UNITED STATES DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
NATIONAL MARINE FISHERIES SERVICE
1315 East-West Highway
Silver Spring, Maryland 20910
THE DIRECTOR

Ms. Debbie Edwards
Director, Office of Pesticide Programs
U.S. Environmental Protection Agency
One Potomac Yard
2777 S. Crystal Drive
Arlington, Virginia 22202

APR 20 2009

Dear Ms. Edwards:

Enclosed is the National Oceanic Atmospheric Administration (NOAA) National Marine Fisheries Service's (NMFS) final biological opinion (Opinion), issued under the authority of section 7(a)(2) of the Endangered Species Act (ESA), on the effects of the U.S. Environmental Protection Agency's (EPA) proposed registration of pesticide products containing the active ingredients carbaryl, carbofuran, and methomyl on endangered species, threatened species, and critical habitat that has been designated for those species. This Opinion assesses the effects of all pesticides containing carbaryl, carbofuran, or methomyl 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 carbaryl and carbofuran are likely to jeopardize the continuing existence of 22 listed Pacific salmonids as described in the attached Opinion. NMFS also concluded that the effects of carbaryl and carbofuran are likely to destroy or adversely modify designated habitat for 20 of 26 listed salmonids. NMFS has not designated critical habitat for two listed salmonids. NMFS determinations for no jeopardy and no adverse modification of critical habitat apply to Ozette Lake sockeye salmon, Snake River sockeye salmon, Northern California steelhead, Columbia River chum salmon, Hood Canal summer-run chum salmon, and Oregon Coast coho salmon. We further conclude that pesticide products containing methomyl are likely to jeopardize 18 listed Pacific salmonids and likely to destroy or adversely modify critical habitat for 16 of 26 salmonids with designated critical habitat. NMFS determinations for no jeopardy and no adverse modification of designated critical habitat apply to California Coastal Chinook salmon, Snake River fall-run Chinook salmon, Snake River spring/summer-run Chinook salmon, Ozette Lake sockeye salmon, Snake River sockeye salmon, Northern California steelhead, Columbia River chum salmon, Hood Canal summer-run chum salmon, Oregon Coast coho salmon, and Snake River steelhead. As NMFS has not designated critical habitat for the Lower Columbia River coho salmon or Puget Sound steelhead, the action area contains no designated critical habitat for these species. Thus, the Opinion presents no further critical habitat analysis for the Lower Columbia River coho salmon and Puget Sound steelhead.

As required by section 7 of the ESA, NMFS provides an incidental take statement with the Opinion. The incidental take statement describes reasonable and prudent measures NMFS considers necessary or appropriate to minimize incidental take associated with this action. The incidental take statement also sets forth nondiscretionary terms and conditions, including reporting requirements that EPA and any person who performs the action must comply with to carry out the reasonable and prudent measures. Incidental take from actions by EPA and the applicants that meets these terms and conditions will be exempt from the ESA section 9 prohibitions for take.

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

Enclosure

National Marine Fisheries Service

Endangered Species Act Section 7 Consultation

Biological Opinion

Environmental Protection Agency Registration of Pesticides Containing Carbaryl, Carbofuran, and Methomyl

Photo Credit: Desmond Maynard NMFS

Photo Credit: Tom Maurer USFWS

April 20, 2009

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Biological Opinion

Agency: United States Environmental Protection Agency

Activities Considered: Authorization of pesticide products containing the a.i.s carbaryl, carbofuran, and methomyl, and their formulations in the United States and its affiliated territories

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

Approved by: Helen M Golde for James H. Lecky

Date: April 20, 2009

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 that an action "may affect but is not likely to adversely affect" endangered species, threatened species or designated critical habitat (50 CFR §420.14(b)).

The United States (U.S.) Environmental Protection Agency (EPA) initiated consultation with NMFS on its proposal to authorize use, pursuant to the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA), 7 U.S.C. 136 *et seq.*, of pesticide products containing the a.i.s (a.i.s) of carbaryl, and methomyl on April 1, 2003, and of carbofuran on 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" most of the 26 Evolutionarily Significant Units (ESUs) of Pacific salmonids listed as endangered or threatened and designated critical habitat for the ESUs. This document represents NMFS' biological opinion (Opinion) on the impacts of EPA's authorization of pesticide products containing the above-mentioned a.i.s on the listed ESUs, plus on two newly listed salmonids. This is a partial consultation because pursuant to the court's order, EPA sought consultation on only this group of listed species under NMFS' jurisdiction. However, even though the court's order did not address the two more recently listed salmonids, NMFS analyzed the impacts of EPA's action to them because they belong to the same taxon. NMFS analysis requires consideration of the same information. Consultation with NMFS will be completed when EPA makes effect determinations on all remaining species and consults with NMFS as necessary.

This Opinion is prepared in accordance with section 7(a)(2) of the ESA and implementing regulations at 50 CFR §402. However, consistent with the decision in Gifford Pinchot Task Force v. USFWS, 378 F.3d 1059 (9th 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 authorization of the above a.i.s. It also includes our review of recovery plans for listed Pacific salmonids, past and current research and population dynamics modeling efforts, monitoring reports from prior research, Opinions on similar research, published and unpublished scientific information on the biology and ecology of threatened and endangered salmonids in the action area, and other sources of information gathered and evaluated during the consultation on the proposed authorization of a.i.s for carbaryl, carbofuran, and methomyl. NMFS also considered information and comments provided by EPA and by the registrants identified as applicants by EPA.

Background

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

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

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

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

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

On January 22, 2004, the court enjoined application of pesticides within 20 (for ground) and 100 (for aerial) feet (ft) of streams supporting salmon. Washington Toxics Coalition v. EPA, 357 F.Supp. 2d 1266 (W.D. Wash. 2004). The court imposed several additional restrictions on pesticide use in specific settings.

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

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

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

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

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

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

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

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. This second consultation evaluates three carbamate insecticides: carbaryl, carbofuran, and methomyl. EPA consultations on pesticide products currently focus on their effects to listed Pacific salmonids. EPA consultations remain incomplete until all protected species under NMFS' jurisdiction are covered.

Consultation History

On April 1, 2003, the EPA sent a letter to NMFS' Office of Protected Resources (OPR) requesting section 7 consultation for the registration of the a.i. carbaryl and detailing its effects determinations on 26 ESUs of Pacific salmonids listed at that time. In that same letter, EPA's Office of Pesticide Programs (OPP) determined that the use of carbaryl will have "no effect" for 4 ESUs, "may affect but is not likely to adversely affect" 2 ESUs, and "may affect" 20 ESUs of listed salmonids. EPA's "no effect" determinations for carbaryl applied to Northern California steelhead, SONCC coho salmon, Hood Canal Summer-run chum salmon, and Ozette Lake sockeye salmon.

On April 1, 2003, the EPA sent a letter to NMFS' OPR requesting section 7 consultation for the registration of the a.i. methomyl and detailing its effects determinations on 26 ESUs of Pacific salmonids listed at that time. In that same letter, the EPA's OPP determined that the use of methomyl will have "no effect" for 2 ESUs, and "may affect" 24 ESUs of listed salmonids. EPA's "no effect" determinations for methomyl applied to the Northern California steelhead and California Coastal Chinook salmon ESUs.

On December 1, 2004, the EPA sent a letter to NMFS' OPR requesting section 7 consultation for the registration of the a.i. carbofuran and detailing its effects determinations on 26 ESUs of Pacific salmonids listed at that time. In that same letter, EPA's OPP determined that the use of carbofuran will have "no effect" for 3 ESUs; "may affect but is not likely to adversely affect" 18 ESUs, and "may affect" 3 ESUs of listed salmonids. EPA's "no effect" determinations applied to the California Coastal Chinook salmon, Central California coho salmon, and Northern California steelhead.

On June 28, 2005, NMFS listed the Lower Columbia River coho salmon ESU as endangered. Given this recent listing, EPA's 2003 and 2004 effects determinations for carbaryl, carbofuran, and methomyl 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 carbaryl, carbofuran, and methomyl on listed Pacific salmonids lack an effects determination for the Puget Sound steelhead.

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

On February 11, 2008, NMFS listed the Oregon Coast coho salmon ESU as threatened. EPA's 2003 and 2004 initiation packages for carbaryl, carbofuran, and methomyl provided an effects determination for the Oregon Coast coho salmon ESU. This ESU was previously listed in 1998 and its ESA status was in-flux until 2008.

On August 20, 2008, NMFS met with EPA and requested EPA to identify applicants for this and subsequent pesticide consultations. NMFS also requested information on EPA's cancellation of carbofuran and of existing stocks of carbofuran.

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 16, 2008, NMFS requested EPA to confirm the status of EPA's cancellation of carbofuran and for existing stocks of that same compound during a conference call.

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 informing it of the roles of the action agency and applicants during formal consultation. NMFS also requested incident reports and label information for subsequent pesticide consultations from EPA. The specified timeline for NMFS' receipt of incident report and label information for carbaryl, carbofuran, and methomyl was November 3, 2008.

On October 3, 2008, NMFS received post-2002 incident reports for carbaryl, carbofuran, and methomyl from EPA.

On November 5, 2008, NMFS sent an e-mail to EPA requesting it to identify applicants for upcoming pesticide consultations and label and incident report information for carbaryl, carbofuran, and methomyl. NMFS also requested information regarding whether final cancellation of carbofuran or any if its uses had occurred.

On November 13, 2008, EPA provided an interim e-mail response to NMFS' November 5, 2008 query. EPA stated that it was developing a process to identify applicants for carbaryl, carbofuran, and methomyl. No applicants were identified in EPA's response. EPA also stated that incident data for carbaryl, carbofuran, and methomyl were sent via FedEx to NMFS on October 2, 2008, which we received. Finally, EPA confirmed that no final cancellations for carbofuran have occurred subsequent to the Scientific Advisory Panel (SAP) meeting required for action relative to the Notice of Intent to cancel this compound. The SAP meeting occurred on February 5, 2008.

On December 1, 2008, NMFS repeated its request to EPA to identify applicants for carbaryl, carbofuran, and methomyl via e-mail. NMFS also requested EPA to provide technical staff contact information for these same chemicals so NMFS staff may request information from them during this consultation.

On December 15, 2008, EPA informed NMFS via e-mail that it would send letters to the technical registrants of carbaryl, carbofuran, and methomyl. EPA also stated that it would inform the parties that they may submit information relative to the consultation directly to NMFS with a copy to EPA. EPA also offered to include additional information requested by NMFS pertinent to the consultation into that same letter.

On December 16, 2008, NMFS and EPA discussed each agency's notification strategy of prospective applicants for this consultation. As the action agency, EPA indicated it would identify and contact prospective applicants. EPA limited applicant status to those technical registrants who have all information pertinent to the consultation.

On December 18, 2008, EPA sent formal correspondence to four technical registrants. EPA's letter requested confirmation on their desire to have applicant status and for parties to submit data not already provided with EPA's consultations that may inform the outcome of the consultation. That information includes any toxicity data, field studies or mesocosm studies not part of the consultation package, or EPA's Interim Registration Eligibility Decision (IRED) or Registration Eligibility Decision (RED) documents for the pesticide a.i.; and current labels for end use products or if available, a master label that includes all use instructions for all products containing the a.i.s. These data would be submitted to NMFS and EPA.

On December 19, 2008, EPA identified technical staff contact information and four applicants to NMFS for this consultation via formal correspondence. In that same letter, EPA referred NMFS to the IRED and RED documents for any changes to the three subject a.i.s since consultation was initiated in 2003 and 2004.

On that same date, NMFS received electronic copies of EPA letters sent to the applicants. EPA identified the following applicants: Bayer CropScience LP, Drexel Chemical Company, E.I., duPont de Nemours and Company (DuPont), and FMC Corporation.

On January 5, 2009, NMFS requested clarification on EPA's registration action for carbofuran via e-mail. Questions pertained to the six carbofuran uses that have not been proposed for voluntary cancellation; the duration of the cancellation process for these same six uses as well as for 22 crop uses proposed for voluntary cancellation; and whether other current or pending registrations for carbofuran (FIFRA sections 3, 4, 18, or 24 (c)) are anticipated.

On January 7, 2009, EPA notified NMFS via email that it was conducting an internal policy review and would provide a full response to NMFS' query on carbofuran as soon as possible.

On January 8 and 9, 2009, EPA and NMFS exchanged e-mails scheduling a meeting at the end of January with identified applicants for this consultation.

On January 9, 2009, NMFS provided EPA with an electronic draft of the Description of the Proposed Action and associated Appendix for this consultation. NMFS requested EPA comments on these documents by January 23, 2009.

On January 21, 2009, the agencies agreed to meet with the applicants on January 30, 2009, at OPP Headquarters in Crystal City, Virginia.

On January 22, 2009, NMFS requested the following information from EPA: the bibliography for the carbofuran IRED (August 3, 2006), the report study cited as Table 16 within the carbofuran IRED; and the methomyl study cited in the Science Chapter – Master Record Identification Number (MRID) #00131255.

On that same date, DuPont informed NMFS that it would submit information to NMFS and EPA to support the consultation for methomyl.

On that same date, EPA informed NMFS that it would provide comments to NMFS on the draft Description of the Proposed Action by January 30, 2009.

On January 23, 2009, EPA instructed DuPont to send any data in support of the methomyl consultation to both EPA and NMFS.

On January 26, 2009, NMFS received data on methomyl from DuPont. The package included a cover letter, analysis of risk to methomyl to listed Pacific salmonids; copies of four studies, including three toxicity tests with formulated material and an environmental fate study (dissipation of methomyl in a simulated pond); and copies of methomyl product labels held by DuPont.

On that same date, NMFS also received confidential information on sales of Lannate (DuPont methomyl product) in Washington and Oregon from 2004-2006.

On January 27, 2009, NMFS asked EPA via e-mail to identify agenda topics and a list of participants/applicants for the January 30, 2009, meeting. EPA provided the requested information on that same date.

On January 29, 2009, NMFS received information from Bayer CropScience, the registrant for carbaryl. The information included a summary of Section 3 label restrictions, a “master label table”, and copies of current 24(c) and Section 3 labels.

On January 30, 2009, NMFS met with EPA, Bayer CropScience, DuPont, and FMC Corporation. At this meeting, NMFS explained the consultation procedure and timelines for this consultation. The applicants also presented information to NMFS and EPA on these a.i.s. DuPont presented the risk of methomyl on endangered salmonids. FMC presented the U.S. registration status and use in the Pacific Northwest for carbofuran. Bayer CropScience presented use pattern summaries for carbaryl. This venue facilitated a question-answer session between the applicants and the agencies.

