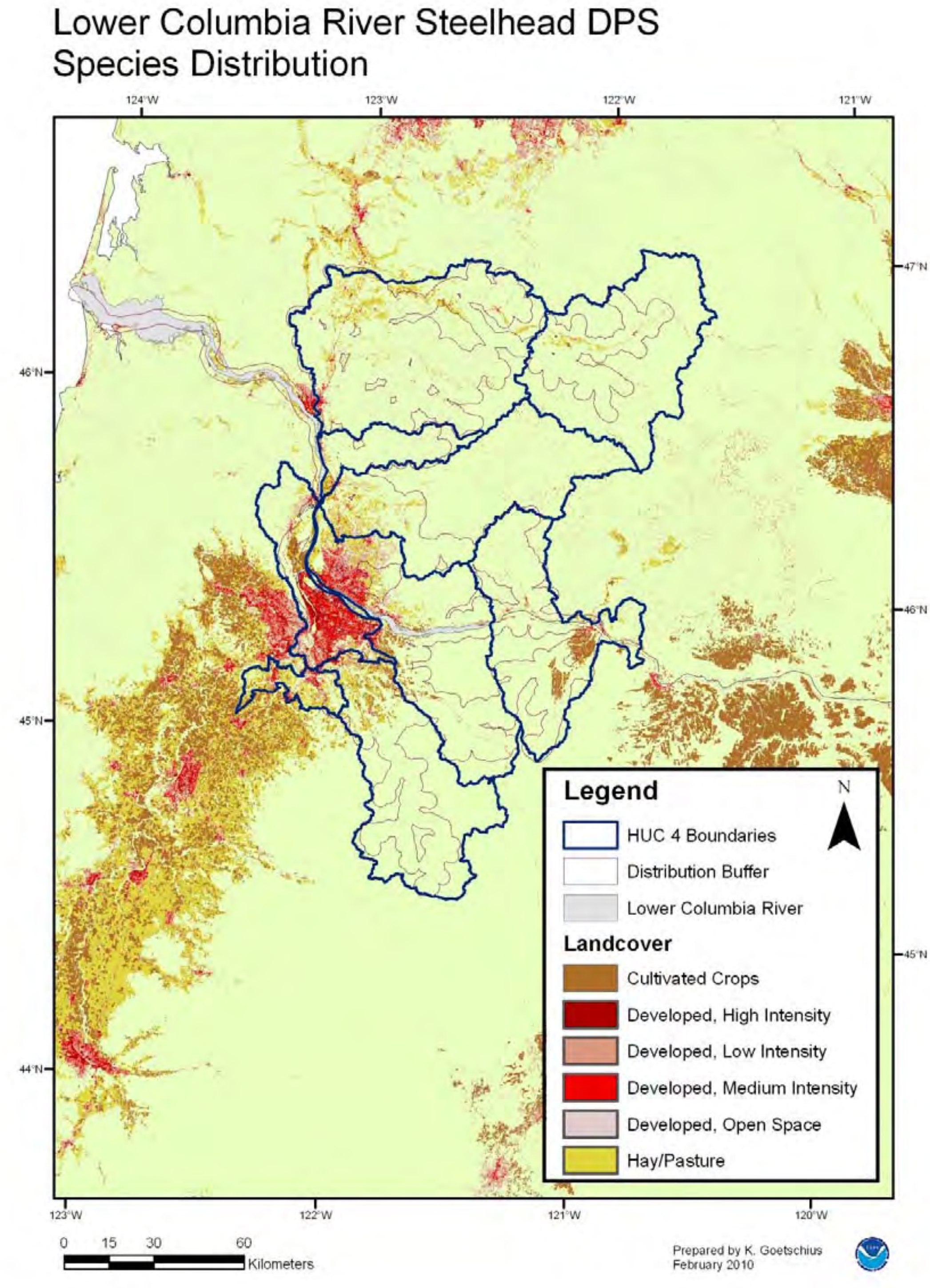


Snake River Sockeye
Critical Habitat, close up of lakes

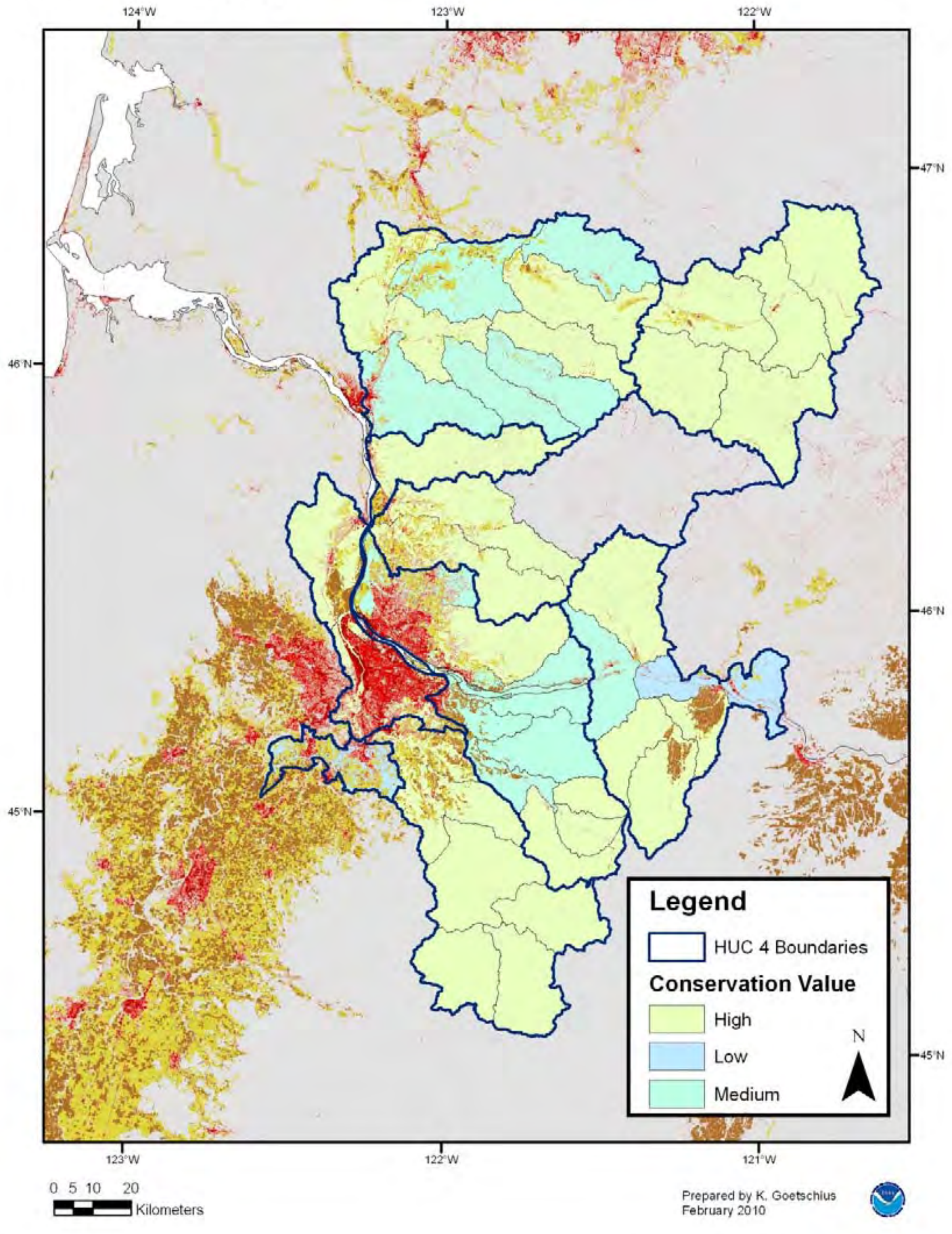


Puget Sound Steelhead DPS
Species Distribution

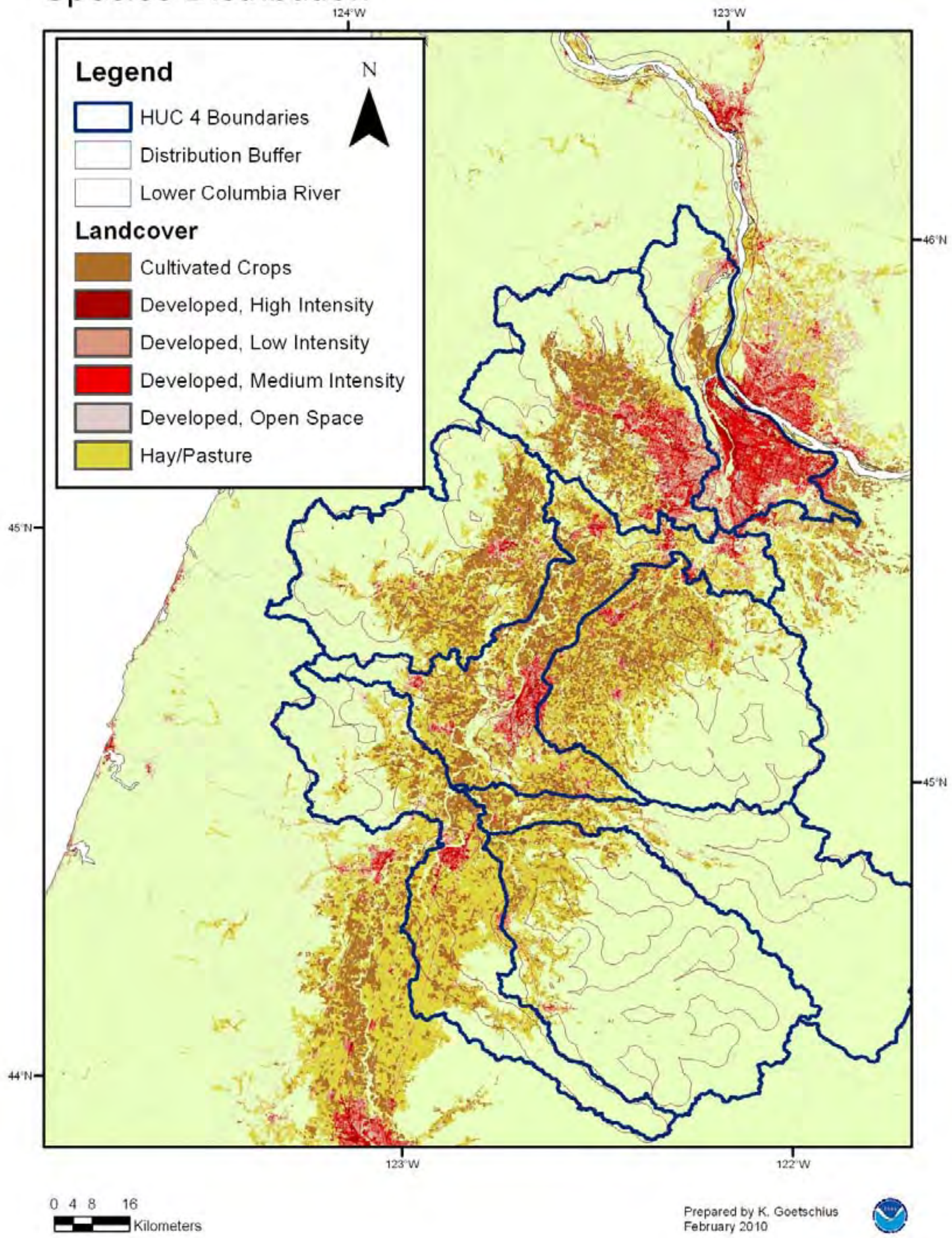



Lower Columbia River Steelhead DPS
Species Distribution
124°W
123°W
122°W
121°W
47°N
46°N
45°N
44°N
123°W
122°W
121°W
120°W
Legend
N
HUC 4 Boundaries
Distribution Buffer
Lower Columbia River
Landcover
Cultivated Crops
Developed, High Intensity
Developed, Low Intensity
Developed, Medium Intensity
Developed, Open Space
Hay/Pasture
0 15 30 60
Kilometers
Prepared by K. Goetschius
February 2010

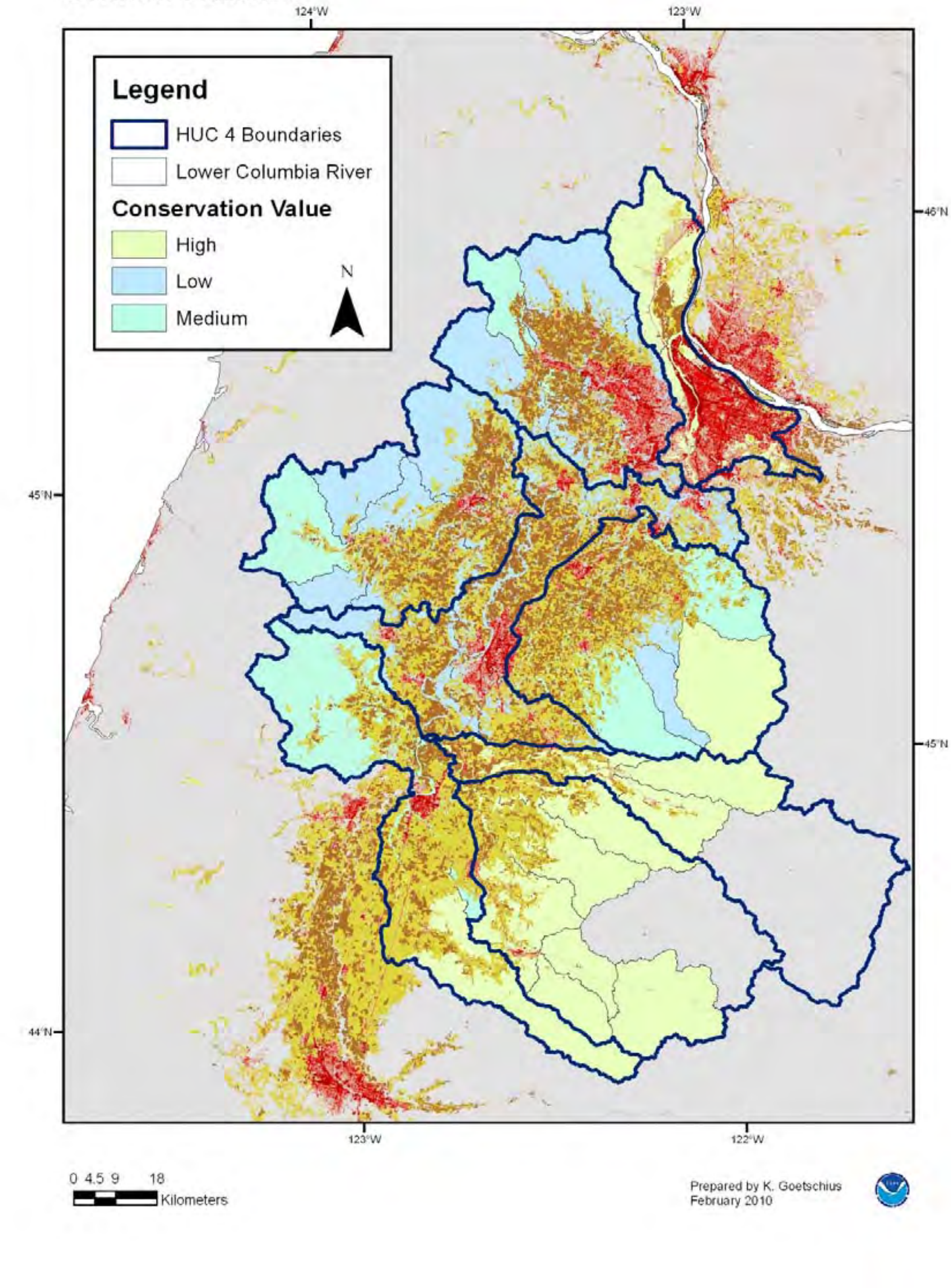
Lower Columbia River Steelhead DPS Critical Habitat



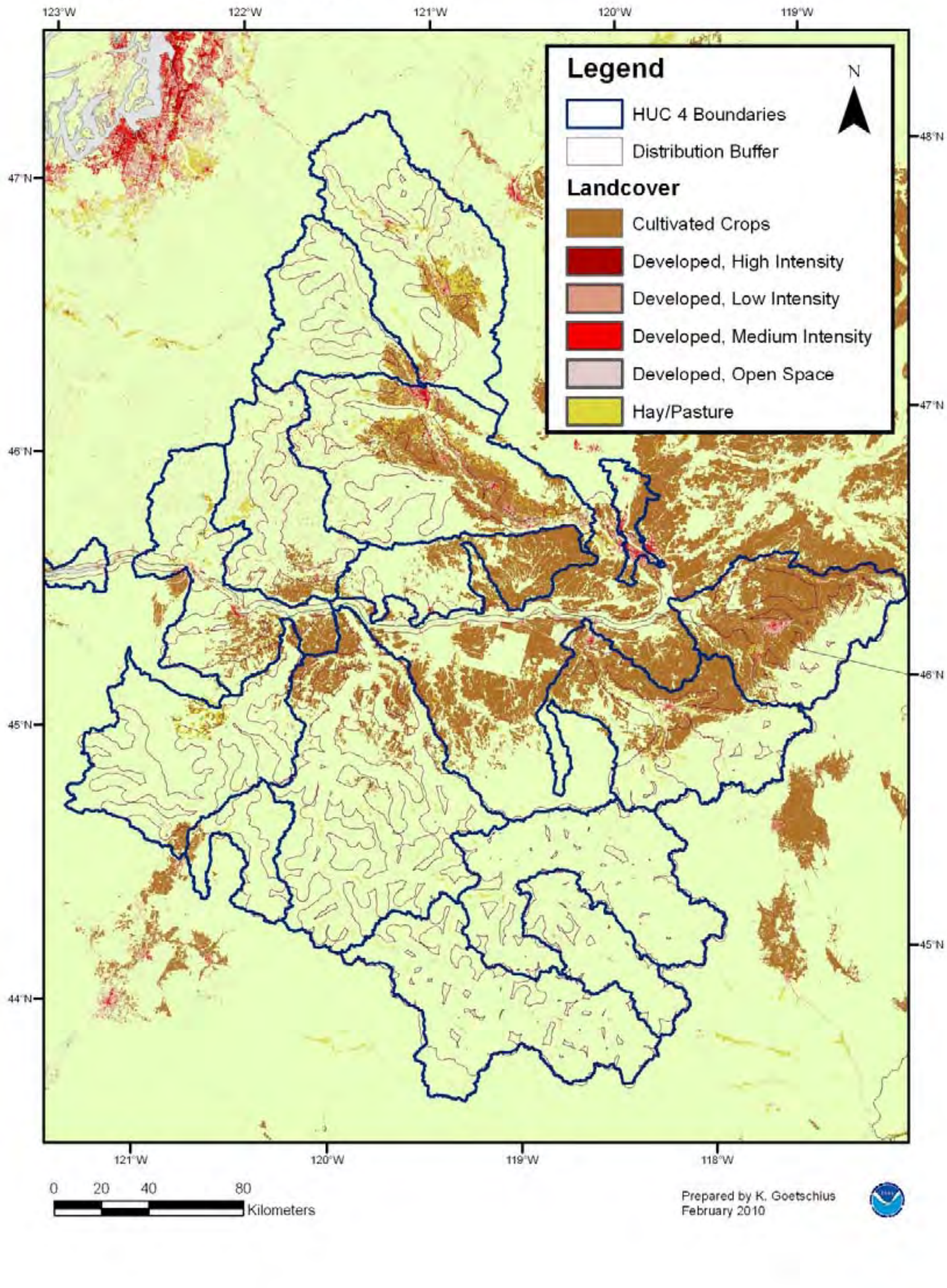
Upper Willamette River Steelhead DPS Species Distribution



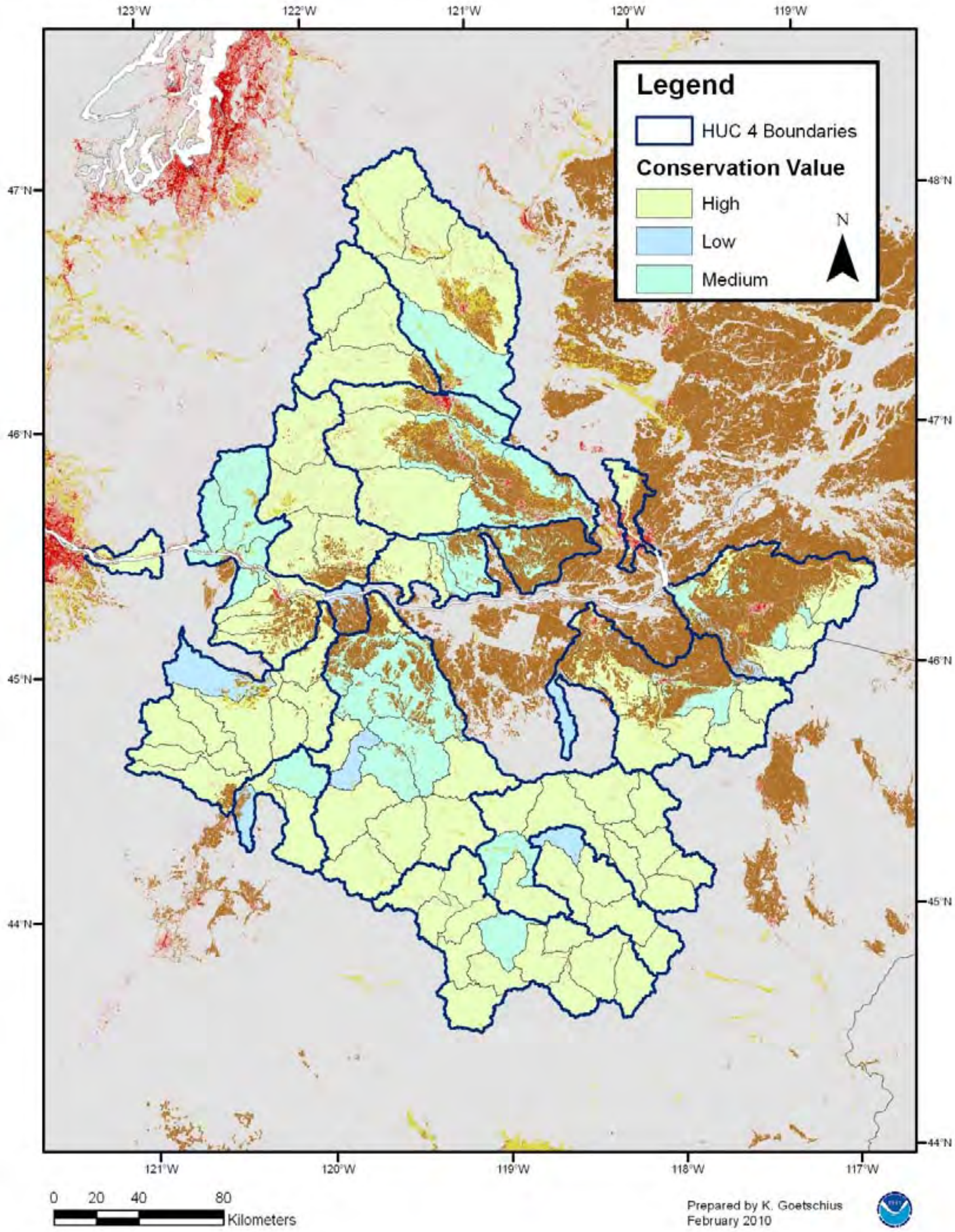
Upper Willamette River Steelhead DPS
Critical Habitat



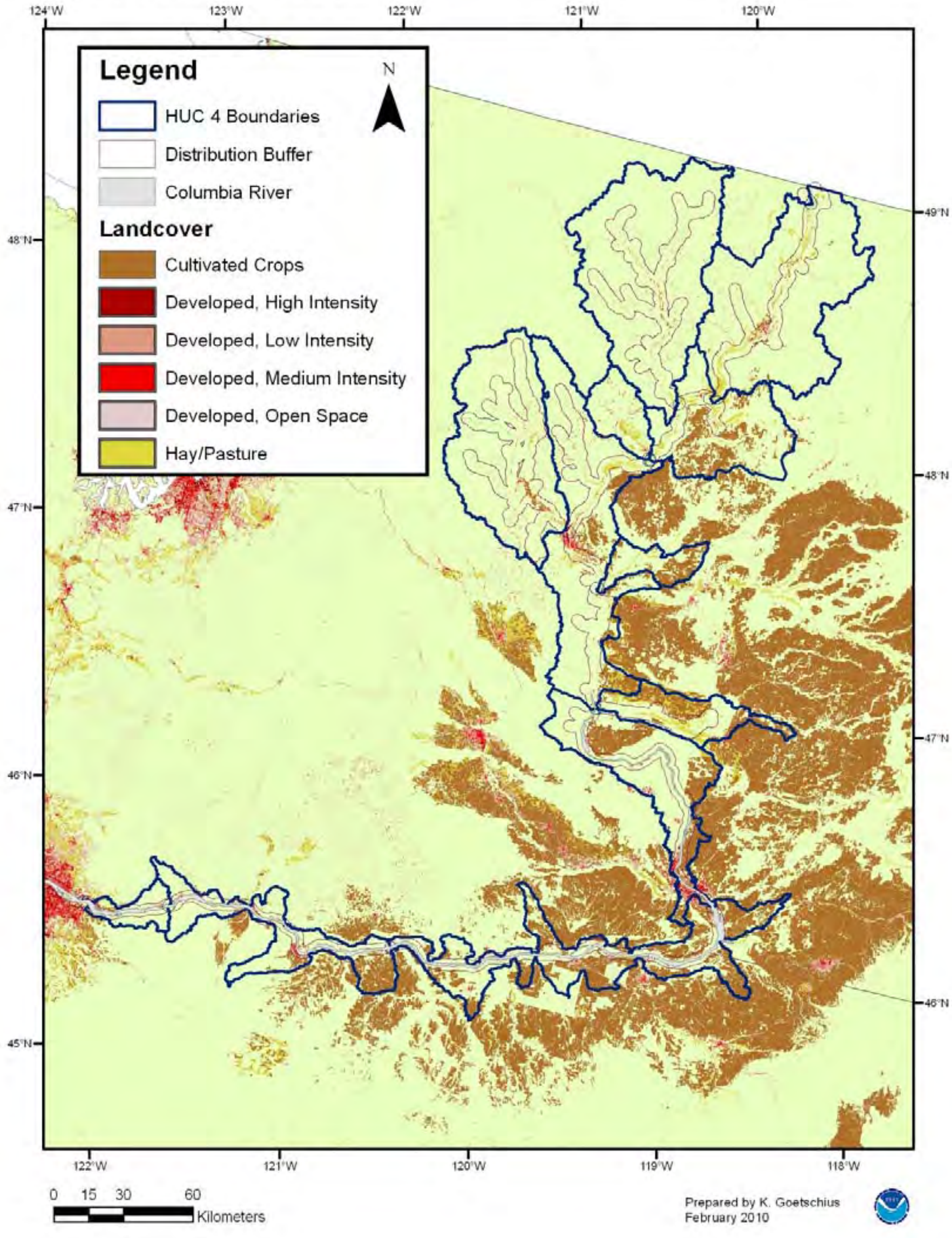
Middle Columbia River Steelhead DPS
Species Distribution

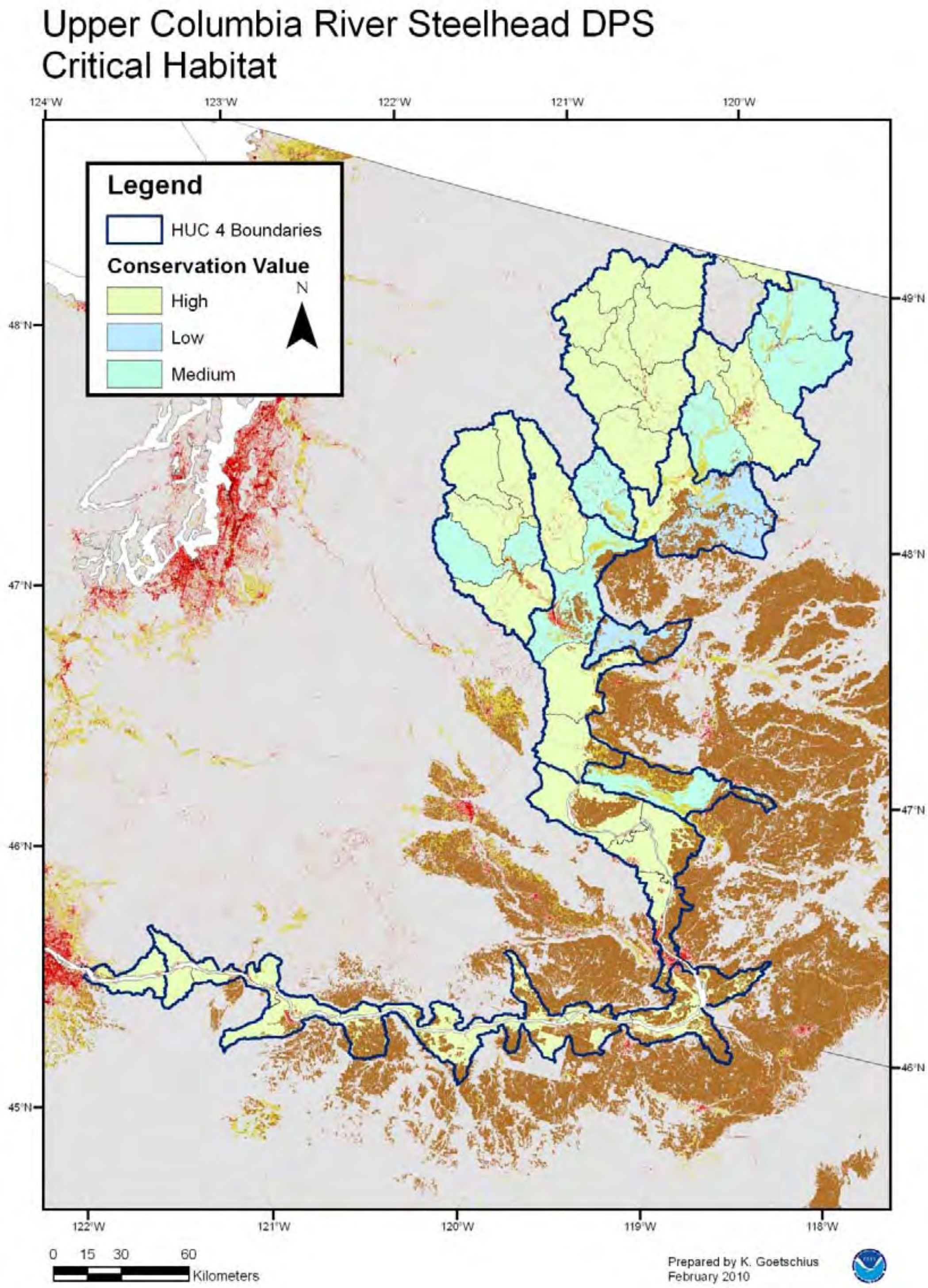


Middle Columbia River Steelhead DPS
Critical Habitat

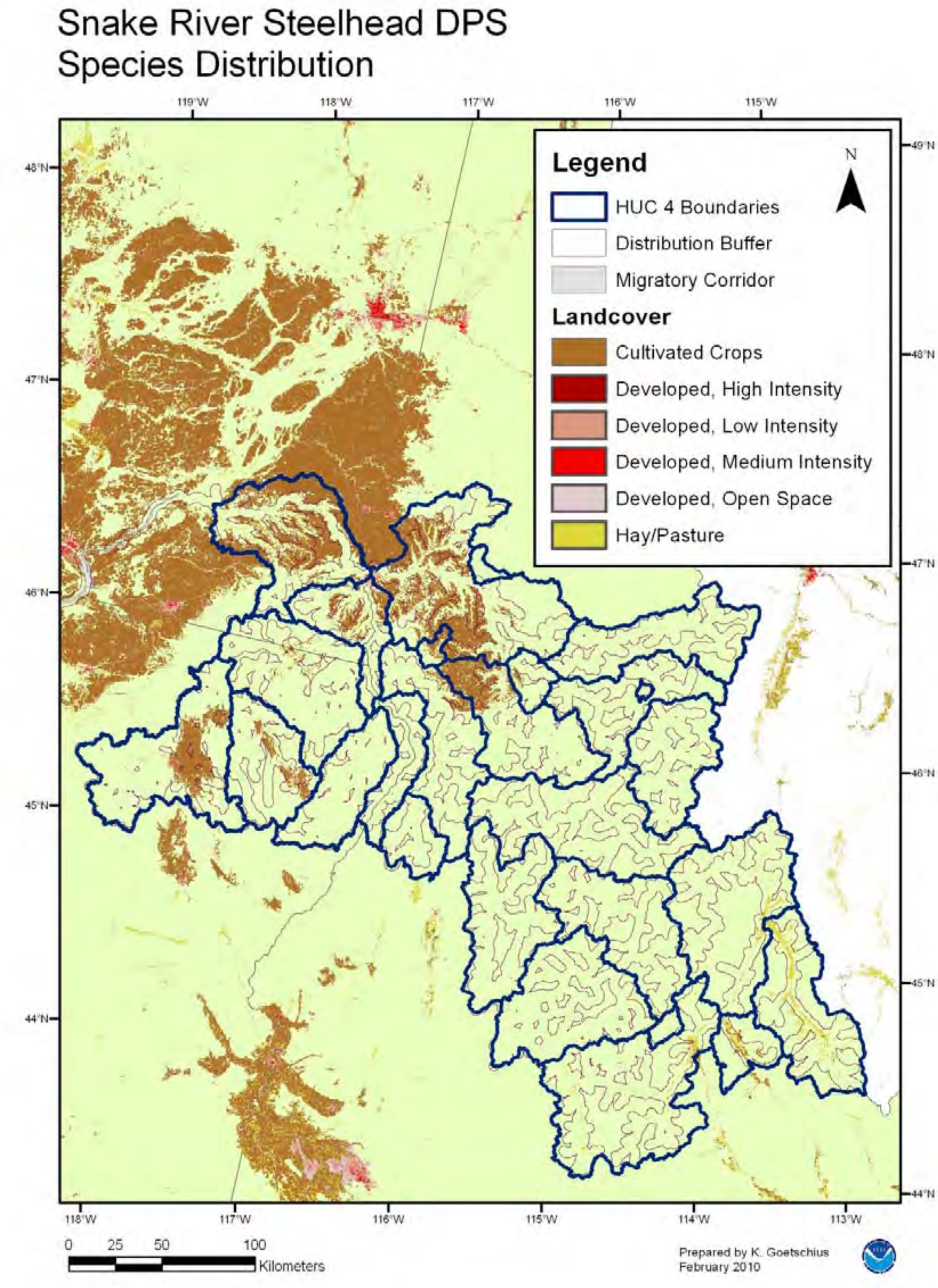


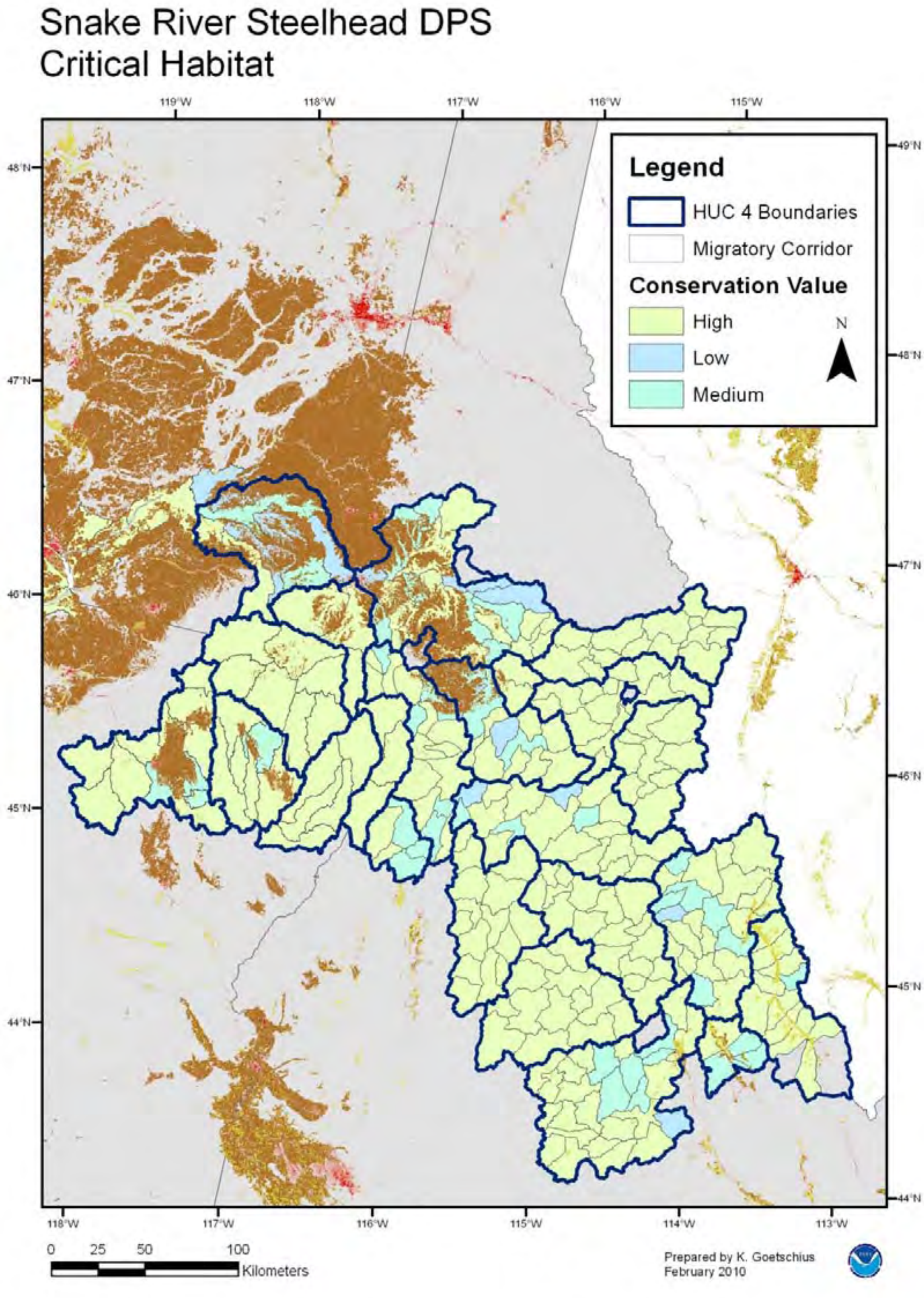
Upper Columbia River Steelhead DPS Species Distribution