On that same date, NMFS received electronic files of applicant presentation materials on carbaryl, carbofuran, and methomyl. EPA also provided NMFS with a PDF file of the Environmental Fate and Effects Division’s (EFED) RED Science Chapter for methomyl.

On February 2, 2009, NMFS received carbofuran data from FMC Corporation. Materials included the status on furadan registration and use in the relevant Pacific Northwest, the proposed federal label for furadan, current special local needs label for potato use in Oregon and spinach grown for seed use in Washington, and the Federal Register notice for proposed voluntary cancellation of most uses.

On that same date, NMFS queried FMC Corporation via e-mail regarding when EPA’s response on the proposed federal label for furadan is expected. NMFS also requested information whether the proposed permitted uses for foliar application of furadan on cotton would apply in California, Idaho, Oregon, and Washington.

On February 3, 2009, NMFS received comments from EPA on the draft Description of the Proposed Action relevant to carbofuran and methomyl. EPA disagreed with FMC’s statement that use of carbofuran has been discontinued on field corn in the Pacific Northwest. According to EPA, carbofuran use is specified on federal labels and is used in California, Idaho, Oregon, and Washington. Carbofuran is also used on sunflowers grown in these states. EPA feedback for methomyl pertained to the SLN in California and the status of some section 24(c) actions. EPA provided no response regarding carbaryl.

On that same date, NMFS requested clarification on registered carbofuran uses and EPA's verification of NMFS' draft Description of the Proposed Action for this ingredient.

On February 5, 2009, NMFS received the two early life stage studies for fish (MRID 131255 and 126862) from EPA. On that same date, FMC Corporation responded to NMFS' questions regarding uses of furadan raised in it February 2, 2009, e-mail.

On February 9, 2009, EPA provided responses via e-mail regarding questions posed in NMFS' February 3, 2009, e-mail, on carbaryl. EPA provided no comments on NMFS' Description of the Proposed Action for carbaryl and no response towards NMFS' questions on carbofuran.

On February 11, 2009, NMFS received a study on carbaryl from Bayer CropScience.

On February 13, 2009, NMFS contacted EPA by phone and requested EPA comments on the draft Description of the Proposed Action.

On that same date, NMFS received a copy of a position paper on key issues related to carbaryl use, ecotoxicology, and aquatic exposure from Bayer CropScience. Bayer Crop Science also sent this document to EPA.

On February 20, 2009, EPA e-mailed NMFS information on carbaryl use patterns for Idaho, Oregon, and Washington. NMFS previously requested this information for the draft Description of the Proposed Action.

On February 20, 2009, EPA e-mailed NMFS information on carbaryl use patterns for Idaho, Oregon, and Washington.

On March 5, 2009, NMFS received an extension from the court for this consultation from March 31, 2009, to April 20, 2009.

On March 12, 2009, NMFS requested clarification from EPA on the carbaryl use data provided in its February 20, 2009, e-mail. NMFS also requested EPA forward cancellation notices for carbaryl for several crop uses once they are available.

On March 13, 2009, NMFS met with and provided EPA a list of questions pertaining to past, ongoing, and future pesticide consultations with EPA.

On March 17, 2009, NMFS e-mailed EPA instructions to access a pdf copy of the draft Opinion from NMFS' ftp site. NMFS also FedExed a pdf copy of the draft Opinion to EPA on that same day. Although this draft did not include Reasonable and Prudent Alternatives (RPAs), NMFS conveyed to EPA that RPAs will be provided on the following day.

On that same date, EPA downloaded the draft Opinion file from NMFS' ftp site and requested NMFS provide word version files for future draft Opinions.

On March 18, 2009, NMFS provided EPA instructions to access a pdf copy of the draft Opinion, including RPAs from NMFS' ftp site. NMFS also FedExed a word file of a full draft Opinion, including RPAs and a separate RPA file to EPA. On this same day, EPA responded to NMFS' query of SLN use of carbaryl in Washington State. EPA's e-mail further stated that Bayer CropScience does not support carbaryl use for adult mosquito treatments although carbaryl is registered for use on a number of sites where it could kill mosquitoes.

On that same date, EPA e-mailed NMFS information on the voluntary cancellation of carbofuran. NMFS replied to EPA's message and reminded EPA of NMFS questions raised in a February 3, 2009, e-mail regarding uses of carbofuran.

On March 24, 2009, EPA notified NMFS via e-mail of receipt of the cd containing the word versions of a complete draft Opinion and separate RPA document.

On March 30, 2009, EPA and NMFS confirmed their availability to meet with the three identified applicants for this consultation on April 7, 2009.

On April 7, 2009, EPA, NMFS, and the applicants met at EPA's office in Crystal City, VA. NMFS presented its assessment and conclusions on the draft Opinion. The applicants provided a combined presentation on their general and specific comments on NMFS' draft Opinion. Afterwards, all parties discussed potential Reasonable and Prudent Alternatives (RPAs) as NMFS concluded that EPA's proposed action will likely result in jeopardy of listed species and adverse modification of critical habitat for listed Pacific salmonids under NMFS' jurisdiction.

On that same date, NMFS and EPA discussed RPAs for this consultation and for NMFS' November 18, 2008, Opinion for chlorpyrifos, diazinon, and malathion.

On April 9, 2009, NMFS received copies of the three presentations made by the applicants at the April 7, 2009, meeting.

On April 10, 2009, NMFS received written comments from EPA on the March 18, 2009 draft Opinion for carbaryl, carbofuran, and methomyl. On that same date, NMFS received comments on the draft Opinion from Bayer CropScience on carbaryl and from DuPont on methomyl.

On April 15, 2009, NMFS received additional comments from Bayer CropScience on the draft Opinion.

On April 18, 2009, NMFS received additional comments from Bayer CropScience on the draft Opinion.

Description of the Proposed Action

The Federal Action

The proposed action encompasses EPA's registration of the uses (as described by product labels) of all pesticides containing carbaryl, carbofuran, and methomyl. 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 (re-registrations and special review), 18 (emergency use), or 24(c) 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 ingredients may include active and other ingredients, 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 Federal Food, Drug, and Cosmetic Act (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 cancelled whenever "a pesticide or its labeling or other material… does not comply with the provisions of FIFRA or, when used in accordance with widespread and commonly recognized practice, generally causes unreasonable adverse effects on the environment."

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

Pesticide Labels. For this consultation, EPA's proposed action encompasses all approved product labels containing carbaryl, carbofuran, or methomyl; their degradates, metabolites, and formulations, including other ingredients within the formulations; adjuvants; tank mixtures; and their individual and collective interactions when applied in agricultural, urban, and residential landscapes throughout the U.S. and its territories. These activities comprise the stressors of the action (Figure 1). The three biological evaluations (BEs) indicate that carbaryl, carbofuran, and methomyl are labeled for a variety of uses including applications to residential areas and crop lands (EPA 2003; EPA 2003; EPA 2004). Modifications have been made or are planned for new product labels containing carbaryl, carbofuran, and methomyl as a result of reregistration activities that have occurred since the release of the BEs.

The Food Quality Protection Act (FQPA) of 1996 required EPA to complete an assessment of the cumulative effects on human health resulting from exposure to multiple chemicals that have a common mechanism of toxicity. In 2001, EPA identified the *N*-methyl carbamate (NMC) pesticides as a group which shares a common mechanism of toxicity. This group includes carbaryl, carbofuran, methomyl, and seven other cholinesterase-inhibiting pesticides. (http://www.epa.gov/pesticides/cumulative/carbamate_risk_mgmt.htm). EPA published a preliminary Cumulative Risk Assessment for NMC pesticides in 2005 and revised the risk assessment in 2007 (EPA 2007). Concurrent with completing the revised assessments, EPA completed tolerance reassessments and REDs for the NMC pesticides (EPA 1998; EPA 2006; EPA 2007; EPA 2008). EPA has identified measures to address cumulative risk of NMC pesticides (http://www.epa.gov/pesticides/cumulative/carbamate_risk_mgmt.htm). Some of the risk reduction measures for carbaryl, carbofuran, and methomyl follow:

Carbaryl – EPA intends to evaluate the revised worker assessment, which may require an amendment to the RED. EPA continues to respond to petitions requesting that carbaryl be cancelled and its tolerances revoked.

Carbofuran – EPA is pursuing cancellation of all carbofuran uses in the U.S. EPA has received a request from the carbofuran registrant, FMC Corporation, for voluntary cancellation of 22 crop uses of this pesticide. FMC Corporation has six uses not proposed for voluntary cancellation that EPA indicates still present risk concerns and are subject to future regulatory action by EPA. In July 2008, EPA initiated action to revoke existing carbofuran tolerances (residue limits in food) due to unacceptable dietary risks, especially to children, from consuming food or water alone or from a combination of food and water with carbofuran residues. Following resolution of the tolerance revocations, EPA plans to proceed with cancellation of remaining carbofuran uses due to unreasonable ecological and worker risks (Federal Register / Vol. 73, No. 245 / December 19, 2008 /77690-77693). In March 2008, EPA announced an order for the cancellation of some registrations and termination of certain uses, voluntarily requested by the registrant (Federal Register/Vol. 74, No. 51/ March 18, 2009/11551-11553).

Methomyl – The intent of registrants for voluntary cancellation of methomyl use on strawberry and grapes were incorporated in the *N*-methyl carbamate revised cumulative risk assessment. With these and other mitigation measures for these individual pesticides, EPA concluded that the cumulative risks to humans associated with the *N*-methyl carbamates are below EPA's regulatory level of concern (http://www.epa.gov/pesticides/cumulative/carbamate_risk_mgmt.htm). The FQPA does not address cumulative risk of pesticides to aquatic resources.

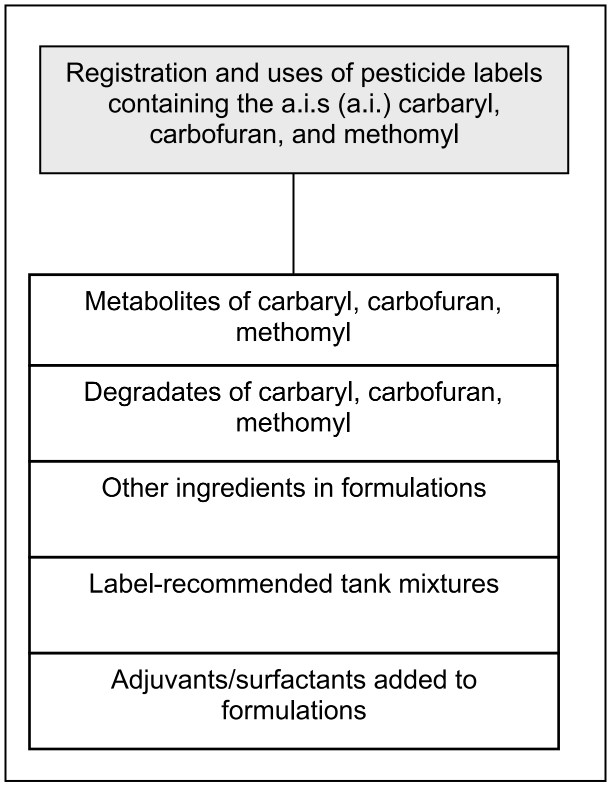


Figure 1. Stressors of the Action

Mode of Action of Carbamate Insecticides. NMC insecticides are neurotoxicants, affecting the central and peripheral nervous systems of animals. Similar to other carbamate and organophosphate (OP) insecticides, these a.i.s inhibit the enzyme acetylcholinesterase (AChE) found in brain and muscle tissue of invertebrates and vertebrates. Thus, NMCs belong to a class of insecticides known as AChE inhibitors. Inhibition of AChE results in a build-up of the neurotransmitter, acetylcholine, which can lead to continued stimulation. Normally, acetylcholine is broken down rapidly in the nerve synapse by AChE. Chemical neurotransmission and communication are impaired when acetycholine is not quickly degraded in animals, which ultimately may result in a number of adverse responses from physiological and behavioral modification to death.

NMFS batched the consultations on carbaryl, carbofuran, and methomyl into one Opinion because these compounds have the same mechanism of action, *i.e.*, they target the same site of action in the exact same way. However, NMFS evaluated the effects of each a.i. independently. Additionally, cumulative exposure to the three a.i.s is expected given they have overlapping uses and detections in surface water samples.

Active and Other Ingredients. Carbaryl, carbofuran, and methomyl 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, and powders. 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 results of EPA's reviews are summarized in RED documents. EPA issued REDs for carbaryl and methomyl in 2007 and 1998, respectively. The IRED for carbofuran was issued in August 2006. EPA considered the registration eligibility determination for carbofuran complete upon issuance of the cumulative assessment for the NMC pesticides in 2007 (http://www.epa.gov/pesticides/reregistration/REDs/carbofuran_red.pdf). Accordingly, EPA treated the carbofuran IRED as its RED. The REDs for all three a.i.s include various mitigation measures, including the cancellation of all registered uses of carbofuran due to ecological and occupational risks. These mitigation components were considered part of the proposed action.

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

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

Registration Information of Pesticide A.i.s under Consultation. As discussed above, the proposed action encompasses EPA's registration of the uses (as described by product labels) of all pesticides containing carbaryl, carbofuran, or methomyl. However, EPA did not provide copies of all product labels containing these a.i.s. The following descriptions represent information acquired from review of a sample of current product labels as well as information conveyed in the BEs, EPA REDs, and other documents.

Carbaryl

Carbaryl, also known by the trade name *Sevin*, is an NMC insecticide which was first registered in 1959 for use on cotton. In 2001, EPA identified the NMC insecticides as a group which shares a common mechanism of toxicity. Therefore, EPA was required to consider the cumulative effects on human health resulting from exposure to this group of chemicals when considering whether to establish, modify, or revoke a tolerance for pesticide residues in food, in accordance with the FQPA (EPA 2008).

Several regulatory documents concerning carbaryl were issued after EPA's BE of the analysis of risk of carbaryl to threatened and endangered salmonids (EPA 2003). An IRED for carbaryl that addressed the potential human health and ecological risks was signed on June 30, 2003. EPA amended the IRED on October 22, 2004, to incorporate clarifications and corrections, updated the residential risk assessment to reflect the voluntary cancellation of the liquid broadcast use of carbaryl on residential turf to address post-application risk to toddlers identified in the 2003 IRED, and addressed issues regarding labeling of carbaryl formulations for mitigating potential hazards to bees. In addition, mitigation measures required in the 2004 amended IRED included cancellation of certain uses and application methods, reduction of application rates, application prohibitions, personal protective equipment (PPE) and engineering control (EC) requirements, and extension of restricted-entry intervals (REIs) for post-application exposure (EPA 2008).