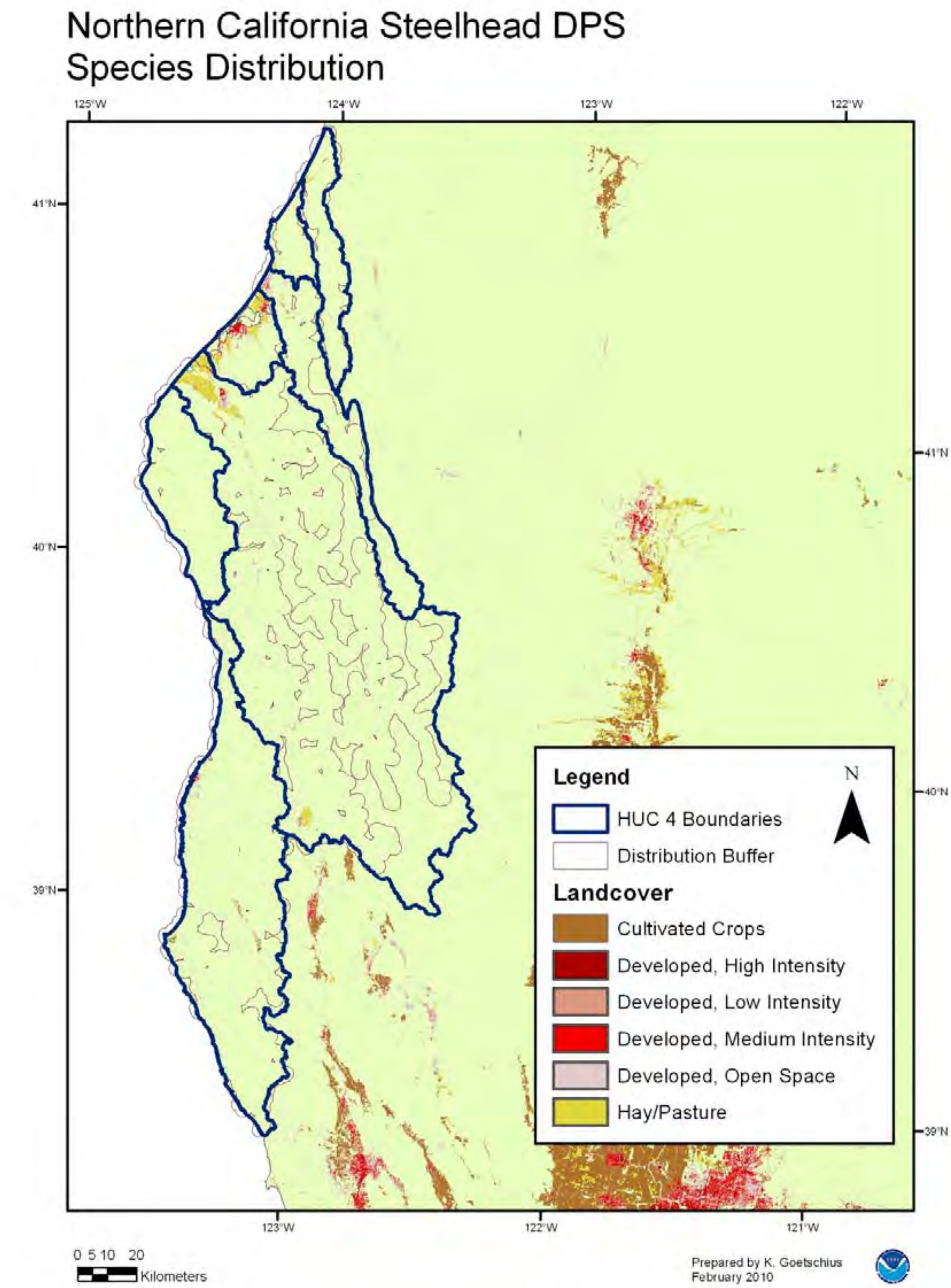




Upper Columbia River Steelhead DPS
Critical Habitat
Legend
HUC 4 Boundaries
Conservation Value
High
Low
Medium
N
124°W
123°W
122°W
121°W
120°W
49°N
48°N
47°N
46°N
45°N
118°W
119°W
0 15 30 60
Kilometers
Prepared by K. Goetschius
February 2010

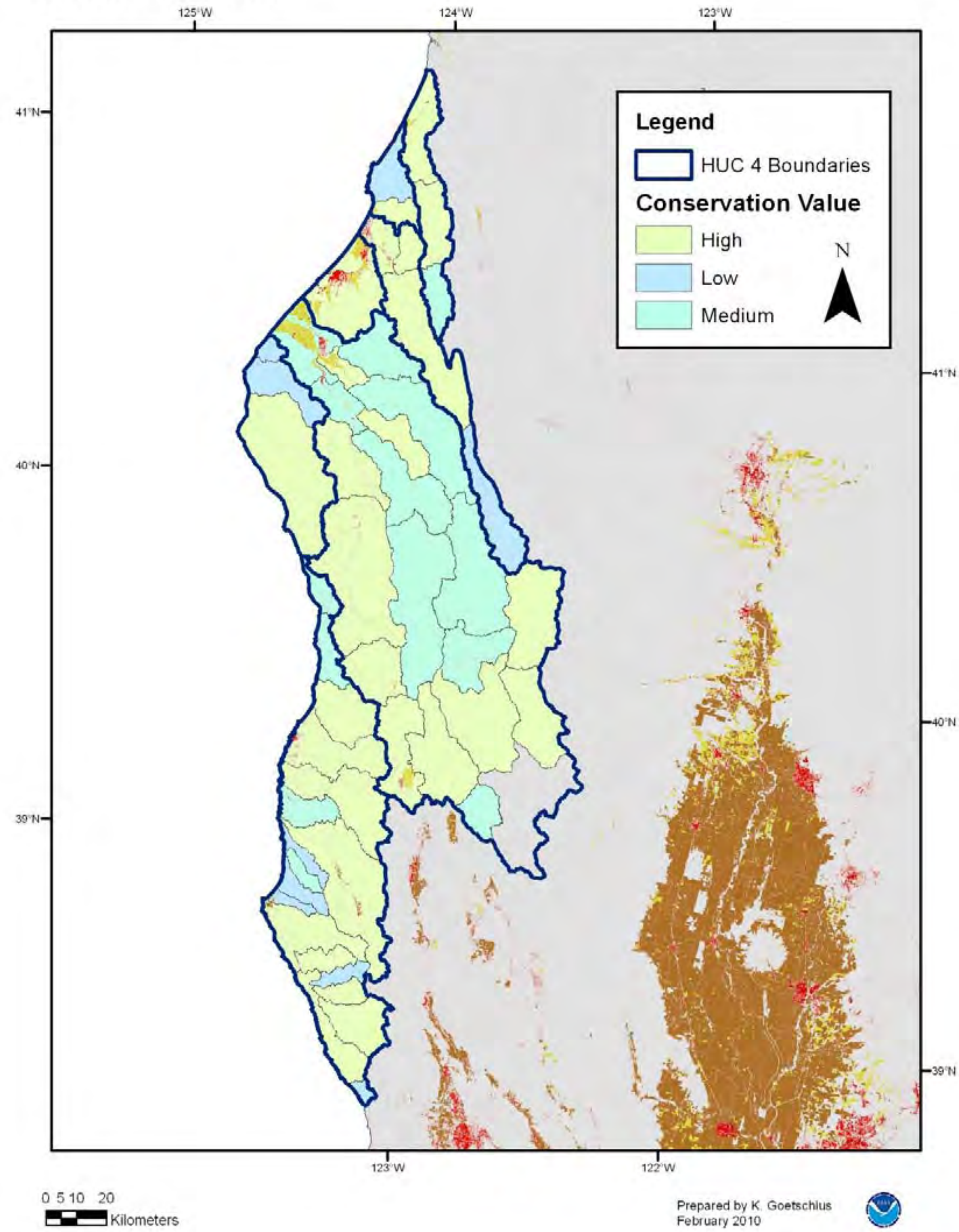

Snake River Steelhead DPS
Species Distribution
Legend
N
HUC 4 Boundaries
Distribution Buffer
Migratory Corridor
Landcover
Cultivated Crops
Developed, High Intensity
Developed, Low Intensity
Developed, Medium Intensity
Developed, Open Space
Hay/Pasture
0 25 50 100
Kilometers
Prepared by K. Goetschius
February 2010


Snake River Steelhead DPS
Critical Habitat
Legend
HUC 4 Boundaries
Migratory Corridor
Conservation Value
High
Low
Medium
N
0 25 50 100
Kilometers
Prepared by K. Goetschius
February 2010

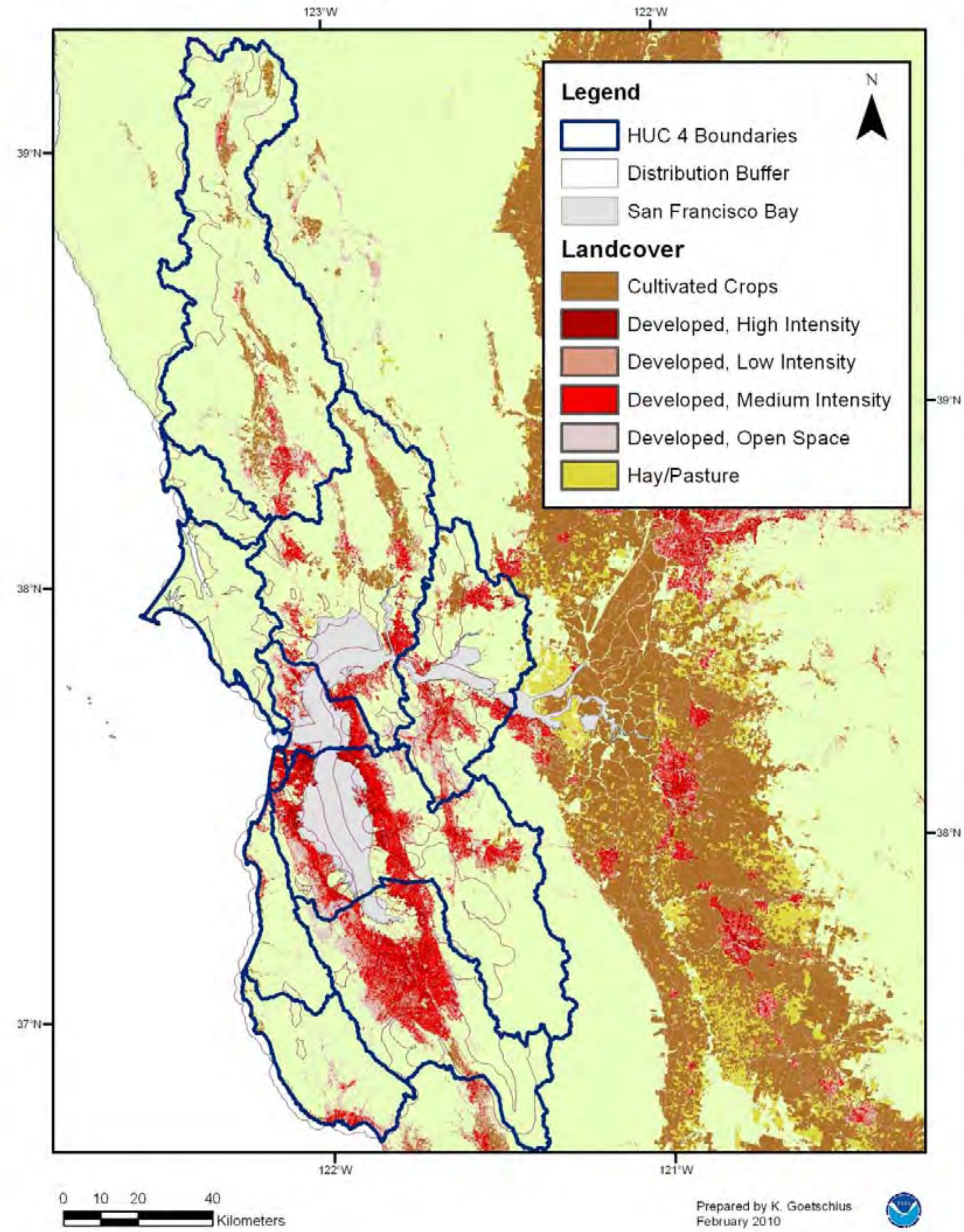


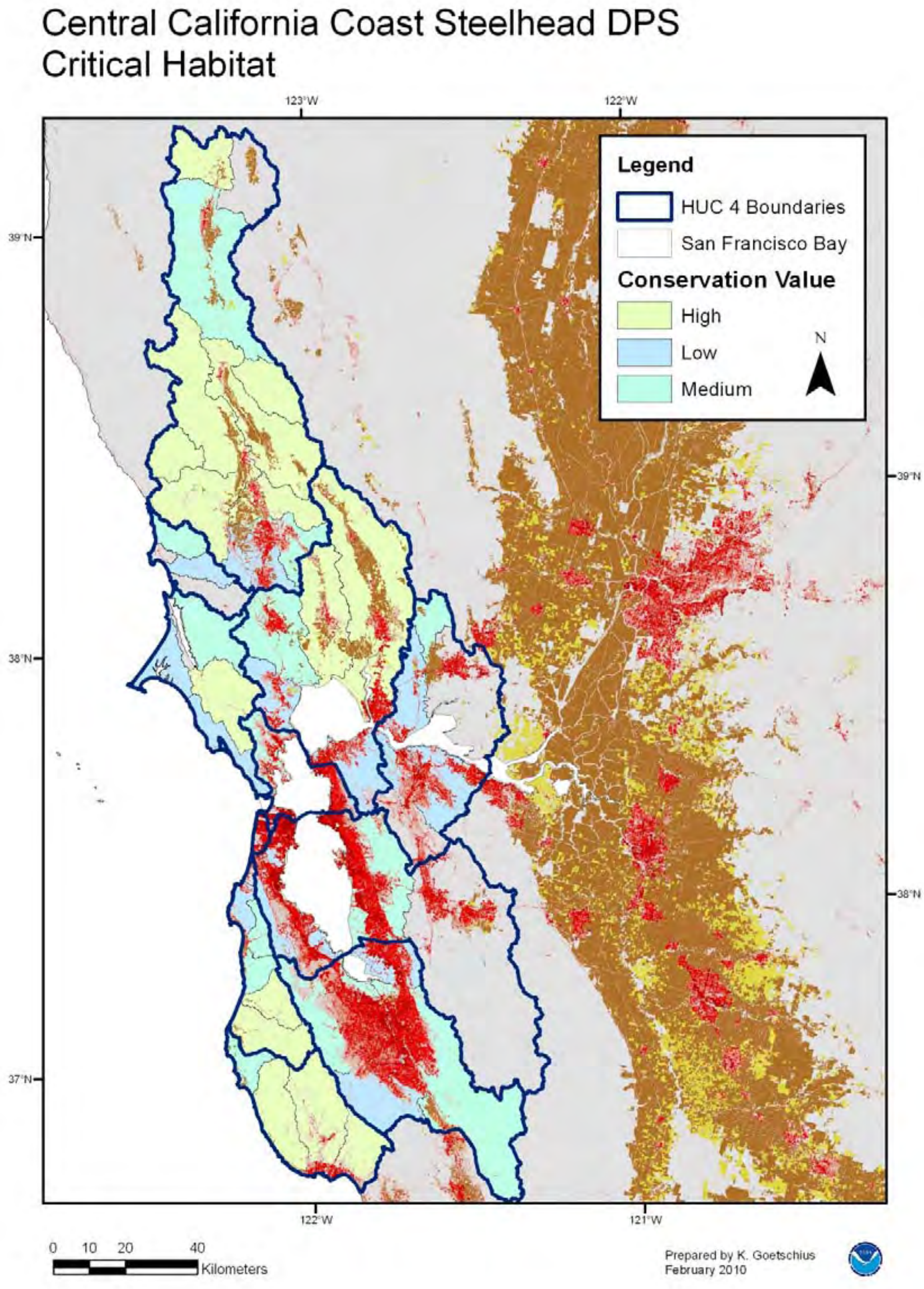
Northern California Steelhead DPS
Species Distribution
125°W
124°W
123°W
122°W
41°N
40°N
39°N
123°W
122°W
121°W
41°N
40°N
39°N
Legend
HUC 4 Boundaries
Distribution Buffer
Landcover
Cultivated Crops
Developed, High Intensity
Developed, Low Intensity
Developed, Medium Intensity
Developed, Open Space
Hay/Pasture
N
0 5 10 20
Kilometers
Prepared by K. Goetschius
February 2010

Northern California Steelhead DPS
Critical Habitat

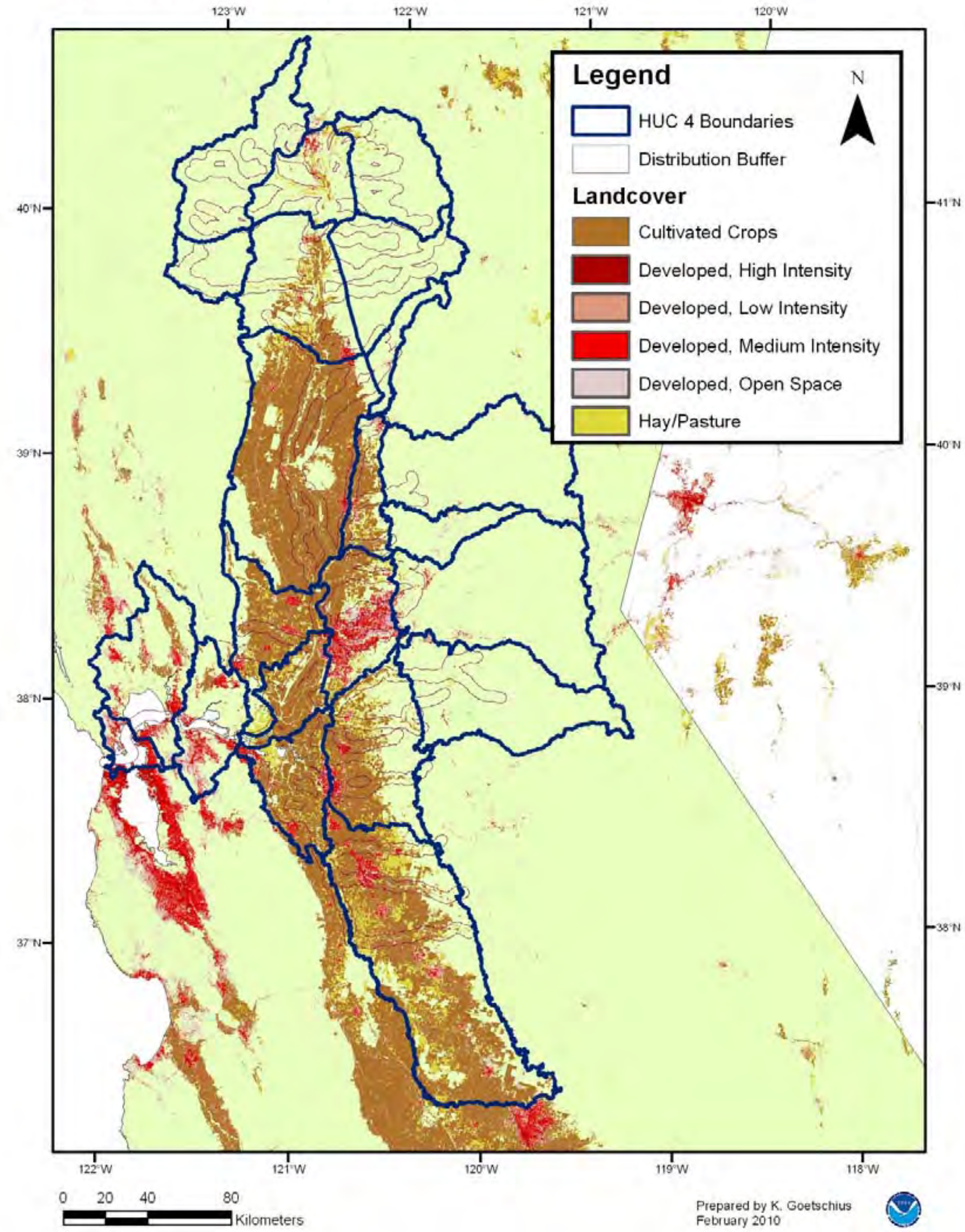


Central California Coast Steelhead DPS
Species Distribution

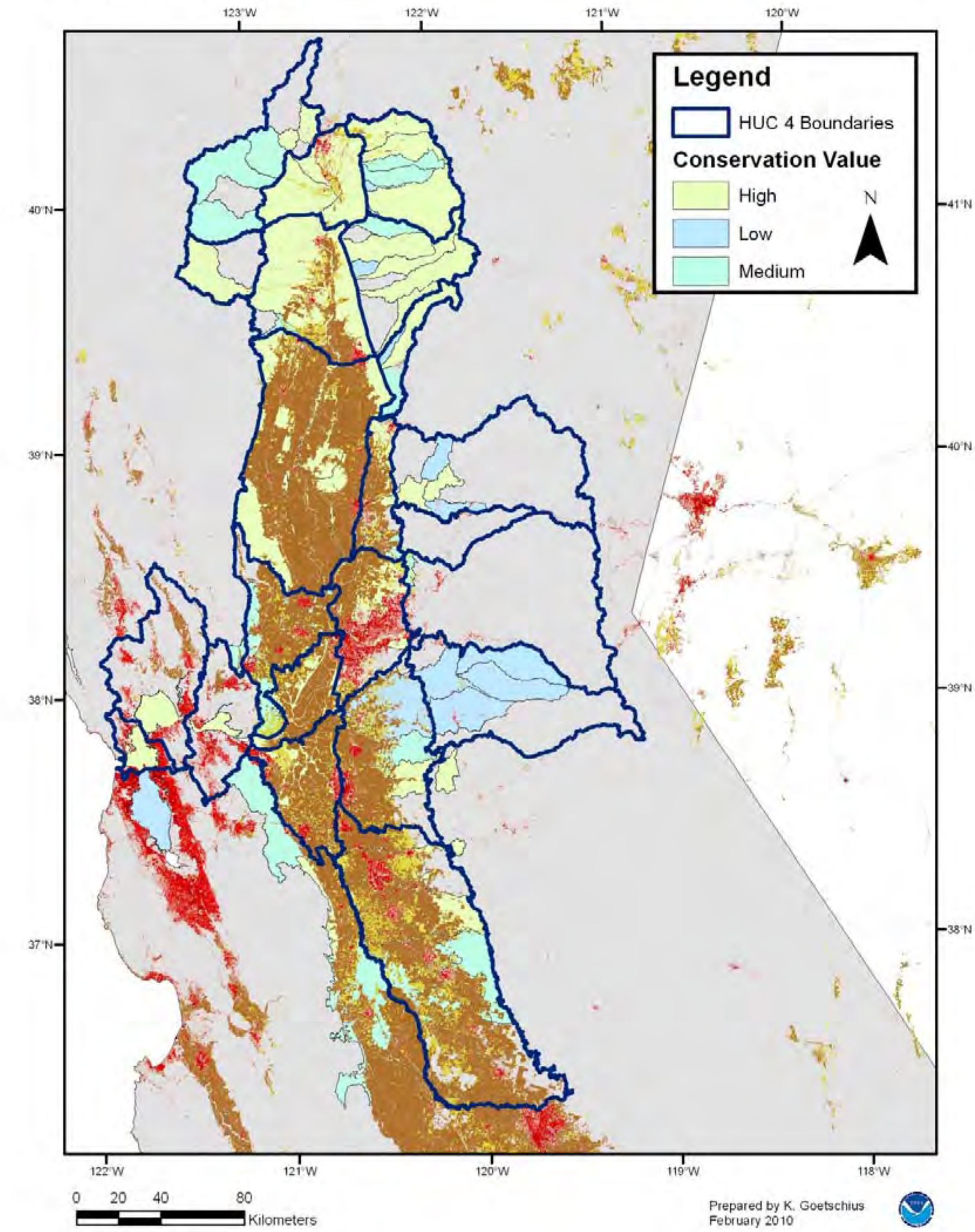



Central California Coast Steelhead DPS
Critical Habitat
123°W
122°W
121°W
39°N
38°N
37°N
Legend
HUC 4 Boundaries
San Francisco Bay
Conservation Value
High
Low
Medium
N
0 10 20 40
Kilometers
Prepared by K. Goetschius
February 2010

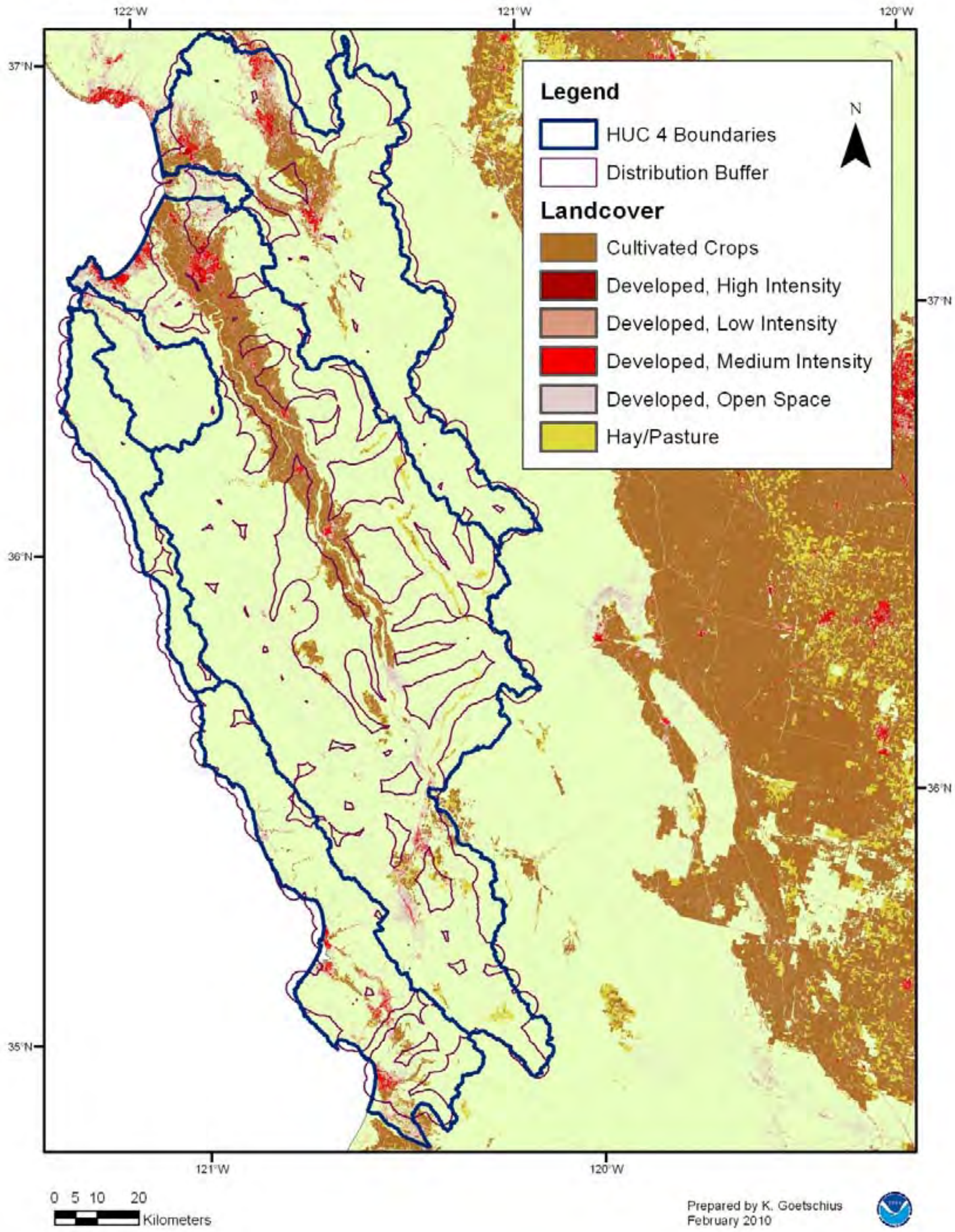
California Central Valley Steelhead DPS
Species Distribution



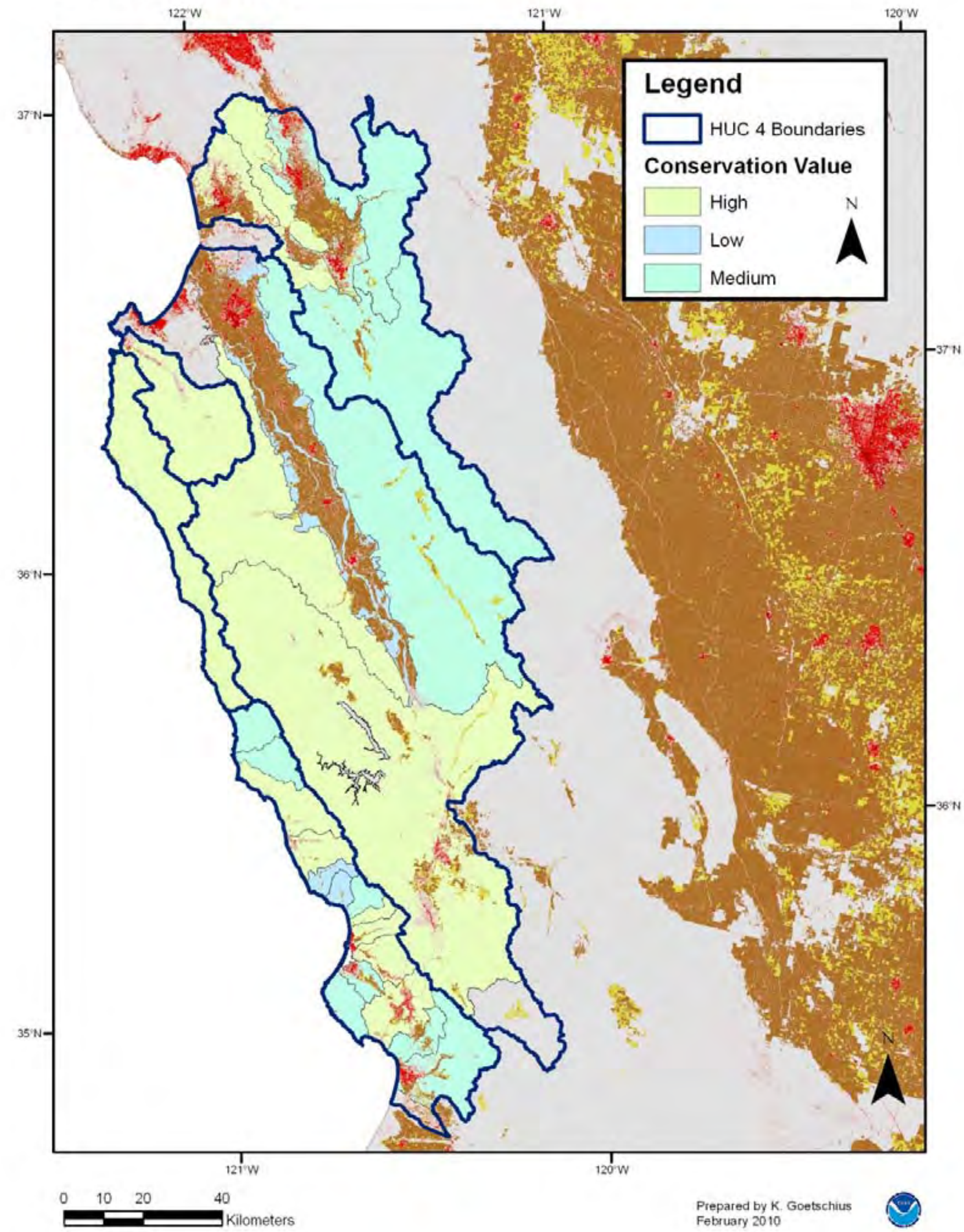
California Central Valley Steelhead DPS Critical Habitat



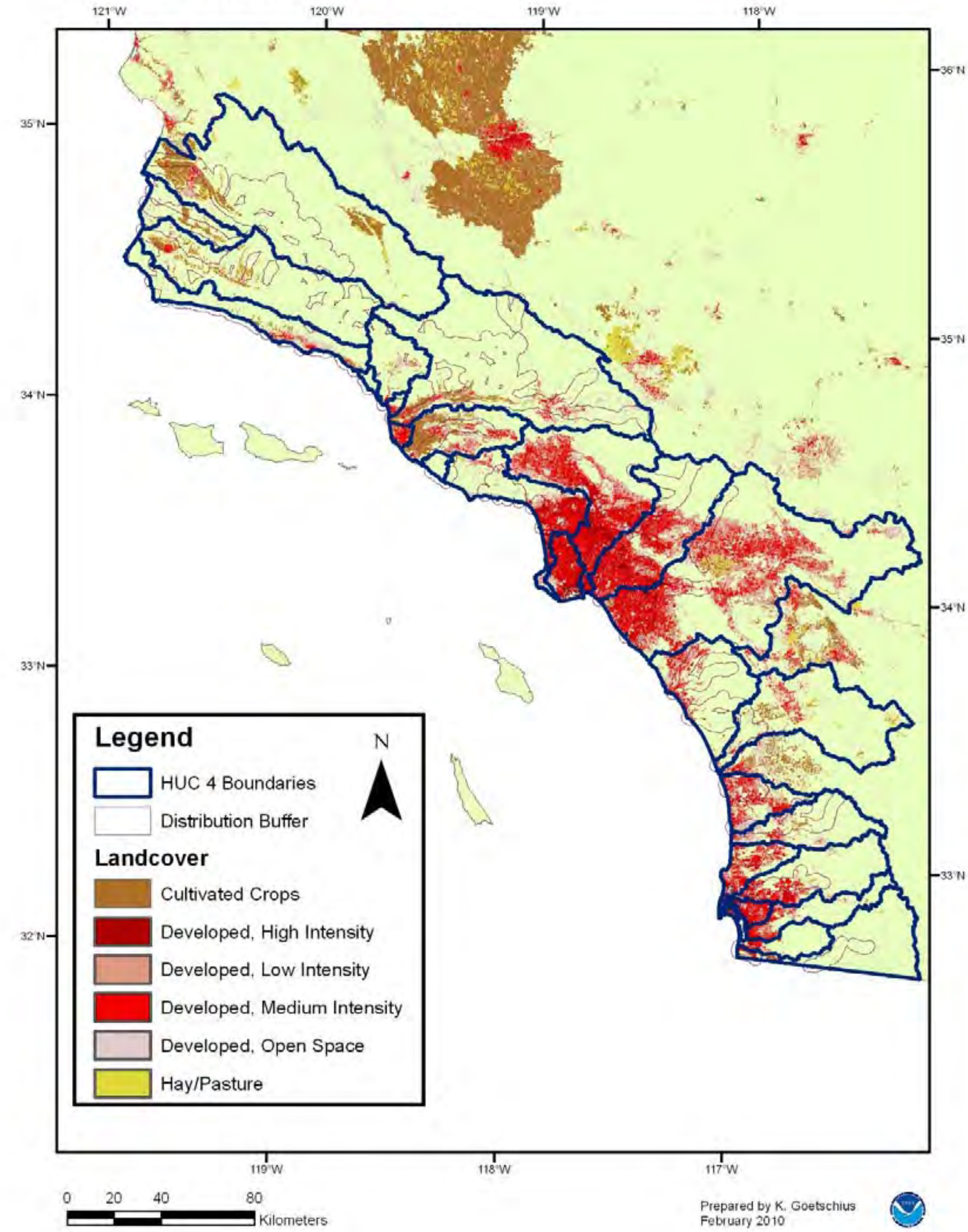
South-Central California Coastal Steelhead DPS Species Distribution



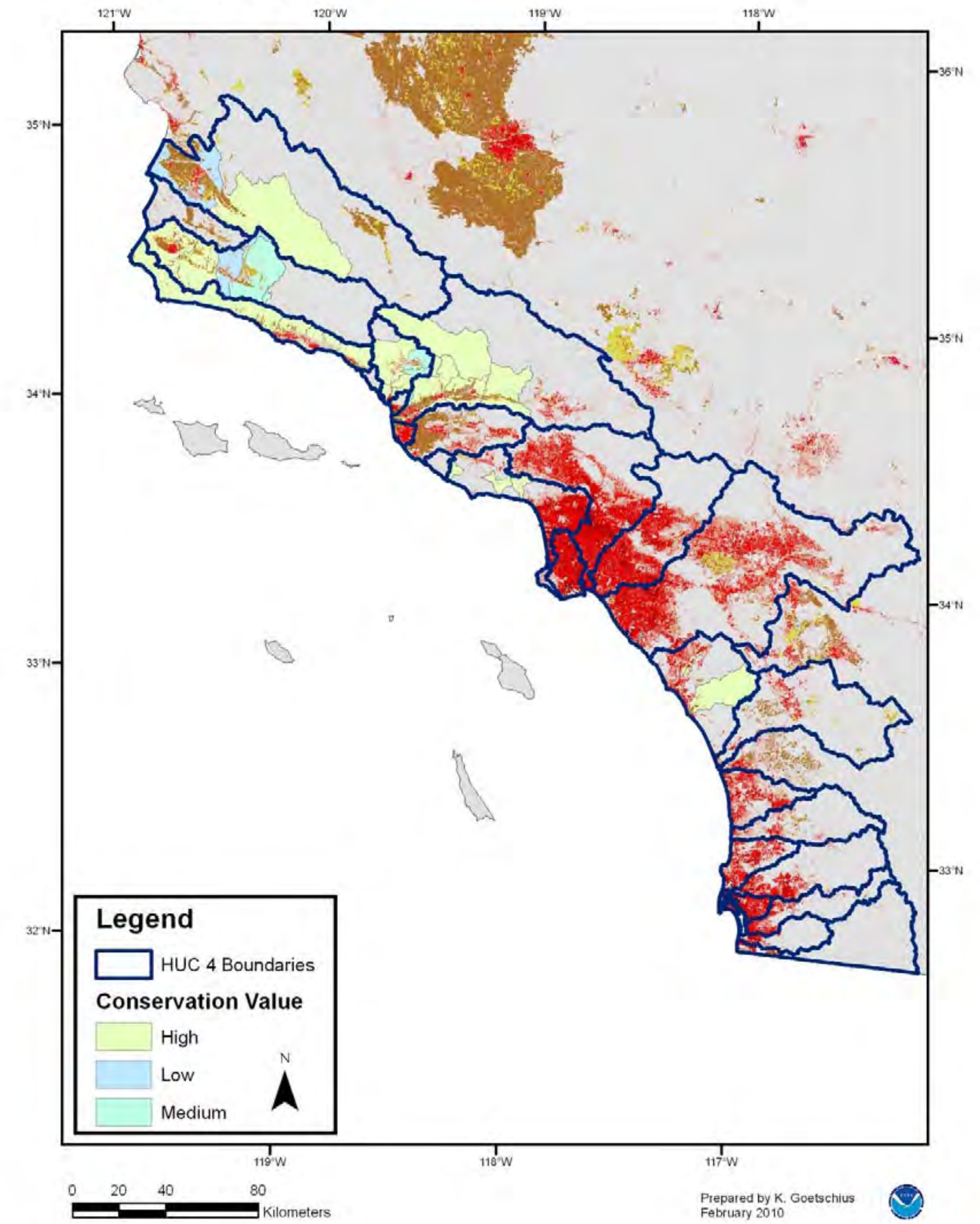
South-Central California Coastal Steelhead DPS Critical Habitat



Southern California Steelhead DPS
Species Distribution



Southern California Steelhead DPS Critical Habitat



Appendix 6: Generalized Average Annual Run-timing for ESA listed Pacific Coast Salmon and Steelhead

Chinook Salmon

Puget Sound Chinook (spring/summer, fall combined)

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases												
Rearing and migration (juveniles)												

Lower Columbia River Chinook

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases												
Rearing and migration (juveniles)												

Upper Columbia River Spring-run Chinook (Endangered)

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases												
Rearing and migration (juveniles)												

Chum Salmon

Hood Canal Summer-run

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases												
Rearing and migration (juveniles)												

Columbia River Chum

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases												
Rearing and migration (juveniles)												

Coho Salmon

Lower Columbia River Coho

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases												
Rearing and migration (juveniles)												

Sockeye Salmon

Ozette Lake Sockeye

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases												
Rearing and migration (juveniles)												

Steelhead

Puget Sound Steelhead (winter/summer runs)

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases												
Rearing and migration (juveniles)												

Lower Columbia River Steelhead (winter/summer runs)

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases												
Rearing and migration (juveniles)												

Middle Columbia River Steelhead

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases												
Rearing and migration (juveniles)												

Upper Columbia River Steelhead

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases												
Rearing and migration (juveniles)												

Steelhead

Upper Willamette River

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases												
Rearing and migration (juveniles)												

Snake River Basin Steelhead

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases												
Rearing and migration (juveniles)												

Chinook Salmon

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)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases												
Rearing and migration (juveniles)												

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)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases												
Rearing and migration (juveniles)												

Upper Willamette River Chinook

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases												
Rearing and migration (juveniles)												

Sockeye Salmon

Snake River Sockeye Salmon (Endangered)

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases												
Rearing and migration (juveniles)												

Coho Salmon

Oregon Coast Coho

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases												
Rearing and migration (juveniles)												

California Listed ESU / DPS Life Histories

Coho

Central California Coast Coho

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases												
Rearing and migration (juveniles)												

Southern Oregon / North California Coast Coho

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases												
Rearing and migration (juveniles)												

Chinook

California Coastal

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases												
Rearing and migration (juveniles)												

Central Valley Spring-run Chinook

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases												
Rearing and migration (juveniles)												

Sacramento River Winter-run Chinook (endangered)

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases												
Rearing and migration (juveniles)												

Steelhead

Northern California

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases												
Rearing and migration (juveniles)												

Central California Coast Steelhead

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases												
Rearing and migration (juveniles)												

California Central Valley Steelhead

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases												
Rearing and migration (juveniles)												

South- Central California Coast Steelhead

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases												
Rearing and migration (juveniles)												

Southern California Steelhead (endangered)

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases												
Rearing and migration (juveniles)												

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Appendix 7. AgDrift Tier III aerial assessment input values for mixture calculation

Aircraft: Air Tractor AT-401

Boom length: 76.3%

Boom height: 10 feet

Flight lines: 1

Swath width definition: Fixed, 60 feet

Swath width displacement: 0.3702 fractional displacement

Droplet size distribution: ASAE course

Nonvolatile rate: 13.5 lb/A

Active Rate: 12 lbs diuron/A, 1.5 lbs imazapyr/A

Spray volume rate: 20 gallons/A

Carrier: water

Windspeed: 10 mph

Temperature: 86 degrees F

Relative humidity: 50%

Flux plane: 0 ft

Distance to water body from edge of application area: 0 and 300 feet

Aquatic definition: EPA-defined pond and NMFS-defined floodplain habitat (2 m wide, 0.1 m deep)

Appendix 8 – Toxicityof Six Herbicides and Fungicides to Embryonic Zebrafish

May 2, 2011

Introduction

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 select herbicides and fungicides 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 linuron, diuron, triclopyr butoxyethyl ester (triclopyr BEE), chlorothalonil, 2,4-D and captan individually in 5-day static exposures. Toxicity endpoints included mortality, developmental abnormalities, and body length on the final day of the experiment. Two of the compounds tested, captan and 2,4-D, did not have an adverse effect on zebrafish survival, morphology or length at the tested concentrations. However, linuron, diuron, triclopyr BEE, and chlorothalonil negatively impacted zebrafish survival, and all but diuron increased the frequency of physical deformity. Additionally, body lengths were significantly smaller following exposure to triclopyr BEE and chlorothalonil.

Methods

Fish: Zebrafish (*D. rerio*) embryos were obtained from a colony maintained at the Northwest Fisheries Science Center according to standard operational 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 ^{o}C for the duration of the experiment.

Pesticide stock solutions: Herbicides and fungicides 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 ^{o}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 pesticide concentrations tested are listed in Table 1.

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 (n= 45). Exposures were conducted in batches comprised of two pesticides, water controls, and 0.1% acetone controls each run in triplicate. Exposure solutions were renewed every 24 hours. Dead embryos were removed from the dishes each day to prevent fungal growth and contamination of healthy embryos.

Anatomical screening and measurement of fish body length: Embryos were scored every 24 hours 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/).

Results

We found that, at the concentrations tested, linuron, diuron, triclopyr BEE, and chlorothalonil adversely affected survival of zebrafish embryos, while 2,4-D and captan did not (Figures 1 – 7). Furthermore, exposure to linuron, triclopyr BEE and chlorothalonil produced an increase in the frequency of developmental abnormalities in surviving embryos. Fish exposed to 0.1 mg/l chlorothalonil had deformed fins, an abnormality not present in any other exposure. Because of this result, an additional chlorothalonil exposure was conducted to document the extent of the fin deformity (Figure 6). In the length analysis, a two-factor ANOVA comparing batch, water and acetone controls showed a significant result of batch only. Subsequent analyses of exposures were thus compared to their corresponding batch controls (Table 3). One-way ANOVAs determined that chlorothalonil and triclopyr BEE exposures significantly affected length (Table 3). A Dunnett's post hoc test revealed that only the highest concentrations of chlorothalonil and triclopyr BEE were significantly shorter than water controls. A significant decrease in length was expected for the highest concentration of triclopyr BEE, as all fish displayed edema, which is known to decrease body length. Chemical-specific mortality and abnormality data, as well as their respective controls, are presented in Figures 1 - 7. Both water and acetone controls showed consistently low rates of both mortality and abnormality.

Table 1. Herbicide and fungicide concentrations tested.

Compound Name	Type	Exposure Concentrations (mg/l)
Linuron	Herbicide	0.01, 0.1, 1, 10
2,4-D	Herbicide	0.01, 0.1, 1, 10
Diuron	Herbicide	0.01, 0.1, 1, 10
Triclopyr BEE	Herbicide	0.01, 0.1, 1, 10
Chlorothalonil	Fungicide	0.001, 0.01, 0.1, 1
Captan	Fungicide	0.001, 0.01, 0.1, 1

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 so much that a line drawn from the posterior tip of the notochord to the mouth of the fish would yield a gap between line and body
Deformed fins	The absence or improper formation of fin tissue
Bent	A bend in the body or tail of the embryo in the coronal plane

Table 3. Average lengths (mm) of control and exposed fish. The highest concentrations of triclopyr BEE and chlorothalonil produced significantly shorter fish (one-way ANOVA, $p < 0.005$, $n = 3$; Dunnett's post hoc $p < 0.05$). Fungicides are listed in italics. NA indicates that lengths were not measured as all embryos died. Shaded areas indicate concentrations not tested. Vertical lines separate batches.

Result (*p*) and concentration	Linuron	2,4-D	Diuron	Triclopyr BEE	*Chlorothalonil*	*Captan*
P	0.32	0.58	0.07	<0.005*	<0.005*	0.46
water control	3.63		3.80		3.86	
0.1% acetone	3.63		3.82		3.87	
0.001 mg/l					3.88	3.85
0.01 mg/l	3.62	3.63	3.85	3.84	3.85	3.86
0.1 mg/l	3.61	3.66	3.81	3.83	3.60**	3.84
1 mg/l	3.57	3.58	3.84	3.48**	NA	3.85
10 mg/l	NA	NA	NA	NA		

* Significant result of one-way ANOVA

** Significant result of Dunnett's post hoc test

Linuron

Linuron exposure produced a concentration-dependent increase in zebrafish embryo mortality. At a concentration of 1 mg/l, 51% of the zebrafish died, while 10 mg/l produced 100% mortality (Figure 1). Abnormalities were present in the embryos, including curvature and edema. The highest incidence was observed for the 1 mg/l concentration, where 9% of surviving embryos displayed one or both of the abnormalities. No abnormalities could be noted at the highest concentration of 10 mg/l as none of the embryos survived the exposure. Data are means of 3 replicate dishes (± one standard deviation).

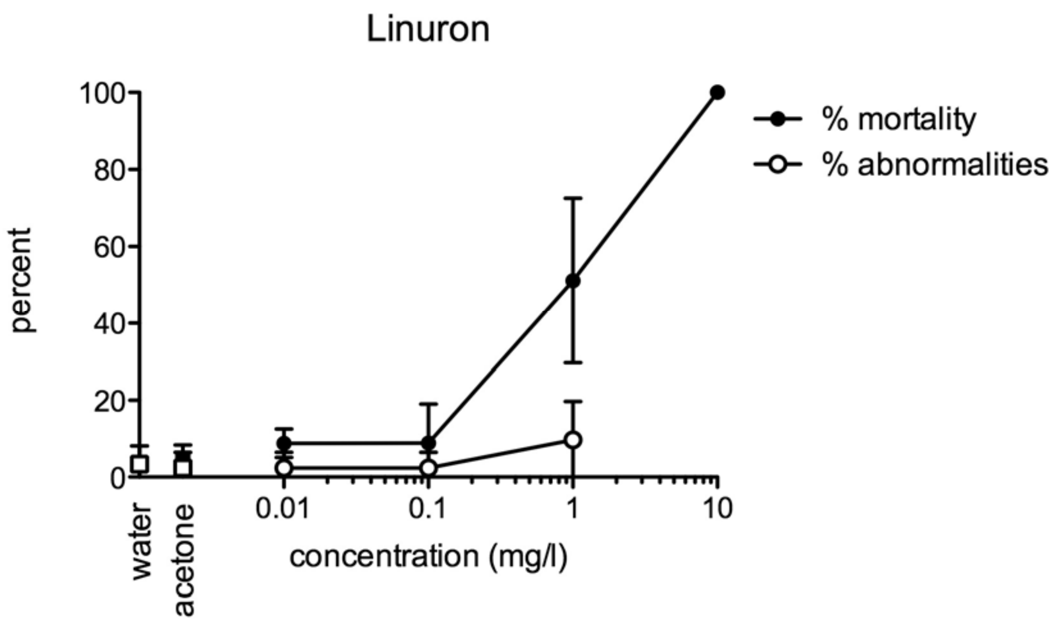

Figure 1. Percentage of mortality and abnormalities observed in zebrafish embryos during the 5-day linuron exposure.

2,4-D

2,4-D exposure did not significantly alter mortality or developmental abnormality rates. The highest rate of mortality (6.7%) was observed for the 10 mg/l concentration. The 1 mg/l exposure group had the highest rate of abnormality (4.7%) (Figure 2). Data are means of 3 replicate dishes (± one standard deviation).

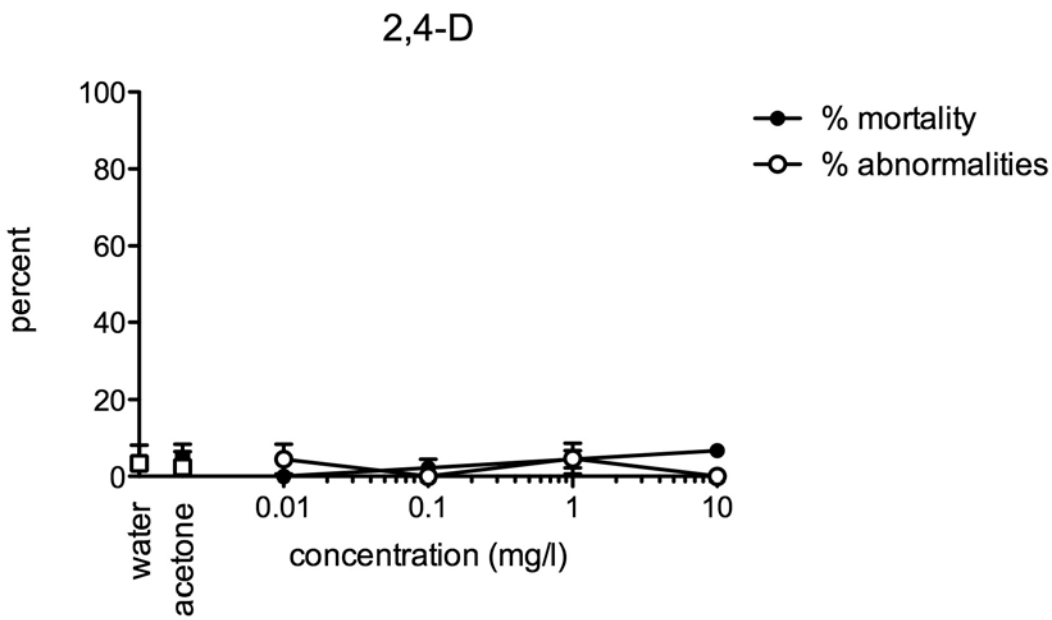

Figure 2. Percentage of mortality and abnormality observed in zebrafish embryos during the 5-day 2,4-D exposure.

Diuron

Exposure to diuron did not increase rates of mortality except for the highest concentration (10 mg/l), in which all embryos died (Figure 3). Diuron did not produce abnormalities in any of the surviving embryos from the other exposure groups (Figure 4). Data are means of 3 replicate dishes (± one standard deviation).

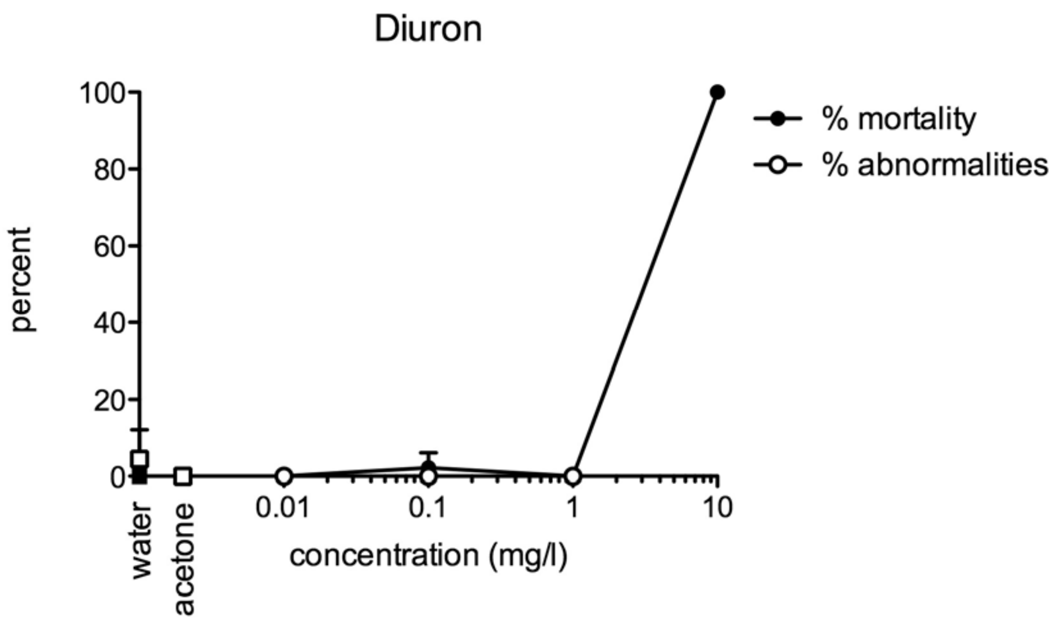

Figure 3. Percentage of mortality and abnormality observed in zebrafish embryos during the 5-day diuron exposure.

Triclopyr BEE

No mortality was observed in embryos exposed to the lower concentrations of Triclopyr BEE. However, 100% mortality was observed at 10 mg/l (Figure 4). Triclopyr BEE did have sublethal developmental effects; all zebrafish exposed to 1 mg/l showed edema. Data are means of 3 replicate dishes (± one standard deviation).

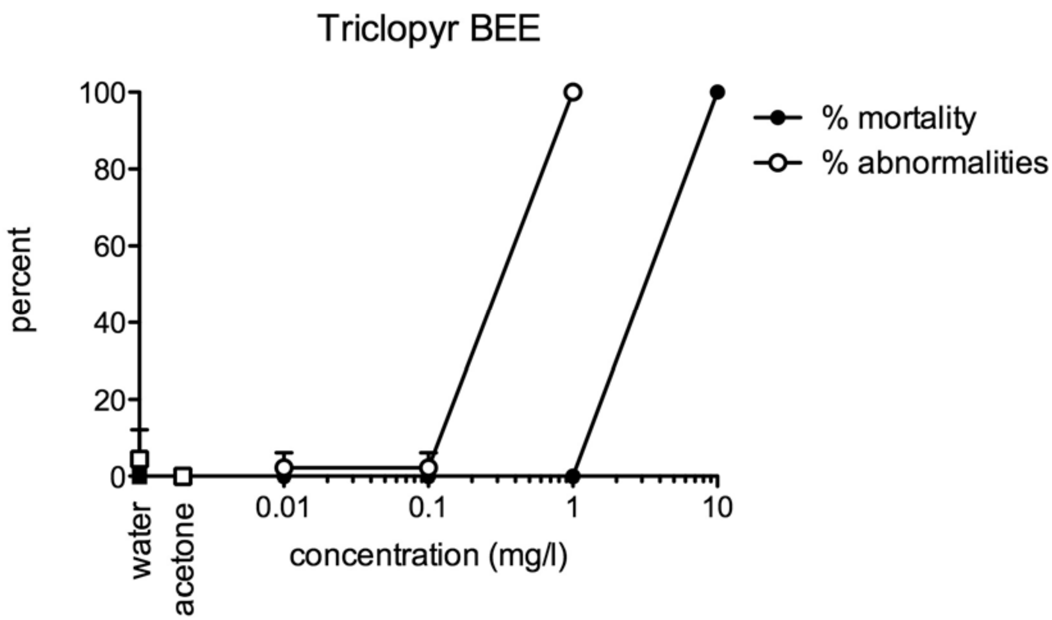

Figure 4. Percentage of mortality and abnormality observed in zebrafish embryos during the 5-day triclopyr BEE exposure.

Chlorothalonil

Embryos exposed to increasing concentrations of chlorothalonil had increased mortality (Figure 5). At 0.1 mg/l, 75.6% of the zebrafish had died, and mortality was 100% at the highest exposure concentration (1 mg/l). Data are means of 3 replicate dishes (± one standard deviation). Also, chlorothalonil produced an unusual abnormality of fin deformity not seen with the other pesticides. At 0.1 mg/l, 93% of the surviving fish displayed fin abnormalities (Figure 6).

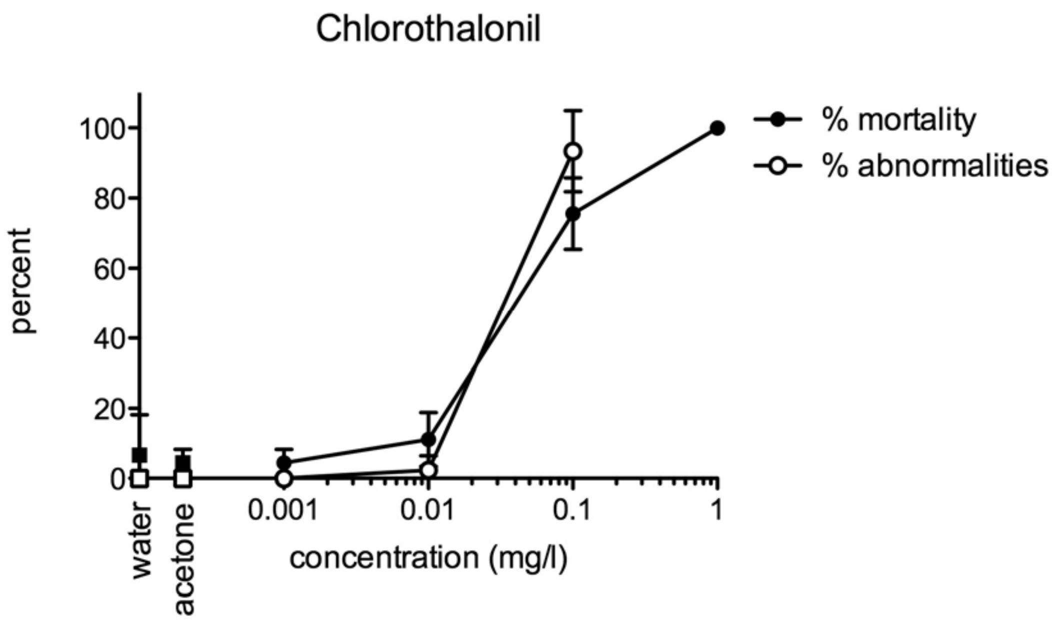

Figure 5. Percentage of mortality and abnormalities observed in zebrafish embryos during the 5-day chlorothalonil exposure.

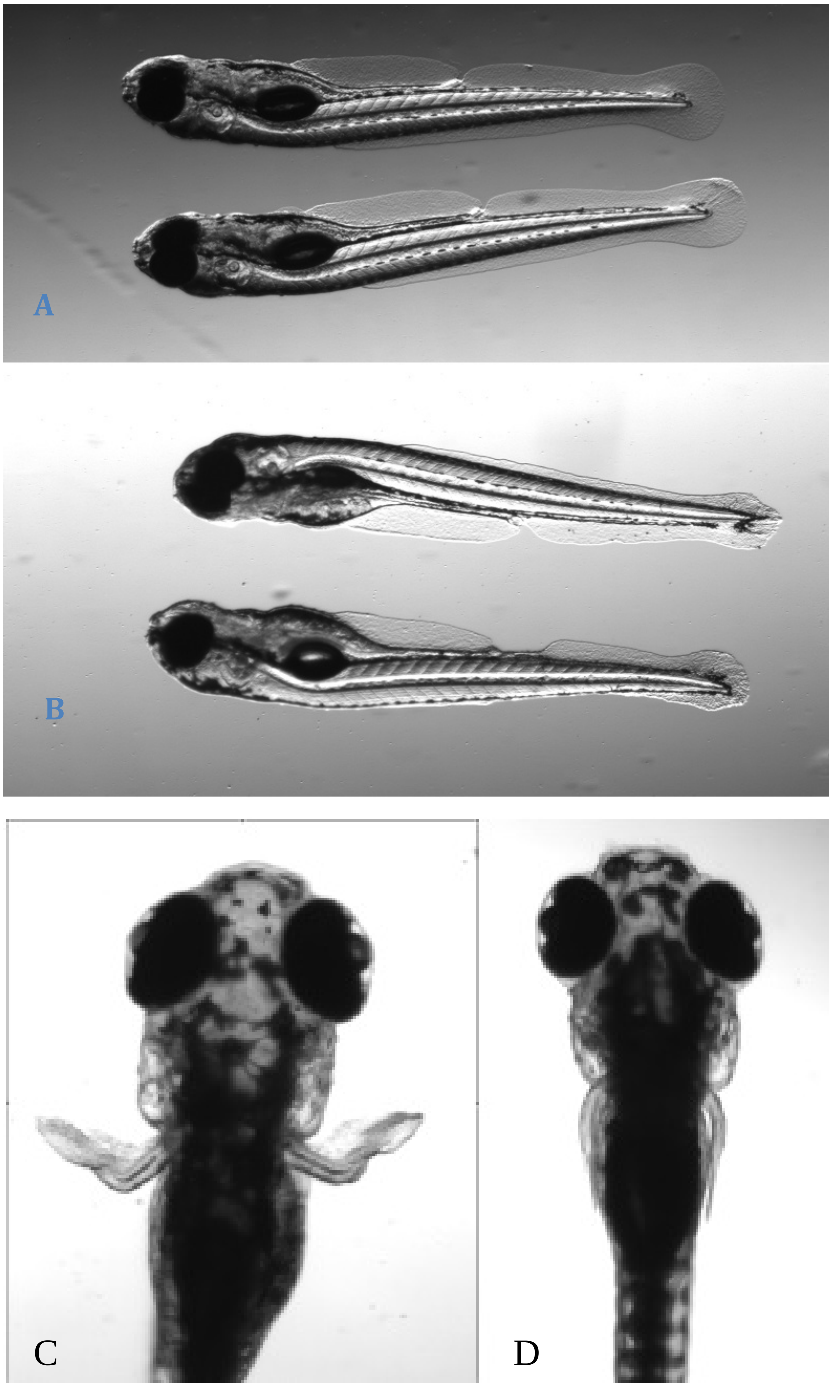

Figure 6. A comparison of fin development between control and chlorothalonil treated zebrafish. A) Two normally developing zebrafish from the system water control group. Note that the caudal fin rays create a regular fan shape, and there is defined fin tissue on both the ventral and dorsal sides of the fish. B) Two fish exposed to 0.1 mg/l chlorothalonil from 24-48 hpf, followed by recovery in system water from 48-120 hpf. The caudal, ventral and dorsal fins are irregular and partially deteriorated. Fish in this treatment group lacked fin tissue entirely immediately after exposure, indicating total fin deterioration. Fin growth resumed after the fish were transferred to clean system water. C) Larval fish exposed to 0.1 mg/l chlorothalonil for 5 days displaying abnormal development of pectoral fins. D) Control fish at 5 dpf showing normal pectoral fins resting along side the body.

Captan

Captan did not show an increased level of mortality with increasing concentration (Figure 7). The highest level of mortality was 8.9% at 0.1 mg/l. No abnormalities were observed in any treatment group, suggesting that captan does not adversely impact zebrafish development at the concentrations tested here.

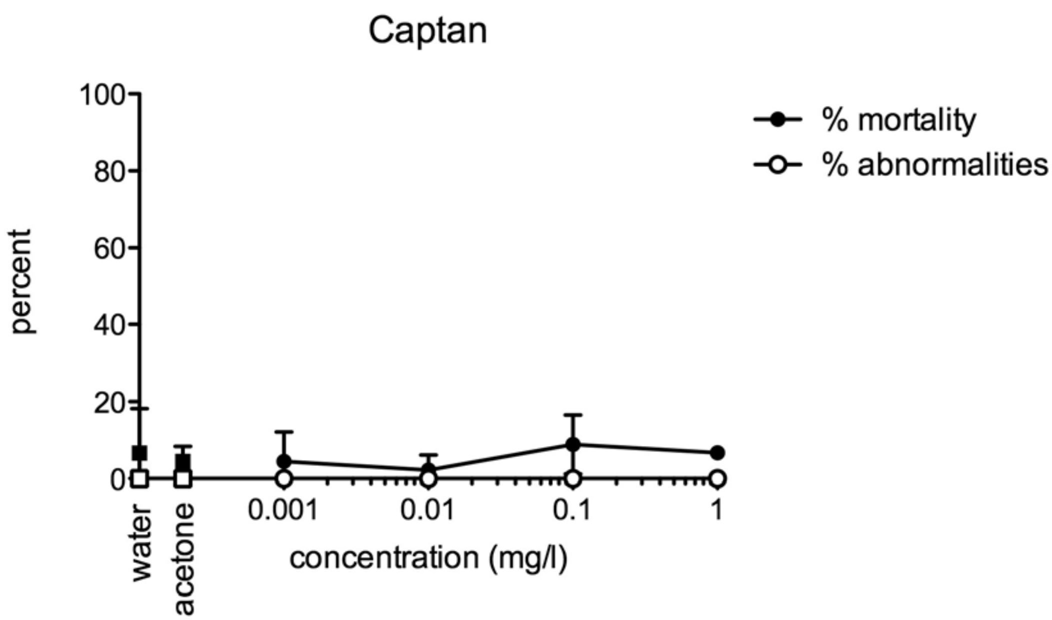

Figure 7. Percentage of mortality and abnormality observed in zebrafish embryos during the 5-day captan exposure.

References

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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.

Appendix 9 – 2,4-D Aquatic Weed Control Treatment Windows to Minimize Effects to Threatened and Endangered Pacific Salmonids

See Table on following page.

Appendix 9. Allowable application windows to control invasive and exotic aquatic plant species with 2,4-D per Term and Condition 2.b.

Species	ESU	Allowable Application Period
Chinook	Puget Sound	July 15 – August 31
	Lower Columbia River	July 15 – August 31
	Upper Columbia River Spring-Run	July 15 – August 31
	Snake River Fall-Run	September 1 – September 30
	Snake River Spring/Summer-Run	July 15 – August 15
	Upper Willamette River	July 15 – August 31
	California Coastal	July 15 – August 15
	Central Valley Spring-Run	Legal Delta only: June 15 – September 15; elsewhere July 15 – August 30
	Sacramento River Winter-Run	Legal Delta only: June 15 – September 15; elsewhere July 15 – August 30
Chum	Hood Canal Summer-Run	June 15 – August 15
	Columbia River	June 1 – August 31
Coho	Lower Columbia River	August 1 – September 15
	Oregon Coast	July 1 – August 31
	Southern Oregon / Northern California Coast	July 15 – August 15
	Central California Coast	July 15 – August 15
Sockeye	Ozette Lake	June 1 – September 30
	Snake River: Lakes: Red Fish, Alturas, Yellowbelly, Pettit, and Stanley	July 15 – August 15
Steelhead	Puget Sound	July 15 – September 30
	Lower Columbia River	August 1 – August 31
	Upper Willamette River	July 15 – August 31
	Middle Columbia River	August 1 – August 15
	Upper Columbia River	August 15 – August 31
	Snake River	August 1 – September 30
	Northern California	July 15 – August 15
	Central California Coast	July 15 – August 15
	California Central Valley	Legal Delta only: June 15 – September 15; elsewhere July 15 – August 15
	South-Central California Coast	July 15 – August 15
	Southern California	July 15 – August 15

Appendix 10 – Ecological Effects Data from EPA's Red Legged Frog BE's

Information in this Appendix is taken directly from the ecological effects appendices in EPA's California Red-legged Frog BEs[17]. Rather than restating all the information contained in those appendices in the main body of this Opinion, we have summarized information appropriate to NMFS's analyses. In some cases, endpoints presented in the *Response* section of the Opinion have been summarized as median and range and/or only certain endpoints have been used in the analysis. Details on how the information was summarized and used are provided in the a.i. specific discussions.

In some cases, the appendices had a title page, and or page numbers and in other cases they did not. If there were page number, they are listed in the reference as given in the original document.

Additionally, NMFS downloaded and considered the original data in Mayer and Ellersieck 1986. The downloaded data, in original form, is also included in this appendix. Page number within this appendix listed below.

[17] Available at http://www.epa.gov/oppfead1/endanger/litstatus/effects/redleg-frog/index.html

Sections included in this appendix (listed in order of appearance) are:

EPA 2009. Risks of 2,4-D use to the federally threatened California red-legged frog *(Rana aurora draytonii)* and Alameda whipsnake (*Masticophis lateralis euryxanthus).* Environmental Protection Agency, Office of Pesticide Programs, Environmental Fate and Effect Division, Washington, DC. *Appendix F, Ecological Effects Data (pages F-1 to F-32)*

EPA 2009. Risks of triclopyr use to the federally threatened California red-legged frog *(Rana aurora draytonii)*. Environmental Protection Agency, Office of Pesticide Programs, Environmental Fate and Effect Division, Washington, DC. *Appendix A, Ecological Effects Data, (no page numbers)*

EPA 2009. Risks of diuron use to the federally threatened California red-legged frog *(Rana aurora draytonii)*. Environmental Protection Agency, Office of Pesticide Programs, Environmental Fate and Effect Division, Washington, DC. *Appendix L, Ecological Effects Data*, *(no page numbers)*

EPA 2009. Risks of linuron use to the federally threatened California red-legged frog *(Rana aurora draytonii)*. Environmental Protection Agency, Office of Pesticide Programs, Environmental Fate and Effect Division, Washington, DC. *Appendix A, Ecological Effects, (no page numbers)*

EPA 2007. Risks of captan use to the federally listed California red-legged frod *(Rana aurora draytonii).* Environmental Protection Agency, Office of Pesticide Programs, Environmental Fate and Effect Division, Washington, DC. *Appendix A, Ecological Effects Data for Captan, (no page numbers)*

EPA 2007. Potential risks of labeled chlorothalonil uses to the federally listed California red-legged frog *(Rana aurora draytonii).* Environmental Protection Agency, Office of Pesticide Programs, Environmental Fate and Effect Division, Washington, DC. *Appendix B, Chlorothalonil Ecological Effects Characterization, (pages 1-26)*

Mayer and Ellersieck 1986. Manual of acute toxicity, interpretation and database for 410 chemicals and 66 species of freshwater animals. USFWS. Resource Publication 160. Data accessed and downloaded from the USGS website. Explanatory information available at: http://www.cerc.usgs.gov/clearinghouse/data/brdcerc0003.html. Data available at: http://www.cerc.usgs.gov/data/acute/acute.html

Appendix F

Ecological Effects Data

Table F-1. Acute freshwater fish toxicity to 2,4-D							
Species	**% ai**	**96-hour LC50 (95% CI)**		**Measured/ Nominal Test Concentrations**	**Toxicity Category (based on ae)**	**MRID No. Author/Year**	**Study Classification**
		(mg ai/L)	**(mg ae/L)**				
2,4-D acid (PC Code: 030001)							
Rainbow trout *(Oncorhynchus mykiss)*	98.7	358	358	Measured	Practically non-toxic	411583-01, Alexander et. al., 1983	Supplemental
Bluegill sunfish *(Lepomis macrochirus)*	98.7	263	263	Measured	Practically non-toxic	411583-01, Alexander et. al., 1983	Supplemental
Fathead minnow (*Pimephales promelas*)	98.7	320	320	Measured	Practically non-toxic	411583-01, Alexander et. al., 1983	Supplemental
2,4-D Sodium Salt (PC Code: 030004)							
Rainbow trout *(Oncorhynchus mykiss)*	80	>100	>91	Nominal	Practically non-toxic	53986, McCann, 1973	Acceptable
2,4-D Diethanolamine (DEA) Salt (PC Code: 030016)							
Rainbow trout *(Oncorhynchus mykiss)*	73.1	>120	>81.6	Measured	Practically non-toxic	419751-05, Graves. et. al., 1991.	Acceptable
Bluegill sunfish *(Lepomis macrochirus)*	73.1	>121	>82.3	Measured	Practically non-toxic	419751-04, Graves. et. al., 1991.	Acceptable
Fatahead minnow (*Pimephales promelas*)	73.1	344	234	Measured	Practically non-toxic	41975`-04, Graves, et. al, 1991	Acceptable
Bluegill sunfish *(Lepomis macrochirus)*	Not Reported	149	101	Measured	Practically non-toxic	0073-091-01, Sleight, B., 1971.	Supplemental
2,4-D Dimethylamine (DMA) Salt (PC Code: 030019)							
Rainbow trout *(Oncorhynchus mykiss)*	67.3	>1000	>830	Measured	Practically non-toxic	233350, Vilkas, A,G,, 1977.	Acceptable

Table F-1. Acute freshwater fish toxicity to 2,4-D							
Species	**% ai**	**96-hour LC50 (95% CI)**		**Measured/ Nominal Test Concentrations**	**Toxicity Category (based on ae)**	**MRID No. Author/Year**	**Study Classification**
		(mg ai/L)	**(mg ae/L)**				
Bluegill sunfish (*Lepomis macrochrus*)	73.1	>121	>100	Measured	Practically non-toxic	419751-04, Graves, et. al., 1991.	Acceptable
Bluegill sunfish *(Lepomis macrochirus)*	67.3	250	207.5	Measured	Practically non-toxic	411583-11, Alexander, et. al., 1983.	Acceptable
Bluegill sunfish *(Lepomis macrochirus)*	51.1 9TEP)	>1000	>830	Measured	Practically non-toxic	234027, Vilkas, A.G., 1978.	Supplemental
Fathead minnow (*Pimephales promelas*)	67.3	318	264	Measured	Practically non-toxic	419751-04, Graves. et. al., 1991.	Acceptable
2,4-D Isoproylamine (IPA) (PC Code: 030025)							
Rainbow trout (Oncorhynchus mykiss)	48.7 (TEP)	2840	2244	Measured	Practically non-toxic	01338869, 1983.	Acceptable
Bluegill sunfish *(Lepomis macrochirus)*	48.7 (TEP)	1700	1343	Measured	Practically non-toxic	01338869, 1983.	Acceptable
Fathead minnow (*Pimephales promelas*)	48.7 (TEP)	2180	1722	Measured	Practically non-toxic	01338869, 1983.	Supplemental
2,4-D Triisopropanolamine (TIPA) Salt (PC Code: 030035)							
Rainbow trout (Oncorhynchus mykiss)	69.2	300	162	Measured	Practically non-toxic	413538-03, Mayes, et. al., 1989.	Acceptable
Bluegill sunfish *(Lepomis macrochirus)*	69.2	401	217	Measured	Practically non-toxic	413538-04, Mayes, et al., 1989	Acceptable
2,4-D Butoxyethyl (BEE) Ester (PC code: 030053)							
Rainbow trout *(Oncorhynchus mykiss)*	97.4	2.09	1.44	Measured	Moderately toxic	413538-01, Alexander, et. al, 1983.	Acceptable

Table F-1. Acute freshwater fish toxicity to 2,4-D							
Species	**% ai**	**96-hour LC50 (95% CI)**		**Measured/ Nominal Test Concentrations**	**Toxicity Category (based on ae)**	**MRID No. Author/Year**	**Study Classification**
		(mg ai/L)	**(mg ae/L)**				
Rainbow trout *(Oncorhynchus mykiss)* static	77.5	0.65 (56-hour LC50)	0.45	Measured	Highly toxic	00050674, Pitcher, F.G., 1974.	Supplemental
Bluegill sunfish *(Lepomis macrochirus)*	Not reported	1.2	0.828	Measured	Moderately toxic	400980-01, Mayer, 1986.	Supplemental
Bluegill sunfish *(Lepomis macrochirus)*	97.4	0.62	0.428	Measured	Highly toxic	413538-01, Alexander, et. al, 1983.	Acceptable
Bluegill sunfish *(Lepomis macrochirus)*	29.0 (TEP)	>100	0.69	Measured	Practically non-toxic	400980-01, Mayer, 1986 J.A.,1969	Supplemental
Fathead minnow (*Pimephales promelas*)	97.4	2.60	1.79	Measured	Moderately toxic	413538-01, Alexander, et. al, 1983.	Acceptable
2,4-D 2-Ethylhexyl Ester (2-EHE) (PC code: 030063)							
Rainbow trout *(Oncorhynchus mykiss)*	66.9 (TEP)	4.82	3.2	Measured EHE	Moderately toxic	417373-03, Mayes, et. al., 1990.	Acceptable
Rainbow trout *(Oncorhynchus mykiss)*	92	22	14.5	Nominal	Slightly toxic	45068, Buccafusco, R.J., 1976.	Acceptable
Bluegill sunfish *(Lepomis macrochirus)*	92	18	11.9	Nominal	Slightly toxic	45069, Buccafusco, R.J., 1976.	Acceptable
2,4-D Isopropyl Ester (IPE) (PC code: 030066)							
Rainbow trout *(Oncorhynchus mykiss)* static	98.2	0.69	0.58	Measured	Highly toxic	439331-01, Drottar, et. al., 1996.	Acceptable

Table F-1. Acute freshwater fish toxicity to 2,4-D							
Species	**% ai**	**96-hour LC50 (95% CI)**		**Measured/ Nominal Test Concentrations**	**Toxicity Category (based on ae)**	**MRID No. Author/Year**	**Study Classification**
		(mg ai/L)	**(mg ae/L)**				
Rainbow trout *(Oncorhynchus mykiss)* static	45.8 (TEP)	0.78	0.66	Measured	Highly toxic	439332-01, Drottar, et. al., 1996.	Acceptable
Bluegill sunfish *(Lepomis macrochirus)* static	98.2	0.31	0.26	Measured	Highly toxic	439307-01, Drottar, et. al., 1996.	Acceptable
Bluegill sunfish *(Lepomis macrochirus)*	45.8 (TEP)	0.31	0.26	Measured	Slightly toxic	439103-01, Drottar, et. al., 1996.	Acceptable

Table F-2. Chronic (early life cycle) freshwater fish toxicity to 2,4-D						
Species	**% ai**	**Toxicity Value (mg ae/L)**		**Most sensitive endpoint**	**MRID No. Author/Year**	**Study Classification**
		NOAEC	**LOAEC**			
2,4-D acid (PC Code: 030001)*						
Fathead minnow (*Pimephales promelas*)	96.1	63.4	102	Larval survival	417373-04, Mayes, et. al.,1990.	Acceptable
2,4-D Diethanolamine (DEA) Salt (PC Code: 030016)*						
Fathead minnow (*Pimephales promelas*)	73.8	19.8	66.6	Larval survival	420183-04, Graves, et. al., 1991.	Acceptable
2,4-D Dimethylamine (DMA)Salt (PC Code: 030019)*						
Fathead minnow (*Pimephales promelas*)	66.5	14.2	23.6	Length	417677-01, Dill, et. al., 1990.	Acceptable

* No early life cycle freshwater fish studies using forms of 2,4-D other than acid, DEA, and DMA were submitted to the Agency.

Table F-3. Chronic (full life cycle) freshwater fish toxicity to 2,4-D						
Species	**% ai**	**Toxicity Value (mg ae/L)**		**Most sensitive endpoint**	**MRID No. Author/Year**	**Study Classification**
		NOAEC	**LOAEC**			
2,4-D Ethylhexyl Ester (EHE) (PC code: 030063)*						
Fathead minnow *(Pimephales promelas)*	94.7	0.0792	0.1452	Larval fish survival	417373-05, Mayes, et. al., 1990.	Supplemental

* No full life cycle freshwater fish studies using forms of 2,4-D other than EHE were submitted to the Agency.