EPA also issued generic and product-specific data call-ins (DCIs) for carbaryl in March 2005. The carbaryl generic DCI required several studies for the a.i. carbaryl, including additional toxicology, worker exposure monitoring, and environmental fate data. The product DCI required acute toxicity and product chemistry data for all pesticide products containing carbaryl. In response to the 2005 DCIs, many carbaryl registrants chose to voluntarily cancel their carbaryl products. Approximately 80% of all of carbaryl end-use products registered at the time of the 2003 IRED have since been cancelled through this process or other voluntary cancellations (EPA 2008).

On September 26, 2007, EPA published a revised NMC cumulative risk assessment (EPA 2007), which concluded that the cumulative risks associated with the NMC pesticides meet the safety standard set forth in the FFDCA. Concurrently, on September 26, 2007, the RED for carbaryl was completed. The 2007 RED presents EPA's revised carbaryl human health risk assessment under FQPA and EPA's final tolerance reassessment decision for carbaryl. EPA amended the carbaryl RED in August of 2008. The amendment updated the 2007 RED to reflect the Revised Occupational Exposure and Risk Assessment, dated July 9, 2007 (EPA 2008).

The National Pesticide Information Retrieval System (NPIRS) website (http://ppis.ceris.purdue.edu/htbin/epachem.com) suggests that there are currently 24 registrants with active registrations of 87 pesticide products containing carbaryl. "Carbaryl is nationally registered for over 115 uses in agriculture, professional turf management, ornamental production, and residential settings (EPA 2007). Carbaryl is also registered for use as a mosquito adulticide. (http://www.umass.edu/fruitadvisor/NEAPMG/145-149.pdf)(EPA 2007)."

Several product labels indicate carbaryl is commonly formulated with other a.i.s. For example, there are active registrations of carbaryl products that also contain copper sulfate, rotenone, malathion, captan, metaldehyde, and bifenthrin (NPIRS website). According to EPA's BE, 26 carbaryl products are registered to individual states under

SLN provisions in Section 24(c) of FIFRA (EPA 2003). Section 24(c) registrations include control of shrimp in oyster beds in two tideland areas (Willapa Bay and Grays Harbor) in Washington and, in California, insecticidal use on fruits and nuts, prickly pear cactus, ornamental plants, and non-food crops. Idaho and Oregon do not have any 24(c) registrations for carbaryl (EPA 2003).

Usage Information.

The insecticide carbaryl is used in agriculture to control pests on terrestrial food crops including fruit and nut trees, many types of fruit and vegetables, and grain crops; cut flowers; nursery and ornamentals; turf, including production facilities; greenhouses; golf courses; and in oyster beds. Carbaryl is also registered for use on residential sites (*e.g.*, annuals, perennials, shrubs) by professional pest control operators and by homeowners on gardens, ornamentals, and turfgrass (EPA 2008). EPA estimated over 1.4 million pounds (lbs) of carbaryl are applied each year on agricultural crops and over 200,000 lbs are applied annually for turf, landscape, and horticultural uses in the U.S. (EPA 2008).

The California Department of Pesticide Regulation (CDPR) indicated approximately 150,000 – 250,000 lbs of carbaryl were applied annually in California between 2002 and 2006 based on agricultural and other "reportable uses" (CDPR 2007). The 1999 Oregon Legislature authorized development of the Oregon Pesticide Use Reporting System (PURS). In 2006, information on household pesticide use was collected through a pesticide use survey. The first full year of collecting non-household pesticide use in PURS was 2007. Over 37,000 lbs of carbaryl were reported as applied in Oregon in 2007 (ODA 2008). Approximately 189,600 lbs of carbaryl are used annually for agriculture in Washington State (WSDA 2004). Similar data on pesticide use were not found for Idaho.

Examples of Registered Uses.

***Agricultural Uses*.** Carbaryl is used on a myriad of crops. Examples of crops currently proposed for continued carbaryl use and which are grown in areas with Pacific salmon and steelhead include cranberries, cucumbers, beans, eggplant, grapefruit, grapes, hay, lemons, lettuce, nectarines, olives, onions, oranges, parsley, peaches, peanuts, pears, pecans, peppers, pistachios, plums, potatoes, prunes, pumpkins, rice, sod, spinach, squash, strawberries, sugar beets, sunflowers, sweet corn, sweet potatoes, tangelos, tangerines, tomatoes, walnuts, watermelons, wheat (EPA 2008). Carbaryl is also used to thin fruit in orchards to enhance fruit size and enhance repeat bloom.

Non-agricultural Uses. Carbaryl is used extensively by homeowners, particularly for lawn care (EPA 2008). Examples of non-agricultural use sites include home and commercial lawns, flower beds around buildings, recreation areas, golf courses, sod farms, parks, rights-of-way, hedgerows, Christmas tree plantations, oyster beds, rural shelter belts, and applications to control ticks, grasshoppers, and adult mosquitoes (EPA 2007). Carbaryl is also used for pet care (pet collars, powders and dip, in kennels, and on pet sleeping quarters).

Examples of Registered Formulation Types. Carbaryl products are manufactured as granular, liquid, wettable powder, and dust formulations. All dry flowable (water dispersible granule) products have been voluntarily cancelled. The use of dust formulation in agriculture and backpack sprayers are not supported by Bayer CropScience, the carbaryl technical registrant, who is amending its carbaryl registrations to delete these uses (EPA 2008).

Methods and Rates of Application.

Methods. Groundboom, airblast, and aerial applications are typical for agricultural uses of carbaryl. Other applications can also be made using handheld equipment, such as low pressure hand wand sprayers, turf guns, and various ready-to-use products. Applications by aerosol cans, hand, spoon, shaker can, and front- and back-mounted spreaders are prohibited (EPA 2008).

Application Rates. The maximum single application rate allowed on the labels for agricultural uses in California, Idaho, Oregon, and Washington is 12 lb carbaryl/acre (Table 1). Many agricultural uses allow repeated application of carbaryl at intervals of 7-14 days. Application intervals are not specified for some uses (*e.g.*, flower beds, home use on fruits, and vegetables). Additionally, some uses do not specify the maximum number of applications (*e.g.*, prickly pear, ticks, and grasshoppers).

Table 1. Registered uses and application rates for carbaryl in California, Idaho, Oregon, and Washington (EPA 2007)

Use Site	Application			
	Maximum Single Application Rate (lb a.i./acre)	Maximum Number of Applications	Minimum Application Interval (days)	Maximum per year (lb a.i./acre)
Home Lawn	9.1	2	7	Not specified
Fire ants	7.4	2	7	Not specified
Flower beds around buildings	8	4	None	6
Lawns, recreation areas, golf courses, sod farms, commercial lawns	8	2	7	8
Parks	8	1	-	8
Citrus	12	1	-	12
Citrus	7.5	8	14	20
Olives	7.5	2	14	Not specified
Almonds, chestnuts, pecans, filberts, walnuts, pistachio	5	4	7	15
Flowers, shrubs	4.3	3	7	Not specified
Apricot, cherries, nectarines, peaches, plums, prunes	4 (5 dormant)	3 + 1 dormant	15	9 + 5 dormant spray
Apple, pear, crabapple, oriental pear, loquat	3	8	14	Not specified
Sweet corn	2	8	3	Not specified
Caneberries, blueberries, grapes, strawberries	2	5	7	Not specified
Tomatoes, peppers, eggplant	2	7	7	8
Peanuts	2	5	7	8
Broccoli, cauliflower, cabbage, kohlrabi, Chinese cabbage, collards, kale, mustard greens, brussel sprouts, hanover salad	2	4	6	7
Sweet potato	2	8	7	8
Field corn, pop corn	2	4	14	8
Leaf lettuce, head lettuce, dandelion, endive, parsley, spinach, swiss chard	2	5	7	6
Celery, garden beets, carrots, horseradish, parsnip, rutabaga, potato, salsify, root turnip, radish	2	6	7	Not specified
Prickly pear	2	As needed	7	6
Rice	1.5	2	7	4
Fresh beans, dry beans, fresh peas, dry peas, cowpeas, fresh southern peas, soybeans	1.5	4	7	6
Sugar beets, pasture, grass for seed	1.5	2	14	3
Alfalfa, birdsfoot trefoil, clover	1.5	1/cutting	None	Not specified
Rangeland	1	1	-	1
Cucumber, melon, pumpkin, squash, roses, other herbaceous	1	6	7	Not specified

Use Site	Application			
	Maximum Single Application Rate (lb a.i./acre)	Maximum Number of Applications	Minimum Application Interval (days)	Maximum per year (lb a.i./acre)
plants, woody plants				
CRP acreage, set-aside acreage, rights-of-way, hedgerows, ditch banks, roadsides, wasteland	1	2	14	3
Non-urban forests, tree plantations, Christmas trees, parks, rangeland trees, rural shelter belts	1	2	7	Not specified
adult mosquitoes[10]	1	*	*	Not specified
Ticks	2	As needed	None	Not specified
Grasshoppers	1.5	As needed	None	Not specified
flax	1.5	2	14	Not specified
Home fruits and vegetables	1.95	6	None	12.1
Proso millet, wheat	1.5	2	14	Not specified
lentils	1.5	4	7	Not specified
Oyster beds	8	Not specified	None	Not specified
Sunflower	1.5	2	7	Not specified
Tobacco	2	4	7	Not specified

*Number of applications and interval as specified for use site (pastures, rangeland, forests and wastelands, etc.).

Metabolites and Degradates.

The major metabolite of carbaryl degradation by both abiotic and microbially mediated processes is 1-naphthol. This degradate represented up to 67% of the applied carbaryl in degradation studies. It is also formed in the environment by degradation of naphthalene and other polyaromatic hydrocarbon compounds. EPA reports that only limited information on the environmental transport and fate of 1-naphthol is available and indicates this compound is less persistent and less mobile than the parent carbaryl (EPA 2003).

Carbofuran

Carbofuran is a NMC systemic pesticide first registered in the U.S. in 1969. The BE for carbofuran indicates it is registered as a restricted use broad spectrum insecticide, nematicide, and miticide for use on a wide variety of agricultural and non-agricultural crops (EPA 2004). Carbofuran is classified as a restricted use pesticide and is formulated into flowable, wettable powder, and granular forms. Through an agreement between EPA and the technical registrant in 1991, granular carbofuran has been limited to the sale of 2,500 lbs of a.i. per year in the U.S. since 1994, for use only on certain crops. Today granular carbofuran is limited to use on spinach grown for seed, pine seedlings, bananas (in Hawaii only), and cucurbits only (EPA 2006).

In the late 1990s, the technical registrant made a number of changes to labels in order to reduce human health (drinking water) and ecological risks of concern. These included reducing application rates and numbers of applications for alfalfa, cotton, corn, potatoes, soybeans,sugarcane, and sunflowers. Numbers of applications were also restricted on some soils to reduce groundwater concentrations (EPA 2006).

Several regulatory documents concerning carbofuran were issued after EPA's BE of the analysis of risk of carbofuran to threatened and endangered salmonids (EPA 2004). An IRED for carbofuran was published in August 2006. As previously indicated, EPA concluded the NMC cumulative risk assessment in September 2007. All tolerance reassessment and REDs for individual NMC pesticides were considered complete. The carbofuran IRED, therefore is considered a completed RED.

The carbofuran IRED (EPA 2006), and draft Notice of Intent to Cancel Carbofuran (January 2008) indicate EPA proposes cancellation of all uses of carbofuran, due to ecological, occupational, and human dietary risks of concern from some crops. Economic benefits are low to moderate for all of these uses, and do not outweigh the risks (EPA 2006). There are several uses for which residues do not pose dietary risks of concern *and* which have moderate benefits to growers [artichokes, chile peppers in the Southwestern U.S., cucurbits (granular formulation only), spinach grown for seed, sunflowers, and pine seedlings in the Southeastern U.S.]. For these uses, EPA is allowing a four-year phase-out in order to allow time for new alternatives to become available to growers (EPA 2006).

Although EPA determined that all uses of carbofuran are ineligible for reregistration, use of carbofuran will continue for an undetermined period of time. EPA has initiated cancellation procedures for product uses of low economic benefits. The remaining uses are subject to a four year phase-out. However, final cancellation of all carbofuran uses may take several years and the decision to cancel carbofuran registrations could be subject to legal challenges. Additionally, EPA indicated that FMC wishes to retain registrations for six uses: corn, potatoes, pumpkins, sunflowers, pine seedlings, and spinach grown for seed (Jones 2009). FMC also proposed to phase out use of artichokes over two years. EPA plans to consider FMC's proposal for the continued registration of carbofuran at a future date.

The IRED indicated there are currently one technical, two manufacturing-use, and six end-use products registered under Section 3 of FIFRA. There are also 77 active SLN registrations under Section 24(c) of FIFRA (EPA 2006). The NPIRS website suggests one registrant holds nine active registrations of pesticide products containing carbofuran (EPA registration numbers: 279-2712, 279-2862, 279-2874, 279-2876, 279-2922, 279-3023, 279-3038, 279-3060, and 279-3310). Many carbofuran uses and products were recently canceled through voluntary requests of the product registrant. Under the conditions of the cancelation order, existing stocks of these carbofuran products may be sold or used until they are depleted, opr until the effective date for the revocation of the associated tolerances (Federal Register/Vol. 74, No. 51/ March 18, 2009/11551-11553).

Usage Information

Nearly one million lbs a.i. are applied annually from the application of liquid carbofuran formulations (EPA 2006). The major use of liquid formulations of carbofuran is on corn, alfalfa, and potatoes. Under the existing terms and conditions of the registration, sale of the granular formulation is limited to 2,500 lbs a.i. per year, and use is limited to pine seedlings, cucurbits, bananas (in Hawaii only), and spinach grown for seed (EPA 2006). Carbofuran use has decreased significantly in California over the last decade. CDPR indicates agricultural uses of carbofuran exceeded 200,000 lbs in 1996 and use declined each preceding year with approximately 23,000 lbs of carbofuran applied in California in 2006 (CDPR 2007). Multiyear use statistics for describing temporal trends of carbofuran were unavailable for Idaho, Oregon, and Washington. It is anticipated that use of carbofuran will decrease across the action area as cancellation orders are implemented and existing stocks are depleted.

Examples of Registered Uses.