Table F-4. Acute freshwater invertebrate toxicity to 2,4-D							
Species	**% ai**	**96-hour LC50 (95% CI)**		**Measured/ Nominal Test Concentrations**	**Toxicity Category (based on ae)**	**MRID No. Author/Year**	**Study Classification**
		(mg ai/L)	**(mg ae/L)**				
2,4-D acid (PC Code: 030001)							
Waterflea *(Daphnia magna)*	98.7	25	25	Measured	Slightly toxic	411583-01, Alexander et. al., 1983	Acceptable
2,4-D Sodium Salt (PC Code: 030004)							
No data							
2,4-D Diethanolamine (DEA) Salt (PC Code: 030016)							
Waterflea *(Daphnia magna)*	71.3	>100	>68	measured	Practically non-toxic	419751-06, Graves, et. al., 1991.	Acceptable
2,4-D Dimethylamine (DMA) Salt (PC Code: 030019)							
Waterflea *(Daphnia magna)*	51.1 (TEP)	774.5	642.8	NA	Practically non-toxic	232630, Vilkas, A.G., 1977.	Acceptable
Waterflea (*Daphnia magna*)	67.3	184	153	NA	Practically non-toxic	411583-11	Acceptable
2,4-D Isoproylamine (IPA) (PC Code: 030025)							
Waterflea *(Daphnia magna)*	48.7	583	461	NA	Practically non-toxic	00138869, Alexander et. al., 1983.	Acceptable
2,4-D Triisopropanolamine (TIPA) Salt (PC Code: 030035)							
Waterflea *(Daphnia magna)*	69.2	630	340.2	measured	Practically non-toxic	413538-05, Mayes, 1989	Acceptable
2,4-D Butoxyethyl (BEE) Ester (PC code: 030053)							

Table F-4. Acute freshwater invertebrate toxicity to 2,4-D							
Species	**% ai**	**96-hour LC50 (95% CI)**		**Measured/ Nominal Test Concentrations**	**Toxicity Category (based on ae)**	**MRID No. Author/Year**	**Study Classification**
		(mg ai/L)	**(mg ae/L)**				
Waterflea *(Daphnia magna)*	97.4	7.2	4.97	measured	Moderately toxic	413538-01, Alexander, et. al, 1983.	Acceptable
2,4-D 2-Ethylhexyl Ester (2-EHE) (PC code: 030063)							
Waterflea *(Daphnia magna)*	92	18	11.88	Measured	Slightly toxic	67328, Kuc, W.J., 1977.	Acceptable
Waterflea *(Daphnia magna)*	96.2	5.2	3.4	measured	Moderately toxic	411583-06, Alexander, et. al., 1983.	Acceptable
2,4-D Isopropyl Ester (IPE) (PC code: 030066)							
Waterflea *(Daphnia magna)*	98.2	2.6	2..2	Measured	Moderately toxic	439306-01, Drottar, et.al., 1996.	Acceptable
NA = Not available							

Table F-5. Chronic (life cycle) freshwater invertebrate toxicity to 2,4-D						
Species	**% ai**	**Toxicity Value (mg ae/L)**		**Most sensitive endpoint**	**MRID No. Author/Year**	**Study Classification**
		NOAEC	**LOAEC**			
2,4-D acid (PC Code: 030001)*						
Waterflea *(Daphnia magna)*	91.3	79	151	No of young	418352-11, Ward T.J. et.al., 1991	Acceptable
2,4-D Diethanolamine (DEA) Salt (PC Code: 030016)*						
Waterflea *(Daphnia magna)*	73.8	16.05	25.64	Survival & Reproduction	420183-03, Holmes, et. al., 1991	Acceptable
2,4-D Dimethylamine (DMA)Salt (PC Code: 030019)*						
Waterflea *(Daphnia magna)*	66.8	LC50=75.7	N/A	Survival (NOAEC not established)	418352-10, Ward, S. C., 1991.	Supplemental
2,4-D Butoxyethyl (BEE) Ester (030053)*						
Waterflea *(Daphnia magna)*	96	LC50>0.869 NOAEC = 0.20	0.483	Survival and reproduction	413538-02, Gersich, et. al.,, 1989.	Acceptable

* No full life cycle freshwater invertebrate studies using forms of 2,4-D other than acid, DEA, DMA, and BEE were submitted to the Agency.

Table F-6. Acute freshwater amphibian toxicity to 2,4-D							
Species	% ai	96-hour LC50 (95% CI)		Measured/ Nominal Test Concentrations	Toxicity Category (based on ae)	MRID No. Author/Year	Study Classification
		(mg ai/L)	(mg ae/L)				
2,4-D acid (PC Code: 030001)*							
Leopard frog tadpoles *(Rana pipiens)*	97.5	359	359	Measured	Practically non-toxic	445173-07, Palmer, S.J. et. al., 1997.	Supplemental
2,4-D Dimethylamine (DMA) Salt (PC Code: 030019)*							
Leopard frog tadpoles *(Rana pipiens)*	67.3	337	278	Measured	Practically non-toxic	445173-06, Palmer, S.J. et. al., 1997.	Supplemental

* No acute freshwater amphibian studies using forms of 2,4-D other than acid and DMA were submitted to the Agency.

Table F-7. Non-vascular aquatic plant toxicity to 2,4-D (Tier 1 studies)						
Species	**% ai**	**Tested Concentration**		**Percent Response (inhibition)**	**MRID No. Author/Year**	**Study Classification**
		(mg ai/L)	**(mg ae/L)**			
2,4-D acid (PC Code: 030001)*						
Green algae *Selenastrum capricornutum*	96.1	26.4	26.4	24	414200-01, Hughes, 1990	Acceptable
Blue-green algae *Anabaena flos-aquae*	96.9	>2.02	>2.02	0.488	433079-01, Hughes, 1994	Acceptable
Freshwater diatom *Navicula pelliculosa*	96.9	>2.13	>2.13	24	433079-02, Hughes, 1990	Acceptable
Marine diatom *Skeletonema costatum*	96.9	2.08	2.08	-10	433079-03, Hughes, 1990	Acceptable
2,4-D Isopropyl Ester (IPE) (PC code: 030066)*						
Green algae *Selenastrum capricornutum*	98.2		0.13	-11 %	437680-01, Hughes, et. al., 1995.	Acceptable

* No Tier I non-vascular aquatic plant studies using forms of 2,4-D other than acid and IPA were submitted to the Agency.

Table F-8. Non-vascular aquatic plant toxicity to 2,4-D (Tier II studies)					
Species	**% ai**	**EC50 / NOAEC**		**MRID No. Author/Year**	**Study Classification**
		(mg ai/L)	**(mg ae/L)**		
2,4-D acid (PC Code: 030001)					
No data					
2,4-D Sodium Salt (PC Code: 030004)					
No data					
2,4-D Diethanolamine (DEA) Salt (PC Code: 030016)					
Green Algae *Selnastrum capricornutum*	73.8	11/ 0.50	7.48/ 0.34	427122-05, Thompson et. al., 1993.	Acceptable
Marine diatom *Skeletonema costatum*	73.8	>95/ 95	>64.6/ 64.6	427122-01, Thompson et. al., 1993	Acceptable
Freshwater diatom *Navicula pelliculosa*	73.8	>97/ 97	>66/ 66	427122-02, Thompson et. al., 1993.	Acceptable
Blue-green algae *Anabaena flos-aquae*	73.8	>96/ 96	>65.3/ 65.3	427122-03, Thompson et. al., 1993.	Acceptable
2,4-D Dimethylamine (DMA) Salt (PC Code: 030019)					
Green Algae *Selnastrum capricornutum*	66.7	51.2/ 19.2	42.5/ 16	414200-02, Hughes, J.sS, 1990.	Acceptable
Marine diatom *Skeletonema costatum*	66.7	148.5/ 96.25	123.3/ 79.89	415059-01, Hughes, J.sS, 1990.	Acceptable
Freshwater diatom *Navicula pelliculosa*	66.7	4.67/ 1.70	3.88/ 1.41	415059-03, Hughes, J.S., 1990.	Acceptable
Blue-green algae *Anabaena flos-aquae*	66.7	188.5/ 67.86	156.5/ 56.32	415059-02, Hughes, J.S., 1990.	Acceptable

Table F-8. Non-vascular aquatic plant toxicity to 2,4-D (Tier II studies)					
Species	**% ai**	**EC50 / NOAEC**		**MRID No. Author/Year**	**Study Classification**
		(mg ai/L)	**(mg ae/L)**		
2,4-D Isoproylamine (IPA) (PC Code: 030025)					
Green Algae *Selenastrum capricornutum*	51.3	43.4/ 13.9	34.29/ 10.98	417321-02, Hughes, J.S., 1990.	Acceptable
2,4-D Triisopropanolamine (TIPA) Salt (PC Code: 030035)					
Green Algae *Selenastrum capricornutum*	73.8	75.7 55.4	40.88/ 29.92	417321-01, Hughes, J.S., 1990.	Acceptable
Marine diatom *Skeletonema costatum*	70.9	79.7 50.4	38.29/	434886-03, Hughes, et. al., 1994	Acceptable
Freshwater diatom *Navicula pelliculosa*	70.9	94.4 5.35	50.98/ 2.89	434886-01, Hughes, et. al., 1994.	Acceptable
Blue-green algae *Anabaena flos-aquae*	70.9	133 47.9	71.82/ 25.87	434886-04, Hughes, et. al., 1994.	Acceptable
2,4-D Butoxyethyl (BEE) Ester (PC code: 030053)					
Green Algae *Selenastrum capricornutum*	96	24.9/ 12.5	17.14/ 8.6	431882-01, Hughes, J.S., 1990.	Acceptable
Marine diatom *Skeletonema costatum*	96	1.48/ 0.78	1.02/ 0.538	42-684-04, Hughes,J.S., 1990.	Acceptable
Freshwater diatom *Navicula pelliculosa*	96	1.86/ 0.86	1.28/ 0.59	420684-03, Hughes, J.S., 1990	Acceptable
Blue-green algae *Anabaena flos-aquae*	96	6.37/ 3.14	4.4/ 2.2	420684-03, Hughes, J.S., 1990.	Acceptable
2,4-D 2-Ethylhexyl Ester (2-EHE) (PC code: 030063)					

Table F-8. Non-vascular aquatic plant toxicity to 2,4-D (Tier II studies)					
Species	**% ai**	**EC50 / NOAEC**		**MRID No. Author/Year**	**Study Classification**
		(mg ai/L)	**(mg ae/L)**		
Green algae *Selenastrum capricornutum*	94.7 (62.8 a. eq.)	>30 / 3.75	19.8/ 2.48	417352-06, Hughes, J.S., 1990.	Acceptable
Marine diatom *Skeletonema costatum*	94.7 (62.8 a. eq.)	0.10 / 0.0938	0.066/ 0.062	417352-04, Hughes, J.S., 1990.	Acceptable
Freshwater diatom *Navicula pelliculosa*	94.7 (62.8 a. eq.)	1.9 / 1.875	1.25/ 1.24	417352-05, Hughes, J.S., 1990.	Acceptable
Blue-green algae *Anabaena flos-aquae*	94.7 (62.8 a. eq.)	>0.32 / 0.32	>0.21/ 0.21	417352-02, Hughes, J.S., 1990.	Acceptable
2,4-D Isopropyl Ester (IPE) (PC code: 030066)					
No data					

Table F-9. Vascular aquatic plant toxicity to 2,4-D (Tier II studies)					
Species	**% ai**	**EC50 / NOAEC**		**MRID No. Author/Year**	**Study Classification**
		(mg ai/L)	**(mg ae/L)**		
2,4-D acid (PC Code: 030001)					
Duckweed ,*Lemna gibba*	96.2	0.695 / 0.0581	0.695 / 0.0581	442951-01, Hughes et al, 1997	Acceptable
2,4-D Sodium Salt (PC Code: 030004)					
No data					
2,4-D Diethanolamine (DEA) Salt (PC Code: 030016)					
Duckweed ,*Lemna gibba*	73.8	0.44/ 0.07	0.2992/ 0.0476	427122-04, Thompson et. al., 1993.	Acceptable
2,4-D Dimethylamine (DMA) Salt (PC Code: 030019)					
Duckweed ,*Lemna gibba*	66.7	0.58/ 0.27	0.48/ 0.23	415059-04, Hughes, J.S., 1990.	Acceptable
2,4-D Isoproylamine (IPA) (PC Code: 030025)					
No data					
2,4-D Triisopropanolamine (TIPA) Salt (PC Code: 030035)					
Duckweed ,*Lemna gibba*	70.9	2.37/ 2.38	1.28/ 1.28	434886-02, Hughes, et. al., 1994.	Acceptable
2,4-D Butoxyethyl (BEE) Ester (PC code: 030053)					
Duckweed,*Lemna gibba*	96	0.576/ 0.204	0.3974/ 0.141	4206884-02, Hughes, J.S., 1990.	Acceptable
2,4-D 2-Ethylhexyl Ester (2-EHE) (PC code: 030063)					
Duckweed ,*Lemna gibba*	94.7	0.50 / <0.0938	0.33/ 0.062	417352-03, Hughes, J.S., 1990.	Acceptable
2,4-D Isopropyl Ester (IPE) (PC code: 030066)					
No data					

Table F-10. Acute bird toxicity (gavage) to 2,4-D						
Species	**% ai**	**LD50**		**Toxicity Category (based on ae)**	**MRID No. Author/Year**	**Study Classification**
		(mg ai/kg-bwt)	**(mg ae/kg-bwt)**			
2,4-D acid (PC Code: 030001)						
Mallard duck *(Anas platyrhynchos)*	96.1	>5620	>5620	Practically non-toxic	415462-02, Culotta et.al., 1990	Acceptable
2,4-D Sodium Salt (PC Code: 030004)						
No data						
2,4-D Diethanolamine (DEA) Salt (PC Code: 030016)						
Northern bobwhite quail *(Colinus virginianus)*	73.1	595	404.6	Moderately toxic	419751-01, Cambell, et. al, 1991.	Acceptable
2,4-D Dimethylamine (DMA) Salt (PC Code: 030019)						
Northern bobwhite quail *(Colinus virginianus)*	66.8	500	415	Moderately toxic	415462-01, Hoxter et. all., 1990.	Acceptable
Mallard duck *(Anas platyrhynchos)*	100	>4640	>3851.2	Practically non-toxic	233351, Fink, R., 1978	Acceptable
2,4-D Isoproylamine (IPA) (PC Code: 030025)						
Mallard duck *(Anas platyrhynchos)*	48.7	>398	>314.4	Moderately toxic	00138871, Beavers, et. al., 1983,	Supplemental
Northern bobwhite quail (*Colinus virginianus*)	49%	377	298	Moderately toxic	442757-01 Beavers 1985	Acceptable
2,4-D Triisopropanolamine (TIPA) Salt (PC Code: 030035)						
Northern bobwhite quail *(Colinus virginianus)*	70.4	>405	>218.7	Moderately toxic	416444-01, Hoxter, K.A., 1990.	Acceptable
2,4-D Butoxyethyl (BEE) Ester (PC code: 030053)						

Table F-10. Acute bird toxicity (gavage) to 2,4-D						
Species	**% ai**	**LD50**		**Toxicity Category (based on ae)**	**MRID No. Author/Year**	**Study Classification**
		(mg ai/kg-bwt)	**(mg ae/kg-bwt)**			
Northern bobwhite quail *(Colinus virginianus)*	96	>2000	>1380	Slightly toxic	414541-01, Lloyd, D., 1989.	Acceptable
2,4-D 2-Ethylhexyl Ester (2-EHE) (PC code: 030063)						
Northern bobwhite quail *(Colinus virginianus)*	96.2	633	417.78	Moderately toxic	411583-03, Beavers, J.B., 1984.	Acceptable
Mallard duck *(Anas platyrhynchos)*	92	.>3000	>1980	Slightly toxic	72472, Fink, R., 1976.	Acceptable
Mallard duck *(Anas platyrhynchos)*	92	.>4640	>3062	Practically non-toxic	226397, Fink, R., 1976.	Acceptable
2,4-D Isopropyl Ester (IPE) (PC code: 030066)						
Northern bobwhite quail *(Colinus virginianus)*	98.2	1879	1578	Slightly toxic	439350-01, Palmer, et. al., 1996.	Acceptable

Table F-11. Acute bird toxicity (dietary) to 2,4-D						
Species	**% ai**	**LD50**		**Toxicity Category (based on ae)**	**MRID No. Author/Year**	**Study Classification**
		(mg ai/kg-diet)	**(mg ae/kg-diet)**			
2,4-D acid (PC Code: 030001)						
Northern bobwhite quail *(Colinus virginianus)*	96.1	>5620	>5620	Practically nontoxic	415861-01, Culotta J., 1989.	Acceptable
Mallard duck *(Anas platyrhynchos)*	96.1	>5620	>5620	Practically non-toxic	415462-02, Culotta et.al., 1990	Acceptable
2,4-D Sodium Salt (PC Code: 030004)						
No data						
2,4-D Diethanolamine (DEA) Salt (PC Code: 030016)						
Northern bobwhite quail *(Colinus virginianus)*	73.1	>5620	>3821.6	Slightly toxic	419751-02, Hoxter, et. al., 1991.	Acceptable
Mallard duck *(Anas platyrhynchos)*	73.1	>5620	>3820.6	Slightly toxic	419751-03, Hoxter, et. al., 1991.	Acceptable
2,4-D Dimethylamine (DMA) Salt (PC Code: 030019)						
Northern bobwhite quail *(Colinus virginianus)*	66.8	>5620	>4665	Slightly toxic	417495-01, Long, et. al., 1990.	Acceptable
Northern bobwhite quail *(Colinus virginianus)*	100	>10,000	>8300	Practically non-toxic	233351, Fink, R., 1978.	Acceptable
Mallard duck *(Anas platyrhynchos)*	66.8	>5620	>4665	Slightly toxic	417495-02, Long, et. al., 1990.	Acceptable
2,4-D Isoproylamine (IPA) (PC Code: 030025)						
Northern bobwhite quail *(Colinus virginianus)*	48.7	>5620	>4440	Slightly toxic	00138870, Beavers, J.B., 1983.	Acceptable

Table F-11. Acute bird toxicity (dietary) to 2,4-D						
Species	**% ai**	**LD50**		**Toxicity Category (based on ae)**	**MRID No. Author/Year**	**Study Classification**
		(mg ai/kg-diet)	**(mg ae/kg-diet)**			
Mallard duck *(Anas platyrhynchos)*	48.7	>5620	>4440	Slightly toxic	00138872, Beavers, J.B., 1983.	Acceptable
2,4-D Triisopropanolamine (TIPA) Salt (PC Code: 030035)						
Northern bobwhite quail *(Colinus virginianus)*	70.4	>5620	>3035	Slightly toxic	416444-02, Driscoll, et., al. 1990.	Acceptable
Mallard duck *(Anas platyrhynchos)*	70.4	>5620	>3035	Slightly toxic	416444-03, Driscoll, et., al. 1990.	Acceptable
2,4-D Butoxyethyl (BEE) Ester (PC code: 030053)						
Northern bobwhite quail *(Colinus virginianus)*	96	>5620	>3878	Slightly toxic	414484-01, Grimes, J.., 1989.	Acceptable
Mallard duck *(Anas platyrhynchos)*	96	>5620	>3866	Slightly toxic	414290-07, Grimes, J., 1989.	Acceptable
2,4-D 2-Ethylhexyl Ester (2-EHE) (PC code: 030063)						
Northern bobwhite quail *(Colinus virginianus)*	96	>5620	>3878	Slightly toxic	414484-01, Grimes, J.., 1989.	Acceptable
Mallard duck *(Anas platyrhynchos)*	96	>5620	>3866	Slightly toxic	414290-07, Grimes, J., 1989.	Acceptable
2,4-D Isopropyl Ester (IPE) (PC code: 030066)						
Northern bobwhite quail *(Colinus virginianus)*	98.2	>5456	>4583	Slightly toxic	439349-01, Palmer, et. al., 1996.	Acceptable
Mallard duck *(Anas platyrhynchos)*	98.2	>5218	>4383	Slightly toxic	439352-01, Palmer, et. al., 1996.	Acceptable

Table F-12. Chronic bird toxicity (reproductive) to 2,4-D						
Species	**% ai**	**NOAEC / LOAEC**		**Most sensitive endpoints**	**MRID No. Author/Year**	**Study Classification**
		(mg ai/kg-diet)	**(mg ae/kg-diet)**			
2,4-D acid (PC Code: 030001) *						
Northern bobwhite quail *(Colinus virginianus)*	96.9	962/>962	962/>962	No effects	415861-01, Culotta J., 1989.	Acceptable

* No avian reproduction studies using forms of 2,4-D other than the acid were submitted to the Agency.

Table F-13. Acute mammal toxicity (gavage) to 2,4-D						
Species	**% ai**	**LD_{50} (mg ae/kg-bwt)[1]**	**Toxicity Category (based on ae)**	**MRID No.**	**Study Classification[2]**	**Comments**
2,4-D acid (PC Code: 030001)						
Laboratory rat *(Rattus norvegicus)*	NA	699	Slightly toxic	00101605	Acceptable	
2,4-D Sodium Salt (PC Code: 030004)						
No data						
2,4-D Diethanolamine (DEA) Salt (PC Code: 030016)						
Laboratory rat *(Rattus norvegicus)*	100% (assumed by study author)	619	Slightly toxic	41920901	Acceptable	Study author assumed test material was 100% DEA salt, dose listed in terms of test material administered.
2,4-D Dimethylamine (DMA) Salt (PC Code: 030019)						
Laboratory rat *(Rattus norvegicus)*	57.9% a.e.	>579	Slightly toxic	00157512	Acceptable	Chemical analysis confirmed 57.9% a.e., administered dose listed in terms of test material (an end-use product), total mortality at highest dose was 4/12 (no other mortality)
2,4-D Isoproylamine (IPA) Salt (PC Code: 030025)						
Laboratory rat *(Rattus norvegicus)*	39.4% a.e.	747	Slightly toxic	00252291	Acceptable	Administered dose listed in terms of test material (technical), technical label indicated 39.4% a.e.
2,4-D Triisopropanolamine (TIPA) Salt (PC Code: 030035)						

Table F-13. Acute mammal toxicity (gavage) to 2,4-D						
Species	**% ai**	**LD_{50} (mg ae/kg-bwt)[1]**	**Toxicity Category (based on ae)**	**MRID No.**	**Study Classification[2]**	**Comments**
Laboratory rat *(Rattus norvegicus)*	37.7% a.e.	441	Slightly toxic	41413501	Acceptable	Administered dose listed in terms of test material. Study author stated test material was 37.7% ae.
2,4-D Butoxyethyl (BEE) Ester (PC code: 030053)						
Laboratory rat *(Rattus norvegicus)*	NA	573	Slightly toxic	40629801	Acceptable	Administered dose listed in terms of test material, chemical analysis confirmed 66.1% ae
2,4-D 2-Ethylhexyl Ester (EHE) (PC code: 030063)						
Laboratory rat *(Rattus norvegicus)*	NA	NA	NA	41209001	Acceptable	Study conducted using end-use product EPA Reg. 34704-607. Label stated product contained 33.18% 2,4-D EHE and 32.52% 2,4-DP EHE (PCcode 31464). Because this product is a mixture, it will not be considered.
2,4-D Isopropyl Ester (IPE) (PC code: 030066)						
Laboratory rat *(Rattus norvegicus)*	37% a.e.	458	Slightly toxic	41709901	Acceptable	Administered dose listed in terms of test material, material was an end-use product, EPA Reg. 400-444. Label stated material 37% ae.

Table F-13. Acute mammal toxicity (gavage) to 2,4-D						
Species	**% ai**	**LD_{50} (mg ae/kg-bwt)[1]**	**Toxicity Category (based on ae)**	**MRID No.**	**Study Classification[2]**	**Comments**

[1] All LD_{50}s were calculated by EFED using data provided in the original study reports. Clarifications regarding the calculations are made in the "Comments" column.

[2] Classifications determined by HED.

Table F-14. Chronic mammalian toxicity (reproductive) to 2,4-D							
Species	% ai	Endpoint (mg ae/kg-bwt/day)			Affected parameters (most sensitive)	MRID No.	Study Classification
			NOAEL	LOAEL			
2,4-D acid (PC Code: 030001) *							
Laboratory rat *(Rattus norvegicus)*		Parental	Target =5 Actual (3.5-13.5)	Target =20 Actual (14-48)	Decreased female body wt gain(F1) and male renal tubule alteration (F0 and F1)	00150557; 00163996	Acceptable, determined by HED
		Reproductive	Target =20 Actual (18-35)	Target =80 Actual (69-114)	increase in gestation length		
		Offspring	Target =5 Actual (7.2-13.5)	Target =20 Actual (26-48)	decreased pup body weight; increase in pup deaths at 80 mg ae/kg-bwt/day		

* No mammalian reproduction studies using forms of 2,4-D other than the acid were submitted to the Agency.

Table F-15. Acute honey bee toxicity (contact) to 2,4-D						
Species	% ai	96-hour LD50 (95% CI)		Toxicity Category (based on ae)	MRID No. Author/Year	Study Classification
		(µg ai/bee)	(µg ae/bee)			
2,4-D Dimethylamine (DMA) Salt (PC Code: 030019) *						
Honey bee *(Apis mellifera)*	67.3	>100	>83	Practically non-toxic	445173-04, Palmer S. et al., 1997	Acceptable
2,4-D 2-Ethylhexyl Ester (2-EHE) (PC code: 030063) *						
Honey bee *(Apis mellifera)*	96.96	>100	>66	Practically non-toxic	445173-01, Palmer S. et al., 1997	Acceptable

* No contact honey bee studies using forms of 2,4-D other than DMAS and EHE were submitted to the Agency.

Table F-16. Terrestrial plant toxicity (seedling emergence, most sensitive monocot) to 2,4-D - all available forms, technical only					
Species	% ai	EC25 (lbs ae/acre)	Most sensitive endpoint	MRID No. Author/Year	Study Classification
2,4-D acid (PC Code: 030001)					
Onion and sorghum	96.7	2.1	Fresh weight	424168-02, Backus, 1992	Acceptable
2,4-D Sodium Salt (PC Code: 030004)					
No data					
2,4-D Diethanolamine (DEA) Salt (PC Code: 030016)					
Onion	50.2	0.38	Fresh weight	426091-01, Backus, 1992	Acceptable
2,4-D Dimethylamine (DMA) Salt (PC Code: 030019)					
Sorghum	55.5	0.026	Fresh weight	423895-01, Backus, 1992	Acceptable
2,4-D Isoproylamine (IPA) Salt (PC Code: 030025)					
No data					
2,4-D Triisopropanolamine (TIPA) Salt (PC Code: 030035)					
No data					
2,4-D Butoxyethyl (BEE) Ester (PC code: 030053)					
Onion	65.6	0.36	Survival	431970-01, Narnish, 1994	Supplemental
2,4-D 2-Ethylhexyl Ester (2-EHE) (PC code: 030063)					
Onion	63.5	0.218	Fresh shoot weight	435269-01, Backus, 1995	Acceptable
2,4-D Isopropyl Ester (IPE) (PC code: 030066)					
Onion	98.2	0.010	Shoot length	439821-01, Hoberg, 1996	Acceptable

Table F-17. Terrestrial plant toxicity (seedling emergence, most sensitive dicots) to 2,4-D - all available forms, technical only					
Species	% ai	EC25 (lbs ae/acre)	Most sensitive endpoint	MRID No. Author/Year	Study Classification
2,4-D acid (PC Code: 030001)					
Mustard	96.7	0.033	Fresh weight	424168-02, Backus, 1992	Acceptable
2,4-D Sodium Salt (PC Code: 030004)					
No data					
2,4-D Diethanolamine (DEA) Salt (PC Code: 030016)					
Mustard	50.2	0.045	Fresh weight	426091-01, Backus, 1992	Acceptable
2,4-D Dimethylamine (DMA) Salt (PC Code: 030019)					
Mustard	55.5	0.00953	Fresh weight	423895-01, Backus, 1992	Acceptable
2,4-D Isoproylamine (IPA) Salt (PC Code: 030025)					
No data					
2,4-D Triisopropanolamine (TIPA) Salt (PC Code: 030035)					
No data					
2,4-D Butoxyethyl (BEE) Ester (PC code: 030053)					
Tomato	65.6	0.05	Dry weight	431970-01, Narnish, 1994	Supplemental
2,4-D 2-Ethylhexyl Ester (2-EHE) (PC code: 030063)					
Radish	63.5	0.037	Fresh shoot weight	424492-01, Backus, 1992	Acceptable
2,4-D Isopropyl Ester (IPE) (PC code: 030066)					
Lettuce	98.2	0.00081	Shoot length	439821-01, Hoberg, 1996	Supplemental

Table F-18. Terrestrial plant toxicity (vegetative vigor, most sensitive monocot) to 2,4-D - all available forms, technical only					
Species	% ai	EC25 (lbs ae/acre)	Most sensitive endpoint	MRID No. Author/Year	Study Classification
2,4-D acid (PC Code: 030001)					
Onion	96.7	<0.0075	Fresh weight	424168-01, Backus, 1991	Acceptable
2,4-D Sodium Salt (PC Code: 030004)					
No data					
2,4-D Diethanolamine (DEA) Salt (PC Code: 030016)					
Onion	50.2	0.04	Fresh weight	426091-02 Backus 1992	Acceptable
2,4-D Dimethylamine (DMA) Salt (PC Code: 030019)					
No data					
2,4-D Isoproylamine (IPA) Salt (PC Code: 030025)					
No data					
2,4-D Triisopropanolamine (TIPA) Salt (PC Code: 030035)					
No data					
2,4-D Butoxyethyl (BEE) Ester (PC code: 030053)					
Onion	65.6	0.19	Dry weight	430671-03, Narnish, 1993	Supplemental
2,4-D 2-Ethylhexyl Ester (2-EHE) (PC code: 030063)					
Sorghum	63.5	0.218	Fresh weight	423439-02, Backus, 1992	Acceptable
2,4-D Isopropyl Ester (IPE) (PC code: 030066)					
Corn	82.7	0.2016	Shoot weight	437882-01, Hoberg, 1995	Acceptable

Table F-19. Terrestrial plant toxicity (vegetative vigor, most sensitive dicots) to 2,4-D - all available forms, technical only					
Species	% ai	EC25 (lbs ae/acre)	Most sensitive endpoint	MRID No. Author/Year	Study Classification
2,4-D acid (PC Code: 030001)					
Tomato	96.7	0.0075	Fresh weight	424168-01, Backus, 1991	Acceptable
2,4-D Sodium Salt (PC Code: 030004)					
No data					
2,4-D Diethanolamine (DEA) Salt (PC Code: 030016)					
Tomato	50.2	0.003	Fresh weight	426091-02 Backus, 1992	Acceptable
2,4-D Dimethylamine (DMA) Salt (PC Code: 030019)					
No data					
2,4-D Isoproylamine (IPA) Salt (PC Code: 030025)					
No data					
2,4-D Triisopropanolamine (TIPA) Salt (PC Code: 030035)					
No data					
2,4-D Butoxyethyl (BEE) Ester (PC code: 030053)					
Radish	65.6	0.02	survival	430671-03, Narnish, 1993	Supplemental
2,4-D 2-Ethylhexyl Ester (2-EHE) (PC code: 030063)					
Soybean	63.5	0.02	Fresh weight	423439-02, Backus, 1992	Acceptable
2,4-D Isopropyl Ester (IPE) (PC code: 030066)					
Radish	82.7	0.0042	Root weight	437882-01, Hoberg, 1995	Acceptable

Table F-20. Summary of seedling emergence study conducted with Gordon's Amine 4000 2,4-D Weed Killer, a TEP of 2,4-D DMAS[1]					
Species	**Most sensitive Endpoint**	**Value (lbs ae/acre)**			
		NOAEC	**EC_{05}**	**EC_{25}**	**EC_{50}**
Corn	Plant Height	4.0	2.4	>4.0	>4.0
Onion	Dry Weight	0.091	<0.0014	0.097	>0.35
Ryegrass	Plant Height	4.0	2.2	>4.0	>4.0
Wheat	Dry Weight	0.35	0.0054	0.20	>0.35
Cabbage	Dry Weight	0.020	0.010	0.043	0.12
Lettuce	Dry Weight	0.020	0.0054	0.026	0.078
Radish	Dry Weight	0.020	0.0087	0.033	0.082
Soybean	Dry Weight	0.26	<0.26	0.37	1.2
Tomato	None	0.34	N.D.	>0.34	>0.34
Turnip	Dry Weight	0.020	0.035	0.13	0.34

[1] MRID 471060-01, Porch et al. 2006, 39.21%ai, Acceptable

Table F-21. Summary of vegetative vigor study conducted with Gordon's Amine 4000 2,4-D Weed Killer, a TEP of 2,4-D DMAS[1]					
Species	**Most sensitive Endpoint**	**Value (lbs ae/acre)**			
		NOAEC	**EC_{05}**	**EC_{25}**	**EC_{50}**
Onion	Dry Weight	0.0335	0.034	0.14	0.37
Ryegrass	Dry Weight	2.07	1.0	>2.07	>2.07
Wheat	Dry Weight	0.133	0.32	1.1	>2.03
Cabbage	Dry Weight	0.0981	0.079	0.18	0.31
Lettuce	Dry Weight	0.0017	<0.0017	0.0038	0.015
Radish	Dry Weight	0.0016	<0.0016	0.0012	>0.0266
Soybean	Dry Weight	0.0072	0.0093	0.039	>0.0998
Tomato	Dry Weight	0.0016	0.0018	0.0074	0.020
Turnip	Dry Weight	0.0015	<0.0015	0.011	0.065

[1] MRID 471060-02, Porch et al. 2006, 39.21%ai, Acceptable

Table F-22. Summary of seedling emergence study conducted with Gordon's LV400 2,4-D Weed Killer, a TEP of 2,4-D EHE[1]					
Species	**Most sensitive Endpoint**	**Value (lbs ae/acre)**			
		NOAEC	**EC_{05}**	**EC_{25}**	**EC_{50}**
Corn	Dry Weight	1.9	1.7	3.4	>3.8
Onion	Plant height	0.019	0.047	0.17	>0.34
Ryegrass	Dry Weight	0.26	<0.26	0.27	2.8
Wheat	None	4.0	<0.26	>4.0	>4.0
Cabbage	Dry Weight	0.0015	0.0056	0.021	0.053
Lettuce	Dry Weight	0.0058	0.0061	0.018	0.039
Radish	Dry Weight	0.0058	0.0061	0.036	0.12
Soybean	Dry Weight	0.47	<0.25	0.85	2.3
Tomato	Dry Weight	0.0058	<0.0015	0.012	0.29
Turnip	Dry Weight	0.0058	0.021	0.062	0.13
[1] MRID 471060-03, Porch et al. 2006, 44.9%ai, Acceptable					

Table F-23. Summary of vegetative vigor study conducted with Gordon's LV400 2,4-D Weed Killer, a TEP of 2,4-D EHE[1]					
Species	**Most sensitive Endpoint**	**Value (lbs ae/acre)**			
		NOAEC	**EC_{05}**	**EC_{25}**	**EC_{50}**
Corn	Dry Weight	0.0289	<0.00803	0.17	1.6
Onion	Dry Weight	0.0254	0.025	0.088	0.21
Ryegrass	None	2.03	N.D.	>2.03	>2.03
Wheat	Dry Weight	0.0356	0.043	0.34	1.4
Cabbage	Dry Weight	0.00167	0.012	0.027	0.047
Lettuce	Dry Weight	0.00167	<0.00167	0.0021	0.0076
Radish	Survival	0.00527	0.0026	0.0068	0.013
Soybean	Dry Weight	0.0259	0.014	0.058	>0.0992
Tomato	Dry Weight	<0.00134	<0.00134	0.0044	0.016
Turnip	Dry Weight	<0.00134	<0.00134	0.0021	0.042
[1] MRID 471060-04, Porch et al. 2006, 44.9%ai, Acceptable					

Appendix A. Ecological Effects Data

Comparison of Toxicity of Organisms to different forms of Triclopyr in terms of the acid equivalent (TEA, BEE, & degradate TCP)

Taxa	Effect Type	Endpoint	Triclopyr Acid - TGAI	TEA (ae)	BEE (ae)	TCP (ae)
Freshwater fish Rainbow Trout (*Oncorhynchus mykiss*)	Acute	96h LC50	117 mg/L	79.2 mg/L	0.47 mg/L	1.9 mg/L
Freshwater fish Bluegill sunfish (*Lepomis macrochirus*)	Acute	96h LC50	148 mg/L	155.4 mg/L	0.26 mg/L	16.1 mg/L
Freshwater fish Fathead Minnow (*Pimephales promelas*)	Chronic	NOEC LOEC	No Data	> 32.2 mg/L < 50.2 mg/L	No Data	No Data
Freshwater fish Rainbow Trout (*Oncorhynchus mykiss*)	Chronic	NOEC LOEC	No Data	No Data	0.019 mg/L 0.034 mg/L	No Data
Freshwater invertebrate Water flea (*Daphnia magna*)	Acute	48h EC50	132.9 mg/L	346 mg/L	0.25 mg/L	13.4 mg/L
Freshwater invertebrate Water flea (*Daphnia magna*)	Chronic	NOEC LOEC	No Data	25 mg/L 46.2mg/L	No Data	No Data
Vascular aquatic plant Duckweed (*Lemna gibba*)	Acute	14d EC50	No Data	6.1 mg/L	0.86 mg/L	Invalid study
Non-vascular aquatic plant Green algae (*Kirchneria subcapitata*) (Formerly *Selenastrum capricortum*)	Acute	5d EC50	29.8 mg/L	12.1 mg/L	2.5 mg/L	2.3 mg/L
Non-vascular aquatic plant Blue-green algae (*Anabeana flos-aquae*)	Acute	5d EC50	No Data	(7d EC50) 4.1 mg/L	1.42 mg/L	2.3 mg/L
Non-vascular aquatic plant Freshwater diatom (*Navicula pelliculosa*)	Acute	5d EC50	No Data	(4d EC50) 10.6 mg/L	0.07 mg/L	No Data
Avian Mallard duck (*Anas platyrhynchos*)	Acute oral	LD50	1698 mg/kg bw	(14d LC50) 1418 mg/kg bw	No Data	No Data

Taxa	Effect Type	Endpoint	Triclopyr Acid	TEA (ae)	BEE (ae)	TCP (ae)
Avian Northern bobwhite Quail (*Colinus virginianus*)	Acute oral	21d LD50	No Data	No Data	529 mg/kg bw	(8d LD50) > 2585 mg/kg bw
Avian Northern bobwhite Quail (*Colinus virginianus*)	Subacute dietary	8d LC50	2934 ppm	5,189 ppm	3385 ppm	No Data
Avian Mallard duck (*Anas platyrhynchos*)	Subacute dietary	8d LC50	No Data	> 4,464.8 ppm	> 3885	> 7265 ppm
Avian Mallard duck (*Anas platyrhynchos*)	Chronic	NOAEC LOAEC	100 ppm 200 ppm # of 14 d old survivors	No Data	No Data	No Data
Rodent Rat	Acute oral	LD50	630 (F) mg/kg	572 (M & F) mg/kg	578 (M & F) mg/kg	1026 (M) mg/kg
Rodent Rat	Chronic	NOAEL LOAEL	5 mg/kg bw 25 mg/kg bw	No Data	No Data	No Data
Honeybee (*Apis mellifera*)	Acute	Contact 48h LD50	>100 µg/bee	No Data	> 72 µg/bee	No Data
Terrestrial dicot Sunflower (*Helianthus annus*) (Vegetative Vigor)	Acute	14d EC25	No Data	0.005 lbs ae/A Parameter: shoot length	0.006 lbs ae/ A Parameter: shoot weight	No Data
Terrestrial monocot Onion (*Allium cepa*) (Vegetative Vigor)	Acute	14d EC25	No Data	0.114 lbs ae/A Parameter: shoot weight	0.063 lbs ae/A Parameter: shoot weight	No Data
Terrestrial dicot Alfalfa (*Medicago sativa*) (Seedling Emergence)	Acute	51d EC25	No Data	No Data	0.045 lbs ae/A parameter: emergence	No Data
Terrestrial dicot Soybean (*Glycine max*) (Seedling Emergence)	Acute	14d EC25	No Data	> 0.23 lbs ae/A Parameter: shoot length	< 8.0 lbs ae/A Parameter: emergence	No Data
Terrestrial monocot Onion (*Allium cepa*) (Seedling Emergence)	Acute	51d EC25	No Data	> 0.69 lbs as/A Parameter: shoot length	0.053 lbs ae/A Parameter: shoot weight	No Data

Terrestrial monocot Barley (*Hordeum vulgare*) (Seedling Emergence)	Acute	14d EC25	No Data	> 0.23 lbs ae/A Parameter: shoot length	< 8.0 lbs ae/A Parameter: emergence	No Data

Triclopyr Acid (colored cells are most sensitive for triclopyr acid)

Habitat	Taxa	Effect Type	Endpoint	Value	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author))
Aquatic	Freshwater fish Rainbow Trout (*Oncorhynchus mykiss*)	Acute	96h LC50	117	mg/L	Practically Non-Toxic	Acceptable	00049637 Dow Chemical 1973
Aquatic	Freshwater fish Bluegill sunfish (*Lepomis macrochirus*)	Acute	96h LC50	148	mg/L	Practically Non-Toxic	Acceptable	00049637 Dow Chemical 1973
Aquatic	Freshwater invertebrate Water flea (*Daphnia magna*)	Acute	48h EC50	132.9	mg/L	Practically Non-Toxic	Acceptable	40346504 McCarty 1977
Aquatic plants	Non-vascular aquatic plant Green algae (*Kirchneria subcapitata*) (Formerly *Selenastrum capricortum*)	Acute	5d EC50	29.8	mg/L	Parameter?	Supplemental	41736303 Cowgill 1989
Terrestrial	Avian Mallard duck (*Anas platyrhynchos*)	Acute oral	LD50	1698	mg/kg bw	Slightly Toxic	Acceptable	40346401 Dow Chemical 1976
Terrestrial	Avian Cortunix Quail	Subacute dietary	LC50	3272	ppm	Slightly Toxic	Supplemental	00049638 Dow Chemical 1973
Terrestrial	Avian Northern bobwhite Quail (*Colinus virginianus*)	Subacute dietary	LC50	2934	ppm	Slightly Toxic	Acceptable	40346403 Dow Chemical 1976
Terrestrial	Avian Mallard duck (*Anas platyrhynchos*)	Subacute dietary	LC50	5620	ppm	Practically Non-Toxic	Acceptable	0031249 Wildlife Int'l 1979
Terrestrial	Avian Northern bobwhite Quail (*Colinus virginianus*)	Chronic	NOAEC LOAEC LOAEC	500 >500 200	ppm ppm ppm	N/A	Acceptable	00031251 Beavers 1979
Terrestrial	Avian Mallard duck (*Anas platyrhynchos*)	Chronic	NOAEC LOAEC	100 200	ppm	# of 14 d old survivors	Acceptable	00031250 Beavers 1979

Habitat	Taxa	Effect Type	Endpoint	Value	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Terrestrial	Rodent Rat	Acute oral	LD50	729 (M) 630 (F) (F 95% CI 450-829)	mg/kg mg/kg mg/kg	Slightly Toxic	Acceptable (HED 2002)	00031940 Henck et al. 1979
Terrestrial	Rodent	Chronic	NOAEL LOAEL	25 250	mg/kg bw mg/kg bw	Reproductive/ Systemic	Acceptable (RED 1998)	43545701 Vedula et al. 1995
Terrestrial	Rodent	Chronic	NOAEL LOAEL	5 25	mg/kg bw mg/kg bw	Reproductive/ Offspring	Acceptable (HED 2002)	43545701 Vedula et al. 1995
Terrestrial	Honeybee (*Apis mellifera*)	Acute	Contact 48h LD50	>100	µg/bee	Practically Non-Toxic	Acceptable	40356602 Dingledine 1985

TCP (Degradate Toxicity Information)

Habitat	Taxa	Effect Type	Endpoint	Value (expressed as acid equivalent)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Aquatic	Freshwater fish Bluegill sunfish (*Lepomis macrochirus*)	Acute	96h LC50	16.1	mg/L	Slightly Toxic	Acceptable	41829003 Gorzinski et al. 1991
Aquatic	Freshwater fish Rainbow Trout (*Oncorhynchus mykiss*)	Acute	96h LC50	16.3	mg/L	Slightly Toxic	Acceptable	41829004 Gorzinski et al. 1991
Aquatic	Freshwater fish Rainbow Trout (*Oncorhynchus mykiss*)	Acute	96h LC50	1.9	mg/L	Moderately Toxic	Acceptable	44585404 Wan et al. 1987
Aquatic	Freshwater fish Coho salmon	Acute	96h LC50	2.3	mg/L	Moderately Toxic	Supplemental	44585404 Wan et al. 1987
Aquatic	Freshwater fish Chum salmon	Acute	96h LC50	2.3	mg/L	Moderately Toxic	Supplemental	44585404 Wan et al. 1987
Aquatic	Freshwater fish Sockeye salmon	Acute	96h LC50	3.2	mg/L	Moderately Toxic	Supplemental	44585404 Wan et al. 1987
Aquatic	Freshwater fish Chinook salmon	Acute	96h LC50	2.7	mg/L	Moderately Toxic	Supplemental	44585404 Wan et al. 1987
Aquatic	Freshwater fish Pink salmon	Acute	96h LC50	3.5	mg/L	Moderately Toxic	Supplemental	44585404 Wan et al. 1987
Aquatic	Freshwater invertebrate Water flea (*Daphnia magna*)	Acute	48h EC50	13.4	mg/L	Slightly Toxic	Acceptable	41829003 Gorzinski et al. 1991
Aquatic plants	Vascular aquatic plant Duckweed (*Lemna gibba*)	Acute	----	----	----	Limited critical endpoints measured	**Invalid**	45312002 Kirk et al. 2000
Aquatic plants	Non-vascular aquatic plant Green algae (*Kirchneria subcapitata*) (Formerly *Selenastrum capricortum*)	Acute	4d EC50	2.3	mg/L	Parameter: Growth (Yield)	Supplemental	45312001 Kirk et al. 1999

Habitat	Taxa	Effect Type	Endpoint	Value (expressed as acid equivalent)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Aquatic plants	Non-vascular aquatic plant Green algae (*Kirchneria subcapitata*) (Formerly *Selenastrum capricortum*)	Acute	4d NOAEC (as EC05)	0.84	mg/L	Parameter: Growth (Yield)	Supplemental	45312001 Kirk et al. 1999
Aquatic plants	Non-vascular aquatic plant Blue-green algae (*Anabeana flos-aquae*)	Acute	5d EC50	2.3	mg/L	Parameter: Growth and Reproduction (Yield)	Supplemental	45312003 Kirk et al. 2000
Aquatic plants	Non-vascular aquatic plant Blue-green algae (*Anabeana flos-aquae*)	Acute	5d NOAEC	0.46	mg/L	Parameter: Growth and Reproduction (Yield)	Supplemental	45312003 Kirk et al. 2000
Terrestrial	Avian Northern bobwhite Quail (*Colinus virginianus*)	Acute oral	8d LD50	> 2585	mg/kg bw	Practically Non-Toxic	Acceptable	41829001 Campbell et al. 1990
Terrestrial	Avian Mallard duck (*Anas platyrhynchos*)	Subacute dietary	8d LC50	> 7265	ppm	Practically Non-Toxic	Supplemental	41829002 Long et al. 1990
Terrestrial	Rodent Rat	Acute oral	LD50	1026 (M) 1124 (F)	mg/kg	Slightly Toxic	Acceptable	00064938 Gerbig & Emerson 1970

Triclopyr Triethylamine (TEA) (colored cells are most sensitive for TEA)

Habitat	Taxa	Effect Type	Endpoint	Value (expressed as the acid equivalent)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Aquatic	Freshwater fish Rainbow Trout (*Oncorhynchus mykiss*)	Acute	96h LC50	273.7 (flow-through)	mg/L	Practically Non-Toxic	Acceptable	00151956 McCarty 1978
Aquatic	Freshwater fish Rainbow Trout (*Oncorhynchus mykiss*)	Acute	96h LC50	233.1	mg/L	Practically Non-Toxic	Acceptable	00151956 McCarty 1978
Aquatic	Freshwater fish Rainbow Trout (*Oncorhynchus mykiss*)	Acute	96h LC50	79.2 (flow-through)	mg/L	Practically Non-Toxic	Acceptable (for formulated product)	00049637 Dow Chemical 1973
Aquatic	Freshwater fish Bluegill sunfish (*Lepomis macrochirus*)	Acute	96h LC50	398.7 (flow-through)	mg/L	Practically Non-Toxic	Acceptable	00151956 McCarty 1978
Aquatic	Freshwater fish Bluegill sunfish (*Lepomis macrochirus*)	Acute	96h LC50	155.4 (flow-through)	mg/L	Practically Non-Toxic	Acceptable (for formulated product)	00049637 Dow Chemical 1973
Aquatic	Freshwater fish Fathead Minnow (*Pimephales promelas*)	Acute	96h LC50	422.8 (flow-through)	mg/L	Practically Non-Toxic	Acceptable	00151956 McCarty 1978
Aquatic	Freshwater fish Fathead Minnow (*Pimephales promelas*)	Acute	96h LC50	168.5 (static)	mg/L	Practically Non-Toxic	Acceptable	00151958 Mayes 1983
Aquatic	Freshwater fish Fathead Minnow (*Pimephales promelas*)	Acute	96h LC50	86.4 (flow-through)	mg/L	Practically Non-Toxic	Acceptable	00151958 Mayes 1983
Aquatic	Marine/Estuarine fish Inland silverside (*Menidia beryllina*)	Acute	LC50	40.1	mg/L	Practically Non-Toxic	Acceptable (for formulated product)	41633703 Ward 1989
Aquatic	Freshwater fish Fathead Minnow (*Pimephales promelas*)	Chronic	NOEC LOEC	> 32.2 < 50.2	mg/L mg/L	Parameter: Length	Acceptable	00151958 Mayes 1983
Aquatic	Freshwater invertebrate Water flea (*Daphnia magna*)	Acute	48h EC50	463.5	mg/L	Practically Non-Toxic	Acceptable	00151959 Gerisch 1982

Habitat	Taxa	Effect Type	Endpoint	Value (expressed as the acid equivalent)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Aquatic	Freshwater invertebrate Water flea (*Daphnia magna*)	Acute	48h EC50	346	mg/L	Practically Non-Toxic	Acceptable	00151956 McCarty 1978
Aquatic	Freshwater invertebrate Water flea (*Daphnia magna*)	Chronic	NOEC LOEC	25 46.2	mg/L	total young and mean brood size	Acceptable	00151959 Gerisch 1982
Aquatic	Mollusca Eastern oyster (*Crassostrea virginica*) (shell deposition)	Acute	LC50/EC50	18.4	mg/L	Slightly Toxic	Acceptable (for formulated product)	42646101 Kowalski 1992
Aquatic	Mollusca Eastern oyster (*Crassostrea virginica*) (embryo larvae)	Acute	48h EC50	> 16.9 < 26.3	mg/L mg/L	Parameter: 100% abnormal development at 87 ppm	Acceptable (for formulated product)	00062623 EG & G 1975 40346606?
Aquatic	Crustacea Pink shrimp (*Penaeus duorarum*)	Acute	LC50/EC50	270.5	mg/L	Practically Non-Toxic	Supplemental	00062623 EG & G 1975 40346606?
Aquatic	Crustacea Grass Shrimp (*Palaemontes pugio*)	Acute	LC50/EC50	103.7	mg/L	Practically Non-Toxic	Acceptable (for formulated product)	42646102 Kowalski 1992
Aquatic	Crustacea Fiddler crab (*Uca pugilator*)	Acute	LC50/EC50	> 302.2	mg/L	Practically Non-Toxic	Supplemental	00062623 EG & G 1975 40346606?
Aquatic plants	Vascular aquatic plant Duckweed (*Lemna gibba*)	Acute	14d EC50	6.06	mg/L	Parameter: Growth and Reproduction	Acceptable	41633709 Cowgill 1987
Aquatic plants	Vascular aquatic plant Duckweed (*Lemna gibba*)	Acute	14d EC50	7.6	mg/L	Parameter: Growth and Reproduction	Acceptable	41736302 Cowgill 1988
Aquatic plants	Non-vascular aquatic plant Green algae (*Kirchneria subcapitata*) (Formerly *Selenastrum capricortum*)	Acute	5d EC50	12.1	mg/L	Parameter: Growth	Acceptable	41736305 Cowgill 1987

Habitat	Taxa	Effect Type	Endpoint	Value (expressed as the acid equivalent)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Aquatic plants	Non-vascular aquatic plant Blue-green algae (*Anabeana flos-aquae*)	Acute	7d EC50	4.1	mg/L	Parameter: Growth	Acceptable	41633706 Cowgill 1987
Aquatic plants	Non-vascular aquatic plant Freshwater diatom (*Navicula pelliculosa*)	Acute	4d EC50	10.6	mg/L	Parameter: Growth	Acceptable	41633708 Cowgill 1987
Aquatic plants	Non-vascular aquatic plant Marine diatom (*Skeletonema costatum*)	Acute	5d EC50	4.6	mg/L	Parameter: Growth	Acceptable	41633707 Cowgill 1987
Terrestrial	Avian Mallard duck (*Anas platyrhynchos*)	Acute oral	14d LD50	1418	mg/kg bw	Practically Non-Toxic	Acceptable	40346501 Fink 1978
Terrestrial	Avian Mallard duck (*Anas platyrhynchos*)	Acute oral	8d LD50	1417.6	mg/kg bw	Practically Non-Toxic	Acceptable	00134178
Terrestrial	Avian Northern bobwhite Quail (*Colinus virginianus*)	Subacute dietary	8d LC50	5,189	ppm	Practically Non-Toxic	Acceptable	40346503 Fink 1978
Terrestrial	Avian Mallard duck (*Anas platyrhynchos*)	Subacute dietary	8d LC50	> 4,464.8	ppm	Practically Non-Toxic	Acceptable	40346502 Fink 1977
Terrestrial	Rodent Rat	Acute oral	LD50	572 (M & F)	mg/kg	Slightly Toxic	Acceptable	41443301 Mizell & Lomax 1988
Terrestrial Plant	Terrestrial dicot Sunflower (*Helianthus annus*) (Vegetative Vigor)	Acute	14d EC25	0.0063	lbs ae/A	Parameter: shoot weight	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial dicot Sunflower (*Helianthus annus*) (Vegetative Vigor)	Acute	NOAEC	0.0028	lbs ae/A	Parameter: shoot weight	Acceptable	43129801 Schwab 1993

Habitat	Taxa	Effect Type	Endpoint	Value (expressed as the acid equivalent)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Terrestrial Plant	Terrestrial dicot Sunflower (*Helianthus annus*) (Vegetative Vigor)	Acute	14d EC25	0.005	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial dicot Sunflower (*Helianthus annus*) (Vegetative Vigor)	Acute	NOAEC	0.0028	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial dicot Sunflower (*Helianthus annus*) (Seedling Emergence)	Acute	14d EC25	> 0.69	lbs ae/A	Parameter: shoot length	Acceptable	43129801
Terrestrial Plant	Terrestrial dicot Sunflower (*Helianthus annus*) (Seedling Emergence)	Acute	21d EC25	< 9.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990
Terrestrial Plant	Terrestrial dicot Sugarbeet (*Beta vulgaris*) (Vegetative Vigor)	Acute	14d EC25	0.030	lbs ae/A	Parameter: shoot weight	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial dicot Sugarbeet (*Beta vulgaris*) (Vegetative Vigor)	Acute	14d EC25	0.11	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial dicot Sugarbeet (*Beta vulgaris*) (Seedling Emergence)	Acute	14d EC25	> 0.69	lbs ae/A	Parameter: shoot length	Acceptable	43129801
Terrestrial Plant	Terrestrial dicot Sugarbeet (*Beta vulgaris*) (Seedling Emergence)	Acute	21d EC25	< 9.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990
Terrestrial Plant	Terrestrial dicot Tomato (*Lycopersicon esculentum*) (Vegetative Vigor)	Acute	14d EC25	0.0135	lbs ae/A	Parameter: shoot weight	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial dicot Tomato (*Lycopersicon esculentum*) (Vegetative Vigor)	Acute	14d EC25	0.018	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993

Habitat	Taxa	Effect Type	Endpoint	Value (expressed as the acid equivalent)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Terrestrial Plant	Terrestrial dicot Tomato (*Lycopersicon esculentum*) (Seedling Emergence)	Acute	14d EC25	> 0.69	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial dicot Tomato (*Lycopersicon esculentum*) (Seedling Emergence)	Acute	21d EC25	< 9.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990
Terrestrial Plant	Terrestrial monocot Wheat (*Triticum aestivum*) (Vegetative Vigor)	Acute	14d EC25	> 0.69	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial monocot Wheat (*Triticum aestivum*) (Seedling Emergence)	Acute	14d EC25	> 0.69	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial monocot Wheat (*Triticum aestivum*) (Seedling Emergence)	Acute	21d EC25	< 9.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990
Terrestrial Plant	Terrestrial dicot Oilseed rape (*Brassica napus*) (Vegetative Vigor)	Acute	14d EC25	0.051	lbs ae/A	Parameter: shoot weight	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial dicot Oilseed rape (*Brassica napus*) (Vegetative Vigor)	Acute	14d EC25	0.064	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial dicot Oilseed rape (*Brassica napus*) (Seedling Emergence)	Acute	14d EC25	> 0.69	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial dicot Oilseed rape (*Brassica napus*) (Seedling Emergence)	Acute	21d EC25	< 9.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990
Terrestrial Plant	Terrestrial dicot Radish (*Raphanus sativus*) (Vegetative Vigor)	Acute	14d EC25	0.125	lbs ae/A	Parameter: shoot weight	Acceptable	43129801 Schwab 1993

Habitat	Taxa	Effect Type	Endpoint	Value (expressed as the acid equivalent)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Terrestrial Plant	Terrestrial dicot Radish (*Raphanus sativus*) (Vegetative Vigor)	Acute	14d EC25	0.132	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial dicot Radish (*Raphanus sativus*) (Seedling Emergence)	Acute	14d EC25	> 0.69	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial dicot Radish (*Raphanus sativus*) (Seedling Emergence)	Acute	21d EC25	< 9.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990
Terrestrial Plant	Terrestrial monocot Corn (*Zea mays*) (Vegetative Vigor)	Acute	14d EC25	0.121	lbs ae/A	Parameter: shoot weight	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial monocot Corn (*Zea mays*) (Vegetative Vigor)	Acute	14d EC25	0.32	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial monocot Corn (*Zea mays*) (Seedling Emergence)	Acute	14d EC25	> 0.23	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial monocot Corn (*Zea mays*) (Seedling Emergence)	Acute	21d EC25	< 9.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990
Terrestrial Plant	Terrestrial monocot Barley (*Hordeum vulgare*) (Seedling Emergence)	Acute	14d EC25	> 0.69	lbs ae/A	Parameter: shoot weight	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial monocot Barley (*Hordeum vulgare*) (Seedling Emergence)	Acute	14d EC25	> 0.23	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial monocot Barley (*Hordeum vulgare*) (Seedling Emergence)	Acute	NOAEC	0.230	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial monocot Barley (*Hordeum vulgare*) (Seedling Emergence)	Acute	14d EC25	> 0.69	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial monocot Barley (*Hordeum vulgare*) (Seedling Emergence)	Acute	21d EC25	< 9.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990

Habitat	Taxa	Effect Type	Endpoint	Value (expressed as the acid equivalent)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Terrestrial Plant	Terrestrial monocot Onion (*Allium cepa*) (Vegetative Vigor)	Acute	14d EC25	0.114	lbs ae/A	Parameter: shoot weight	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial monocot Onion (*Allium cepa*) (Vegetative Vigor)	Acute	14d EC25	0.24	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial monocot Onion (*Allium cepa*) (Seedling Emergence)	Acute	14d EC25	> 0.69	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial monocot Onion (*Allium cepa*) (Seedling Emergence)	Acute	21d EC25	< 9.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990
Terrestrial Plant	Terrestrial dicot Soybean (*Glycine max*) (Vegetative Vigor)	Acute	14d EC25	0.0057	lbs ae/A	Parameter: shoot weight	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial dicot Soybean (*Glycine max*) (Vegetative Vigor)	Acute	NOAEC	0.0003	lbs ae/A	Parameter: shoot weight	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial dicot Soybean (*Glycine max*) (Vegetative Vigor)	Acute	14d EC25	0.028	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial dicot Soybean (*Glycine max*) (Seedling Emergence)	Acute	14d EC25	> 0.23	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial dicot Soybean (*Glycine max*) (Seedling Emergence)	Acute	NOAEC	0.0028	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial dicot Soybean (*Glycine max*) (Seedling Emergence)	Acute	21d EC25	< 9.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990
Terrestrial Plant	Monocots & Dicots Veg.Crops (10 Sp.) (Vegetative Vigor)	Acute	6 wks EC25	> 9.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990

Triclopyr Butoxyethyl Ester (BEE) (colored cells are most sensitive for BEE)

Habitat	Taxa	Effect Type	Endpoint	Value (expressed as the acid equivalent)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Aquatic	Freshwater fish Rainbow Trout (*Oncorhynchus mykiss*)	Acute	96h LC50	0.47	mg/L	Highly Toxic	Acceptable	42884501 Woodburn et al. 1993
Aquatic	Freshwater fish Rainbow Trout (*Oncorhynchus mykiss*)	Acute	96h LC50	1.29 (Propylene glycol butyl ether)	mg/L	Moderately Toxic	Acceptable	00134181 Acc# 229783?
Aquatic	Freshwater fish Rainbow Trout (*Oncorhynchus mykiss*)	Acute	96h LC50	0.70	mg/L	Highly Toxic	Acceptable	43442602 Weinberg et al. 1994
Aquatic	Freshwater fish Rainbow Trout (*Oncorhynchus mykiss*)	Acute	24h LC50	0.35-1.2	mg/L	Highly to Moderately Toxic	Supplemental	41971603 Gorzinski 1991
Aquatic	Freshwater fish Bluegill sunfish (*Lepomis macrochirus*)	Acute	96h LC50	1.46 (Propylene glycol butyl ether)	mg/L	Moderately Toxic	Acceptable	00134181 Acc# 229783?
Aquatic	Freshwater fish Bluegill sunfish (*Lepomis macrochirus*)	Acute	96h LC50	0.25	mg/L	Highly Toxic	Acceptable	42917901 Woodburn et al. 1993
Aquatic	Freshwater fish Bluegill sunfish (*Lepomis macrochirus*)	Acute	24h LC50	0.59	mg/L	Moderately Toxic	Supplemental	41971604 Gorzinski 1991
Aquatic	Freshwater fish Bluegill sunfish (*Lepomis macrochirus*)	Acute	96h LC50	0.31	mg/L	Highly Toxic	Acceptable	43442601 Weinberg et al. 1994
Aquatic	Freshwater fish Coho Salmon (*Oncorhyncus kissutch*)	Acute	96h LC50	0.32-0.33 (yolk-sac fry) 1.0 (juvenile fry)	mg/L mg/L	Highly Toxic Moderately Toxic	Supplemental	41736304 Barron 1987
Aquatic	Freshwater fish Fathead Minnow (*Pimephales promelas*)	Acute	24h LC50	1.7	mg/L	Moderately Toxic	Supplemental	00151965 Batchelder 1981
Aquatic	Marine/Estuarine fish Inland silverside (*Menidia beryllina*)	Acute	96h LC50	0.32	mg/L	Highly Toxic	Acceptable	42053901 Ward 1991

Habitat	Taxa	Effect Type	Endpoint	Value (expressed as the acid equivalent)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Aquatic	Marine/Estuarine fish Inland silverside (*Menidia beryllina*)	Acute	96h LC50	0.34	mg/L	Highly Toxic	Acceptable (for formulated product)	41969901 Ward 1991
Aquatic	Freshwater fish Rainbow Trout (*Oncorhynchus mykiss*)	Chronic	NOEC LOEC	0.019 0.034	mg/L mg/L	Very Highly Toxic Parameter: growth effects	Acceptable	43230201 Weinberg et al. 1994
Aquatic	Freshwater invertebrate Water flea (*Daphnia magna*)	Acute	48h EC50	1.2	mg/L	Moderately Toxic	Supplemental	00151963 Batchelder 1980
Aquatic	Freshwater invertebrate Water flea (*Daphnia magna*)	Acute	48h EC50	8.63	mg/L	Slightly Toxic	Acceptable	00151965 47006032 Milazzo 1981
Aquatic	Freshwater invertebrate Water flea (*Daphnia magna*)	Acute	48h EC50	0.25	mg/L	Highly Toxic	Acceptable	43442603 Weinberg et al. 1994
Aquatic	Mollusca Eastern oyster (*Crassostrea virginica*) (shell deposition)	Acute	96h EC50	0.33 (Species)	mg/L	Highly Toxic	Acceptable	41971602 Boeri 1991
Aquatic	Mollusca Eastern oyster (*Crassostrea virginica*) (shell deposition)	Acute	96h EC50	0.23	mg/L	Highly Toxic	Acceptable (for formulated product)	41969903 Boeri 1991
Aquatic	Crustacea Grass Shrimp (*Palaemontes pugio*)	Acute	96h LC50	1.8	mg/L	Moderately Toxic	Acceptable	41971601 Boeri 1991
Aquatic	Crustacea Grass Shrimp (*Palaemontes pugio*)	Acute	96h LC50	0.77	mg/L	Highly Toxic	Acceptable (for formulated product)	41969902 Ward 1991
Aquatic plants	Vascular aquatic plant Duckweed (*Lemna gibba*)	Acute	14d EC50	0.86	mg/L	Parameter: Growth and Reproduction	Supplemental (downgraded)	42719101 Milazzo 1993

Habitat	Taxa	Effect Type	Endpoint	Value (expressed as the acid equivalent)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Aquatic plants	Vascular aquatic plant Duckweed (*Lemna gibba*)	Chronic	14d NOAEC	< 0.111	mg/L	Parameter: Growth and Reproduction	Supplemental	42719101 Milazzo 1993
Aquatic plants	Non-vascular aquatic plant Green algae (*Kirchneria subcapitata*) (Formerly *Selenastrum capricortum*)	Acute	5d EC50	2.5	mg/L	Parameter: Growth	Acceptable	41633704, 42090422 Cowgill & Millazzo 1989
Aquatic plants	Non-vascular aquatic plant Blue-green algae (*Anabeana flos-aquae*)	Acute	5d EC50	1.42	mg/L	Parameter: Growth	Acceptable	42721101 Hughes 1993
Aquatic plants	Non-vascular aquatic plant Freshwater diatom (*Navicula pelliculosa*)	Acute	5d EC50	0.073	mg/L	Parameter: Growth	Acceptable	42721102 Hughes 1993
Aquatic plants (BEE)	Non-vascular aquatic plant Freshwater diatom (*Navicula pelliculosa*)	Chronic	5d NOAEC	0.0014	mg/L	Parameter: Growth	Acceptable	42721102 Hughes 1993
Aquatic plants	Non-vascular aquatic plant Marine diatom (*Skeletonema costatum*)	Acute	5d EC50	0.84	mg/L	Parameter: Growth	Acceptable	42721103 Hughes 1993
Terrestrial	Avian Northern bobwhite Quail (*Colinus virginianus*)	Acute oral	21d LD50	529	mg/kg bw	Practically Non-Toxic	Acceptable	41902002 Campbell & Lynn 1991
Terrestrial	Avian Northern bobwhite Quail (*Colinus virginianus*)	Acute oral	14d LD50	611	mg/kg bw	Practically Non-Toxic	Acceptable	41902003 Campbell & Lynn 1991
Terrestrial	Avian Northern bobwhite Quail (*Colinus virginianus*)	Subacute dietary	8d LC50	6038	ppm	Slightly Toxic	Acceptable	00134180 Wildlife Int'l 1978
Terrestrial	Avian Northern bobwhite Quail (*Colinus virginianus*)	Subacute dietary	8d LC50	3885	ppm	Slightly Toxic	Acceptable	41905501 Lynn et al. 1991
Terrestrial	Avian Mallard duck (*Anas platyrhynchos*)	Subacute dietary	8d LC50	> 6689	ppm	Practically Non-Toxic	Acceptable	00134179 Wildlife Int'l 1977

Habitat	Taxa	Effect Type	Endpoint	Value (expressed as the acid equivalent)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Terrestrial	Avian Mallard duck (*Anas platyrhynchos*)	Subacute dietary	8d LC50	> 3885	ppm	Practically Non-Toxic	Acceptable	41905501 Lynn et al. 1992
Terrestrial	Rodent Rat	Acute oral	LD50	578 (M & F)	mg/kg	Slightly Toxic	Acceptable	40557004 Wall et al. 1987
Terrestrial	Honeybee (*Apis mellifera*)	Acute	Contact 48h LD50	> 72	µg/bee	Practically Non-Toxic	Acceptable	41219109 Dingledine 1985
Terrestrial Plant	Terrestrial dicot Soybean (*Glycine max*) (Seedling Emergence)	Acute	21d EC25	< 8.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990
Terrestrial Plant	Terrestrial dicot Sunflower (*Helianthus annus*) (Seedling Emergence)	Acute	21d EC25	< 8.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990
Terrestrial Plant	Terrestrial dicot Sunflower (*Helianthus annus*) (Vegetative Vigor)	Acute	51d EC25	0.006	lbs/A	parameter: shoot weight	Acceptable	43650001 Schwab 1995
Terrestrial Plant	Terrestrial dicot Sunflower (*Helianthus annus*) (Vegetative Vigor)	Acute	NOAEC	0.028	lbs/A	parameter: shoot weight	Acceptable	43650001 Schwab 1995
Terrestrial Plant	Terrestrial dicot Tomato (*Lycopersicon esculentum*) (Seedling Emergence)	Acute	21d EC25	< 8.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990
Terrestrial Plant	Terrestrial dicot Radish (*Raphanus sativus*) (Seedling Emergence)	Acute	21d EC25	< 8.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990
Terrestrial Plant	Terrestrial monocot Onion (*Allium cepa*) (Seedling Emergence)	Acute	21d EC25	< 8.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990
Terrestrial Plant	Terrestrial monocot Onion (*Allium cepa*) (Seedling Emergence)	Acute	51d EC25	0.053	lbs/A	parameter: shoot weight	Acceptable	43650001 Schwab 1995

Habitat	Taxa	Effect Type	Endpoint	Value (expressed as the acid equivalent)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Terrestrial Plant (BEE)	Terrestrial monocot Onion (*Allium cepa*) (Seedling Emergence)	Acute	NOAEC	0.0021	lbs/A	parameter: shoot weight	Acceptable	43650001 Schwab 1995
Terrestrial Plant	Terrestrial monocot Onion (*Allium cepa*) (Vegetative Vigor)	Acute	51d EC25	0.063	lbs/A	parameter: shoot weight	Acceptable	43650001 Schwab 1995
Terrestrial Plant	Terrestrial monocot Onion (*Allium cepa*) (Vegetative Vigor)	Acute	NOAEC	< 0.063	lbs/A	parameter: shoot weight	Acceptable	43650001 Schwab 1995
Terrestrial Plant	Terrestrial dicot Sugarbeet (*Beta vulgaris*) (Seedling Emergence)	Acute	21d EC25	< 8.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990
Terrestrial Plant	Terrestrial monocot Barley (*Hordeum vulgare*) (Seedling Emergence)	Acute	21d EC25	< 8.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990
Terrestrial Plant	Terrestrial monocot Corn (*Zea mays*) (Seedling Emergence)	Acute	21d EC25	> 8.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990
Terrestrial Plant	Terrestrial monocot Wheat (*Triticum aestivum*) (Seedling Emergence)	Acute	21d EC25	< 8.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990
Terrestrial Plant	Terrestrial dicot Oilseed rape (*Brassica napus*) (Seedling Emergence)	Acute	21d EC25	< 8.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990
Terrestrial Plant	Terrestrial dicot Alfalfa (*Medicago sativa*) (Seedling Emergence)	Acute	51d EC25	0.045	lbs/A	parameter: emergence	Acceptable	43650001 Schwab 1995
Terrestrial Plant (BEE)	Terrestrial dicot Alfalfa (*Medicago sativa*) (Seedling Emergence)	Acute	NOAEC	0.0026	lbs/A	parameter: emergence	Acceptable	43650001 Schwab 1995

Triclopyr Butoxyethyl Ester (BEE)/Picloram ethyl ester Mixture

Habitat	Taxa	Effect Type	Endpoint	Value (% ai)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Aquatic plants	Vascular aquatic plant Duckweed (*Lemna gibba*)	Acute	14d EC50 NOEC	6.6 (9.2; 99.8) 1.03 (1.43; 15.9) (TBEE = 9.2% ai)	mg/L mg/L	Parameter: Growth (plants)	Acceptable	43230310 Milazzo 1994
Aquatic plants	Non-vascular aquatic plant Blue-green algae (*Anabeana flos-aquae*)	Acute	5d EC50	0.20 (0.276; 3.0) (TBEE = 9.2% ai)	mg/L	Parameter: Growth	Invalid	43230307 Boeri et al. 1994
Aquatic plants	Non-vascular aquatic plant Green algae (*Kirchneria subcapitata*) (Formerly *Selenastrum capricortum*)	Acute	5d EC50 NOEC	0.32 (0.441; 4.9) 0.211 (0.294; 3.2) (TBEE = 9.2% ai)	mg/L mg/L	Parameter: Growth	Supplemental	42645901 Hughes et al. 1993
Aquatic plants	Non-vascular aquatic plant Marine diatom (*Skeletonema costatum*)	Acute	5d EC50	0.12 (0.166; 1.8) (TBEE = 9.2% ai)	mg/L	Parameter: Growth	Invalid	43230304 Boeri et al. 1994
Aquatic plants	Non-vascular aquatic plant Freshwater diatom (*Navicula pelliculosa*)	Acute	5d EC50	0.36 (0.50; 5.4) (TBEE = 9.2% ai)	mg/L	Parameter: Growth	Invalid	43230301 Boeri et al. 1994
Terrestrial	Honeybee (*Apis mellifera*)	Acute	Contact 44h LD50	> 1.62 (> 2.25; > 25) (TBEE = 9.2% ai)	µg/bee	Practically Non-Toxic	Acceptable	42625901 Hoxter et al.1992
Terrestrial Plant	Terrestrial dicot Radish (*Raphanus sativus*) (Vegetative Vigor)	Acute	21 d EC25 NOEL	0.008 (0.035) 0.0035 (0.015) (TBEE = 23.5% ai)	lbs ae/A lbs ae/A	Parameter: height	Supplemental	41296501 Weseloh and Stockdale 1989
Terrestrial Plant	Terrestrial dicot Soybean (*Glycine max*) (Vegetative Vigor)	Acute	55 d EC25 NOEL	0.00005 (0.00021) 0.000026 (0.00011) (TBEE = 23.5% ai)	lbs ae/A lbs ae/A	Parameter: height	Supplemental	41296501 Weseloh and Stockdale 1989

Habitat	Taxa	Effect Type	Endpoint	Value (% ai)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Terrestrial Plant	Terrestrial monocot Wheat (*Triticum aestivum*) (Vegetative Vigor)	Acute	42 d EC25 NOEL	0.05 (0.21) 0.0146 (0.062) (TBEE = 23.5% ai)	lbs ae/A lbs ae/A	Parameter: height	Supplemental	41296501 Weseloh and Stockdale 1989
Terrestrial Plant	Terrestrial dicot Drybean (*Phaseolus vulgaris*) (Seedling Emergence)	Acute	14 d EC25 NOEL	0.0000009 (0.000004) < 0.000007 (< 0.00003) (TBEE = 23.5% ai)	lbs ae/A lbs ae/A	Parameter: emergence	Supplemental	41296501 Weseloh and Stockdale 1989
Terrestrial Plant	Terrestrial monocot Onion (*Allium cepa*) (Seedling Emergence)	Acute	14 d EC25 NOEL	0.008 (0.035) 0.004 (0.0156) (TBEE = 23.5% ai)	lbs ae/A lbs ae/A	Parameter: emergence	Supplemental	41296501 Weseloh and Stockdale 1989
Terrestrial Plant	Terrestrial monocot Wheat (*Triticum aestivum*) (Seedling Emergence)	Acute	21d EC25 NOEL	0.006 (0.025) 0.02 (0.08) (TBEE = 23.5% ai)	lbs ae/A lbs ae/A	Parameter: emergence	Supplemental	41296501 Weseloh and Stockdale 1989
Terrestrial Plant	Terrestrial Lima bean (*Phaseolus lunatus*) (Seedling Emergence)	Acute	21d EC25 NOEL	0.00003 (0.00004; 0.00042) (TBEE = 9.2% ai)	lbs ae/A lbs ae/A	Parameter: shoot height	Supplemental	43276601 Schwab 1994
Terrestrial Plant	Terrestrial monocot Barley (*Hordeum vulgare*) (Seedling Emergence)	Acute	21d EC25 NOEL	0.57 (0.791; 8.6) (TBEE = 9.2% ai))	lbs ae/A lbs ae/A	Parameter: shoot weight	Supplemental	43276601 Schwab 1994

Combined –Most sensitive when converted to acid equivalent (Acid, TEA or BEE)

Habitat	Taxa	Effect Type	Endpoint	Form Tested	Value (acid equivalent)	Value (Original Value)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Aquatic (BEE)	Freshwater fish Bluegill sunfish (*Lepomis macrochirus*)	Acute	96h LC50	F (Garlon 4) 62.9% ai TBEE	0.32	0.44	mg/L	Highly Toxic	Acceptable	43442601 Weinberg et al. 1994
Aquatic (BEE)	Freshwater fish Bluegill sunfish (*Lepomis macrochirus*)	Acute	96h LC50	TGAI 96.98% ai TBEE	0.26	0.36	mg/L	Highly Toxic	Acceptable	42917901 Woodburn et al. 1993
Aquatic (BEE)	Freshwater fish Rainbow Trout (*Oncorhynchus mykiss*)	Acute	96h LC50	F (Garlon 4) 62.9% ai TBEE	0.70	0.98	mg/L	Highly Toxic	Acceptable	43442602 Weinberg et al. 1994
Aquatic (BEE)	Freshwater fish Rainbow Trout (*Oncorhynchus mykiss*)	Acute	96h LC50	TGAI 96.98% ai TBEE	0.47	0.65	mg/L	Highly Toxic	Acceptable	42884501 Woodburn et al. 1993
Aquatic (BEE)	Freshwater fish Rainbow Trout (*Oncorhynchus mykiss*)	Chronic	NOEC LOEC	TGAI 96.98% ai TBEE	0.019 0.034	0.0263 0.048	mg/L	Very Highly Toxic Parameter: growth effects	Acceptable	43230201 Weinberg et al. 1994
Aquatic (TEA)	Freshwater fish Fathead Minnow (*Pimephales promelas*)	Chronic	NOEC LOEC	F 44.9% ai TEA	> 32.2 <50.2	>46.7 (>104) <72.7 (<162)	mg/L	Parameter: Length	Acceptable	00151958 Mayes 1983
Aquatic (BEE)	Freshwater invertebrate Water flea (*Daphnia magna*)	Acute	48h EC50	F (Garlon 4) 62.2% ai TBEE	0.25	0.35	mg/L	Highly Toxic	Acceptable	43442603 Weinberg et al. 1994
Aquatic (BEE)	Freshwater invertebrate Water flea (*Daphnia magna*)	Acute	48h EC50	TGAI 96.4% ai TBEE	1.2	1.7 (nominal)	mg/L	Moderately Toxic	Supplemental	00151963 Batchelder 1980
Aquatic (TEA)	Freshwater invertebrate Water flea (*Daphnia magna*)	Chronic	NOEC LOEC	F 44.9% ai TEA	25 46.2	36.2 (80.7) 66.9 (149.0)	mg/L	total young and mean brood size	Acceptable	00151959, 42090411, 92189013 Gerisch 1982
Aquatic plants (BEE)	Vascular aquatic plant Duckweed (*Lemna gibba*)	Acute	14d EC50	TGAI 96.98% ai TBEE	0.86	1.2	mg/L	Parameter: Growth and Reproduction	Supplemental	42719101 Milazzo et al. 1993