Food Crops. Alfalfa, artichoke, banana, barley, coffee, corn (field, pop, and sweet), cotton, cucurbits (cucumber, melons, and squash), grapes, oats, pepper, plantain, potato, sorghum, soybean, sugar beet, sugarcane, sunflower, and wheat (EPA 2006).

Non-food uses. Agricultural fallow land, cotton, ornamental and/or shade trees, ornamental herbaceous plants, ornamental non-flowering plants, ornamental woody shrubs and vines, pine, spinach grown for seed, and tobacco (EPA 2006).

Examples of Registered Formulation Types.

Carbofuran is formulated into flowable, wettable powder, and granular forms. The flowable formulation constitutes the vast majority of the carbofuran currently used (EPA 2004).

Examples of Approved Methods and Rates of Application.

Equipment. Carbofuran is applied by aerial equipment, chemigation systems, groundboom sprayers, airblast sprayers, tractor-drawn spreaders, push-type granular spreaders, and handheld equipment (EPA 2006).

Method and Rate. Carbofuran can be applied as a foliar or soil treatment. Maximum single and seasonal application rates range from 0.002 to 10 lbs a.i./acre, depending on the application scenario (EPA 2006).

Table 2. Registered uses and application rates for carbofuran in California, Idaho, Oregon, and Washington (EPA 2004)

Use Site	Application			
	Maximum Single Application Rate (lb a.i./acre)	Maximum Number of Applications	Minimum Application Interval (days)	Maximum per year (lb a.i./acre)
Flowable Carbofuran – Section 3 registrations				
Alfalfa, corn, cotton	1	1	-	1
Ornamentals	0.06	Not specified	Not specified	0.06
Pine seedlings	0.05	1	-	Prepare slurry: Add 0.05 lbs a.i., 0.5 gallons water, and 2.0 lbs clay; Slurry sufficient to treat roots of 150 to 200 seedlings.
Potatoes	1	2	Not specified	2
Small grains, soybeans	0.25	2	Not specified	0.5
Sugarcane	0.75	2	Not specified	1.5
Sunflowers	1.4	1	-	1.4
Tobacco	6	Not specified	Not specified	6
Granular Carbofuran – Section 3 registrations				
Bananas	0.006	2	Not specified	0.012
Pine seedlings	0.002	Not specified	Not specified	0.002
Rice[1]	0.5	1	-	0.5
California- Flowable Carbofuran- Section 24C				
Artichokes	1	2	Not specified	2
Grapes	10	1	-	10
Ornamentals	10	Not specified	Not specified	10
Idaho- Flowable Carbofuran- Section 24C				
Potatoes	3	2	Not specified	6
Sugar beets	2	1	-	2
Oregon- Flowable Carbofuran- Section 24C				
Potatoes	3	2	Not specified	6
Nursery stock	10	Not specified	Not specified	10
Sugar beets	2	1	-	2
Oregon- Granular Carbofuran- Section 24C				
Watermelons	1	Not specified	Not specified	1
Washington- Flowable Carbofuran- Section 24C				
Potatoes	3	2	Not specified	6
Washington- Granular Carbofuran- Section 24C				
Spinach (grown for seed)	1	1	-	1

[1]The section 3 registration for rice was discontinued in 1997. Additional use of carbofuran on rice since that time has been from existing stock or in connection with emergency exemption requests (EPA 2004a).

Timing. Carbofuran is a contact insecticide applied at planting or post-planting (EPA 2006). The timing is variable among crops.

Metabolites and Degradates. The major transformation product of carbofuran in water and aerobic aquatic metabolism is the hydrolysis product, carbofuran 7-phenol (EPA 2006). It also appears as the transformation endpoint prior to conversion to CO_2 and is shorter lived in water than the parent. Other major expected environmental transformation products in soils that have potential to reach the aquatic environment are 3-hydroxycarbofuran and 3-ketocarbofuran, which typically occur in small amounts (*i.e.*, < 5.0 % of applied) and are relatively short lived as compared to the parent (EPA 2006).

Methomyl

Methomyl was first registered for use in the U.S. in 1968. Methomyl is currently registered for use on a wide variety of sites including field, vegetable, and orchard crops; turf (sod farms only); livestock quarters; commercial premises; and refuse containers (EPA 2007). All methomyl products, except the 1% bait formulations, are classified as restricted use pesticides (EPA 2007). A Registration Standard issued in April 1989 required additional testing, modified tolerances. It also required label modifications related to applicator safety, re-entry intervals, and environmental hazards (EPA 2007). Additional label modifications were required with the publication of the methomyl RED in 1998 (EPA 1998).

EPA's BE of the analysis of risks of methomyl to threatened and endangered salmonids indicated there were 10 end-use products registered under Section 3 of FIFRA (EPA 2003). The NPIRS website suggests that there are currently six registrants with active registrations for nine products containing methomyl (EPA registration numbers: 270-255, 352-342, 352-361, 352-366, 352-384, 2724-274, 7319-6, 53871-3, 5742-2). Eighteen additional methomyl products are registered to individual states under SLN provisions in Section 24(c) of FIFRA (EPA 2003). California has seven SLN for use to control insects on ornamentals, beans, soybeans, radishes, sweet potatoes, Chinese broccoli, broccoli raab, and pumpkins (EPA 2003). Idaho, Oregon, and Washington do not have any SLNs for methomyl (EPA 2003). Methomyl also was previously registered as a molluscicide to control snails and slugs and as a fungicide for control of blights, rots, mildews and other fungal diseases. Those uses, as well as uses on ornamentals and in greenhouses, have been cancelled (EPA 2003).

Usage Information

The BE indicated EPA has no recent national data on the amount of methomyl applied annually (EPA 2003). According to the 1998 RED, an estimated 2.5 to 3.5 million lbs of methomyl a.i. were applied annually in the U.S. between 1987 and 1995. CDPR indicates approximately 262 – 554 thousand lbs of methomyl were applied annually in California between 2000 and 2006 based on agricultural and other "reportable uses" (CDPR 2007). Over 42,000 lbs of methomyl were applied in Oregon in 2007 (ODA 2008). Similar data on pesticide use were not found for Idaho and Washington.

Examples of Registered Uses

Agriculture. Methomyl is used for a variety of agricultural uses including alfalfa, anise, asparagus, barley, beans (succulent and dry), beets, Bermuda grass (pasture), blueberries, broccoli, broccoli raab, Brussels sprouts, cabbage, carrot, cauliflower, celery, chicory, Chinese broccoli, Chinese cabbage, collards (fresh market), corn (sweet), corn (field and popcorn), corn (seed), cotton, cucumber, eggplant, endive, garlic, horseradish, leafy green vegetables, lentils, lettuce (head and leaf), lupine, melons, mint, nonbearing nursery stock (field grown), oats, onions (dry and green), peas, peppers, potato, pumpkin, radishes, rye, sorghum, soybeans, spinach, sugar beet, summer squash, sweet potato, tomatillo, tomato, turf (sod farms only), wheat, and orchards including apple, avocado, grapes, grapefruit, lemon, nectarines, oranges, peaches, pomegranates, tangelo, and tangerine (EPA 2007).

Non-agriculture. Methomyl has several non-crop uses that are outside uses involving scatter bait or bait station formulations including the following use sites: bakeries, beverage plants, broiler houses, canneries, commercial dumpsters which are enclosed, commercial use sites (unspecified), commissaries, dairies, dumpsters, fast food establishments, feedlots, food processing establishments, hog houses, kennel, livestock barns, meat processing establishments, poultry houses, poultry processing establishments, restaurants, supermarkets, stables, and warehouses (EPA 2007).

Examples of Registered Formulations and Types.

End-use formulations of methomyl include soluble concentrate, wettable powder, granular, pelleted/tableted, and water soluble packaged. Products registered as fly baits also contain (Z)-9-tricosene (0.04 to 0.26% a.i.) as an a.i.; labels note that these products contain a sex attractant and feeding synergist (EPA 2003).

Examples of Approved Methods and Rates of Application.

Application Equipment. Methomyl can be applied by aircraft; bait box; brush; cup; duster; glove; granule applicator; ground; high volume ground sprayer; low volume ground sprayer; package applicator; scoop; shaker can; shaker jar; sprayer; and ultra low volume sprayer (EPA 1998).

Application Rates. The maximum single application rate allowed on the labels for agricultural uses in California, Idaho, Oregon, and Washington is 0.9 lb a.i./acre. Many agricultural uses allow repeated application of methomyl at relatively short intervals (1-5 days). For example, the application interval for methomyl for sweet corn is one day and methomyl can be applied 28 times within a single crop of sweet corn. Additionally, several crops of sweet corn may be grown per year in some locations within the action area (EPA 2007). The maximum seasonal labeled application rates (indicated on the label as maximum application rates per crop) for agricultural uses range from 0.9 lb a.i./acre/crop [*i.e.*, Bermuda grass (pasture), avocado, lentils, beans (interplanted with trees), sorghum, and soybeans (interplanted with trees)] to 7.2 lbs a.i./acre/crop [*i.e.*, cabbage, lettuce (head), cauliflower, broccoli raab, celery, and Chinese cabbage]. Several methomyl crops can be grown more than one time per year (*i.e.,* they have multiple crop cycles). Therefore, for those methomyl uses that have more than one crop cycle per year, the maximum allowable yearly application rate will be higher than the maximum seasonal application rate. For perennial crops (*e.g.*, alfalfa), the number of cuttings per year was used to determine the number of crop cycles per year. Based on the labeled application rates and information from EPA's OPP Benefits and Economic Analysis Division (BEAD) on the number of times each crop for which methomyl is registered for use can be grown in California, the maximum yearly application rates for methomyl are 32.4 lb a.i./acre/year (alfalfa) and 21.6 lb a.i./acre/year (broccoli raab, cabbage, and Chinese cabbage) for agricultural crops; 5.4 lb a.i./acre/year (peaches) for orchards; and 0.22 lb a.i./acre/application for nonagricultural uses (no maximum application/acre/year is provided on the nonagricultural use labels). All orchard and most agricultural uses involve foliar application. The only granular agricultural/orchard use is for corn which also has a foliar use (EPA 2007).

All non-agricultural outside uses for methomyl in California, Idaho, Oregon, and Washington are limited to scatter baits and bait stations around agricultural (*e.g.*, animal premises) and commercial structures and commercial dumpsters, where children or animals are not likely to contact the pesticide. The scatter bait can also be mixed with water to form a paste which can be brushed onto walls, window sills, and support beams. The maximum application rate for the scatter bait use is 0.22 lb a.i./acre (0.0025 lb a.i./500 ft^2). However, it is unlikely that applications would involve a full acre as the outside use of the scatter bait is limited to areas around structures and dumpsters. No minimum application interval or maximum application rate per year is provided on the scatter bait labels (EPA 2007).

Table 3. Examples of registered uses and application rates for methomyl in California, Idaho, Oregon, and Washington (EPA 2007).

Use Site	Application			
	Maximum Single Application Rate (lb a.i./acre)	Maximum Number of Applications	Minimum Application Interval (days)	Maximum per year (lb a.i./acre)
Commercial dumpsters; poultry houses; unspecified commercial sites; outside commercial uses: feedlots, dairies, stables, broiler houses, hog houses, livestock barns, meat processing establishments, poultry processing establishments, beverage plants, canneries, food processing establishments, kennels, dumpsters, restaurants, supermarkets, commissaries, and bakeries.	0.22	Not specified	1-3	Not specified
Alfalfa	0.9	10 x 9 crops	5	32.4
Asparagus	0.9	5	7	4.5
Avocado	0.9	2	5	0.9
Barley	0.45	4	5	1.8
Beans	0.9	10	5	4.5
Broccoli	0.9	10 x 3 crops	5	21.6
Cabbage	0.9	15 x 3 crops	2	21.6
Corn	0.45	28 x 3 crops	1	18.9
Lettuce	0.9	8 x 2 crops	2	14.4

Use Site	Application			
	Maximum Single Application Rate (lb a.i./acre)	Maximum Number of Applications	Minimum Application Interval (days)	Maximum per year (lb a.i./acre)
Onions	0.9	8 x 3 crops	5	16.2
Spinach	0.9	8 x 3 crops	5	10.8
Turf	0.9	4 x 2 crops	5	7.2

Timing. The timing of application is dependent on use, but may occur throughout the year. In most cases multiple applications are allowed to maintain pest control. Applications occur on fruit crops during the bloom, petal fall, pre-bloom, and leaf stages and when pest pressure is highest on a "When Needed" basis. On corn, application may occur during the whorl/foliar stages. With other crops, application is during the foliar or leaf stages of the crop (EPA 1998).

Metabolites and Degradates

Several degradates and metabolites have been identified for methomyl. The major degradate in most metabolism studies was CO^2. Another degradate, S-methyl-N-hydroxythioacetamidate, which is highly mobile, appears primarily as a product of alkaline hydrolysis. In an aquatic metabolism study, methomyl degraded with estimated half-lives of four to five days. After seven days, acetonitrile comprised a maximum of 17% and acetamide up to 14% of the amount of methomyl applied. After 102 days, volatilized acetonitrile totaled up to 27% of the applied and CO_2 up to 46% of the applied material (EPA 2003).

Species Addressed in the BEs

EPA's BEs considered the effects of carbaryl, carbofuran, and methomyl to 26 species of listed Pacific salmonids and their designated critical habitat. EPA determined that carbaryl, carbofuran, and methomyl may affect most of these species. Exceptions follow:

EPA concluded that the registration of carbaryl products would have no effect on Northern California steelhead, SONCCal coho, Hood Canal summer-run chum, and Ozette Lake sockeye. EPA also concluded that the registration of carbaryl products would not likely adversely affect California Coastal Chinook and Puget Sound Chinook salmon.

EPA concluded that the registration of carbofuran products would have no effect on Northern California Steelhead, Central California Coast coho, and California Coastal Chinook salmon. EPA also concluded that the registration of carbofuran products would not likely adversely affect Central Valley Spring-run Chinook salmon, Lower Columbia River Chinook salmon, Puget Sound Chinook salmon, Sacramento River winter-run Chinook salmon, Upper Willamette River Chinook salmon, Columbia River chum, Hood Canal summer-run chum, Oregon Coast coho, SONCC coho, Ozette Lake sockeye, Snake River sockeye, Central California Coast steelhead, California Central Valley steelhead, Lower Columbia River steelhead, Snake River steelhead, South-Central California steelhead, Southern California steelhead, and Upper Willamette River steelhead.

EPA concluded that the registration of methomyl products would have no effect on Northern California steelhead and California Coastal Chinook salmon.

Although EPA has determined that its action in registering pesticides containing the three a.i.s is not likely to adversely affect certain ESUs/DPSs and will have no effect on others, EPA initiated formal consultation on its action because EPA concluded that its action may adversely affect other listed ESUs/DPSs. When an action agency concludes that its action will not affect any listed species or critical habitat, then no section 7 consultation is necessary (USFWS and NMFS 1998). If NMFS concurs with a federal agency that its action is not likely to adversely affect any listed species or critical habitat, then formal consultation is not required. Since formal consultation was triggered, NMFS evaluated the federal action and its impacts to all listed Pacific, anadromous salmonids and their designated critical habitat. 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 or not likely to be adversely affected including two species of salmonid listed after EPA provided its BEs.

Approach to this Assessment

Overview of NMFS' Assessment Framework

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

The second step of our analyses identifies the listed resources (endangered and threatened species and designated critical habitat) that are likely to occur in the same space and at the same time as these potential stressors. If we conclude that such co-occurrence is likely, we then try to estimate the nature of co-occurrence (these represent our *Exposure Analyses*). In this step of our analysis, we try to identify the number, age (or life stage), gender, and life history of the individuals that are likely to be exposed to an action's effects and the populations or subpopulations those individuals represent.

Once we identify which listed resources are likely to be exposed to potential stressors associated with an action and the nature of that exposure, in the third step of our analysis we examine the scientific and commercial data available to determine whether and how those listed resources are likely to respond given their exposure (these represent our *Response Analyses*). We integrate the exposure and response analyses to assess the risk to listed individuals and their habitat from the stressors of the action (these represent our *Risk Characterization*). 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. 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 (Mills and Beatty 1979; Stearns 1982; Anderson, Phillips et al. 2006). 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.

Critical habitat analysis focuses on reductions in the quality or quantity of PCEs. The stressors of the action for this Opinion are chemicals and PCEs potentially affected are salmonid prey availability and degradation of water quality in freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, estuarine areas, and nearshore marine areas. Endpoints evaluated for the prey PCE 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. Degradation of water quality was evaluated by considering the information available on the presence of constituents known to adversely affect aquatic organisms (*e.g.,* toxic chemicals, nutrients, sediments), whole effluent test or toxicity indicator evaluations, and/or instances of waterbodies not meeting local, state, or federal water quality criteria.

Evidence Available for the Consultation

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

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

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

Application of Approach in this Consultation

The EPA proposes to authorize the use of over 100 pesticide formulations (pesticide products) containing the a.i.s carbaryl, carbofuran, and methomyl through its authority to register pesticides under FIFRA. Registration by EPA authorizes the use of these formulations in the U.S. and its territories, documented by EPA's approval of registrant-derived pesticide labels. Pursuant to the court's 2002 order in Washington Toxics Coalition v. EPA, EPA initiated consultation on registration of carbaryl, carbofuran, and methomyl for 26 listed ESUs of Pacific salmonids. Since EPA initiated consultation, NMFS has listed one additional Pacific coho ESU and one additional Pacific steelhead DPS. This Opinion represents NMFS' evaluation of whether EPA's authorization of these labels satisfies EPA's obligations to listed salmonids pursuant to section 7(a)(2) of the ESA.

The NMFS evaluates whether endangered species, threatened species, and designated critical habitat are likely to be exposed to the direct and indirect effects of the proposed action. If those listed resources are not likely to be exposed to these activities, we would conclude that EPA's action is not likely to jeopardize the continued existence of threatened species, endangered species, or result in the destruction or adverse modification of designated critical habitat under NMFS' jurisdiction. If, however, listed individuals are likely to be exposed to these actions and individual fitness is reduced, then we evaluate the potential for population-level consequences.

A Viable Salmonid Population (VSP) is an independent population of any Pacific salmonid that has a negligible risk of extinction due to threats from demographic variation, local environmental variation, and genetic diversity changes over a 100-year time frame (McElhaney, Ruckleshaus et al. 2000). The independent population is the fundamental unit of evaluation in determining the risk of extinction of salmon in an ESU. Attributes or metrics associated with a VSP include the abundance, productivity, spatial structure, and genetic diversity of the population. Abundance is defined as the size of the population and can be expressed in a number of ways, *e.g.,* the number of spawning adults, the number of adults surviving to recruit to fisheries, or the number of emigrating smolts. Abundance is a vital measure, as smaller populations run a greater risk of extinction. The second VSP measure is productivity, generally defined as the growth rate of a population. This Opinion discusses productivity in terms of lambda (λ). Appendix 1 contains a more detailed explanation of λ in the context of our population models. The spatial structure of a population is inherently dependant on the quantity and quality of available habitat. A limited spatial structure can hamper the ability of the ESU to respond to evolutionary pressures. Genetic variability within the ESU gives the species the ability to respond to short-term stochastic events, as well as to evolve to a changing environment in the long-term. These VSP parameters provide an indication of the population's capacity to adapt to various environmental conditions and ability to be self-sustaining in the natural environment (McElhaney, Ruckleshaus et al. 2000; McElhaney, Chilcote et al. 2007).

In determining the effect of an action to populations, we first address whether individual fitness level consequences are likely and whether those consequences affect populations. We evaluate whether identified VSP parameters of populations such as abundance and productivity are reduced by individual fitness effects. If populations are likely to be adversely affected by reductions in VSP parameters, we analyze the potential effects to the species as a whole. In parallel, if designated critical habitats are likely to be exposed and PCEs are adversely affected, then we evaluate the potential for reductions in the conservation value of the habitats. We devise risk hypotheses based on identified PCEs that are potentially affected by the stressors of the action. If the best available data indicate that PCE-specific risk hypotheses are supported, then we discuss whether critical habitat will remain functional to serve the intended conservation role for the species in the *Conclusion* section.

General conceptual framework for assessing risk of EPA's pesticide actions to listed resources.

We evaluate the risk to listed species and designated critical habitat in the *Effects of the Proposed Action* section by applying an ecological risk assessment framework that organizes the available information in three phases: problem formulation, analysis, and risk characterization (EPA 1998; McElhaney, Ruckleshaus et al. 2000). We adapted the EPA framework to address ESA-specific considerations (Figure 2). The framework follows a process for organizing, evaluating, and synthesizing the available information on listed resources and the stressors of the action. Below, we briefly describe each phase in the *Effects of the Proposed Action* section.

Problem Formulation

The first phase of the framework is problem formulation. In this phase, we generate conceptual models from our initial evaluation of the relationships between stressors of the action (pesticides and identified chemical stressors and potential receptors (listed species, habitat). We represent these relationships in conceptual models presented as diagrams and written risk hypotheses (EPA 1998). Conceptual model diagrams are constructed to illustrate potential pesticide exposure pathways and associated listed resources' responses. An example of a conceptual model is presented in Figure 3 for Pacific salmonids. In it, we illustrate where the pesticides generally reside in the environment following application, how pesticides may co-occur with listed species and their habitats, and how the individuals/habitat may respond upon exposure to them. In the case of Pacific salmonids, we ascribe exposure and response to specific life stages of individuals and then assess individual fitness endpoints sensitive to the action's stressors.

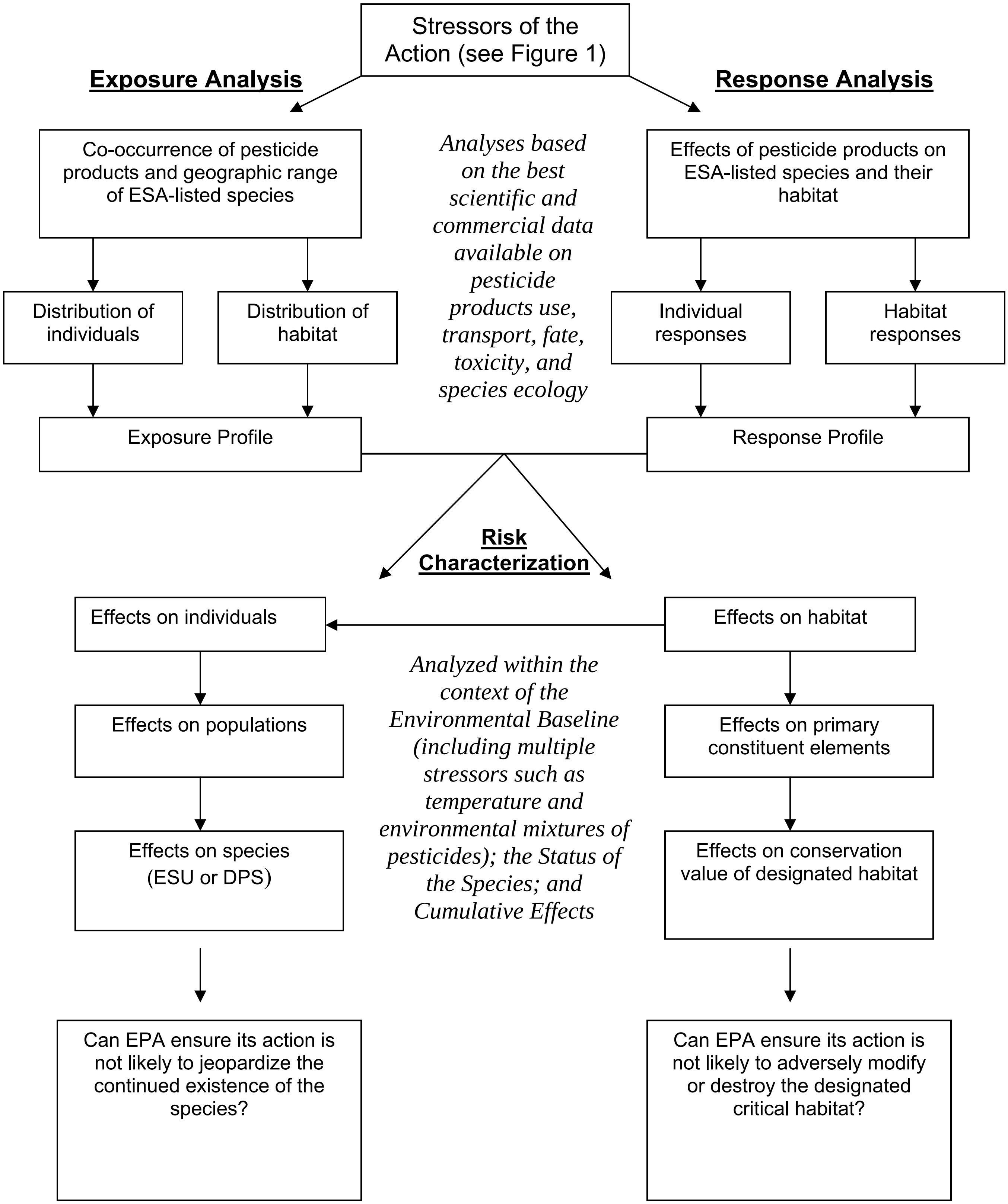


Figure 2. Conceptual framework for assessing risks of EPA's action to listed resources

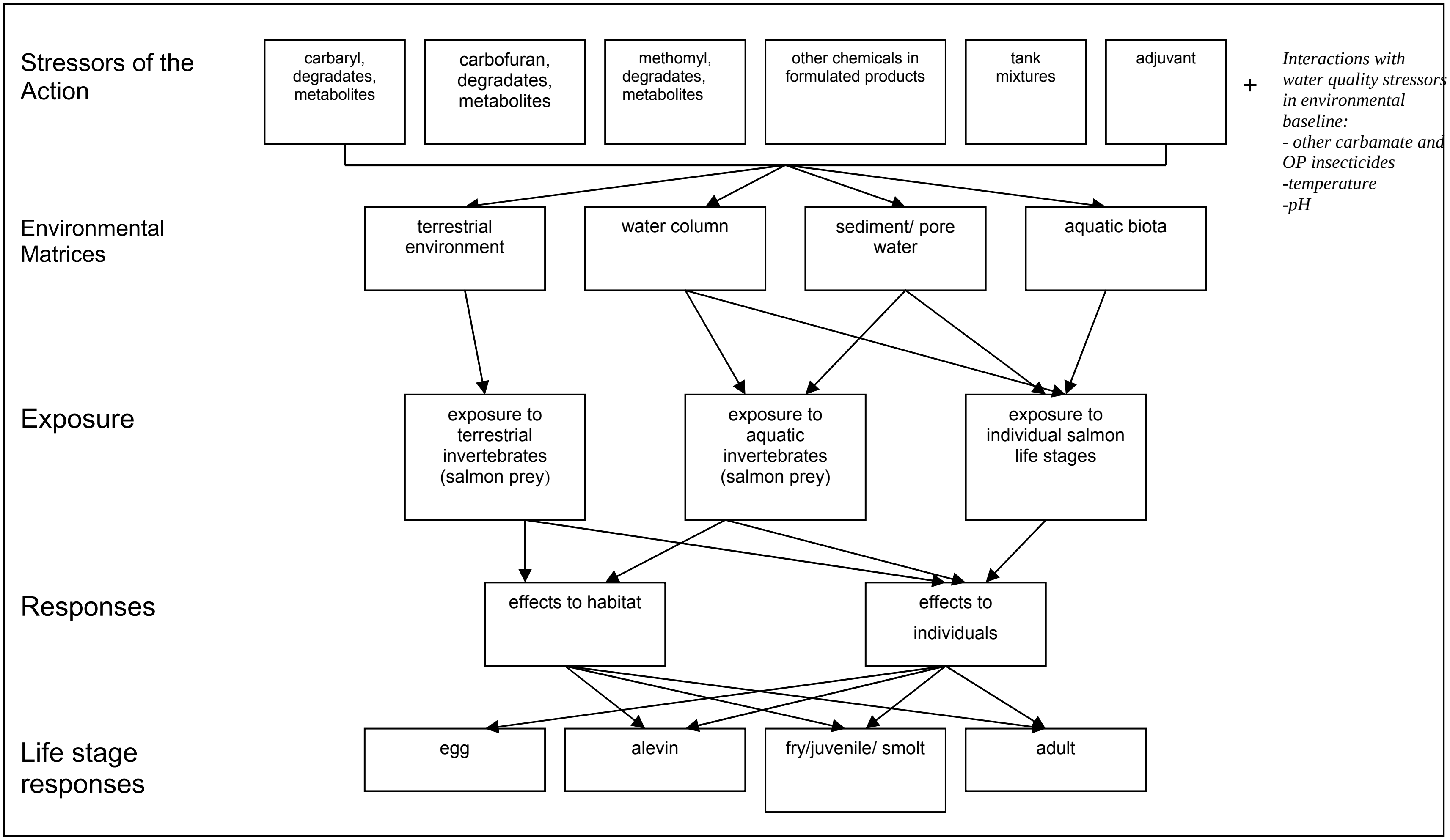


Figure 3. Exposure pathways to carbaryl, carbofuran, and methomyl and general responses of listed Pacific salmonids and habitat.