Habitat	Taxa	Effect Type	Endpoint	Form Tested	Value (acid equivalent)	Value (Original Value)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Aquatic plants (BEE)	Duckweed (*Lemna gibba*)	Chronic	14d NOAEC	TGAI 96.98% ai TBEE	< 0.111	< 0.155	mg/L	Parameter: Growth and Reproduction	Supplemental	42719101 Milazzo et al. 1993
Aquatic plants (TEA)	Vascular aquatic plant Duckweed (*Lemna gibba*)	Acute	14d EC50	F 44.9% ai TEA	6.1	8.8 (19.5)	mg/L	Parameter: Growth and Reproduction	Acceptable	41633709 Cowgill 1987
Aquatic plants (BEE)	Non-vascular aquatic plant Freshwater diatom (*Navicula pelliculosa*)	Acute	5d EC50	TGAI 96.98% ai TBEE	0.073	0.102	mg/L	Parameter: Growth	Acceptable	42721102 Hughes 1993
Aquatic plants (BEE)	Non-vascular aquatic plant Freshwater diatom (*Navicula pelliculosa*)	Chronic	5d NOAEC	TGAI 96.98% ai TBEE	0.0014	0.002	mg/L	Parameter: Growth	Acceptable	42721102 Hughes 1993
Aquatic plants (TEA)	Non-vascular aquatic plant Blue-green algae (*Anabeana flos-aquae*)	Acute	7d EC50	F 45% ai TEA	4.1	5.9	mg/L	Parameter: Growth	Acceptable	41633706 Hughes 1987
Aquatic plants (BEE)	Non-vascular aquatic plant Green algae (*Kirchneria subcapitata*) (Formerly *Selenastrum capricortum*)	Acute	5d EC50	F 61.3% ai TBEE	2.5	3.4 (5.6)	mg/L	Parameter: Growth	Acceptable	41633704, 42090422 Cowgill & Millazzo 1989
Terrestrial (BEE)	Avian Northern bobwhite Quail (*Colinus virginianus*)	Acute oral	21d LD50	TGAI 96.1% ai TBEE	529	735	mg/kg bw	Practically Non-Toxic	Acceptable	41902002 Campbell & Lynn 1991
Terrestrial (BEE)	Avian Northern bobwhite Quail (*Colinus virginianus*)	Acute oral	14d LD50	F (Garlon 4) 62.9% ai TBEE	611	849 (1350)	mg/kg bw	Practically Non-Toxic	Acceptable	41902003 Campbell & Lynn 1991
Terrestrial (Acid)	Avian Northern bobwhite Quail (*Colinus virginianus*)	Subacute dietary	LC50	TGAI Acid	2934	2934	ppm	Slightly Toxic	Acceptable	40346403 Dow Chemical 1976
Terrestrial (BEE)	Avian Northern bobwhite Quail (*Colinus virginianus*)	Subacute dietary	8d LC50	TGAI 96.1% ai TBEE	3885	5401	ppm	Slightly Toxic	Supplemental	41905501 Lynn et al. 1991
Terrestrial (BEE)	Avian Mallard duck (*Anas platyrhynchos*)	Subacute dietary	8d LC50	TGAI 96.1% ai TBEE	> 3,885	> 5,401	ppm	Practically Non-Toxic	Acceptable	41905502 Lynn et al. 1992

Habitat	Taxa	Effect Type	Endpoint	Form Tested	Value (acid equivalent)	Value (Original Value)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Terrestrial (Acid)	Avian Mallard duck (*Anas platyrhynchos*)	Chronic	NOAEC LOAEC	TGAI 98.9% ai Acid	100 200	100 200	ppm	# of 14 d old survivors	Acceptable	00031250 Beavers & Fink 1980
Terrestrial (Acid)	Avian Northern bobwhite Quail (*Colinus virginianus*)	Chronic	NOAEC LOAEC LOAEC	TGAI 98.9% ai Acid	500 >500 200	500 >500 200	ppm ppm ppm	No sign. reproductive impairment	Acceptable	00031251 Beavers & Fink 1979
Terrestrial (Acid)	Rodent Rat	Acute oral	LD50	TGAI Acid	729 (M) 630 (F)	729 (M) 630 (F)	mg/kg	Slightly Toxic	Acceptable	00031940 Henck et al. 1979
Terrestrial (TBEE)	Rodent Rat	Acute oral	LD50	TGAI 96% ai TBEE	578 (M &F)	803 (M & F)	mg/kg	Slightly Toxic	Acceptable	40557004 Wall et al. 1987
Terrestrial (TEA)	Rodent Rat	Acute oral	LD50	TGAI 44.9 % ai TEA	572 (M &F)	1847 (M & F)	mg/kg	Slightly Toxic	Acceptable	41443301 Mizell & Lomax 1988
Terrestrial (Acid)	Rodent	Chronic	NOAEL LOAEL	TGAI 99.4% ai Acid	25 250	25 250	mg/kg bw mg/kg bw	Reproductive/ Systemic	Acceptable (RED 1998)	43545701 Vedula et al 1995
Terrestrial (Acid)	Rodent	Chronic	NOAEL LOAEL	TGAI 99.4% ai Acid	5 25	5 25	mg/kg bw mg/kg bw	Reproductive/ Offspring	Acceptable (HED 2002)	43545701 Vedula et al 1995
Terrestrial (Acid)	Honeybee (*Apis mellifera*)	Acute	Contact 48h LD50	TGAI 99.2% ai Acid	>100	>100	µg/bee	Practically Non-Toxic	Acceptable	40356602 Dingledine 1985
Terrestrial (BEE)	Honeybee (*Apis mellifera*)	Acute	Contact 48h LD50	TGAI 97.7% ai TBEE	> 72	> 100	µg/bee	Practically Non-Toxic	Acceptable	41219109 Dingledine 1985
Terrestrial Plant (TEA)	Terrestrial dicot Sunflower (*Helianthus annus*) (Vegetative Vigor)	Acute	14d EC25	F 46.2% ai TEA	0.005	0.0076	lbs/A	parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant (BEE)	Terrestrial dicot Sunflower (*Helianthus annus*) (Vegetative Vigor)	Acute	51d EC25	F (Garlon 4) 62.2% ai TBEE	0.006	0.0089	lbs/A	parameter: shoot weight	Acceptable	43650001 Schwab 1995

Habitat	Taxa	Effect Type	Endpoint	Form Tested	Value (acid equivalent)	Value (Original Value)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Terrestrial Plant (BEE)	Terrestrial monocot Onion (*Allium cepa*) (Vegetative Vigor)	Acute	51d EC25	F (Garlon 4) 62.2% ai TBEE	0.063	0.0888	lbs/A	parameter: shoot weight	Acceptable	43650001 Schwab 1995
Terrestrial Plant (TEA)	Terrestrial monocot Onion (*Allium cepa*) (Vegetative Vigor)	Acute	14d EC25	F 46.2% ai TEA	0.114	0.166	lbs/A	parameter: shoot weight	Acceptable	43129801 Schwab 1993
Terrestrial Plant (BEE)	Terrestrial dicot Alfalfa (*Medicago sativa*) (Seedling Emergence)	Acute	51d EC25	F (Garlon 4) 62.2% ai TBEE	0.045	0.0622	lbs/A	parameter: emergence	Acceptable	43650001 Schwab 1995
Terrestrial Plant (BEE)	Terrestrial monocot Onion (*Allium cepa*) (Seedling Emergence)	Acute	51d EC25	F (Garlon 4) 62.2% ai TBEE	0.053	0.0732	lbs/A	parameter: shoot weight	Acceptable	43650001 Schwab 1995

Appendix L Ecological Effects Data

Aquatic Animals

Coho salmon (Oncorhynchus kisutch), Static, LC50, / Diuron, 95% ai	96 hr LC50 < 2.4 ppm ai (slope = N.R.)	40098001, 1986 /Supplemental
Cutthroat trout (Oncorhynchus clarki), Static, LC50, / Diuron, 95% ai	96 hr LC50 1.4 ppm ai (slope = N.R.)	40094602, 1980 /Core
Cutthroat trout (Oncorhynchus clarki), Static, LC50, / Diuron, 95% ai	96 hr LC50 0.71 ppm ai (slope = N.R.)	40098001, 1986 /Supplemental
Fathead minnow (Pimephales promelas), Static, LC50, / Diuron, 98.6% ai	96 hr LC50 14.2 ppm ai (slope = N.R.)	00141636, 1975 /Supplemental
Fathead minnow (Pimephales promelas), Static, LOEC, / Diuron, 98.6% ai	60day LOEC 61.8 ppb ai (slope = N.R.)	00141636, 1975 /Core

Lake trout (Salvelinus namaycush), Static, LC50, / Diuron, 95% ai	96 hr LC50 2.7 ppm ai (slope = N.R.)	40094602, 1980 /Core
Lake trout (Salvelinus namaycush), Static, LC50, / Diuron, 95% ai	96 hr LC50 1.2 ppm ai (slope = N.R.)	40098001, 1986 /Supplemental
Rainbow trout (Oncorhynchus mykiss), Static, LC50, / Diuron, 80% ai	96 hr LC50 19.6 ppm ai (slope = N.R.)	42046002, 1991 /Core
Rainbow trout (Oncorhynchus mykiss), Static, LC50, / Diuron, 28% ai	96 hr LC50 23.8 ppm ai (slope = N.R.)	TN 0897, 1975 /Core
Rainbow trout (Oncorhynchus mykiss), Static, LC50, / Diuron, 80WP% ai	96 hr LC50 16 ppm ai (slope = N.R.)	40094602, 1980 /Supplemental

Rainbow trout (Oncorhynchus mykiss), Static, LC50, / Diuron, 95% ai	96 hr LC50 1.95 ppm ai (slope = N.R.)	TN 1020, 1976 /Core
Sheepshead minnow (Cyprinodon variegatus), Static, LC50, / Diuron, 99% ai	96 hr LC50 6.7 ppm ai (slope = N.R.)	41418803, 1986 /Core
Sheepshead minnow (Cyprinodon variegatus), Flow-through, LOEC, / Diuron, 96.8% ai	38 D LOEC < 0.44 ppm ai (slope = N.A.)	42312901, 1992 /Supplemental
Striped mullet (Mugil cephalus), Static, LC50, / Diuron, 95% ai	48 hr LC50 6.3 ppm ai (slope = N.R.)	40228401, 1986 /Supplemental
Brown shrimp (Penaeus aztecus), Flow-through, LC50, / Diuron, 95% ai	48 hr LC50 > 1.0 ppm ai (slope = N.A.)	40228401, 1986 /Supplemental

Mysid (Americamysis bahia), Static, LC50, / Diuron, 99% ai	96 hr LC50 1.1 ppm ai (slope = 12.35)	41418801, 1987 /Supplemental
Mysid (Americamysis bahia), Static, LOEC, / Diuron, 96.8% ai	28 D LOEC 0.56 ppm ai (slope = N.R.)	42500601, 1992 /Core
Scud (Gammarus fasciatus), Static, LC50, / Diuron, 95% ai	96 hr LC50 0.16 ppm ai (slope = N.R.)	40094602, 1980 /Core
Water flea (Simocephalus sp.), Static, EC50, / Diuron, 95% ai	48 hr EC50 2.0 ppm ai (slope = N.R.)	40094602, 1980 /Core
Water flea (Daphnia magna), Static, EC50, / Diuron, 80% ai	48 hr EC50 8.4 ppm ai (slope = 9.10)	42046003, 1991 /Core
Water flea (Daphnia pulex), Static, EC50, / Diuron, 95% ai	48 hr EC50 1.4 ppm ai (slope = N.R.)	40094602, 1980 /Core

Water flea (Daphnia magna), Static, LOEC, / Diuron, 98.2% ai	28 D LOEC 0.2 ppm ai (slope = N.R.)	TN 2418, 1979 /Supplemental
Stonefly (Pteronarcys sp.), Static, LC50, / Diuron, 95% ai	96 hr LC50 1.2 ppm ai (slope = N.R.)	40094602, 1980 /Core
Eastern oyster (Crassostrea virginica), Flow-through, EC50, / Diuron, 96.8% ai	96 hr EC50 4.8 ppm ai (slope = 4.49)	42217201, 1991 /Core
Eastern oyster (Crassostrea virginica), Flow-through, EC50, / Diuron, 95% ai	96 hr EC50 1.8 ppm ai (slope = N.R.)	40228401, 1986 /Core

Terrestrial Animals

Bobwhite quail (Colinus virginianus), O, LD50, / Diuron, 92.8% ai	21 D LD50 940 mg/kg ai (slope = 4.01)	50150170, 1985 /Core

Bobwhite quail (Colinus virginianus), Dietary, LC50, / Diuron, >95% ai	8 D LC50 1730 ppm ai (slope = 7.22)	00022923, 1975 /Core
Japanese quail (Coturnix japonica), Dietary, LC50, / Diuron, >95% ai	8 D LC50 > 5000 ppm ai (slope = N.R.)	00022923, 1975 /Supplemental
Mallard duck (Anas platyrhynchos), Oral, LD50, / Diuron, 95% ai	14 D LD50 > 2000 mg/kg ai (slope = N.R.)	00160000, 1970 /Core
Mallard duck (Anas platyrhynchos), Dietary, LC50, / Diuron, >95% ai	8 D LC50 > 5000 ppm ai (slope = N.R.)	00022923, 1975 /Core
Ring-necked pheasant (Phasianus colchicus), Dietary, LC50, / Diuron, >95% ai	8 D LC50 > 5000 ppm ai (slope = N.R.)	00022923, 1975 /Core

Honey bee (Apis mellifera), Contact, LD50, / Diuron, Tech% ai	48 hr LD50 > 145.03 ug/Bee ai (slope = N.R.)	36935, 1975 /Core

Plants

Algae (Nitzschia closterium), Static, EC50, / Diuron, 95% ai	72 hr EC50 50 ppb ai (slope = N.R.)	40228401, 1986 /Supplemental
Algae (Amphora exigua), Static, EC50, / Diuron, 95% ai	72 hr EC50 31 ppb ai (slope = N.R.)	40228401, 1986 /Supplemental
Algae (Stauroneis amphoroides), Static, EC50, / Diuron, 95% ai	72 hr EC50 31 ppb ai (slope = N.R.)	40228401, 1986 /Supplemental
Algae (Achnanthes brevipes), Static, EC50, / Diuron, 95% ai	72 hr EC50 24 ppb ai (slope = N.R.)	40228401, 1986 /Supplemental
Algae (Cyclotella nana), Static, EC50, / Diuron, 95% ai	72 hr EC50 39 ppb ai (slope = N.R.)	40228401, 1986 /Supplemental
Algae (Isochrysis galbana), Static, EC50, / Diuron, 95% ai	240hr EC50 10 ppb ai (slope = N.R.)	40228401, 1986 /Supplemental
Algae (Monochrysis lutheri), Static, EC50, / Diuron, 95% ai	72 hr EC50 18 ppb ai (slope = N.R.)	40228401, 1986 /Supplemental
Algae (Porphyridium cruentum), Static, EC50, / Diuron, 95% ai	72 hr EC50 24 ppb ai (slope = N.R.)	40228401, 1986 /Supplemental
Diatom (Thalassiosira fluviatilus), Static, EC50, / Diuron, 95% ai	72 hr EC50 95 ppb ai (slope = N.R.)	40228401, 1986 /Supplemental

Green algae (Selenastrum capricornutum), Static, EC50, / Diuron, 96.8% ai	96 hr EC50 2.4 ppb ai (slope = 2.96)	42218401, 1991 /Core
Green algae (Chlorella sp.), Static, EC50, / Diuron, 95% ai	72 hr EC50 19 ppb ai (slope = N.R.)	40228401, 1986 /Supplemental
Green algae (Chlorococcum sp.), Static, EC50, / Diuron, 95% ai	72 hr EC50 10 ppb ai (slope = N.R.)	40228401, 1986 /Supplemental
Green algae (Chlamydomonas sp.), Static, EC50, / Diuron, 95% ai	72 hr EC50 37 ppb ai (slope = N.R.)	40228401, 1986 /Supplemental
Green algae (Dunaliella tertiolecta), Static, EC50, / Diuron, 95% ai	240hr EC50 20 ppb ai (slope = N.R.)	40228401, 1986 /Supplemental
Green algae (Platymonas sp.), Static, EC50, / Diuron, 95% ai	72 hr EC50 17 ppb ai (slope = N.R.)	40228401, 1986 /Supplemental
Green algae (Neochloris sp.), Static, EC50, / Diuron, 95% ai	72 hr EC50 28 ppb ai (slope = N.R.)	40228401, 1986 /Supplemental
Marine diatom (Phaeodactylum tricornutum), Static, EC50, / Diuron, 95% ai	240hr EC50 10 ppb ai (slope = N.R.)	40228401, 1986 /Supplemental
Marine diatom (Navicula incerta), Static, EC50, / Diuron, 95% ai	72 hr EC50 93 ppb ai (slope = N.R.)	40228401, 1986 /Supplemental
Rape (Brassica sp.), Veg. Vigor sw, EC25, / Diuron, 97.3% ai	21 D EC25 0.0331 lb/A ai (slope = N.R.)	44113401, 1996 /Core
Sorghum (Sorghum halepense), Seed Emerg. sh, EC25, / Diuron, 96.8% ai	14 D EC25 < 12 lb/A ai (slope = NA)	42398501, 1992 /Core
Sorghum (Sorghum bicolor), Veg. Vigor sw, EC25, / Diuron, 97.3% ai	21 D EC25 0.0753 lb/A ai (slope = N.R.)	44113401, 1996 /Core

Sorghum (Sorghum bicolor), Seed Emerg. sh, EC25, / Diuron, 97.3% ai	14 D EC25 0.81 lb/A ai (slope = N.R.)	44113401, 1996 /Core
Soybean (Glycine max), Seed Emerg. , EC25, / Diuron, 96.8% ai	14 D EC25 > 12 lb/A ai (slope = NA)	42398501, 1992 /Core
Soybean (Glycine max), Veg. Vigor sw, EC25, / Diuron, 97.3% ai	21 D EC25 0.012 lb/A ai (slope = N.R.)	44113401, 1996 /Core
Soybean (Glycine max), Seed Emerg. sh, EC25, / Diuron, 97.3% ai	14 D EC25 > 12 lb/A ai (slope = N.R.)	44113401, 1996 /Core
Sugarbeet (Beta vulgaris), Seed Emerg. sh, EC25, / Diuron, 96.8% ai	14 D EC25 < 12 lb/A ai (slope = NA)	42398501, 1992 /Core
Sugarbeet (Beta vulgaris), Veg. Vigor sw, EC25, / Diuron, 97.3% ai	21 D EC25 0.0087 lb/A ai (slope = N.R.)	44113401, 1996 /Core
Sugarbeet (Beta vulgaris), Seed Emerg. sw, EC25, / Diuron, 97.3% ai	14 D EC25 0.092 lb/A ai (slope = N.R.)	44113401, 1996 /Core
Tomato (Lycopersicon esculentum), Seed Emerg. , EC25, / Diuron, 96.8% ai	14 D EC25 > 0.038 lb/A ai (slope = NA)	42398501, 1992 /Core
Tomato (Lycopersicon esculentum), Seed Emerg. sw, EC25, / Diuron, 97.3% ai	14 D EC25 0.075 lb/A ai (slope = N.R.)	44113401, 1996 /Core
Tomato (Lycopersicon esculentum), Veg. Vigor sw, EC25, / Diuron, 97.3% ai	21 D EC25 0.0017 lb/A ai (slope = N.R.)	44113401, 1996 /Core
Wheat (Triticum aestivum), Seed Emerg. sh, EC25, / Diuron, 96.8% ai	14 D EC25 < 12 lb/A ai (slope = NA)	42398501, 1992 /Supplemental
Wheat (Triticum aestivum), Veg. Vigor sw, EC25, / Diuron, 97.3% ai	21 D EC25 0.0208 lb/A ai (slope = 0.90)	44113401, 1996 /Core

Wheat (Triticum aestivum), Seed Emerg. sw, EC25, / Diuron, 97.3% ai	14 D EC25 1.05 lb/A ai (slope = N.R.)	44113401, 1996 /Core

APPENDIX A

Ecological Effects

a. Ecological Effects Data

(1) Terrestrial Animal Data

Avian Acute Toxicity

Avian Acute Oral Toxicity Findings			
Species	**% Test Material (TGAI)**	**LD_{50}**	**Conclusion**
Bobwhite Quail	92.8	940 mg/kg	slightly toxic

These results show that linuron is slightly toxic to birds on an acute basis. The guideline requirement for the avian acute oral LD_{50} study is fulfilled. (MRID 00150170)

Avian Subacute Dietary Toxicity

No acceptable avian dietary toxicity studies on technical linuron have been submitted for review. However, the following data from the USFWS (United States Fish and Wildlife Service) using a 50% formulation were considered. Some toxicity in formulation testing may be due to ingredients other than the active ingredient. Other formulations may be more or less toxic, depending on their ingredients. Technical testing allows prediction of the toxicity due to the active ingredient across all formulations. Therefore, tests with the technical material are still required.

Avian Subacute Dietary Toxicity Findings			
Species	**% Test Material**	**LC_{50}**	**Conclusions**
Mallard Duck	50	1700ppm	slightly toxic
Japanese Quail	50	>5,000 ppm	practically nontoxic
Ring-necked Pheasant	50	3438 ppm	slightly toxic

The USFWS extrapolation suggests that 100 precent active ingredient material would be considered "slightly toxic" to the mallard and ring-necked pheasant and "practically nontoxic" to the Japanese quail. (MRID 00034769).

Avian Reproductive Toxicity

Avian reproduction studies are required when birds may be exposed repeatedly or continuously through persistence, bioaccumulation, or multiple applications, or if mammalian reproduction tests indicate reproductive hazard. Because linuron is persistent and can be applied more than one time during a season these studies were required.

Avian Reproductive Toxicity		
Species	**% Test Material**	**Results**
Mallard Duck	98.4	NOEL = 100 ppm LOEL = 300ppm(1)
Bobwhite Quail	98.4	NOEL = 100 ppm LOEL = 300 ppm(2)

(1) Treatment-related effects in adult body weight, feed consumption, egg production, and eggshell thickness.
(2) Treatment-related effects in egg production, hatchability, and offspring survival.

There are sufficient data to characterize the effects of linuron on avian reproduction. The No Observable Effects Level for the mallard duck is 100 ppm and the Lowest Observable Effects Level is 300 ppm. (MRID 42541802)

The No Observable Effects Level for the bobwhite quail is 100 ppm and the Lowest Observable Effects Level is 300 ppm. (MRID 42541801)

Toxicity to Mammals

Mammalian Acute Oral Toxicity Findings		
Species	LD_{50}(mg/kg)	Conclusion
Rat	2600	practically nontoxic

The available data indicate that at a lowest acute oral LD50 of 2600 mg/kg, linuron is practically nontoxic to the rat.

Toxicity to Insects

The minimum data required to establish the acute toxicity to honey bees is an acute contact LD_{50} study with the technical material.

Acute Toxicity to Insects			
Species	% Test Material	LD_{50}	Conclusion
Apis mellifera	not reported	>120.86 ug/bee	practically nontoxic

There is sufficient information to characterize linuron as practically non-toxic to bees. (MRID 00018842).

(2) Aquatic Animal Data

Freshwater Fish Toxicity

Acute testing with the TGAI

In order to establish the toxicity of a pesticide to freshwater fish, the minimum data required on the technical grade of the active ingredient are two freshwater fish toxicity studies. One study should use a coldwater species (preferably the rainbow trout), and the other should use a warmwater species (preferably the bluegill sunfish).

Freshwater Fish Acute Oral Toxicity			
Species	**% Test Material (TGAI)**	**LC_{50}**	**Conclusions**
Rainbow trout	96.2	3 ppm	moderately toxic
Bluegill sunfish	96.2	9.6 ppm	moderately toxic

The results of the 96-hour acute toxicity studies indicate that linuron can be characterized as being moderately toxic to both cold and warm water fish. (MRIDs 40445501 and 40354201).

Acute testing with the formulated product

Formulated product testing is specified if there is direct application to an aquatic environment or if EECs are greater than or equal to the LC50. Linuron is registered for use on rights-of-way (ROWs) which can result in a direct application to aquatic environments.

Freshwater Fish Acute Testing with the Formulated Product			
Species	**% A.I.**	**Result LC50**	**Conclusions**
Rainbow trout	Lorox 50 (WP)	16.4 ppm	slightly toxic
Bluegill sunfish	Lorox 50 (WP)	16.2 ppm	slightly toxic
Bluegill sunfish	Lorox 50 (DF)	9.2 ppm	moderately toxic

The results of the 96-hour EC50 studies indicate that Lorox 50 WP (wettable powder) is slightly toxic to rainbow trout and bluegill sunfish. Lorox 50 DF (dry flowable) is considered moderately toxic to bluegill sunfish. (MRIDs 00018165, 00018165, and 00018198).

Chronic Test-Early Life Stage

The fish early life stage is required to support reregistration of a chemical if exposure is expected to be continuous, recurrent or persistent, and multiple applications of the chemical may occur. The minimum data required to establish chronic toxicity of linuron to fish is the early life stage toxicity test based on survival of fish embryos and post-hatch larvae. There are no data available.

Freshwater Invertebrate Toxicity

Acute testing with the TGAI

The minimum testing required to assess the hazard of a pesticide is a freshwater aquatic invertebrate toxicity test, preferably using first instar *Daphnia magna* or early instar amphipods, stoneflies, mayflies, or midges.

Freshwater Invertebrate Toxicity Findings			
Species	**% Test Material (TGAI)**	**EC_{50}**	**Conclusion**
Daphnia magna	94.4	0.12 ppm	highly toxic

There is sufficient information to characterize linuron as highly toxic to aquatic invertebrates. (MRID 00142932).

Chronic Test-life cycle

The *Daphnia* Life Cycle is required to support reregistration if the chemical's presence in water is likely to be continuous, recurrent or persistent, and multiple applications of the chemical may occur. The minimum data required to establish chronic toxicity of linuron to invertebrates is the Daphnia life cycle test based on reproduction, growth and survival.

Chronic Test-Life Cycle		
Species	**% A.I.**	**Results**
Daphnia magna	98.4	MATC > 0.13 < 0.24 ppm

Based on the data submitted, the MATC is greater than 0.13 and less than 0.24 ppm. The Agency has chronic invertebrate data that appear inconsistent with acute data: chronic effects were not seen until levels higher than those causing acute effects. Also, invertebrates were more sensitive than fish in acute tests, but appear considerably less sensitive in the chronic test. Therefore, additional testing is required based on inconsistent results with the acute toxicity data. (MRID 42153401)

Estuarine/Marine Toxicity

Acute testing with the TGAI

Acute toxicity testing with estuarine and marine organisms is required when an end-use product is intended for direct application to the marine/estuarine environment or is expected to reach this environment in significant concentrations.

The requirements under this category include a 96-hour LC_{50} for an estuarine fish, a 96-hour LC_{50} for shrimp, and either a 48-hour embryo-larvae study or a 96-hour shell deposition study with oysters.

Estuarine/Marine Acute Toxicity Findings			
Species	**% Test Material (TGAI)**	**LC_{50}**	**Conclusions**
Sheepshead minnow	98.4	0.89 ppm	highly toxic
Eastern oyster	98.4	5.4 ppm	moderately toxic
Mysid shrimp	98.4	3.3 ppm	moderately toxic

There is sufficient information to characterize the TGAI of linuron as highly toxic to the sheepshead minnow and moderately toxic to the eastern oyster and mysid shrimp. (MRIDs 42061801, 42061802, and 42061803).

Acute testing with the formulated product

Marine and estuarine testing using the formulated products is required due to the ROW (Rights-of-way) use. ROWs could cross virtually any habitat, including marine aquatic habitat such as salt marshes. Data are not currently available. Testing is required with at least the most sensitive species in acute testing (sheepshead minnow) using the DF (dry flowable) formulation. A DF formulation was found to be more toxic than expected based on active ingredient testing. Because of the ROW (right-of-way) use, there could be direct exposure to the aquatic environment by the formulated product. TEP testing will enable the Agency to assess the risk of specific formulation(s) actually used on ROWs. Additional species and/or formulations may also be required.

Chronic effects

Chronic marine and estuarine testing are indicated based on the same criteria as freshwater species. In the case of linuron, these indications include (1) LC50 value less than 1 mg/l, (2) EEC 0.01 LC50 and (3) aquatic half-life of less than 4 days. Sheepshead minnow and mysid shrimp should be tested.

(3) Non-Target Plants Data

Toxicity to Terrestrial Plants

Data requirements for determining toxicity to terrestrial plants (Tier 2) remain outstanding. These data are required for linuron because it is an herbicide registered for use on terrestrial food and nonfood sites and the vapor pressure is 1.0 X 10^{-5}. Labeling indicate that aerial application can be used for soybeans, as well as ground boom spray for other crops. However, a plant risk assessment for linuron cannot be performed without the phytotoxicity data.

Toxicity to Aquatic Plants

Only one of the five required species for testing for toxicity to aquatic plants has been submitted. Testing for *Lemna gibba*, *Skeletonema costatum*, *Anabaena flos-aquae*, and a freshwater diatom remain outstanding. These data are required for linuron as it is an herbicide registered for use on terrestrial food/nonfood sites, has a vapor pressure $ 1.0 X 10^{-5} mm Hg, and a water solubility greater than 10 ppm. These data are required to conduct the plant risk assessment for linuron.

Aquatic Plant Toxicity		
Species	**% A.I.**	**EC_{50}**
Selenastrum capricornutum	**100**	**5-day = 0.067 mg ai/l**

With a 5-day exposure of 0.067 mg active ingredient per liter of linuron, *S. capricornutum* can be expected to sustain a 50% reduction in density or number of cells. (MRID 42086801).

Appendix A. Ecological Effects Data for Captan

Table A.1. Freshwater Fish Data - Captan Parent

Species	% A.I.	LC50, µg/L (confidence interval)	Measured/ Nominal Flow-through /static	Toxicity Classification	MRID (study year)	Class-ification
Brook Trout	88.4	34 (22 - 52)	Measured, Flow-through 8-day test	Very Highly toxic	00057846 (Hermanutz, 1973)	Supplemental
Fathead Minnow	88.4	65 (59 – 72)	Measured, Flow-through 6-day test	Very Highly toxic	00057846 (Hermanutz, 1973)	Supplemental
Bluegill sunfish	88.4	72 (47 – 111)	Measured, Flow-through 5-day	Very Highly toxic	00057846 (Hermanutz, 1973)	Supplemental
Coho Salmon	90	137 (117-160)	Static	Highly Toxic	(Johnson & Finley, 1980)* 40098001	Supplemental
Coho Salmon	90	56.5 (52.3-61)	Flow-through	Very Highly toxic	(Johnson & Finley, 1980)* 40098001	Supplemental
Chinook Salmon	90	120 (103-140)	Static	Highly Toxic	40098001	Supplemental
Cutthroat trout	90	56.4 (42.2-75.4)	Static	Very Highly toxic	(Johnson & Finley, 1980)* 40098001	Supplemental
Rainbow Trout	90	73.2 (66.6-80.4)	Static	Very Highly toxic	(Johnson & Finley, 1980)* 40098001	Supplemental
Brown Trout	90	80 (63.8– 100)	Static	Very Highly toxic	(Johnson & Finley, 1980)* 40098001	Supplemental
Brown Trout	90	26.2 (21.9-31.3)	Flow-through	Very Highly toxic	40098001	Supplemental
Lake Trout	90	49 (40.1-59.9)	Static	Very Highly toxic	(Johnson & Finley, 1980)* 40098001	Supplemental

Lake Trout	90	63.2 (49.6-80.5)	Static	Very Highly toxic	40098001	Supplemental
Lake Trout	90	51 (39.2-66.2)	Flow-through	Very Highly toxic	40098001	Supplemental
Fathead Minnow	90	200 (168-238)	Static	Highly Toxic	(Johnson & Finley, 1980)* 40098001	Supplemental
Fathead Minnow	90	134 (100-178)	Flow-through	Highly Toxic	40098001	Supplemental
Channel catfish		77.5 (70.5-85.2)	Static	Very Highly toxic	(Johnson & Finley, 1980)* 40098001	Supplemental
Bluegill sunfish	90	141 (119 – 167)	Static	Highly Toxic	(Johnson & Finley, 1980)* 40098001	Supplemental
Yellow Perch	90	120 (97.3-147)	Flow-through	Highly Toxic	(Johnson & Finley, 1980)* 40098001	Supplemental
Bluegill sunfish	90	310 (280 – 340) Slope = 1.17	Static	Highly Toxic	GS0120-042 (1979)	Supplemental
Harlequin Fish (Rasbora heteromorpha)	89	300	Static	Highly Toxic	00034713 Tooby et al. 1975	Supplemental (26 hr test, daily change of test water, no mortality data, test species)

* In Mayer and Ellersieck (MRID 40098001)
* Original source: Johnson, W. W., and M. T. Finley. 1980. Handbook of acute toxicity of chemicals to fish and aquatic invertebates. U.S.F.W.S., Resource. Pub. 137.98 pp.

Table A.2 Freshwater Fish Data - Captan Degradates						
Species	**% A.I.**	**96-hr LC_{50}, µg/L (confidence interval)**	**Measured/ nominal Flow-through /static**	**Toxicity Classification**	**MRID (study year)**	**Satisfies Guideline/ Comments**
Rainbow Trout	96% THPI	> 120,000	Measured, Static Renewal	Practically non-toxic	43869806	Acceptable
Rainbow Trout	96% THPAm	> 126,000	Measured, Static	Practically non-toxic	44738801	Supplemental (used 10 fish, but 30 are required for limit test

A.3. Chronic Toxicity to Freshwater Fish			
Species	**Toxicity**	**Source**	**Effects**
Fathead minnow	NOAEC = 16.5 µg/L LOAEC = 39.5 µg/L	MRID 00057846	Acceptable Reductions in adult and larval survival, growth and overall larval-juvenile development, survival of the juvenile species, a reduction in eggs laid, and an inability for juveniles to reproduce

Table A.4. Aquatic Invertebrate Captan Data						
Species	**% A.I.**	**Toxicity**	**Measured/ nominal Flow-through /static**	**Toxicity Classification**	**MRID (study year)**	**Classification**
Daphnia magna	90	48-hr LC_{50} = 8400 (7060-9960) µg/L Slope= 1.187	Static	Moderately Toxic	GS0120041	Acceptable
Daphnia magna	96% THPI	48-hr LC_{50} >113,000 µg/L	Static	Practically non-toxic	438698-08	Acceptable
Daphnia magna	technical	NOAEC = 560 µg/L LOAEC = 1000 µg/L	Static	--	441488-01	Supplemental (based on nominal concentrations)

Table A.5. Aquatic Plant Captan Data					
Species	**% A.I.**	**EC_{50}, µg/L (confidence interval)**	**Toxicity Classification**	**MRID (study year)**	**Classification**
Scenedesmus subspicatus Green algae (96-hr)	92.7	320	Highly toxic	00137688	Supplemental (based on nominal concentrations)
Selenastrum capricornutum Green Algae (96 hr)	90	1770 (1550-2030)	Moderately Toxic	438698-09	Acceptable
Anabaena flos-aquae Freshwater Algae (96 hours)	99.8	1200 (830-1600)	Moderately Toxic	448065-01	Acceptable
Lemna gibba Duckweed (7 days)	99.8	> 12,700	Slightly Toxic	448065-03	Acceptable
Selenastrum capricornutum Green Algae (72 hours)	96% THPI	> 180,000	Practically non-toxic	438698-10	Supplemental (short test duration)

Note: *Skeletonema costatum* (marine diatom), *Isochrysis galbana*, *Pavlova gyrans*, *Pavlova lutheria*, and *Dunaliella tertiolecta* (marine algae) are marine species and not applicable to RLF assessment (MRID 40228401).

U.S. EPA. 1986. Acute Toxicity Handbook of Chemicals to Estuarine Organisms., *U.S.EPA*, *Gulf Breeze,FL* (US EPA MRID 40228401).

Table A. 6. Captan Bird Data						
Species	**LD_{50} mg/kg bw**	**Acute Oral Toxicity (MRID)**	**LC_{50} (mg/kg diet)**	**Subacute Dietary Toxicity (MRID)**	**NOAEC mg/kg diet MRID**	**Affected Endpoints**
Northern bobwhite Quail *Colinus virginianus*	> 2150	00151236 Beavers, 1978	> 2400	GS0120 Fiche/Master ID 00022923 Hill, 1975	1000 (00098295 Fink, 1980)	No affected endpoints
Mallard Duck *Anas platyrhynchos*	> 2000	GS999-001 Hudson, 1984	>5000	GS0120 Fiche/Master ID 00022923 Hill, 1975	1000 (00098296 Fink, 1980)	No affected endpoints

Table A.7. Mammalian Captan Data					
Species	**Test Type**	**LC_{50} (mg/kg diet)**	**NOAEL/ LOAEL (mg/kg diet)**	**Citation (MRID)**	**Comments**
Rat	Acute Oral	> 5000	--	00265785 (1984)	Two males died. One death occurred on day 1 and one on day 12. One female died on day 4. The deaths were treatment related according to necropsy.
Rat	Acute Oral	Male: 5400 (4290-6800) Female: 5500 (4370-6930)	--	ACC# 241805	--
Rat	Acute Oral	9000	--	00054789 (1949)	--
Rat	One generation	--	> 500/ >500	00120315	
Rat	Three Generation	--	250 / 500	00125293 246101 241001	decreases in the mean litter weights of pups and severe sexual organ atrophy in adults and pups, signs of severe changes in liver weights in the adult males as well as abdominal and intestinal atrophy. In females, there were signs of stomach atrophy and esophageal atrophy

Table A. 8. Terrestrial Invertebrate Data				
Species	**Test Type**	**LD_{50} (µg/kg bee)**	**Citation (MRID)**	**Comments**
Apis mellifera Honeybee	Acute Contact	> 10	Fiche/Master ID 05001991 Stevenson, 1978	
Apis mellifera Honeybee	Acute Contact	> 215	Fiche/Master ID 00080871 Atkins, 1972	
Osmia lignaria Bee	72-hr Acute Oral	46.26 (32.75 – 77.44)	Ecotox # 87252 Ladurner et al, 2005	Captan 50WP 48.9% a.i.
	72-hr Acute Contact	269.68 (151.32 – 2841.84)		

Summary of Amphibian Larvae Study

Chemical Name: Captan
PC Code: 081301

ECOTOX Record Number and Citation: 90515. Mouchet, F., Gauthier, L., Mailhes, C., Ferrier, V, and Devaux, A. 2006. Comparative evaluation of genotoxicity of captan in amphibian larvae (*Xenopus laevis* and *Pleurodeles waltl*) using the comet assay and the micronucleus test. Environmental Toxicology 21(3): 264-277.

Purpose of Review: Litigation (California Red-Legged Frog)

Date of Review: October 2007

Brief Summary of Study Findings:
The toxic and genotoxic potentials of captan were evaluated with the micronucleus test (MNT) and the comet assay (CA).

Adult pairs of *Xenopus* and *Pleurodeles* were mated. Viable eggs were maintained until they reached a development stage appropriate for testing (3 weeks for *Xenopus* and 6 weeks for *Pleurodeles*). Experimental conditions generally followed the French Standard AFNOR (French National Organization for Quality Regulation) NF T90-325. Amphibians were exposed to either reconstituted water (RW) to which nutritive salts were added or mineral water (MW). Nominal captan concentrations were: 2000, 1000, 500, 250, 125, 65.5, 31.25, and 15.60 µg/L. Actual concentrations in water were not measured. Negative controls were either RW or MW. Positive controls were benzo[a]pyrene (B[a]P, [50-32-8], purity: 96.0%, Sigma France) at 0.125 mg/L for MNT and methyl methanesulfonate (MMs, [66-27-3], purity: 99%, Sigma France) at 1.56 mg/L for CA. Captan was dissolved in DMSO at a final concentration of 0.05% before addition to water. Media in all flasks was renewed daily.

Acute toxicity was examined for 12 days by visual inspection (death, abnormal behavior, reduced size, diminished food intake. No signs of toxicity or mortality were observed in any of the negative controls (personal communication with F. Mouchet, October 2007).

Captan flasks containing RW became turbid between 12-24 hours after renewal. The study author hypothesized that this turbidity was probably caused by amphibian residues/excretion or by the suspended captan or the degradation products that may interact with mineral ions, which make up a larger proportion of RW than MW.

Results of acute toxicity to *Xenopus* and *Pleurodeles* larvae exposed to captan (µg/L) in mineral water (MW) and reconstituted water (RW) for 12 days

Conc(µg/L)		2000	1000	500	250	125	62.5	31.25	15.60
Xenopus	MW	++(100%)	++(100%)	++(100%)	++(100%)	++(55%)	-	-	-
	RW	++(100%)	++(100%)	++(100%)	+	-	-	-	-
Pleurodeles	MW	++(100%)	++(100%)	++(75%)	++(45%)	-	-	-	-
	RW	++(100%)	++(100%)	++(50%)	+	-	-	-	-

%: percent dead (of 20 larvae); - No toxicity of larvae; + weak toxicity; ++ severe toxicity.

Genotoxicity was only assayed in MW at those concentrations where there was no acute toxicity. At 12 days for MNT and 1, 2, 4, 8, or 12 days for CA a blood sample was taken. Genotoxicity was assessed to the highest concentration that did not lead to signs of acute toxicity of the exposed larvae.

The results of the *Xenopus* MNT showed that a captan concentration of 62.50 µg/L induced a significant genotoxic response. The lowest concentrations (15.60 and 31.25 µg/L) were not genotoxic to *Xenopus* larvae. The results of the *Pleurodeles* MNT showed no genotoxicity regardless of the concentration of captan tested: 125, 62.50, 31.25, or 15.60 µg/L.