Species Risk Hypotheses

We construct risk hypotheses by identifying biological requirements or assessment endpoints (Table 4) for listed resources in the action area. We designate assessment endpoints as those biological properties of species and their habitat essential for successful completion of a species life cycle. We integrate the listed resources information with what is known about the stressors of the action, including their physical properties, use, presence in aquatic habitats, and their toxicity. We then evaluate how listed salmonids and their habitat are potentially affected by the stressors of the action and integrate this information with exposure information to develop risk hypotheses. Below are the risk hypotheses (written as affirmative statements) we evaluate in the *Effects of the Proposed Action* section:

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

We discuss an example of one risk hypothesis to show the relationship between assessment endpoints and measures with species responses. In risk hypothesis 1 (d), aquatic exposure to carbaryl, carbofuran, and methomyl can impair a salmonid's nervous system and consequently affect swimming ability of fish. Behavioral modifications, such as changes in swimming performance, are regularly considered in NMFS' Opinions. Swimming performance therefore is an assessment endpoint. Measurable changes in swimming speed are the assessment measure used to evaluate this endpoint. Reductions in swimming performance could also affect other assessment endpoints such as migration and predator avoidance. We may or may not have empirical data that address these endpoints, resulting in a recognized data gap. This uncertainty would be identified during the problem formulation phase, and discussed in the risk characterization phase.

Critical Habitat Risk Hypotheses:

To determine potential effects to designated critical habitat, NMFS evaluates the effects of the action by first looking at the effects on PCEs of critical habitat. 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. As such, the key PCEs that are potentially affected are salmonid prey availability and degradation of water quality in freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, estuarine areas, and nearshore marine areas. We developed two risk hypotheses based on these PCEs:

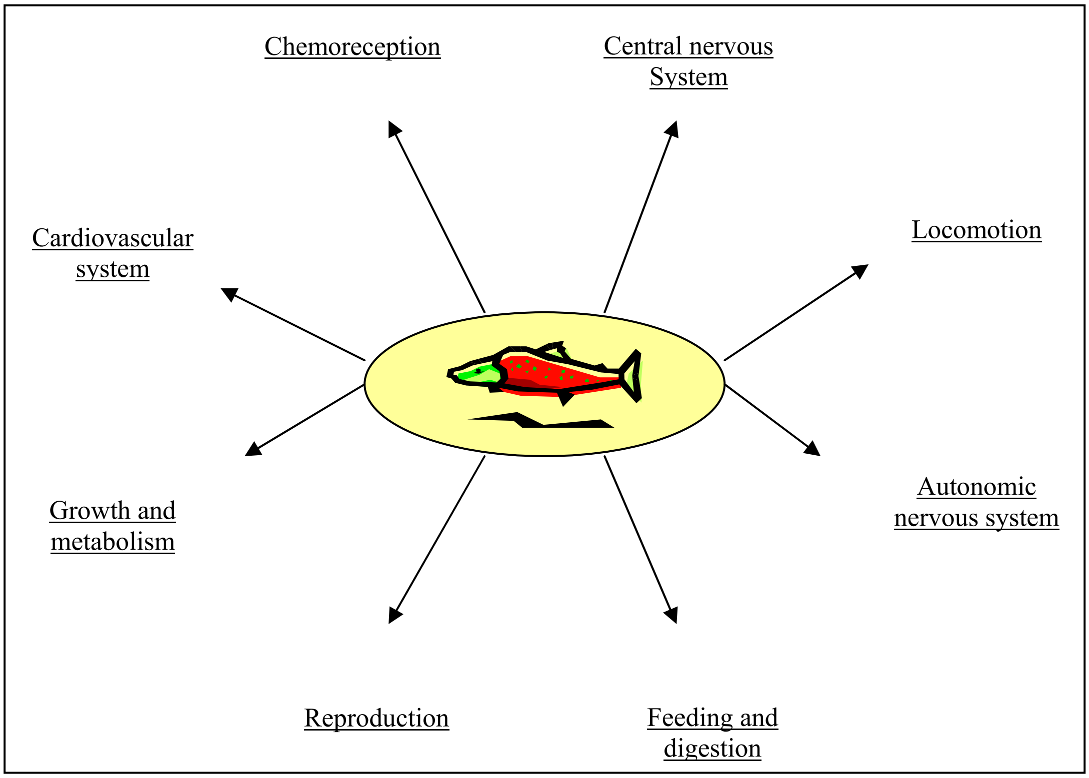
1. Exposure to the stressors of the action is sufficient to reduce abundance of aquatic prey items of salmonids; and
2. Exposure to the stressors of the action is sufficient to degrade water quality in designated critical habitat.

These hypotheses are evaluated using the best scientific and commercial data available presented in the *Response* section. Examples of assessment endpoints evaluated include prey survival, prey growth, prey drift, prey reproduction, abundance of prey, health of invertebrate aquatic communities, recovery of aquatic communities following pesticide exposure, etc. If the available evidence supports the risk hypotheses, then NMFS evaluates whether the potential reductions in PCEs are localized or widespread. The potential reduction of PCEs affect on the conservation value of designated critical habitats is then assessed. This portion of the analysis is conducted in the *Integration and Synthesis* section.

Below we discuss an example of one risk hypothesis to show the relationship between assessment endpoints and measures with species responses. In risk hypothesis 1 (d), aquatic exposure to carbaryl, carbofuran, and methomyl can impair a salmonid's nervous system and consequently affect swimming ability of fish. Behavioral modifications, such as changes in swimming performance, are regularly considered in NMFS' Opinions. Swimming performance therefore is an assessment endpoint. Measurable changes in swimming speed are the assessment measure used to evaluate this endpoint. Reductions in swimming performance could also affect other assessment endpoints such as migration and predator avoidance. We may or may not have empirical data that address these endpoints, resulting in a recognized data gap. This uncertainty would be identified during the problem formulation phase, and discussed in the risk characterization phase.

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

Figure 4. Physiological systems potentially affected by acetylcholinesterase inhibition



In Table 4, assessment endpoints and assessment measures are identified for particular life stages. We focused on the following physiological systems identified in Figure 4: chemoreception, locomotion, feeding, reproduction, and growth. We did not locate any information on the remaining systems in Figure 4. Thus, they were not specifically addressed in our analysis.

We assess the likelihood of these fitness level consequences occurring from exposure to the action. In the exposure analysis (Figure 2), we select exposure estimates for our listed resources derived from reviewing the available exposure data. Depending on the chemicals being evaluated, data may or may not be available for all endpoints and measures, and available data may vary in reliability. Thus, we use a weight-of-evidence approach in this Opinion.

The problem formulation phase as articulated in EPA's 1998 Guidelines for Conducting Ecological Risk Assessment concludes with the development of an analysis plan. In this Opinion, the *Approach to the Assessment* section is the general analysis plan. This section identifies how exposure will be assessed and which assessment endpoints will be evaluated. Therefore, the *Approach to the Assessment* is a road map for evaluating the effects of EPA's registration actions with carbaryl, carbofuran, and methomyl.

Table 4. Examples of salmonid lifestage assessment endpoints and measures

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

Risk Characterization

We follow the framework presented in Figure 2 to conduct the analysis and risk characterization phases. First we conduct exposure and response analyses to estimate/determine the type, likelihood, magnitude, and frequency of adverse responses resulting from predicted exposure based on the best available information. We evaluate species information and pesticide information to determine when, where, and at what concentrations listed salmonids and their habitat may be exposed. We then correlate those exposure estimates with probable response based on available toxicity data. Once we have conducted the analysis phase, we move to the risk characterization phase (Figure 2).

In the risk characterization phase, we revisit the risk hypotheses and apply tools to address whether any individual fitness consequences assessed in the analysis phase would be expected to impact populations and ultimately species. One of the tools we employ is individual-based population models predicated on a juvenile salmonids' probability of survival in its first year of life. We also assess interactions between the stressors of the action and stressors in the *Environmental Baseline* (Figure 2). Some pesticides' toxicity profiles are influenced by environmental parameters such as pH and temperature. Temperature can affect pesticide metabolism in fish and is seasonally elevated in many salmonid supporting watersheds. As described earlier in this section we translate expected effects to identified PCEs by evaluating the available information to support risk hypotheses. If we expect PCEs to be reduced we discuss whether the expected reductions translate to reductions in the conservation value of designated critical habitat.

To conclude consultation, cumulative effects are described and the extent to which species and habitat are affected is documented. Cumulative effects as defined in 50 CFR §404.2 include the effects of future, state, tribal, local, or private actions that are reasonably certain to occur in the action area of this Opinion. Integrating the *Effects of the Proposed Action*, the *Status of Listed Resources*, and the *Environmental Baseline*, NMFS determines whether EPA's pesticide registration action jeopardizes the continued existence of the species. NMFS also determines whether the action results in the 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. Ultimately, our 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, Solomon et al. 1999; Giddings 2009). There is utility in information generated by probabilistic approaches if supported by robust data. We compared the species sensitivity distributions presented in Giddings 2009 with the probability distributions of salmonid prey acute lethality values that we developed to highlight differences in outcomes. The assessment with carbaryl did not address many of the species-specific risk hypotheses. We found no other probabilistic assessments that addressed risk to salmonids affected by short-term sublethal exposures, mixtures, or affects on growth from reduced feeding ability and reduced abundances of prey.

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. In the *Risk Characterization* section, the distribution of the sensitivity of salmonid prey items was used to determine selection of a survival toxicity value used in population modeling exercises. Probabilistic techniques were not otherwise utilized in the Opinion due to issues with data collection, paucity of data, non-normal distributions of data, and quality assurance and quality control. For example, it was not deemed appropriate to pair the salmonid prey responses with exposure probabilities based on monitoring results given the limitations of that data set discussed in the *Effects of the Proposed Action*. To evaluate population consequences associated with potential lethality from pesticide exposure in salmon, NMFS selected the lowest reported salmonid LC50 from the available information to ensure risk was not underestimated. When we consider the data limitations coupled with the inherent complexity of EPA's proposed action (Figure 1) in California, Idaho, Oregon, and Washington, we find that probabilistic assessments at population and species scales introduce an unquantifiable amount of uncertainty that undermines confidence in derived risk estimates. These same studies do not factor the status of the existing health and baseline conditions of the environment into their assessment. At this time, the best available data do not support such an analysis and conclusions from such an analysis would be highly speculative.

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 surface waters of California, Oregon, and Washington. The action area also includes all freshwater surface waters in Idaho (Figure 5).

Carbaryl, carbofuran, and methomyl are the second set of three insecticides identified in the consultation schedule established in the settlement agreement and are analyzed in this Opinion. NMFS' analysis focuses only on the effects of EPA's action on listed Pacific salmonids in the above-mentioned states. It includes the effects of these pesticides on the recently listed Lower Columbia River coho salmon, Puget Sound steelhead, and Oregon Coast coho salmon. The Lower Columbia River coho salmon was listed as endangered in 2005. The Puget Sound steelhead and the Oregon Coast coho salmon were listed as threatened in 2007 and 2008, respectively.

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

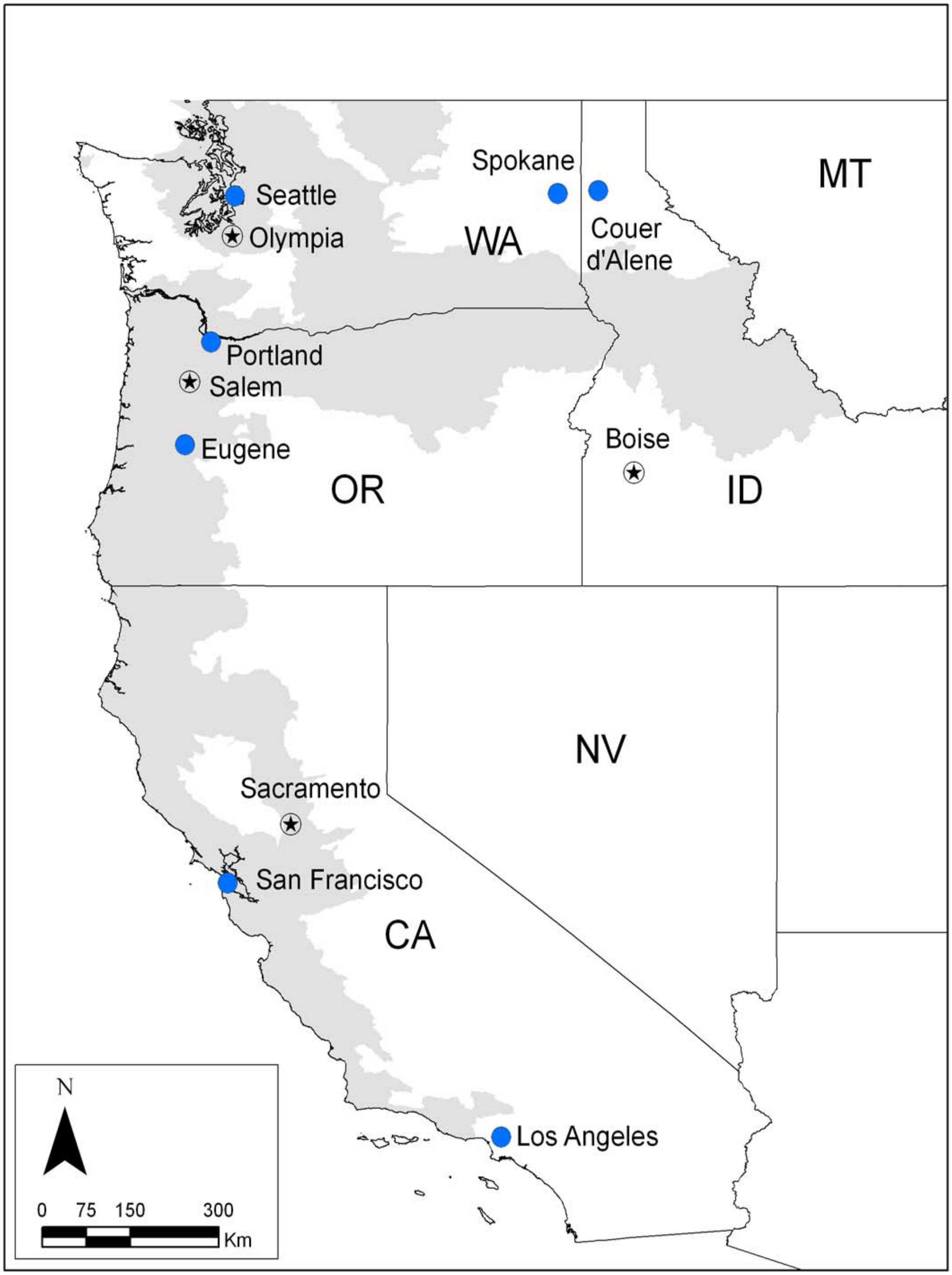


Figure 5. Map showing extent of inland action area with the range of all ESU and DPS boundaries for ESA listed salmonids highlighted in gray.