Results of the *Xenopus* CA showed that captan had genotoxic effects at all concentrations tested (15.60 µg/L after 8 and 12 days; 31.25 and 62.5 µg/L after 1, 2, 4, and 8 days; and 125 µg/L after 1, 2, and 4 days). The results of the Pleurodeles CA showed genotoxic effects at captan concentrations of 62.5 and 125 µg/L after 1 and 2 days of exposure, whatever the parameter, except with tail DNA after 2 days of exposure to 62.5 µg/L.

LC_{50} and slope (when possible) was estimated by the reviewer using TOXANAL software.

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RESULTS CALCULATED USING THE PROBIT METHOD
ITERATIONS    G              H            GOODNESS OF FIT PROBABILITY
 16            .1202498      1             .8025137
SLOPE    =     4.58749
95 PERCENT CONFIDENCE LIMITS  = 2.996684     AND    6.178296
LC50 =         311.0651
95 PERCENT CONFIDENCE LIMITS = 253.3895 AND  381.8606
```

		LD50, (µg/L) (confid int)	**Method**
Xenopus	MW	119.4 (62.5, 250)	Binomial
	RW	353.6 (250, 500)	Binomial
Pleurodeles	MW	311.1 (253.4, 381.9)	Probit
	RW	500 (250, 1000)	binomial

Description of Use in Document: Qualitative
Rationale for Use: This is the only known study evaluating the toxicity of captan to amphibians.

Limitations of Study:
1. Detailed raw data not available.
2. LC_{50} and slope (using probit model) not estimable for 3 survival curves as there was only one concentration with partial mortality.
3. Captan concentrations not measured.
4. Turbidity in RW flasks containing captan not definitively explained.

Reviewers: Christine Hartless, Wildlife Biologist (ERB1)

Summary of Wheat Study

Chemical Name: Captan
PC Code: 081301

ECOTOX Record Number and Citation: 91168. Mantecon, J. D. (1989). Persistence of Systemic and Non-Systemic Fungicides in the Control of Seedling Blight of Wheat (Fusarium graminearum). *Tests Agrochem.Cultiv.* 10: 76-77.

Purpose of Review: Litigation (California Red-Legged Frog)

Date of Review: September 2007

Brief Summary of Study Findings:
This study was conducted in a greenhouse at the Experiment Station INTA Balcarce, Buenos Aires Province, Argentina. Highly infected seeds of a durum wheat (Triticum durum Desf.) cv. Buck Patacon were sown in an artificially infested soil. The treatments were arranged in a randomized complete block design with three replications of 100 seeds each. Greenhouse temperatures averaged 20 ± 5 C. Fungicides were applied one day before sowing the seed. Four fungicides (including captan) were evaluated. Captan was applied at a rate of 120 g ai/kg-seed (0.26 lbs ai/cwt) using a wettable powder product from Stauffer Chemicals that was 83% ai.

The measured response variable was number of seedlings present after 7, 14, 21, and 28 days. Data were analyzed for each day by ANOVA and Tukey's test. Only the results from captan and the control are included in the table below.

	Time (days) after sowing			
	7	**14**	**21**	**28**
Captan	67	89	79	69
Control	39	43	28	26

At each time point, there was a statistically significant difference between the mean number of seedlings in the captan and the control groups.

Description of Use in Document: Qualitative

Rationale for Use: One of several seed treatment studies used in lieu of seedling emergence studies.

Limitations of Study:
1. Detailed raw data not available.
2. Only one treatment level evaluated (EC_{25} cannot be determined).
3. Exposure is by seed treatment, rather than by spray on top of soil surface.
4. Watering regime not available.

Reviewers: Christine Hartless, Wildlife Biologist (ERB1)

Summary of Sorghum Study # 91004

Chemical Name: Captan
PC Code: 081301

ECOTOX Record Number and Citation: 91004. Mc Laren, N. W. and Rijkenberg, F. H. J. (1989). Efficacy of Fungicide Seed Dressings in the Control of Pre- and Post-Emergence Damping-Off and Seedling Blight of Sorghum. *S.Afr.J.Plant Soil* 6 : 167-170.

Purpose of Review : Litigation (California Red-Legged Frog)

Date of Review: September 2007

Brief Summary of Study Findings:

This field study was conducted in Potchefstroom, Republic of South Africa in a field in which seedling diseases had been previously recorded. The seed cultivars DC34, DC99, NK283, and PNR8311 were used. A randomized split plot design with five replications was used. Cultivar was the whole plot factor and seed treatment was the sub-plot factor. Captan was applied as a seed dressing at a rate of 135 mg ai/kg-seed (0.30 lbs ai/cwt). Each subplot consisted of three rows, 11 m in length, spaced 1 m apart. After application of 2:3:2 fertilizer (300 kg/ha) seeds were planted to a depth of ± 5 cm and spaced 15 cm apart. A total of 17 fungicide treatments was used, only captan and control results are reported here.

To facilitate recovery of seed from the soil for determination of germination and pre-emergence damping-off, samples of 20 seeds were planted in cocoons, 30 cm in length, folded from single ply cheesecloth. Two cocoons with the relevant seed treatment and cultivar were randomly placed in each subplot row. Cocoons were recovered after 7 days and the percentage germination and per-emergence damping-off were assessed. Pre-emergence damping-off was measured as the percentage germinated seeds in which rotting was so severe that growth had ceased prior to emergence of seedlings from soil.

Twenty-one days after planting the percentage post-emergence damping-off (as a percentage of emerged seedlings) was determined in each sub-plot. Thereafter, 25 seedlings were removed from each sub-plot row and washed to remover adhering soil particles. Visual assessments of the percentage mesocotyl and primary root discoloration were made. Seedlings were also dried and weighed.

For each of the measured parameters, there were no statistically significant differences between the captan and the control group.

	Pre-emergence damping off (%)	Mesocotyl discoloration (%)	Root discoloration (%)	Post-emergence damping off (%)	Seedling mass (g)
Captan	19.4	58.2	19.8	10.5	3.8
Control	18.0	64.9	21.4	11.3	3.5
Least Significant Difference LSD (0.05)	8.2	16.4	7.5	2.3	0.5

Description of Use in Document: Qualitative

Rationale for Use: One of several seed treatment studies used in lieu of seedling emergence studies.

Limitations of Study:
1. Detailed raw data not available.
2. Only one treatment level evaluated (EC_{25} cannot be determined).
3. Exposure is by seed treatment, rather than by spray on top of soil surface.
4. Watering regime not available.

Reviewers: Christine Hartless, Wildlife Biologist (ERB1)

Summary of Sorghum Study # 90836

Chemical Name: Captan
PC Code: 081301

ECOTOX Record Number and Citation: 90836. Davis, M. A. and Bockus, W. W. (2001). Evidence for a *Pythium* sp. as a Chronic Yield Reducer in a Continuous Grain Sorghum Field. *Plant Dis.* 85: 780-784.

Purpose of Review (DP Barcode or Litigation): Litigation (California Red-Legged Frog)

Date of Review: September 2007

Brief Summary of Study Findings:

Field experiments
Two field experiments were conducted (planting dates of 11 May 1995 and 7 June 1995) in which there were three treatment groups (control, captan, and metalaxyl). A high vigor commercial hybrid seed (germination rate > 90%, Cargill 618Y) was planted. Prior to planting, seed was treated. A glass canning jar (1 liter) was "seasoned" by adding 2.5 ml water, the correct amount of chemical, and 100 g of seed. Jar was shaken until all liquid was absorbed by seed. This seed was discarded; procedure was repeated to produce treated seed for experiments. Treated seed was placed in paper bags to dry before sowing. Captan 400D at 3.0 fl oz/cwt (73g ai/kg-seed or 0.16 lbs ai/cwt) was used. Stand counts (plants/m^2) were taken on 12 and 23 June, vigor ratings (scale of 1 to 5) were taken on 3 July (boot and growing point differentiation growth stages), and grain yields (kg/acre) were measured on 17 and 20 October. There was a statistically significant increase or no difference in the captan treated seed responses relative to the control seeds in all measured parameters for both experiments.

		11 May 1995	7 June 1995
Stand			
	Control	3.8 c	6.0 b
	captan	7.3 a	7.9 a
	Metalaxyl	6.3 b	7.7 a
Vigor			
	Control	2.3 b	4.2 a
	captan	2.9 a	4.0 a
	Metalaxyl	3.1 a	3.6 a
Grain yield			
	Control	2592 b	5651 b
	captan	2754 b	6302 ab
	Metalaxyl	3947 a	6742 a

Values within a column and parameter followed by a common letter are not significantly different according to analysis of variance followed by least significant difference (P = 0.05).

Greenhouse experiment

Seed (Cargill 618Y) was treated or not treated with captan or metalaxyl at 0.16 lbs ai/cwt using the same method as described above. The experiment was arranged in a randomized complete block design using 10 plastic tubes 2.5 cm in diameter by 15 cm long. Each treatment had four replications. Soil was collected from the field experiment site above and left nontreatd or autoclaved at 121C for 2 hrs and placed I the tubes. One seed was sown per tube and plants were maintained in a greenhouse at 15-27 C Plant counts (out of 10 seeds planted) and shoot fresh weight per plant were recorded after 28 days. . There was a statistically significant increase or no difference in the captan treated seed responses relative to the control seeds in all measured parameters for either naturally infested soil or autoclaved soil.

		Experiment 1		Experiment 2	
Seed trt	**Soil trt**	**Stand**	**Fresh shoot wt**	**Stand**	**Fresh shoot wt**
Nontreated	Autoclaved	5.8 a	0.83 a	7.8 a	1.03 a
Captan	Autoclaved	7.5 a	0.85 a	8.0 a	1.18 a
Metalaxyl	Autoclaved	7.0 a	0.87 a	8.0 a	1.15 a
Nontreated	Nonautoclaved	3.0 b	0.57 b	5.5 b	0.52 b
Captan	Nonautoclaved	7.0 a	0.60 b	7.8 a	0.69 b
Metalaxyl	Nonautoclaved	7.8 a	0.90 a	8.0 a	1.07 a

Values within a column followed by a common letter are not significantly different according to analysis of variance followed by least significant difference (P = 0.05).

Description of Use in Document: Qualitative

Rationale for Use: One of several seed treatment studies used in lieu of seedling emergence studies.

Limitations of Study:
1. Detailed raw data not available.
2. Only one treatment level evaluated (EC_{25} cannot be determined).
3. Exposure is by seed treatment, rather than by spray on top of soil surface.

Reviewers: Christine Hartless, Wildlife Biologist (ERB1)

Summary of Lupine Study

Chemical Name: Captan
PC Code: 081301

ECOTOX Record Number and Citation: 91007. Fahim, M. M., Osman, A. R., Sahab, A. F., and El-Kader, M. M. A. (1983). Agricultural Practices and Fungicide Treatments for the Control of Fusarium Wilt of Lupine. *Egypt.J.Phytopathol.* 15: 35-46.

Purpose of Review (DP Barcode or Litigation): Litigation (California Red-Legged Frog)

Date of Review: September 2007

Brief Summary of Study Findings:

In vivo experiments were carried out in unsterilized 25-cm diameter clay pots containing clay sand mixture (1:1, w/w), referred to as loamy soil. The seeds were treated with the tested fungicides by shaking them in polyethylene bags until an even dressing was observed. Captan was applied at 0.50 lbs ai/cwt as the enduse product Orthocide (75% captan, recommended rate of 3 g Orthocide/kg-seed). Each treatment had five replicates. A total of eight fungicides and the control were evaluated in the experiment; only the captan results are summarized below.

Soil infestation was conducted by mixing cultures of *Fusarium oxysporum* with the soil at a rate of 5%, w/w. The inoculum was a 2-week-old growth of a virulent isolate, obtained from Alquam, Giza Governorate, on barley/sand (3:1, w/w) medium at 30 C. visual observations were made during the growth season. Macroscopic checks were also carried out at maturity. Seeds were air-dried for several days.

At the end of growing season, average weight of 100 seeds in the treated group was the same or greater than in the control. Percent occurrence of diseased plants was less in treated group than in control group.

		Diseased plants, %			**Avg wt of 100 seeds, g.**
		Pre-emergence	**Post-emergence**	**total**	
Captan	Infested	0	10	10	19.0
	Uninfested	0	7.5	8	20.2
Control	Infested	12.5	68.9	73	14.7
	Uninfested	7.5	19.1	25	16.7
Least Significant Difference (LSD) at P=0.05					
Main effect of fungicide		3.5	6.2	-	3.3
Main effect of infestation		1.6	2.9	-	1.5
Interaction (fungicide x infestation)		4.9	8.6	-	4.7

Description of Use in Document: Qualitative

Rationale for Use: One of several seed treatment studies used in lieu of seedling emergence studies.

Limitations of Study:
1. Detailed raw data not available.
2. Only one treatment level evaluated (EC_{25} cannot be determined).
3. Exposure is by seed treatment, rather than by spray on top of soil surface.
4. Rainfall/watering regime not available.

Reviewers: Christine Hartless, Wildlife Biologist (ERB1)

Summary of Blueberry Study

Chemical Name: Captan
PC Code: 081301

ECOTOX Record Number and Citation: 63909. Polavarapu, S. (2000). Evaluation of Phytotoxicity of Diazinon and Captan Formulations on Highbush Blueberries. *Horttechnology* 10: 308-314.

Purpose of Review (DP Barcode or Litigation): Litigation (California Red-Legged Frog)

Date of Review: September 2007

Brief Summary of Study Findings:

Experiments were conducted during the 1997 and 1998 growing seasons at Rutgers University Blueberry and Cranberry Research and Extension Center, Chatsworth, NJ, on highbush blueberries planted in 1994. Bushes were 4-5 yrs old, approx 5 ft tall, and spaced 9 x 4 ft apart on light sandy organic matter soil wth pH of 4.5. Two formulations of diazinon (Diazinon AG600 and Diazanon 50W) and of captan (Captec 4L and Captan 80WP) as well as an adjuvant, LI-700 were evaluated. Results pertaining to the adjuvant will not be reported here. All experiments described below had a negative control group. Application rates (author stated maximum labeled rates were used) are below:

formulation	Rate/acre	lbs ai/acre
Diazinon AG600	22.5 fl oz	NA
Diazanon 50W	2 lb	NA
Captec 4L	3.12 lb	2.43 lbs ai/acre
Captan 80WP	2.5 qt	2.5 lbs ai/acre

NA – not applicable, reviewer did not calculate as only captan is under review in this summary.

Treatments were arranged in a randomized complete block design. Treatments within a block were separated by at least 4 bushes and blocks were arranged 50-133 ft apart.

Pesticides were applied with a CO_2 pressurized backpack sprayer equipped with a hollowcone nozzle calibrated to deliver 30 gal/acre. At each evaluation, samples of foliage and fruit were collected in polyethylene bags and transported to lab for phytotoxicity evaluations. A fruit or foliage cluster was determined to have phytotoxicity even if only one fruit or leaf was injured.

Phytotoxicity injury occurred within 24 to 36 hrs after application of pesticides. Phytotoxicity on berries ranged from deep purple blotches to circular depressions, especially where residues accumulated. In the most severe cases, fruit had 2 to 3 mm diameter circular depressions filled with apparent pesticide residue. Phytotoxicity on leaves was typically brownish purple spots on the underside of the leaf surface. The degree of phytotoxicity severity caused by the mixtures of captan and diazinon was much greater than the phytotoxicity when captan or diazinon was applied alone.

Data were analyzed using ANOVA and Duncan's multiple range test (P=0.05). Data were transformed before analysis using square root (number of clusters with phytotoxicity, number of berries, and berry weight) or arcsin (percent phytotoxicity) transformations.

Experiment 1

- Conducted in 1997, treated on 11 June 1997
- 5 single bush reps per treatment, variety Ellicot
- single treatment was Diazinon AG600 and Captec 4L
- 5 fruit and 5 foliage clusters collected from each side of each bush - 10 days after treatment
- the combined treatment had a significantly greater proportion of berries exhibiting phytotoxicity and lighter weight berries; although, the number of berries per 10 clusters was not different than the control.

	Berries with phytotoxicity (%)		Number of berries/10 clusters	Wt of 100 berries (g)
treatment	green	Blue		
Diazinon AG600 + Captec 4L	99.6±0.4 a	97.7±1.7 a	103±6.6 a	108.4±7.2 a
Untreated	0.0±0 b	1.5±1.0 b	99.2±5 a	145.5±11.2 b

For each response variable, treatment means followed by different letters are significantly different at P=0.05.
Listed response is mean ± standard error

Experiment 2

- Conducted in 1997, treated on 12 June 1997
- Three replications, each consisting of 6 bushes in a row, variety Bluecrop
- Treatments were combinations of the 4 listed pesticides.
- First evaluation 7 days after trt, 10 clusters from 3 randomly selected bushes within each replication
- Second evaluation 13 July with 25 fruit clusters per rep (during harvest)
- For all responses, the single pesticide applications were not significantly different from the control. Responses with no significant differences (means not listed in summary, are available in paper) were number of berries per 30 clusters 7 days after treatment, blue berries with phytotoxicity/25 clusters (%) at harvest, all berries with phytotoxicity/25 clusters (%) at harvest, and number of berries/25 clusters at harvest.

	Clusters with phytotoxicity (no/30 clusters) 7 d after treatment		Green berries with phytotoxicity/25 clusters (%) at harvest
	fruit	leaf	
Diazinon AG600	0.0±0 c	0.0±0 d	0.7±0.7 b
Captec 4L	0.0±0 c	0.3±0.3 d	1.9±0.5 ab
Diazinon AG600 + Captec 4L	9.3±1.9 a	22.7±1.2 a	5.2±1.4 a
Diazinon AG600 + Captan 80WP	7.7±1.3 a	16.0±2.0 b	1.1±0.5 b
Diazinon 50W + Captec 4L	3.0±0 b	3.0±1.0 c	0.4±0.4 b
control	0.0 ±0 c	0.0±0 d	0.6±0.6 b

For each response variable, treatment means followed by different letters are significantly different at P=0.05, Duncan's multiple range test.
Listed response is mean ± standard error

Experiment 3

- Conducted in 1997, treatment applied on 25 June 1997
- Three replications, each consisting of three bushes, variety Ellicott
- Treatments were combinations of the 4 listed pesticides.
- 20 fruit and leaf clusters per rep (10 each from two randomly selected bushes) sampled 8 d after treatment.
- In addition to responses reported below, percent phytoxicity/20 clusters was also analyzed, results were similar to the number of clusters (reported below). There were no significant differences in the number of berries per 20 clusters among treatments.
- Relative to control, captan alone or with diazinon resulted in no significant change or an increase in the observed phytotoxicity in fruit and leaves.

	Clusters with phytotoxicity (no/20 clusters) 8 d after treatment	
	fruit	leaf
Diazinon AG600	0.0±0 c	4.0±1.5 c
Diazinon 50W	0.0±0 c	0.3±0.3 d
Captec 4L	0.3±0.3 bc	14.0±1.5 b
Captan 80WP	0.7±0.3 bc	0.3±0.3 d
Diazinon AG600 + Captec 4L	4.0±2.5 a	20.0±0 a
Diazinon AG600 + Captan 80WP	1.0±1.0 bc	15.3±0.9 b
Diazinon 50W + Captec 4L	1.0±0.6 bc	4.7±1.7 c
control	0.0±0 c	0.0±0 e

For each response variable, treatment means followed by different letters are significantly different at P=0.05, Duncan's multiple range test.
Listed response is mean ± standard error

Experiment 4

- Conducted in 1998, treatments applied on 18 May. For some trts, diazinon applied first, followed by captan 8 hrs later.
- 4 reps, each consisted of 6 bushes in a single row, variety Weymouth

- 30 fruit and leaf clusters sampled from each rep 9 days after trt.
- In addition to responses reported below, percent phytoxicity/30 clusters was also analyzed, results were similar to the number of clusters (reported below).
- Relative to control, captan alone or with diazinon resulted in no significant change or an increase in the observed phytotoxicity in fruit and leaves. Applying captan 8 hrs after diazinon did demonstrate a significant reduction in phytotoxicity relative to applying both simultaneously.

	Clusters with phytotoxicity (no/30 clusters) 9 d after treatment	
	fruit	**leaf**
Diazinon AG600	0.0±0 e	0.0±0 c
Diazinon 50W	0.3±0.3 de	0.3±0.3 c
Captec 4L	1.8±0.5 c	8.0±1.5 b
Diazinon AG600 + Captec 4L	25.8±2.5 a	14.5±2.0 a
Diazinon 50W + Captec 4L	13.5±0.6 b	7.2±0.9 b
Diazinon AG600 first + Captec 4L 8 hrs later	2.8±1.1 c	7.0±1.1 b
Diazinon 50W first + Captec 4L 8 hrs later	1.5±0.6 cd	5.0±1.5 b
control	0.0±0 e	0.0±0.0 c

For each response variable, treatment means followed by different letters are significantly different at P=0.05, Duncan's multiple range test.
Listed response is mean ± standard error

Experiment 5

- Conducted in 1998, treatments applied on 26 May. For some trts, chemicals were applied with an 8 h interval between them.
- 4 reps, each consisted of 6 bushes in a single row, variety Bluecrop
- 30 fruit and leaf clusters sampled from each rep 8 days after trt.
- Relative to control, captan alone resulted in no significant change or an increase in the observed phytotoxicity in fruit and leaves. Using an 8 hr interval between pesticide applications (with either captan or diazinon first) resulted in a significant reduction in phytotoxicity relative to applying both simultaneously.

	Phytotoxicity /30 clusters (%) 8 d after treatment	
	fruit	**Leaf**
Captan 80WP	0.2±0.2 b	1.4±0.5 b
Diazinon AG600 + Captan 80WP	18.1±3.1 a	9.0±1.3 a
Captec 4L first + Diazinon AG600 8 hrs later	2.2±1.8 b	3.5±1.1 b
Captan 80WP first + Diazinon AG600 8 hrs later	1.5±1.0 b	1.8±0.8 b
Captan 80WP first + Diazinon 50W 8 hrs later	0.1±0.1 b	2.3±1.3 b
control	0.0±1 b	0.0±0 c

For each response variable, treatment means followed by different letters are significantly different at P=0.05, Duncan's multiple range test.
Listed response is mean ± standard error

Experiment 6

- Evaluated effect of repeated applications of captan and diazinon applied together
- Conducted in 1998, treatments applied on 22 May, 26 June, 29 July.
- Variety Ellicott was used

- Samples collected 5 to 8 days after treatment.
- Only one treatment (Diazinon AG600 + Captec 4L, applied at same time) that caused most severe phytotoxicity plus control were used.
- Statistical analysis indicated a time*treatment interaction – a greater percentage of fruit and leaves showed phytotoxicity after 22 May application (immediately following petal fall) than after the other two application dates.

Description of Use in Document: Qualitative

Rationale for Use: Foliar spray study used in lieu of vegetative vigor studies.

Limitations of Study:
1. Detailed raw data not available.
2. Only one treatment level evaluated (EC_{25} cannot be determined).
4. Watering regime not available.
5. Impact on growth of plants not measured.
6. Plants were established, not young seedlings.

Reviewers: Christine Hartless, Wildlife Biologist (ERB1)

Summary of Bee Study

Chemical Name: Captan
PC Code: 081301

ECOTOX Record Number and Citation: 87252. Ladurner, E., Bosch, J., Kemp, W. P., and Maini, S. (2005). Assessing Delayed and Acute Toxicity of Five Formulated Fungicides to *Osmia lignaria* Say and *Apis mellifera*. *Apidologie* 36: 449-460.

Purpose of Review (DP Barcode or Litigation): Litigation (California Red-Legged Frog)

Date of Review: October 2007

Brief Summary of Study Findings:

Contact and oral toxicity of five formulated pesticides were evaluated in this study. Only the results for captan (Captan 50WP, 49% ai) will be reported here.

In May 2002, wintering *O. lignaria* females, reared at the Bee Biology and Systematics Laboratory, Logan, Utah, were incubated at 25 C until emergence from cocoons. Unfed females were transferred to a screened flight cage to all them to deposit meconium. Females were then starved overnight and exposed to a specific fungicide treatment the next morning, approximately 24 h after emergence. In June 2002, *A. mellifera* foragers of different ages from a healthy, queen-right colony were captured in a clear plastic jar as they left the hive in the morning. All bees were chilled for a maximum of 30 minutes at 4 C prior to treatment.

In the contact toxicity tests, 1 µL of test solution was applied to the dorsal surface of the thorax with a 50 µL-micro syringe. Test solution was prepared by dissolving fungicide in acetone and purified distilled water (50% v/v) to obtain desired concentrations; fresh test solution was used for all tests.

In the oral toxicity tests known amounts of the fungicide were dissolved in a feeding solution (25% v/v sucrose in purified distilled water) to obtain desired concentrations. *O. lignaria* and *A. mellifera* were fed 10 µL of the test solution using the flower method devised by Ladurner et al (2003). The test solution was pipetted into a plastic ampoule and inserted into the calyx of a flower (cherry for *O. lignaria* and morning glory for *A. mellifera*). Flowers and bees were individually housed in holding cages (waxed cardboard cups, 8 cm diameter x 5 cm height) with a wire mesh screen lid. Flowers and bees in holding cages were kept in an incubator (22 C for *O. lignaria* and 25 C for *A. mellifera*) under artificial light (two 15W Cool White fluorescent tubes 15cm above holding cages) for one hour.

For the contact test, control bees were dosed with the mixture of acetone and purified distilled water (50% v/v). For the oral test, control bees were fed the feeding solution (25% v/v sucrose in purified distilled water).

TEST 1
Three sets of ten bees each were evaluated for delayed toxicity in the form of a single dose (122.5 µg ai/bee) for both oral and contact tests. After exposure, each set of 10 bees was transferred to a holding cage (same as described for oral test) with an artificial feeder. The feeder was a 5 mL-LDPE sample vial containing a sucrose solution (25% v/v sucrose in water) with a soaked cigarette filter inserted through the end of the vial. Fresh solution was provided every 24 hrs. Holding cages for *A. mellifera* were also provided with a piece of wax foundation comb. Holding cages were kept in an incubator (*O. lignaria* – temperature=22 C, relative humidity=60-80%, L:D=12:12hr; *A. mellifera* – temperature=25 C, relative humidity=60-80%, L:D-0:24 hr). Survival was recorded every 24 hrs for 7 days.

In oral exposure trials, 97.7% of *A. mellifera* and 88.2% of *A. mellifera* consumed all the test solution in one hour. Control survival was 100% in the *O. lignaria* studies and was 75-80% in the *A. mellifera* studies. Captan resulted minimal mortality for *A. mellifera* and higher mortalty rates for *O. lignaria.*

For *A. mellifera*, survival was not significantly reduced relative to control at the end of 7 days (Wilcoxon test) in either the oral or contact tests. For *O. lignaria*, survival was significantly reduced relative to control at the end of 7 days: in the contact test, survival was approximately 50%; and in the oral test, survival was approximately 35% on day 1 and approximately 0% by day 3.

TEST 2 – Methods of administration and bee maintenance were the same as described above. Only *O. lignaria* bees were used for captan, as there was minimal mortality for *A. mellifera* in the first test.

This test was designed to provide an estimate of an LD50. Five doses were administered; however, the test concentrations were not provided. Probit analysis was used for LD50 estimation.

	24 hr	48 hr	72 hr	7 days
Contact	NA	NA	269.68 (151.32, 2841.84)	95.26 (79.83, 134.59)
oral	NA	100.45 (63.75, 245.23)	46.26 (32.75, 77.44)	10.87 (5.40, 19.28)

Units are in µg ai/bee
NA – not available (LD50 was > than highest dose)
95% confidence interval in parentheses

Reference:
Ladurner, E., Bosch, J., Maini, S., Kemp, W.P. 2003. A method to feed individual bees (Hymenoptera: Apiformes) known amounts of pesticides. Apidologie 34: 597-602.

Description of Use in Document: Quantitative

Rationale for Use: This study provides a definitive toxicity endpoint for bees.

Limitations of Study:
1. Detailed raw data not available.
2. Dose concentrations not provided for second test.

Reviewers: Christine Hartless, Wildlife Biologist (ERB1)

Appendix B. Chlorothalonil Ecological Effects Characterization

This appendix presents additional details on available registrant-submitted and open literature studies available on chlorothalonil and its major degradate of toxicological concern, SDS-3701. Studies submitted to the Agency in support of pesticide registration or re-registration are categorized as either; acceptable, supplemental, or invalid. Acceptable means that all essential information was reported, the data are scientifically valid, and the study was performed according to recommended protocols. Studies in the "acceptable" category fulfill the corresponding data requirement in 40 CFR Part 158 and are appropriate for use in risk assessment. Supplemental studies are also scientifically valid; however, they were either performed under conditions that deviate from recommended guideline protocols or certain data necessary for complete verification are missing. Supplemental studies may be used quantitatively in the risk assessment and can, at the Agency's discretion, fulfill the corresponding data requirement in 40 CFR Part 158. Invalid studies are not scientifically valid, or deviate substantially from recommended protocols such that they are not useful for risk assessment. Invalid studies do not fulfill the corresponding data requirement in 40 CFR Part 158.

With respect to the open literature, studies may be classified as either; qualitative, quantitative, or invalid. The degree to which open literature data are quantitatively or qualitatively characterized is dependent on whether the information is directly correlated with the assessment endpoints (i.e., maintenance of the survival, reproduction, and growth of the California red-legged frog and PCEs of their designated critical habitat identified in the problem formulation). Open literature studies classified as qualitative are not appropriate for quantitative use but are of good quality, address issues of concern to the risk assessment, and, when appropriate, are discussed qualitatively in the risk characterization discussion. Those open literature studies that are classified as quantitative are appropriate for quantitative use in the risk assessment including calculation of RQs. This appendix includes registrant-submitted studies in addition to studies identified in the open literature. In general, effects data in the open literature that are more conservative than the registrant-submitted data or that add to the weight of evidence on the toxicity to under-represented species or taxa are considered for quantitative use. Open literature studies that were either rejected by ECOTOX or that were not considered in this assessment are in Appendix H. Those appendices also include rationale for rejection of studies that did not pass the ECOTOX screen and those that were not evaluated as part of this endangered species assessment. Further detail on the ECOTOX exclusion categories is provided in the Agency's *Guidance of the*

Evaluation Criteria for Ecological Toxicity Data in the Open Literature (U.S. EPA, 2004).

B.1 Toxicity to Birds, Reptiles, and Terrestrial Phase Amphibians

No studies in reptiles or terrestrial phase amphibians have been submitted to the Agency or were located in the open literature. Therefore, birds were used as surrogates for terrestrial phase amphibians. Acute oral, subacute dietary, and chronic reproduction toxicity studies for birds are discussed in Sections B.1.1 through B.1.3. All avian studies described below are registrant-submitted studies. No additional information on the acute, subacute, and/or chronic toxicity of chlorothalonil or SDS-3701 to birds was located in the open literature that suggests greater sensitivity than the registrant submitted data. In addition, no relevant information on the toxicity of chlorothalonil and/or SDS-3701 to reptiles and/or terrestrial-phase amphibians was located in the open literature.

B.1.1 Birds: Acute and Subacute Studies

Chlorothalonil

An acute oral toxicity study using the technical grade of the active ingredient (TGAI) is required to establish the toxicity of chlorothalonil to birds. The preferred test species is either mallard duck (*Anas platyrhynchos*; a waterfowl) or bobwhite quail (*Colinus virginianus*; an upland gamebird). Results of these studies are summarized below in Table B.1-1 and B.1-2. These studies suggest that chlorothalonil is practically non-toxic to birds on an acute basis.

Table B.1-1. Avian Acute Oral Toxicity Findings for Chlorothalonil

Test Species	% a.i.	LD_{50} mg/kg	Citation (MRID)	Toxicity Category	Fulfills Guideline?
Mallard	96%	> 4640	00068753	Practically non-toxic	Yes
Japanese quail	Tech.	> approx. 2000	40964105	Practically non-toxic	Supplemental

Table B.1-2. Avian Subacute Dietary Toxicity Findings for Chlorothalonil					
Test Species	**% a.i.**	**LC_{50} ppm**	**Citation (MRID)**	**Toxicity Category**	**Fulfills Guideline?**
Northern Bobwhite	96%	> 10,000	00030388	Practically non-toxic	Yes
Mallard	93.6%	> 21,500	00039146	Practically non-toxic	Yes
Mallard	96%	> 10,000	00030389	Practically non-toxic	Yes

SDS-3701

Acute avian LD_{50} data for the chlorothalonil degradate, SDS-3701, are summarized in Table B.1-3. These studies show that SDS-3701 is "moderately toxic" on an acute oral basis and "slightly toxic" on a dietary basis to the test birds on an acute basis. Sublethal effects were seen in some birds at the lowest test level with the mallard, including lethargy, depression, lost reaction to stimuli, lost coordination, and wing droop.

In an acute oral study (MRID 00030395), fourteen-day old mallard ducks were dosed with 46, 100, 215, 464, and 1000 mg/kg of technical SDS-3701. No deaths occurred in the control pens. The acute oral LD50 for SDS-3701 was 158 mg/kg, with 95% confidence limits of 125 to 201 mg/kg. The highest dose at which no deaths occurred was 46 mg/kg. All the birds died in the two highest dosing groups.

Table B.1-3. Avian Acute Oral and Subacute Dietary Toxicity Findings for SDS-3701					
Test Species	**% a.i.**	**Results**	**Citation (MRID #)**	**Toxicity Category**	**Fulfills Guideline?**
Mallard	SDS-3701 (87%)	LD_{50} = 158 mg/kg	00030395	Moderately toxic	Yes
Northern Bobwhite	SDS-3701 (87%)	LC_{50} = 1746mg/kg NOEL = 562 mg/kg	00115109	Slightly toxic	Yes
Mallard	SDS-3701 (87%)	LC_{50} = 2000 mg/kg	00115108	Slightly toxic	Yes

B.1.3 Birds: Reproduction Studies

Chlorothalonil

Avian reproduction studies using the TGAI are required because chlorothalonil is persistent (i.e., half-life exceeds 4 days in aerobic soils) and has multiple applications per growing season. The preferred test species are mallard duck and bobwhite quail. Results of these tests are summarized in Table B.1-4.

Table B.1-4. Avian Reproduction Findings of Chlorothalonil Exposure

Test Species	% a.i.	NOEL PPM	LOEL PPM	Endpoints affected	Citation (MRID #)	Fulfills Guideline ?
Bobwhite	Tech.	1000 (reprod.)	5000 (reprod.)	"Overt signs of toxicity and reduced reproduction" cited at 5000 ppm; "overt signs of toxicity, mortalities, and profound effects upon several reproductive parameters related to egg production, hatching success, and survival of hatchlings" cited at 10,000 ppm.	40964104	Yes
Bobwhite	Tech.	153	624	18% reduction in no. of eggs laid per hen.	45710218	Yes
Bobwhite	99.6	50 ppm	Not established. Highest level of 50 ppm did not cause impairment.	None	00041440	Suppleme ntal
Mallard	Tech.	>10,000 (reprod.)	>10,000 (reprod.)	No reproductive effects cited at any test level (1000, 5000, 10,000 ppm)	40964102	Yes
Mallard	99.6	50 ppm	Not established. Highest	None	00041441	Suppleme

Table B.1-4. Avian Reproduction Findings of Chlorothalonil Exposure						
Test Species	**% a.i.**	**NOEL PPM**	**LOEL PPM**	**Endpoints affected**	**Citation (MRID #)**	**Fulfills Guideline ?**
			level of 50 ppm did not cause impairment.			ntal

In the most sensitive avian reproduction study (MRID 45710218) chlorothalonil was administered to bobwhite quail in the diet at nominal concentrations of 0 (negative control), 40, 160, or 640 ppm. Mean-measured concentrations were <1.5 (<LOD, control), 41, 153, and 624 ppm a.i., respectively. A treatment-related reduction in the number of eggs laid/hen and thus in the number of 14-day old survivors/hen were observed at the 624 ppm a.i. level. The number of eggs laid/hen was 62.0 for the control group, and 62.4, 68.9, and 51.0 for the 41, 153, and 624 ppm a.i. test groups, respectively. The number of 14-day old survivors/hen was 37.2 for the control group, and 42.8, 42.2, and 30.4 for the 41, 153, and 624 ppm a.i. test groups, respectively. Although not statistically significant, these findings were considered to be biological significance by the study authors and the study reviewers.

SDS-3701

Avian reproduction studies have also been required for SDS-3701. These studies are summarized in the following table. The most sensitive NOAEC was 50 ppm based on reduction in eggshell thickness. However, the relevance of this endpoint to terrestrial amphibians is questionable. Therefore, the NOAEC of 100 ppm was chosen for use in risk assessment.

Table B.1-5. Avian Reproduction Findings (SDS-3701)

Test Species	% SDS-3701	NOEL PPM	LOEL PPM	Endpoints affected	Citation (MRID)	Fulfills Guideline?
Mallard	99.6	50	100	Reduction in eggshell thickness seen at 100 ppm; at 250 ppm adult body weight, food consumption, and gonad development affected, as well as effects on numbers of eggs laid, embryonic development, eggshell thickness, hatchability, and hatching survival.	40729402	Yes
Bobwhite	99.6	100	250	Reduction in numbers of eggs laid	40729404	Yes

B.2 Toxicity to Mammals

Wild mammal testing is required on a case-by-case basis, depending on the results of lower tier laboratory mammalian studies, intended use patterns, and pertinent environmental fate characterisitics. For this assessment, registrant-submitted reproduction toxicity data obtained from the Agency’s Health Effects Division (HED) was used. Acute and chronic toxicity data for mammals is presented in Sections B.2.1 and B.2.2, respectively.

B.2.1 Mammals, Acute

Chlorothalonil

Acute mammalian toxicity studies for chlorothalonil are summarized in Table B.2-2. The available mammalian data indicate that chlorothalonil is "practically non-toxic" to small mammals on an acute oral basis, based on the rat oral LD_{50}.

Table B.2-1 Mammalian Acute Toxicity Findings--Chlorothalonil					
Test Species	**% a.i.**	**LD50**	**Citation (MRID #)**	**Toxicity Category**	**Fulfills guidelines?**
Rat (small mammal surrogate)	96%	Oral >10,000	MRID 00094941	practically non-toxic	Yes

SDS-3701

Data on the toxicity of SDS-3701 to mammals are tabulated below. These data indicate that the degradate SDS-3701 is more toxic to mammals than the parent chlorothalonil, and is moderately toxic on an acute oral basis.

Table B.2-2 Mammalian Acute Toxicity Findings--SDS-3701				
Test Species	**LD_{50} mg/kg**	**Comments**	**Citation (MRID #)**	**Toxicity Category**
Rat (small mammal surrogate)	242 (females)	The LD50 for males was 422 mg/kg-bw and was 332 for the combined sexes	MRIDs MRID 00047938, 00047939, and 00095783	moderately toxic

B.2.2 Mammals, Reproduction Studies

Chlorothalonil

When available, 2-generation reproduction toxicity studies are used to estimate chronic risk to mammals. In a two-generation study, Sprague Dawley rats were administered chlorothalonil (98%) in the diet at levels of 0, 500, 1500 or 3000 ppm (0, 38, 115 and 234 mg/kg/day). For parental/systemic toxicity, the NOAEL was less than 500 ppm (<38 mg/kg/day). The LOEL was 500 ppm (38 mg/kg/day) based on hyperplasia of renal and forestomach tissues. For offspring toxicity, the NOEL was 1500 ppm (115 mg/kg/day) and the LOEL was 3000 ppm (234 mg/kg/day) based on lower neonatal body weights by day 21 (MRID 41706201).

Test Species	Offspring NOAEL	Offspring LOAEL	Citation (MRID)	Fulfills Guidelines?
Rat (2 generation reproduction)	1500 ppm	3000 ppm decrease in pup weight	41706201C	Yes

SDS-3701

Data on the toxicity of the SDS-3701 degradate to mammalian reproduction are tabulated below. In a 1-generation reproduction study in Sprague-Dawley rats, SDS-3701 was administered at 0, 10, 20, 30, 60, or 120 ppm (approximately 0, 0.5, 1.0, 1.5, 3.0 or 6.0 mg/kg/day). For parental systemic toxicity, the NOEL was 1.5 mg/kg/day and the LOEL was 3.0 mg/kg/day. No ecologically relevant reproductive or offspring toxicity occurred at up to the highest level tested (MRID 00127845).

In a 3-generation reproduction study in Sprague-Dawley rats, SDS-3701 was administered at 0, 10, 60 or 125 ppm (approximately 0, 0.5, 3.0 or 6.25 mg/kg/day). No ecologically relevant reproductive or offspring toxicity occurred at up to the highest level tested, 6.25 mg/kg/day (MRID 00127844).

Test Species	Reproduction NOAEL	Reproduction LOAEL	Citation (MRID #)
Rat (3-generation reproduction)	120 ppm	None	00127844
Rat (1-generation reproduction)	125 ppm	None	00127845

B.2.3 Mammals: Open Literature

Based on a review of the open literature, no additional information on the acute or chronic toxicity of chlorothalonil or SDS-3701 to mammals was located that produced more sensitive endpoints relevant to ecological risk assessment.

B.3 Toxicity to Non-Target Terrestrial Invertebrates

B.3.1 Honey Bee Acute Contact Study

Hhoney bee acute contact LD_{50} study is required if the proposed use will likely result in exposure to honey bees. The available acute contact toxicity findings for chlorothalonil are summarized in Table B.3-1. The available data suggests that chlorothalonil is practically non-toxic to honey bees.

Table B.3-1: Non-target Insect Acute Contact Toxicity Findings

Test Species	% a.i.	Results	Citation (MRID)	Toxicity Category	Fulfills Guideline?
Honey bee	Tech.	at 181 ug/bee, 14.28% mortality	00036935	Practically non-toxic	Yes
Honey bee	Tech.	non-toxic at 181 ug/bee	00077759	Practically non-toxic	Yes

In addition, a number of studies were located in the open literature that evaluated the toxicity of chlorothalonil to terrestrial invertebrates (Table B.3-2). The available data suggest that sensitive terrestrial invertebrates exist, but many of the species tested were not sensitive to chlorothalonil at the levels tested, which typically approximated maximum labeled application rates.

Table B.3-2: Nontarget Invertebrate Acute Contact Toxicity Findings from the Open Literature

Test Species	Material Tested	Results Summary	Citation ECOTOX#	Comment
Aphid endoparasitoid wasp *(Aphidius rhopalosiphi*)	Daconil 500 Flowable (50% a.i.)	NOEL =1250 g a.i./ hectare (1.1 lbs a.i./Acre;	64665 (Jansen, 1999)	Chlorothalonil was considered “slightly harmful” to *A. rhopalosiphi.* Endpoints evaluated included mortality and reproduction
Earthworms and arthropods	Daconil 2787	NOAEL = 12.6 kg (AI)/ha (11.25 lbs a.i./acre)	71484 (Potter et. al. 1990)	Endpoint studied was abundance and biomass
Carabidae	Daconil (54% a.i.)	NOAEL = 8.2 kg a.i./ha (7.3 lbs a.i./Acre)	89639 (Smitley and Rothwell, 2003)	Transient reduction in abundance occurred after the first week of treatment.
		No reduction in		--

Test Species	Material Tested	Results Summary	Citation ECOTOX#	Comment
Onion Thrips *Thrips tobaci*	Bravo 500	abundance occurred at an application rate of 4.5 pints/acre (approx. 2.3 lbs a.i./Acre).	90255 (Al-Dosari et.al 1996)	
Corn earworm; fall armyworm	Bravo 720	Dietary exposure to the fall armyworm and corn earworm at 800 ppm and higher resulted in high mortality rates.	90193 (Lynch, 1996)	--
Mustard beetle (*Phaedon cochleariae*)	Jupital	No significant effects on survival.	90531 (Cherry et. al. 1992)	--
Rove Beetle (*Aleochara bilineata*)	Daconil 500 (50%)	No effects occurred at levels reportedly equivalent to maximum application rate.	63488 (Samsoe-Petersen, 1995)	Effects evaluated included reproduction, egg production and viability.
Predatory mite (*Amblyseius victoriensis*)	Bravo	<10% mortality at up to 10-times the reported maximum field rate	67984 (James et.al. 1995)	--
Aphid Predatory Midge (*Aphidoletes aphidimyza*)	Repulse	Mortality was approximately 10% in larvae after 72 hours at 1100 mg a.i./L.	89884 (Helyer, 1991)	Test level was reportedly equivalent to the maximum labeled application rate.
Aphareta pallipes E. muscae Delia antique D. platura *Coenosio tigrina*	Bravo 6 F	LC50 = 7.83% 100% mortality at 0.0054% No effects to these species were observed at any test level	71029 (Carruthers et. al. 1985)	Cups sprayed with various concentrations of chlorothalonil from 0.0054% to 10.8%.

B.4 Toxicity to Freshwater Animals

B.4.1 Freshwater Fish, Acute Submitted Data

In order to establish the toxicity of a pesticide to freshwater fish, the minimum data required on the technical grade of the active ingredient are two (one cold water and one warm water) freshwater fish toxicity studies. The freshwater fish acute toxicity findings for the technical grade of the active ingredient are summarized in the Table B.4-1. The registrant submitted studies are consistent with each other and indicate that chlorothalonil is very highly toxic to freshwater fish.

Table B.4-1: Freshwater Fish Acute Toxicity Findings					
Test Species	**% a.i.**	**LC_{50} ppb a.i.**	**Citation (MRID #)**	**Toxicity Category**	**Fulfills Guideline?**
Rainbow trout	96	42.3	00056486	very highly toxic	Supplemental
Rainbow Trout	96	18	45710219	very highly toxic	Yes
Bluegill	96	60	00041439	very highly toxic	Yes
Bluegill	99	84	00029410	very highly toxic	Yes
Bluegill	98	51	RIOCHL01 Pitcher (1976)	very highly toxic	Yes
Channel catfish	96	48	00030390	very highly toxic	Yes
Fathead minnow	96	23	00030391	very highly toxic	Yes

The most sensitive LC50 value for registrant-submitted freshwater fish studies was from MRID 45710219. In this study Rainbow Trout (*Oncorhynchus mykiss*) were exposed under flow-through conditions to chlorothalonil at nominal concentrations of 0 (negative and solvent controls), 0.010, 0.018, 0.032, 0.056, and 0.10 ppm. Mean-measured concentrations were ≤0.0004 (≤LOD, solvent control), 0.0085, 0.0177, 0.0357, 0.0491, and 0.0739 ppm a.i., respectively. Following 96 hours of exposure, cumulative mortality was 0% in control groups and 0, 60, 90, 100, and 100% in the 0.0085, 0.0177, 0.0357, 0.0491, and 0.0739 ppm a.i. treatment groups, respectively. The 96-hour LC_{50} (with 95% C.I.) was 0.018 (0.013-0.023) ppm a.i., which classifies chlorothalonil as very highly toxic to Rainbow Trout (*Oncorhynchus mykiss*) on an acute toxicity basis. Swimming and/or hanging at the surface, lethargy, loss of equilibrium, and/or moribundity were observed in surviving fish from the ≥0.0177 ppm a.i. groups; effects were first observed within 6 hours of exposure and continued through 96 hours in groups with surviving fish. The NOEC (for mortality and sub-lethal effects) was 0.0085 ppm a.i.

B.4.2 Freshwater Fish, Open Literature Data

Acute LC50s reported in the open literature are summarized in Table B.4-2. One study (Davies and White, 1985) located in the open literature reported an LC50 that was more sensitive than the most sensitive registrant submitted studies. Davies and White (1985) reported an LC50 of 10.5 ppb in rainbow trout. The most sensitive rainbow trout LC50 from registrant submitted studies was 18 ppb (MRID 45710219), which is similar to the LC50 reported by Davies and White (1985) of 10.5 ppb. Although the LC50 of 10.5 ppb was from a study that used low oxygen levels, the oxygen levels utilized did not affect controls and were not outside values commonly found in the environment. Therefore, an LC50 of 10.5 ppb was chosen for use in risk estimation.

Table B.4-2: Freshwater Fish Lethal Effect Studies from the Open Literature

Test Species	LC_{50} ppb a.i.	Citation (MRID or Ecotox No.)	Toxicity Category	Comment
Jollytail	16	Ecotox No. 87454	very highly toxic	--
Spotted mountain galaxias	19 – 29	Ecotox No. 87454	very highly toxic	Tested species has not been evaluated in registrant submitted studies.
Threespine stickleback	69	Ecotox No. 7055	very highly toxic	Tested species has not been evaluated in registrant submitted studies.
Rainbow trout	10.5 - 76	Ecotox No. 87454; 7055	very highly toxic	The 10.5 ug/L value is the most sensitive acute LC50 available and is consistent with the range of LC50s in rainbow trout. 10.5 ug/L was chosen for use in risk quotient calculations. Study used an oxygen level of 50%; however, controls were not affected by the low DO, and the oxygen levels are not outside of the range expected to occur in the environment.
Tilapia	100 - 120	Ecotox No. 229772	highly toxic	--

A number of studies were also located in the open literature that evaluated sublethal endpoints such as biochemical endpoints. These studies did not report more sensitive toxicity values on endpoints that are correlated with assessment endpoints (survival and reproduction).

B.4.3. Toxicity of Formulated Products:

Formulated product testing is specified for products with direct application to aquatic habitats and for typical end-use products where the EEC for the active ingredient is $\geq$ LC_{50}. The previous Phase IV Review (1/12/93) specified further testing of a 54% ai flowable concentrate due to a cranberry use. The freshwater fish acute toxicity findings for the 54%, 75%, and Bravo W-75 formulations are summarized in Table B.4-3. The data suggest that studies using chlorothalonil products were of similar toxicity than studies using technical grade chlorothalonil.

Table B.4-3: Freshwater Fish Acute Toxicity Findings—Formulated Products				
Test Species	**% a.i.**	**LC_{50} ppb formulation**	**Citation (MRID #)**	**Toxicity Category (FP)**
Rainbow trout	54 (Bravo 720)	61 (33.2 ppb ai)	43302101	very highly toxic
Bluegill	54 (Bravo 720)	49 (26.3 ppb ai)	42433804	very highly toxic
Rainbow trout	75	152 (114 ppb ai; 48-hr study)	00087304	highly toxic
Rainbow trout	75	103 (77.2 ppb ai)	00087303	highly toxic
Bluegill	Bravo W-75	167 (125 ppb ai)	00087258	highly toxic

B.4.4. Toxicity of SDS-3701

Testing using the degradate SDS-3701 has been previously specified due to its persistence in water. Freshwater fish acute toxicity findings for the degradate SDS-3701 are summarized in Table B.4-3. These studies show that SDS-3701 is "slightly toxic" to the bluegill and therefore is significantly less toxic than parent chlorothalonil.

Table B.4-4: Freshwater Fish Acute Toxicity Findings—SDS-3701

Test Species	% SDS-3701	LC_{50} (ppb)	Citation (MRID #)	Toxicity Category
Bluegill	not avail.	45,000	00029415	slightly toxic
Bluegill	99	15,000	00030393	slightly toxic

B.4.5. Freshwater Fish, Chronic Submitted Data

Data from fish early life-stage testing is required for chlorothalonil since it can be expected to be transported to water from the intended use site, acute LC_{50} values are less than 1 mg/L, and aquatic EECs are $\geq$ 0.01 of LC_{50}s. The fish early life stage data are summarized in Table B.4-5.

In an early life-stage study in fathead minnows (MRID 00030391), fish were exposed to chlorothalonil at 0, 0.6, 1.4, 3.0, 6.5, and 16 ppb under flow-through conditions. Exposure at 16 and 6.5 ppb chlorothalonil produced adverse effects on egg production and on the survival. Specifically, the hatching success in the control groups averaged between 90% and 92.5% compared with 79.5% at 6.5 ppb and 46.5% at 16 ppb. The percent survival of the Fo generation in the control groups averaged 94% compared with 9% at 16 ppb. The results indicate that fathead minnow hatching success and survival were affected between 3 and 6.5 ppb.

Table B.4-5. Chronic Fish Toxicity Findings

Test Species	% a.i.	NOAEL (ppb)	LOAEL (ppb)	Citation (MRID #)	Endpoints Affected
Fathead minnow	96	3	6.5	00030391	Hatching success and survivability

B.4.7. Effects to Amphibians

No relevant data on terrestrial or aquatic phase amphibians were located in the open literature or were submitted by chlorothalonil registrants.

B.4.8. Freshwater Invertebrates, Acute Submitted Data

Registrant submitted freshwater invertebrate toxicity studies for technical grade chlorothalonil are summarized in Table B.4-6. Two studies in daphnids were submitted that reported consistent results. The lowest EC50 was 54 ppb (MRID 45710221).

Table B.4-6: Freshwater Invertebrate Toxicity Findings

Test Species	% a.i.	LC_{50} (ppb)	Citation (MRID #)	Toxicity Category	Fulfills Guideline?
Daphnia magna	Tech.	68	00068754	very highly toxic	Yes
Daphnia magna	Tech.	54	45710221	very highly toxic	Supplemental

B.4.9. Freshwater Invertebrates, Open Literature Data

Table B.4-7 summarizes the information available from the open literature. The data found in the open literature are not inconsistent with the registrant submitted studies; however, the only species tested in registrant submitted studies was the daphnid. Acute studies in several additional species were located in the open literature, which are summarized in Table B.4-7.

Table B.4-7: Freshwater Invertebrate Acute Effect Studies from the Open Literature

Study type / Test material	Test Organism (Common and Scientific Name) and Age and/or Size	Test Design	Endpoint Concentration in ppb	Citation (ECOTOX #)	Rationale for Use in Risk Assessment(1)
Acute (48 hr) Bravo 500 (40% a.i.)	Water flea (Daphnia magna) adult egg-bearing	Static Renewal; 4 treatment levels; 2 replicates, one fed and one not fed; algal food source	LC50 = 129 (95% CI 84-174) EC50 = 97 (95% CI 81-113)	Ernst et. al. 1991 (7055)	Qualitative. EC50 is not more sensitive than the lowest registrant-submitted study. Results based on nominal chlorothalonil levels.
Acute: Lethal and Sublethal 4 and 7 day values reported ≥ 98% a.i.	Giant Tasmanian Freshwater Crayfish (*Astacopsis gouldi*)	Flow through; 4 and 7 day values reported; 5 treatment levels	LC50 = 12.0 (7.9-18.1) 4 Day LC50 = 3.6 (2.1-6.0) 7-Day value	Davies et. al 1994 (64835)	Quantitative (acute lethal studies only)
Acute: Lethal and Sublethal 4 and 7 day values reported ≥ 98% a.i.	Amphipod (*Neoniphargus sp. A*)	Flow through; 4 and 7 day values reported; 5 treatment levels	LC50 = >40 (4 and 7 Day)	Davies et. al 1994 (64835)	QUAL The LC50 value was above the highest test concentration.
Acute: Lethal and Sublethal 4 and 7 day values reported ≥ 98% a.i.	Isopod (*Colubotelson chiltoni minor*)	Flow through; 4 and 7 day values reported; 5 treatment levels	LC50 = >40 (4 and 7 Day)	Davies et. al 1994 (64835)	QUAL The LC50 value was above the highest test concentration.
Acute: Lethal and Sublethal 4 and 7 day values reported ≥ 98% a.i.	Freshwater aytid shrimp Parataya australiensis	Flow through; 4 and 7 day values reported; 5 treatment levels	LC50 = 16 (14.4-17.9) 4-Day LC50 = 10.9 (9.1-13.1) 7-Day	Davies et. al 1994 (64835)	Quantitative (acute lethal data). The LC50 value is definitive and bounded by a 95% confidence interval. Study design appears adequate.

<u>Toxicity of Formulated Product:</u> The freshwater invertebrate toxicity findings for formulated product testing are summarized in the following table.

Table B.4-8: Freshwater Invertebrate Toxicity Findings—Formulated Product

Test Species	% a.i.	LC_{50} (ppb) formulation	Citation (MRID #)	Toxicity Category (FP)	Fulfills Guideline? (for FP tested)
Daphnia magna	54 (Bravo 720)	180 (97; ai) Probit slope = 7.9	42433806	highly toxic	Yes

<u>Toxicity of SDS-3701:</u> The freshwater invertebrate toxicity findings for the degradate, SDS-3701, are summarized in the following table. The data indicate that SDS-3701 is slightly toxic to aquatic invertebrates.

Table B.4-8: Freshwater Invertebrate Toxicity Findings—SDS-3701

Test Species	% SDS-3701	EC_{50} (ppb)	Citation (MRID #)	Toxicity Category	Fulfills Guideline?
Daphnia magna	99	26,000 Probit slope = 5.75	00030394	slightly toxic	Yes (for SDS-3701)

In MRID 00030394, the acute toxicity of SDS-3701 was evaluated in daphnids. Procedures used in this acute toxicity test were based on EPA protocols specified in "Methods for Acute Toxicity Tests with Fish, MAcroinvertebrates, and amphibians (U.S.EPA, 1975). Acetone was used as asolvent and in a solvent control. Nominal concentrations were 10, 15. 22. 32,46, 68 and 100 mg/L (ppm) and were tested in triplicate. The 48-hour EC50 for Daphnids exposed to SDS-3701 was calculated to be 26 ppm with 95% confidence limits of 21 to 31 ppm. The lowest concentration at which 100% mortality occurred was 68 ppm, while highest concentration in which there were no deaths was 10 ppm.

B.4.6 Freshwater Invertebrate, Chronic Submitted Data

The aquatic invertebrate life-cycle toxicity findings are summarized in the following table.

Table B.4-9: Aquatic Invertebrate Life-Cycle Toxicity Findings

Test Species	NOEL (ppb)	LOEL (ppb)	Citation (MRID #)	Endpoints Affected	Fulfills Guideline?
Daphnia magna	39	79	00115107	survival, cumulative numbers of offspring/ female	Yes
	0.6	1.8	45710222	Survival	

In the most sensitive chronic study in aquatic invertebrates (MRID 45710222), the 21-day-chronic toxicity of chlorothalonil to *Daphnia magna* was studied under static renewal conditions. Nominal concentrations were 0 (negative and solvent controls), 0.0010, 0.0032, 0.010, 0.032, and 0.10 ppm. Mean-measured concentrations were <0.0001 (<LOD, solvent control), 0.00060, 0.0018, 0.0058, 0.019, and 0.075 ppm a.i., respectively. However, chlorothalonil was unstable under the static renewal conditions employed in this test, declining to less than the level of detection to 62% of nominal concentrations in expired test media. Because chlorothalonil concentrations declined to < the level of detection at the lower concentrations, there is considerable uncertainty in the chlorothalonil levels associated with toxic effects in this study.

After 21 days of exposure, cumulative adult mortality/immobility was 7% for both control groups, and 5, 25, 40, 55, and 60% in the 0.00060, 0.0018, 0.0058, 0.019, and 0.075 ppm a.i. treatment groups, respectively. There was a significant inhibitory effect on reproduction (number of live young/adult) at the 0.075 ppm a.i. test level. The total number of live young produced per adult was at least 40 for the control through 0.019 ppm a.i. test groups, and was 31 for the 0.075 ppm a.i. group. No dead young were observed during the study in any test group, and the number of unhatched eggs per adult was <1 for all control and test groups. Terminal growth measurements were not performed. The NOAEC for survival was 0.6 ug/L (0.0006 mg/L).

B.5 Toxicity to Non-target Terrestrial Plants

B.5.1 Non-Target Terrestrial Plants: Submitted Data

The required tier 1 plant toxicity data are summarized in Table B.5-1.

Table B.5-1: Nontarget Terrestrial Plant Toxicity Findings

Study	% a.i.	Results (lb ai/A)	Citation (MRID #)	Fulfills Guideline?
Seed germination/seedling emergence--Tier 1 (122-1A); 10 species	97.9	NOEL $\geq$ 16	42433808	Yes
Vegetative vigor--Tier 1 (122-1B); 10 species	97.9	NOEL $\geq$ 16	42433809	Yes

B.5.2 Non-Target Terrestrial Plants: Open Literature Data

Based on a review of the open literature, no additional information was located that indicates greater non-target terrestrial plant sensitivity to chlorothalonil than the submitted data. Studies located in the open literature were predominantly efficacy studies (i.e., studies that evaluated effects of chlorothalonil on fungal diseases) or were studies that did not elicit adverse effects to plants.

B.5.3 Aquatic Plants: Registrant Submitted Data

Available aquatic plant toxicity data summarized in Table B.5-2. The most sensitive aquatic plant species from registrant studies was the diatom with an EC50 of 14 ppb.

Table B.5-2: Nontarget Aquatic Plant Toxicity Findings

Test Species	% a.i.	Results (ppb)	Citation (MRID #)	Meets Guideline Requirements
Freshwater Vascular Plant Duckweed (*Lemna gibba*)	TGAI	Number of fronds EC50 = 730 ppb (670-800ppb Probit slope = 8.08 Biomass (dry weight) EC50 = 630 ppb (550-730ppb) Probit Slope = 5.3 NOEC 290ppb	44908102	The only DER available is a contractor version. The contractor classifies this study as core.
Freshwater non-vascular plant. *Selenastrum capricornutum*	97.9	EC_{50} = 190 NOEC = 50 LOEC = 100 Slope = 4.027 95% CI 1.34-6.71	42432801	Yes

Test Species	% a.i.	Results (ppb)	Citation (MRID #)	Meets Guideline Requirements
Freshwater non-vascular plant. (Diatom) *Navicula pelliculosa*	TGAI	EC50 = 14 ppb (12-17 ppb) Probit slope = 4.49 NOEC = 3.9 ppb	44908105	The only DER available is a contractor version. The contractor classifies this study as core.

B.5.3 Aquatic Plants: Open Literature Data

Aquatic plant studies located in the open literature are summarized in Table B.5-3. The most sensitive aquatic plant study was from Mezcua et al. (2002), which reported a 72-hour EC50 of 6.8 ppb in *Selenastrum capricorotum*. This study reportedly followed OECD 201.

Table B.5-3: Aquatic Plant Toxicity Tests (Laboratory)

Study type/ Test material	Test Organism (Common and Scientific Name)	Test Design	Endpoint Concentration / Results	Citation (ECOTOX)	Study Classification[(1)]
30% SC	Green Algae *Scenedesmus obliqnus*	Laboratory bioassay	EC50 = 100 ppb	65723 (Ma et. al., 2001)	QUAL
30% SC	Green Algae *Chlorella pyrenoidosa*	Laboratory bioassay	EC50 = 8069 ppb		QUAL
Tech.	Green Algae *Selenastrum capricorotum*	Laboratory bioassay using Toxkit Algaltoxkit™ (Creasel Belgium) a growth inhibition assay – study reportedly followed OECD 201.	72-hr EC50 = 6.8 ppb. The EC50 @ 30 h = 42,400ppb	80747 (Fernandez-Alba et. al., 2002) 80359 (Mezcua et. al. 2002)	QUAN.

(1) QUAL = The paper is not appropriate for quantitative use but is of good quality, addresses issues of concern to the risk assessment and is used in the risk characterization discussion.

B.6 References

ECOTOX or MRID	Citation
2478	Kikuchi, M. 1993. Toxicity Evaluation of Selected Pesticides Used in Golf Links by Algal Growth Inhibition Test. Journal of Japanese Society of Water Environment 16(10):704-710
63488	Samsoe-Peterson, L. 1995. Effects of 37 Fungicides on the Rove Beetle *Aleochara bilineata* (Col, : Staphylinidae) in the Laboratory. Entomophaga 40(2):145-152.
64665	Jansen, J.P. 1999. Effects of Wheat Foliar Fungicides and the Aphid Endoparisitoid *Aphidus rhopalosiphi* DeStefani-Perez (Hym. Aphidiidae) on Glass Plates and on Plants. Journal of Applied Entomology. 123:217-223
64835	Davies PE;Cook LSJ;Goenarso D. 1994. Sublethal Responses to Pesticides of Several Species of Australian Freshwater Fish and Crustaceans and Rainbow Trout.Environ Toxicol Chem 13(8): 1341-1354 (OECDG Data File)
65723	MA, J., Zeng, R., Xu, L., Wang, S. 2002. Differential Sensitivity of Two Green Algae, *Scenedesmus obliqnus* and *Chlorella pyrenoidosa*, to 12 Pesticides. Ecotoxicology and Environmental Safety 52(1):57-61
67894	James, D.G and Rayner M. 1995. Toxicity of viticultural pesticides to the predatory mites *Amblyseius victorensis* and *Typhlodromus doreenae*. Plant Protection Quarterly. 10(3):99-102.
7055	Ernst, W., Doe, K., Jonah, P., Young, J., Julien, G., and Hennigar, P. 1991. The Toxicicty of Chlorothalonil to Aquatic Fauna and the impact of its operational Use on a Pond Ecosystem. Archives of Environmental Contaminantion and Toxicology. 21:1-9
71029	Carruthers, R.L., Whitfield, G.H., and Haynes, D.L. 1985. Pesticide-Induced Mortality of Natural Enemies of the Onion Maggot, *Delia antigua.* Entomophaga 30(2):151-161
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80359	Mezcua, M., Hernando, M.D., Piedra, L., Aguera, A., Fernandez-Alba, A.R. 2002. Chromatogrophy-Mass Spectrometry and Toxicity Evaluation of Selected Contaminants in Seawater. Chromatographia 56(3/4):199-205

80747 Fernandez-Alba, A.R., Hernando, M.D., Piedra, L., Christi, Y. 2002. Toxicity Evaluation of Single and Mixed Antifouling Biocides Measured with Acute Toxicity Bioassays. Analytical Chimica 456:303-312.

89639 Smitley, D.R. and Rothwell, N.L. 2003. How the Use of Chlorothalonil on Golf Courses Impacts *Paenibacillus* sp., a Pathogen of *Ataenius spretulus* (Coleoptera: Scarabaeidae) Journal of Economic Entomology 96(3): 792-797

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CHEMICAL	2,4-D	2,4-D	2,4-D	2,4-D	2,4-D	2,4-D
PERCENT	100	100	100	100	100	100
DESCRIP	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL
SPECIES	CUTTHROAT TROUT	CUTTHROAT TROUT	CUTTHROAT TROUT	CUTTHROAT TROUT	CUTTHROAT TROUT	CUTTHROAT TROUT
SIZE						
WEIGHT	0.5	0.8	0.5	0.4	0.6	0.8
STRAIN						
DIET						
TEMP	10	10	10	15	5	10
PH	7.4	7.4	8.5	7.4	7.4	6.5
HARDNESS	300	160	44	44	44	44
TYPE	STATIC	STATIC	STATIC	STATIC	STATIC	STATIC
TEST_UNT	LC	LC	LC	LC	LC	LC
TOX_UNT	MG	MG	MG	MG	MG	MG
LC50_24H	32	41	40	41.5	41.5	43.5
FROM_24H	23.7	32.4	30.7	33.1	32.8	35
TO_24H	43.3	51.9	52.1	52	52.5	54.1
LC50_48H						
FROM_48H						
TO_48H						
LC50_96H	24.5	37	40	41.5	41.5	43.5
FROM_96H	18.4	27.9	30.2	33.1	32.8	35
TO_96H	32.6	49	52.1	42	52.5	54.1

2,4-D	2,4-D	2,4-D	2,4-D	2,4-D	2,4-D	2,4-D
100	100	100	100	100	100	100
TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL
CUTTHROAT TROUT	CUTTHROAT TROUT	CUTTHROAT TROUT	CUTTHROAT TROUT	CUTTHROAT TROUT	CUTTHROAT TROUT	LAKE TROUT
0.5	0.4	0.6	0.5	0.8	0.8	0.3
10	10	10	10	10	10	10
7.4	7.4	6.5	7.5	7.5	8.5	7.4
44	44	44	44	44	44	44
STATIC	STATIC	STATIC	STATIC	STATIC	STATIC	STATIC
LC	LC	LC	LC	LC	LC	LC
MG	MG	MG	MG	MG	MG	MG
49	64	67	130	169	185	44.5
40.4	57.3	57.5	100	146	165	35.3
59.4	71.5	78	170	195	207	56.1
44	64	67	130	169	172	44.5
35.7	57.3	57.5	100	146	148	35.3
54.2	71.5	78	170	195	199	56.1

2,4-D	2,4-D	2,4-D	2,4-D	2,4-D	2,4-D	2,4-D
100	100	100	100	100	98	98
TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL
LAKE TROUT	LAKE TROUT	LAKE TROUT	LAKE TROUT	LAKE TROUT	RAINBOW TROUT	BLUEGILL
0.7	0.7	0.7	0.9	0.7	0.3	0.5
10	15	5	10	10	12	22
6.5	7.4	7.4	7.5	8.5	7.3	7.3
44	44	44	44	44	40	40
STATIC	STATIC	STATIC	STATIC	STATIC	STATIC	STATIC
LC	LC	LC	LC	LC	LC	LC
MG	MG	MG	MG	MG	MG	MG
63	64	65.7	105	127.5		
59.3	47.8	47.2	95.6	110		
67	85.7	91.4	115	149		
62	64	65.7	105	120	110	180
54	47.8	47.2	95.6	103	77	
71.2	85.7	91.4	115	140	157	

2,4-D
98
TECHNICAL MATERIAL
FATHEAD MINNOW
0.9
22
7.3
40
STATIC
LC
MG
133
103
171

CHEMICAL		TRICLOPYR		TRICLOPYR
PERCENT		43		43
DESCRIP		LIQUID		LIQUID
SPECIES		BLUEGILL		RAINBOW TROUT
SIZE				
WEIGHT		0.8		0.9
STRAIN				
DIET				
TEMP		22		12
PH		7.4		7.4
HARDNESS		44		44
TYPE		STATIC		STATIC
TEST_UNT		LC		LC
TOX_UNT		MG		MG
LC50_24H		>100.		>100.
FROM_24H				
TO_24H				
LC50_48H				
FROM_48H				
TO_48H				
LC50_96H		>100.		>100.
FROM_96H				
TO_96H				

CHEMICAL	DIURON	DIURON	DIURON	DIURON
PERCENT	95	95	95	95
DESCRIP	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL
SPECIES	ASELLUS BREVICAUDUS	BLUEGILL	BLUEGILL	BLUEGILL
SIZE	MATURE			
WEIGHT		0.8	1.1	0.8
STRAIN				
DIET				
TEMP	15	13	12	7
PH	7.1	7.1	8	7.1
HARDNESS	44	44	300	44
TYPE	STATIC	STATIC	STATIC	STATIC
TEST_UNT	LC	LC	LC	LC
TOX_UNT	MG	MG	MG	MG
LC50_24H	>10.	27	>30.0	29.8
FROM_24H		24.8		27.3
TO_24H		29.4		32.5
LC50_48H				
FROM_48H				
TO_48H				
LC50_96H	15.5	9.5	8	9.3
FROM_96H	7.2	8.5	5.8	8.1
TO_96H	33.4	10.6	11.7	10.7

DIURON	DIURON	DIURON	DIURON	DIURON
95	95	95	95	95
TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL
BLUEGILL	BLUEGILL	BLUEGILL	BLUEGILL	BLUEGILL
0.8	0.8	0.8	1.1	1.1
18	24	29	12	12
7.1	7.1	7.1	6.5	8
44	44	44	44	44
STATIC	STATIC	STATIC	STATIC	STATIC
LC	LC	LC	LC	LC
MG	MG	MG	MG	MG
16.7	8.4	3.6	>30.0	>30.0
14.9	7.7	3		
18.7	9.3	4.2		
8.2	6.4	2.8	8.6	10
7.4	5.9	2.3	7.1	8.5
9.1	7	3.3	10.4	11.8

DIURON	DIURON	DIURON	DIURON	DIURON
95	95	95	95	95
TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL
BLUEGILL	BLUEGILL	BLUEGILL	BLUEGILL	COHO SALMON
1.1	1.1	1.1	0.9	1.4
12	12	12	12	13
8.5	9.5	8	7.5	7.1
44	44	170	44	44
STATIC	STATIC	STATIC	STATIC	STATIC
LC	LC	LC	LC	LC
MG	MG	MG	MG	MG
>30.0	38.8	>30.0	35	11
	33.2		28.6	
	45.4		42.8	
10.4	7	8.3	10.4	<2.4
7.3	5.4	7	7.6	
14.9	9	9.8	14.2	

DIURON	DIURON	DIURON	DIURON	DIURON
95	95	95	95	95
TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL
CUTTHROAT TROUT	CUTTHROAT TROUT	CUTTHROAT TROUT	CUTTHROAT TROUT	CUTTHROAT TROUT
0.7	0.7	1	0.8	0.9
10	10	10	10	10
7	7.7	6.5	7.7	7.5
44	165	44	295	44
21 DAY DEGRA	STATIC	STATIC	STATIC	STATIC
LC	LC	LC	LC	LC
MG	MG	MG	MG	MG
>30.0	>5.0	>4.5	>5.0	4.2
				3.1
				5.7
12.8	1.9	2.1	1.9	1.4
9	1.5	1.7	1.5	1
18	2.5	2.6	2.5	1.9

DIURON	DIURON	DIURON	DIURON	DIURON
95	95	95	95	95
TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL
CUTTHROAT TROUT	CUTTHROAT TROUT	CUTTHROAT TROUT	CUTTHROAT TROUT	CUTTHROAT TROUT
1	0.6	0.8	0.4	0.3
10	10	15	5	10
8.5	7.8	7.2	7.4	7.2
44	44	44	44	44
STATIC	STATIC	STATIC	STATIC	STATIC
LC	LC	LC	LC	LC
MG	MG	MG	MG	MG
>4.5	>4.0	2.8	>4.0	3.4
		1.9		2.9
		4.2		3.8
2.2	1.7	0.71	1.4	1.4
1.8	1.4	0.53	1.1	1.1
2.7	2.1	0.96	1.7	1.9

DIURON	DIURON	DIURON	DIURON	DIURON
95	95	95	95	95
TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL
CUTTHROAT TROUT	CUTTHROAT TROUT	CUTTHROAT TROUT	CUTTHROAT TROUT	CUTTHROAT TROUT
0.7	0.7	0.7	0.7	3.1
10	10	10	10	10
7	7	7	7	7.4
44	44	44	44	162
28 DAY DEGRA	14 DAY DEGRA	7 DAY DEGRA	0 DAY DEGRA	FLOW THROUGH
LC	LC	LC	LC	LC
MG	MG	MG	MG	MG
30	>20.0	>10.0	>5.0	>4.0
12.3	13.8	11.5	1.5	1.9
9.5	6.3	6.3	1.1	1.6
15.8	29.9	20.9	2	2.1

DIURON	DIURON	DIURON	DIURON	DIURON
95	95	95	95	95
TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL
DAPHNIA PULEX	GAMMARUS FASCIATUS	LAKE TROUT	LAKE TROUT	LAKE TROUT
1ST INSTAR	MATURE			SWIMUP FRY
		1	0.9	
15	21	10	10	10
7.1	7.1	6.5	8	7
44	44	44	175	44
STATIC	STATIC	STATIC	STATIC	STATIC
EC	LC	LC	LC	LC
MG	MG	MG	MG	MG
	0.7	>4.5	3.5	4.2
	0.59		2.8	3.3
	0.83		4.4	5.3
1.4				
1				
1.9				
	0.16	2.5	2.1	1.1
	0.13	1.9	1.5	1
	0.19	3.1	3	1.3

DIURON	DIURON	DIURON	DIURON	DIURON
95	95	95	95	95
TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL
LAKE TROUT	LAKE TROUT	LAKE TROUT	LAKE TROUT	LAKE TROUT
5.1	0.3	1.5	1.5	0.4
10	10	10	10	5
7.4	7	7	7	7.5
162	44	44	44	44
FLOW THROUGH	STATIC	21 DAY DEGRA	14 DAY DEGRA	STATIC
LC	LC	LC	LC	LC
MG	MG	MG	MG	MG
	3.5	>20.0	17.5	5.3
	2.2		10.8	4
	5.5		28.5	7
1.8	1.8	11.5	3.6	2.2
1.6	1.5	7.9	2.5	1.7
2.1	2	16.8	5	2.7

DIURON	DIURON	DIURON	DIURON	DIURON
95	95	95	95	95
TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL
LAKE TROUT	LAKE TROUT	LAKE TROUT	LAKE TROUT	LAKE TROUT
1.2	1.1	1	1	0.4
10	10	10	10	15
8	8	8.5	7.5	7.5
295	44	44	44	44
STATIC	STATIC	STATIC	STATIC	STATIC
LC	LC	LC	LC	LC
MG	MG	MG	MG	MG
>3.0	3.3	4.2	>4.0	2.9
	2	2.8		2.2
	5.3	6.2		3.9
2.6	2.2	2.6	2.4	1.2
2	1.6	1.9	1.9	0.9
3.4	2.9	3.1	2.9	1.5

DIURON	DIURON	DIURON	DIURON	DIURON
95	95	95	95	95
TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL
LAKE TROUT	LAKE TROUT	PTERONARCYS CALIFORNICA	RAINBOW TROUT	RAINBOW TROUT
		2ND YEAR CLASS		
1.5	1.5		1.5	0.8
10	10	15	12	13
7	7	7.1	7.5	7.1
44	44	44	44	44
STATIC	7 DAY DEGRA	STATIC	21 DAY DEGRA	STATIC
LC	LC	LC	LC	LC
MG	MG	MG	MG	MG
>3.5	>5.0	3.6		9
		2.8		7.6
		4.7		11
2.7	3.2	1.2	7.4	4.9
2.4	2.2	0.9	6.2	4.1
3	4.6	1.7	8.7	5.9

DIURON	DIURON	DIURON	DIURON	DIURON
80	95	95	95	95
WETTABLE POWDER	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL
RAINBOW TROUT	RAINBOW TROUT	RAINBOW TROUT	RAINBOW TROUT	RAINBOW TROUT
1.2	1.2	1.2	1.2	1.2
13	2	7	13	18
7.1	7.1	7.1	7.1	7.1
44	44	44	44	44
STATIC	STATIC	STATIC	STATIC	STATIC
LC	LC	LC	LC	LC
MG	MG	MG	MG	MG
71	11.5	15.5	12.5	8.4
61.1	10.5	14.1	11.5	7.9
82.5	12.7	17.1	13.6	9.1
16	7.7	7.2	6.2	5.3
11.3	6.8	6.5	5.8	5.1
22.7	8.9	7.9	6.6	5.7

DIURON	DIURON	DIURON	DIURON	DIURON
95	95	95	95	95
TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL
RAINBOW TROUT	RAINBOW TROUT	RAINBOW TROUT	RAINBOW TROUT	SIMOCEPHALUS SERRULATUS
				1ST INSTAR
1.5	1.5	1.5	1.5	
12	12	12	12	15
7.5	7.5	7.5	7.5	7.1
44	44	44	44	44
0 DAY DEGRA	14 DAY DEGRA	28 DAY DEGRA	7 DAY DEGRA	STATIC
LC	LC	LC	LC	EC
MG	MG	MG	MG	MG
				2
				1.4
				2.8
3.5	13.4	9.4	4.2	
2.7	10.7	8.3	3.1	
4.4	16.7	10.7	5.6	

CHEMICAL	LINURON	LINURON	LINURON	LINURON	LINURON	LINURON
PERCENT	50	95	50	95	50	95
DESCRIP	WETTABLE POWDER	TECHNICAL MATERIAL	WETTABLE POWDER	TECHNICAL MATERIAL	WETTABLE POWDER	TECHNICAL MATERIAL
SPECIES	CHANNEL CATFISH	CHANNEL CATFISH	CHIRONOMUS PLUMOSUS	CHIRONOMUS PLUMOSUS	DAPHNIA MAGNA	DAPHNIA MAGNA
SIZE			3RD INSTAR	3RD INSTAR	1ST INSTAR	1ST INSTAR
WEIGHT	0.6	0.4				
STRAIN						
DIET						
TEMP	22	22	22	22	17	17
PH	7	7.4	7	7	7	7
HARDNESS	40	40	43	43	43	43
TYPE	STATIC	STATIC	STATIC	STATIC	STATIC	STATIC
TEST_UNT	LC	LC	EC	EC	EC	EC
TOX_UNT	MG	MG	MG	MG	MG	MG
LC50_24H	3.2					
FROM_24H	2.3					
TO_24H	4.4					
LC50_48H			4.3	2.9	0.21	0.27
FROM_48H			3.4	1.9	0.16	0.22
TO_48H			5.4	4.2	0.28	0.33
LC50_96H	1.8	2.9				
FROM_96H	1.2	2				
TO_96H	2.6	4.2				