Status of Listed Resources

NMFS has determined that the following species and critical habitat designations may occur in this action area for EPA's registration of carbaryl, carbofuran, and methomyl -containing products (Table 5). More detailed information on the status of these species and critical habitat are found in a number of published documents including recent recovery plans, status reviews, stock assessment reports, and technical memorandums. Many are available on the Internet at http://www.nmfs.noaa.go/pr/species/.

Table 5. 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 (California Coastal*)	*Oncorhynchus tshawytscha*	Threatened
Chinook salmon (Central Valley Spring-run*)		Threatened
Chinook salmon (Lower Columbia River*)		Threatened
Chinook salmon (Upper Columbia River Spring-run*)		Endangered
Chinook salmon (Puget Sound*)		Threatened
Chinook salmon (Sacramento River Winter-run*)		Endangered
Chinook salmon (Snake River Fall-run*)		Threatened
Chinook salmon (Snake River Spring/Summer-run*)		Threatened
Chinook salmon (Upper Willamette River*)		Threatened
Chum salmon (Columbia River*)	*Oncorhynchus keta*	Threatened
Chum salmon (Hood Canal Summer-run*)		Threatened
Coho salmon (Central California Coast*)	*Oncorhynchus kisutch*	Endangered
Coho salmon (Lower Columbia River)		Threatened
Coho salmon (Southern Oregon & Northern California Coast*)		Threatened
Coho salmon (Oregon Coast*)		Threatened
Sockeye salmon (Ozette Lake*)	*Oncorhynchus nerka*	Threatened
Sockeye salmon (Snake River*)		Endangered
Steelhead (Central California Coast*)	*Oncorhynchus mykiss*	Threatened
Steelhead (California Central Valley*)		Threatened
Steelhead (Lower Columbia River*)		Threatened
Steelhead (Middle Columbia River*)		Threatened
Steelhead (Northern California*)		Threatened
Steelhead (Puget Sound)		Threatened
Steelhead (Snake River*)		Threatened
Steelhead (South-Central California Coast*)		Threatened
Steelhead (Southern California*)		Threatened
Steelhead (Upper Columbia River*)		Threatened
Steelhead (Upper Willamette River*)		Threatened

The following brief narratives summarize the biology and ecology of threatened and endangered species in the action area that are relevant to the effects analysis in this

Opinion. Summaries of the status and trends [including (VSP) information] of each species are presented to provide a foundation for the analysis.

One of the important factors defining a viable population is the population's long- and short-term tendency to increase in abundance. In our status reviews of each listed salmonid species, we calculated the median annual population growth rate (denoted as lambda, λ) from available time series of abundance for individual populations. The lambda for each population is calculated using the rate at which four year running sums of available abundance estimates changes through time. Several publications provide a detailed description of the calculation of lambda (McClure, Holmes et al. 2003; Good, Waples et al. 2005). The lambda values for salmonid VSPs presented in these papers are summarized in Appendix 2. Unfortunately, reliable time series of abundance estimates are not available for most Pacific salmon and steelhead populations. In those cases, we made general inferences of long-term change based on what is known of historical and past abundances from snapshot surveys, surveys of a population segments, harvest by commercial and recreational fisheries, and professional judgment. We then compare these to similar information of current populations.

Below, each species narrative is followed by a description of its critical habitat with particular emphasis on any essential features of the habitat that may be exposed to the proposed action, and may warrant special attention.

Chinook Salmon

Description of the Species

Chinook salmon are the largest of the Pacific salmon and historically ranged from the Ventura River in California to Point Hope, Alaska in North America, and in northeastern Asia from Hokkaido, Japan to the Anadyr River in Russia (Healey 1991). In addition, Chinook salmon have been reported in the Canadian Beaufort Sea (McPhail and Lindsey 1970). We discuss the distribution, life history, diversity (when applicable), status, and critical habitat of the nine species of endangered and threatened Chinook salmon separately.

Of the Pacific salmon species, Chinook salmon exhibit one of the most diverse and complex life history strategies. Chinook salmon are generally described as one of two races, within which there is substantial variation. One form, the "stream-type" resides in freshwater for a year or more following emergence from gravel nests. Another form, the "ocean-type" migrates to the ocean within their first year. The ocean-type typifies populations north of 56ºN (Healey 1991). Within each race, there is often variation in age at seaward migration, age of maturity, timing of spawning migrations, male precocity, and female fecundity.

Status and Trends

Over the past few decades, the size and distribution of Chinook salmon populations have declined because of natural phenomena and human activity. Geographic features, such as waterfalls, pose natural barriers to salmon migrating to spawning habitat. Flooding can eliminate salmon runs and significantly alter large regions of salmon habitat. However, these threats are not considered as serious as several anthropogenic threats. Of the various natural phenomena that affect most populations of Pacific salmon, changes in ocean productivity are generally considered most important. Natural variations in freshwater and marine environments have substantial effects on the abundance of salmon populations.

Salmon along the U.S. west coast are prey for a variety of predators, including marine mammals, birds, sharks, and other fishes. In general, Chinook salmon are prey for pelagic fishes, birds, and marine mammals, including harbor seals, sea lions, and killer whales. Chinook salmon are also exposed to high rates of natural predation, during freshwater rearing and migration stages, as well as during ocean migration. There have been recent concerns that the increasing size of tern, seal, and sea lion populations in the Pacific Northwest may have reduced the survival of some salmon ESUs. Human activities include the operation of hydropower systems, over-harvest, hatcheries, and habitat degradation including poor water quality from chemical contamination.

Chinook salmon are dependent on the quantity and quality of aquatic habitats. Juvenile salmonids rely on a variety of non-main channel habitats that are critical to rearing. All listed salmonids use shallow, low flow habitats at some point in their life cycle. Examples of off-channel habitat include alcoves, channel edge sloughs, overflow channels, backwaters, terrace tributaries, off-channel dredge ponds, and braids (Anderson 1999; Swift III 1979). Chinook salmon, like the other salmon NMFS has listed, have declined under the combined effects of overharvests in fisheries; competition from fish raised in hatcheries and native and non-native exotic species; dams that block their migrations and alter river hydrology; gravel mining that impedes their migration and alters the hydrogeomorphology of the rivers and streams that support juveniles; water diversions that deplete water levels in rivers and streams; destruction or degradation of riparian habitat that increase water temperatures in rivers and streams sufficient to reduce the survival of juvenile Chinook salmon; and land use practices (logging, agriculture, urbanization) that destroy or alter wetland and riparian ecosystems. These activities and features introduce sediment, nutrients, biocides, metals, and other pollutants into surface and ground water and degrade water quality in the freshwater, estuarine, and coastal ecosystems throughout the Pacific Northwest.

Salmonids along the west coast of the U.S. share common threats. Therefore, anthropogenic threats for all species and stocks are summarized here (see (NMFS 2005) for a review). Population declines have resulted from several human-mediated causes. However, the greatest negative influence has been the establishment of waterway obstructions such as dams, power plants, and sluiceways for hydropower, agriculture, flood control, and water storage. These structures have blocked salmon migration to spawning habitat or resulted in direct mortality and have eliminated entire salmon runs. Presently, many of these structures have been re-engineered, renovated, or removed to allow for surviving runs to access former habitat. However, success has been limited. Remaining freshwater habitats are threatened from development along waterways as well as sedimentation, pollution run-off, habitat modification, and erosion. These factors can directly cause mortality, affect salmonid health, or modify spawning habitat so as to reduce reproductive success. Immature salmonids remain in freshwater systems and may be exposed to these modifications for years. These conditions reduce juvenile survival.

Salmonids are also a popular commercial resource and have faced significant pressure from fishing. Although currently protected, illegal oceanic driftnet gear is suspected of hindering salmon survival and recovery. Despite the protection of weaker salmonid stocks from fishing, exploitation of more populous stocks may actually harm weaker stocks. Hatchery-reared salmon have been and are still being introduced to bolster stocks. However, the broader effects of this action are unknown.

California Coastal Chinook Salmon

Distribution

California Coastal (CC) Chinook salmon includes all naturally-spawned coastal Chinook salmon spawning from Redwood Creek south through the Russian River as shown in (Figure 6).

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, Spence et al. 2005). Table 6 identifies populations within the CC Chinook salmon ESU, their abundances, and hatchery input.

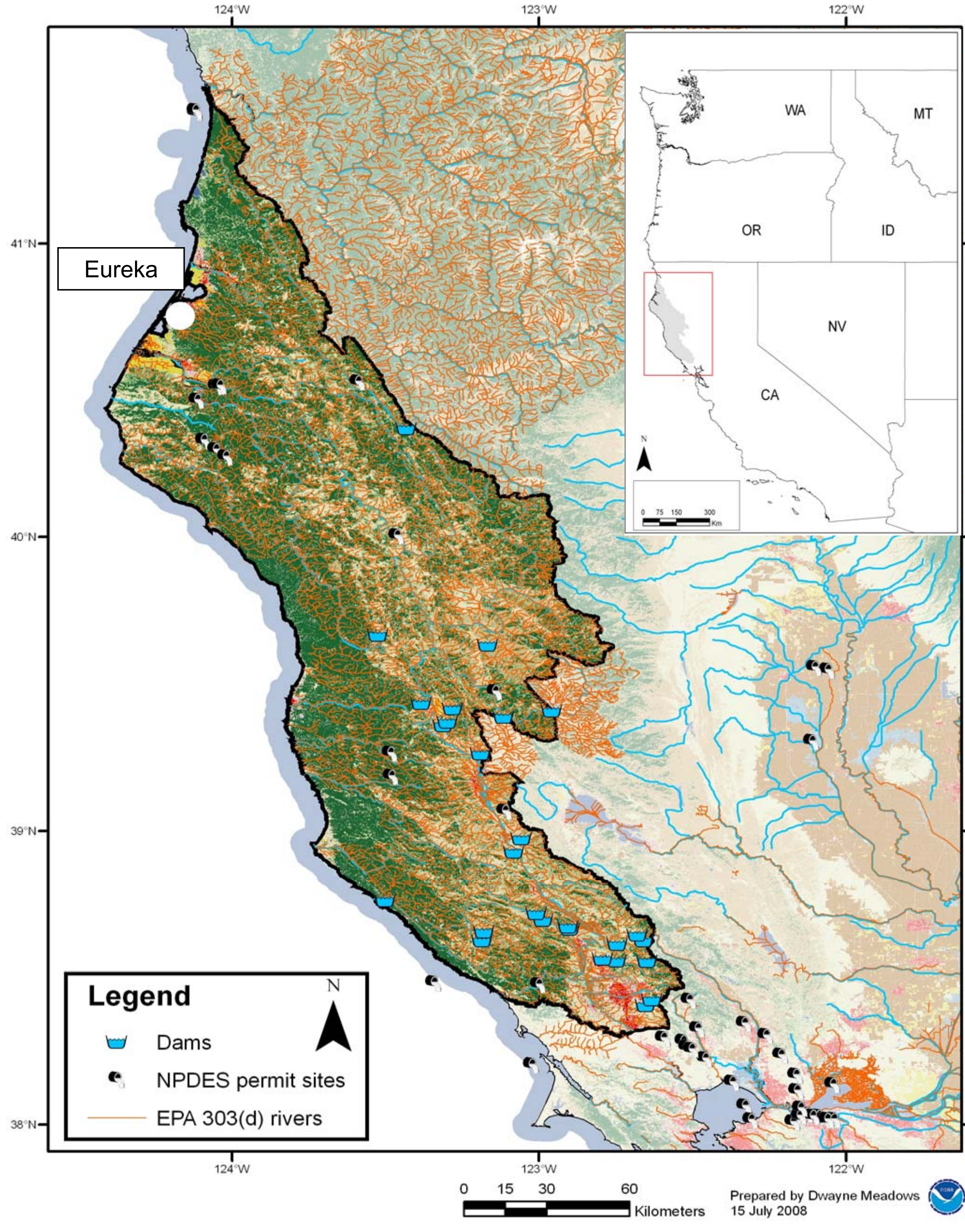


Figure 6. CC Chinook salmon distribution. Land Cover Class Legend in Figure 7.

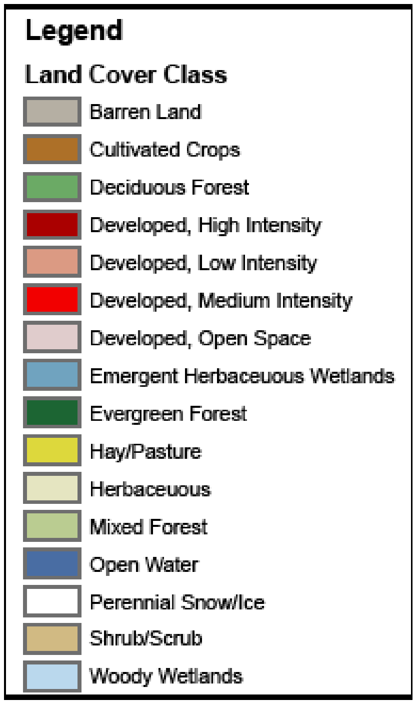


Figure 7. Legend for the Land Cover Class categories found in species distribution maps. Land cover is based on the 2001 National Land Cover Data and classifications. http://www.mrlc.gov/index.php.

Table 6. CC Chinook salmon--preliminary population structure, abundances, and hatchery contributions (Good, Waples et al. 2005).

Population	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Eel River (includes * tributaries below)	17,000-55,000	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	Unknown
Russian River	50-500	200,000	~0%
Humbolt Bay tributaries	40	120	40 (33%)
Tenmile to Gualala coastal effluents	Unknown	Unknown	0
Small Humboldt County rivers	1,500	Unknown	0
Rivers north of Mattole River	600	Unknown	0
Noyo River	50	Unknown	0
Total	**20,750-72,550**	**200,175 (min)**	

Status and Trends

CC Chinook salmon were listed as threatened on September 16, 1999 (64 FR 50393). Their classification was reaffirmed following a status review on June 28, 2005 (70 FR 37160). The outcome was based on the combined effect of dams that prevent individuals from reaching spawning habitat, logging, agricultural activities, urbanization, and water withdrawals in the river drainages that support CC Chinook salmon. Historical estimates of escapement, based on professional opinion and evaluation of habitat conditions, suggest abundance was roughly 73,000 in the early 1960s with the majority of fish spawning in the Eel River [see CDFG 1965 *in* (Good, Waples et al. 2005)]. The species exists as small populations with highly variable cohort sizes and discussion is underway to split Eel River salmon into as many as five separate populations (see Table 3). The Russian River probably contains some natural production. However, the origin of those fish is unclear as a number of introductions of hatchery fish occurred over the last century. The Eel River contains a substantial fraction of the remaining Chinook salmon spawning habitat for this species.

Since the original listing and status review, little new data are available or suitable for analyzing trends or estimating changes in the Eel River population's growth rate (Good, Waples et al. 2005). Historical and current abundance information indicates that independent populations of Chinook salmon are depressed in many of those basins where they have been monitored.

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52630). The critical habitat designation for this ESU identifies PCEs that include sites necessary to support one or more Chinook salmon life stages. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat, and estuarine areas. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity. 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 limited instream cover that does exist is provided mainly by large cobble and overhanging vegetation. Instream large woody debris, needed for foraging sites, cover, and velocity refuges is especially lacking in most of the streams throughout the basin. NMFS has determined that these degraded habitat conditions are, in part, the result of many human-induced factors affecting critical habitat. They include dam construction, agricultural and mining activities, urbanization, stream channelization, water diversion, and logging.

Central Valley Spring-Run Chinook Salmon

Distribution

The Central Valley spring-run (CV) Chinook salmon includes all naturally spawned populations of spring-run Chinook salmon in the Sacramento River and its tributaries in California (Figure 8).

Table 7 identifies populations within the CV spring-run Chinook salmon ESU, their abundances, and hatchery input.

Table 7. CV Chinook salmon--preliminary population structure, abundances, and hatchery contributions (Good, Waples et al. 2005).

Population	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Butte Creek Spring-run Chinook		67-4,513	Unknown
Deer Creek Spring-run Chinook		243-1,076	Unknown
Mill Creek Spring-run Chinook		203-491	Unknown
Total	**~700,000 for all populations**	**513-6,080**	Unknown

Life History

CV Chinook salmon enter the Sacramento River from March to July and spawn from late August through early October, with a peak in September. Spring-run fish in the Sacramento River exhibit an ocean-type life history, emigrating as fry and sub-yearlings. Chinook salmon require cool freshwater while they mature over the summer. This species tends to take advantage of high flows. Adult upstream migration may be blocked by temperatures above 21ºC (McCullough 1999). Temperatures below 21ºC can stress fish by increasing their susceptibility to disease (Berman 1990) and elevating their metabolism (Brett 1979).

Status and Trends

CV Chinook salmon were listed as threatened on September 16, 1999 (64 FR 50393). This classification was retained following a status review on June 28, 2005 (70 FR 37160). The species was listed because dams isolated individuals from most of their historic spawning habitat and the remaining habitat is degraded. Historically, spring-run Chinook salmon were predominant throughout the CV. This species occupied the upper and middle reaches (1,000 to 6,000 ft) of the San Joaquin, American, Yuba, Feather, Sacramento, McCloud and Pit Rivers. Smaller populations occurred in most tributaries with sufficient habitat for over-summering adults (Stone 1874; Rutter 1904; Clarke 1929).

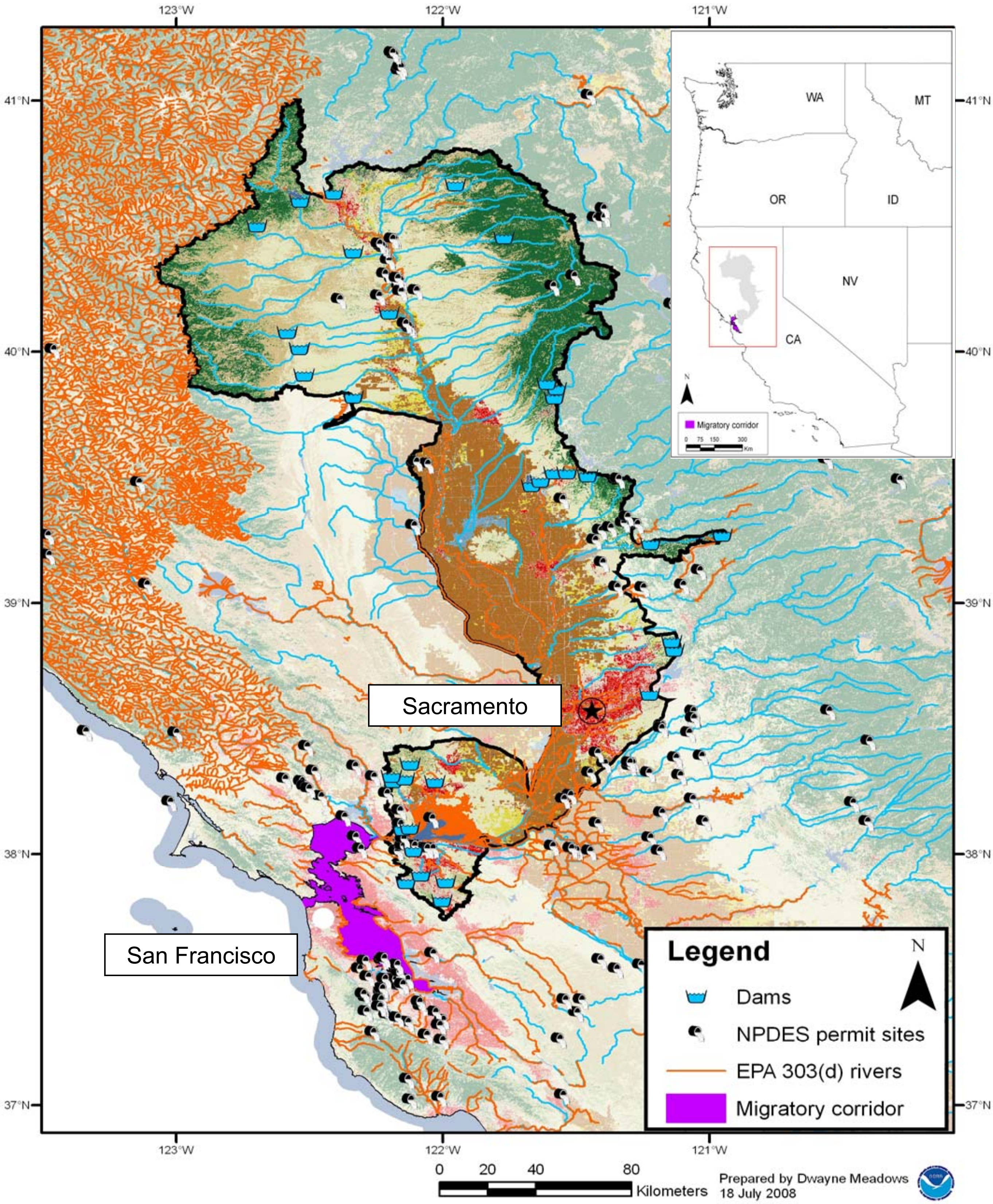


Figure 8. CV Chinook salmon distribution. Land Cover Class Legend in Figure 7.

The CV drainage as a whole is estimated to have supported spring-run Chinook salmon runs as large as 700,000 fish between the late 1880s and the 1940s (Brown, Moyle et al. 1994). Before construction of Friant Dam, nearly 50,000 adults were counted in the San Joaquin River alone (Fry 1961). Following the completion of Friant Dam, the native population from the San Joaquin River and its tributaries (*i.e.*, the Stanislaus and Mokelumne Rivers) was extirpated. Spring-run Chinook salmon no longer exist in the American River due to the operation of Folsom Dam. Naturally spawning populations of CV Chinook salmon currently are restricted to accessible reaches of the upper Sacramento River, Antelope Creek, Battle Creek, Beegum Creek, Big Chico Creek, Butte Creek, Clear Creek, Deer Creek, Feather River, Mill Creek, and Yuba River (CDFG 1998). Since 1969, the CV Chinook salmon ESU (excluding Feather River fish) has displayed broad fluctuations in abundance ranging from 25,890 in 1982 to 1,403 in 1993 (CDFG unpublished data).

The average abundance for the ESU was 12,499 for the period of 1969 to 1979, 12,981 for the period of 1980 to 1990, and 6,542 for the period of 1991 to 2001. In 2003 and 2004, total run size for the ESU was 8,775 and 9,872 adults, respectively. These averages are well above the 1991 to 2001 average.

Evaluating the ESU as a whole, however, masks significant changes that are occurring among populations that comprise the ESU. For example, the mainstem Sacramento River population has undergone a significant decline while the abundance of many tributary populations increased. Average abundance of Sacramento River mainstem spring-run Chinook salmon recently declined from a high of 12,107 for the period 1980 to 1990, to a low of 609 for the period 1991 to 2001 (Good, Waples et al. 2005). Meanwhile, the average abundance of Sacramento River tributary populations increased from a low of 1,227 to a high of 5,925 over the same periods.

According to Good et al. (2005), abundance time series data for Mill, Deer, and Butte creeks spring-run Chinook salmon (updated through 2001) confirm that population increases in the tributary populations seen in the 1990s have continued. During this period, habitat improvements included the removal of several small dams and increases in summer flows in the watersheds, a reduced ocean fisheries, and a favorable terrestrial and marine climate. All three spring-run Chinook populations in the Sacramento River tributaries have long-and short-term lambdas >1, indicating population growth. However, population sizes are relatively small compared to fall-run Chinook salmon populations, and there have been some extreme fluctuations in population size, which is often indicative of an impending collapse in small populations. Additionally, Feather River hatchery and Feather River spring-run Chinook salmon are not closely related to the Mill, Deer, and Butte creek spring-run Chinook salmon populations. This group represents a distinct genetic legacy. Although protective measures and critical habitat restoration likely have contributed to recent increases in spring-run Chinook salmon abundance, the ESU is still below levels observed from the 1960s through 1990. Threats from hatchery production (*i.e.*, competition for food between naturally spawned and hatchery fish, and run hybridization and homogenization), climatic variation, reduced stream flow, high water temperatures, predation, and large scale unscreened water diversions persist.

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52630). The critical habitat designation for this ESU identifies PCEs that include sites necessary to support one or more Chinook salmon life stages. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat, and estuarine areas. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity. Factors contributing to the downward trends in this ESU include: loss of most historical spawning habitat, reduced access to spawning/rearing habitat behind impassable dams, climatic variation, water management activities, hybridization with fall-run Chinook salmon, predation, and harvest. Additional factors include the degradation and modification of remaining rearing and migration habitats in the natal stream, the Sacramento River, and the Sacramento delta. The natal tributaries have many small hydropower dams and water diversions that in some years have greatly reduced or eliminated in-stream flows during spring-run migration periods. Problems in the migration corridor include unscreened or inadequately screened water diversions, predation by nonnative species, and excessively high water temperatures. Collectively, these factors have impacted spring-run Chinook salmon critical habitat and population numbers (CDFG 1998). Several actions have been taken to improve and increase the PCEs of critical habitat for spring-run Chinook salmon, including improved management of CV water (*e.g.*, through use of CALFED EWA and CV Project Improvement Act (b)(2) water accounts), implementing new and improved screen and ladder designs at major water diversions along the mainstem Sacramento River and tributaries, removal of several small dams on important spring-run Chinook salmon spawning streams, and changes in ocean and inland fishing regulations to minimize harvest.

Lower Columbia River Chinook Salmon

Distribution

Lower Columbia River (LCR) Chinook salmon includes all naturally-spawned populations of 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 (Figure 7). Naturally spawned populations also occur along the Willamette River to Willamette Falls, Oregon, exclusive of spring-run Chinook salmon in the Clackamas River (Table 5). 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 eastern boundary for this species occurs at Celilo Falls, which corresponds to the edge of the drier Columbia Basin Ecosystem. Historically, Celilo Falls may have been a barrier to salmon migration at certain times of the year. Table 8 identifies populations within the LCR Chinook salmon ESU, their abundances, and hatchery input

Table 8. LCR Chinook salmon - preliminary population structure, abundances, and hatchery contributions (Good, Waples et al. 2005).

Population	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Youngs Bay	Unknown	Unknown	Unknown
Grays River	2,477	99	38%
Big Creek	Unknown	Unknown	Unknown
Elochoman River	Unknown	676	68%
Clatskanie River	Unknown	Unknown	Unknown
Mill, Abernathy, and German Creeks	Unknown	734	47%
Scappoose Creek	Unknown	Unknown	Unknown
Coweeman River	Unknown	274	0%
Lower Cowlitz River	4,971	1,562	62%
Upper Cowlitz River (fall run)	Unknown	5,682	Unknown
Toutle River (fall run)	53,956	Unknown	Unknown
Kalama River (fall run)	25,392	2,931	67%
Salmon Creek and Lewis River	47,591	256	0%
Clackamas River	Unknown	40	Unknown
Washougal River	7,518	3,254	58%
Sandy River (fall run)	Unknown	183	Unknown
Columbia Gorge-lower tributaries	Unknown	Unknown	Unknown
Columbia Gorge-upper tributaries	Unknown	Unknown	Unknown
Hood River (fall run)	Unknown	18	Unknown
Big White Salmon River	Unknown	334	21%
Sandy River (late fall run)	Unknown	504	3%
Lewis River-North Fork	Unknown	7,841	13%
Upper Cowlitz River (spring run)	Unknown	Unknown	Unknown
Cispus River	Unknown	1,787	Unknown
Tilton River	Unknown	Unknown	Unknown
Toutle River (spring run)	2,901	Unknown	Unknown
Kalama River (spring run)	4,178	98	Unknown
Lewis River	Unknown	347	Unknown
Sandy River (spring run)	Unknown	Unknown	Unknown
Big White Salmon River (spring run)	Unknown	Unknown	Unknown
Hood River (spring run)	Unknown	51	Unknown
Total	**148,984 (min)**	**26,273 (min)**	

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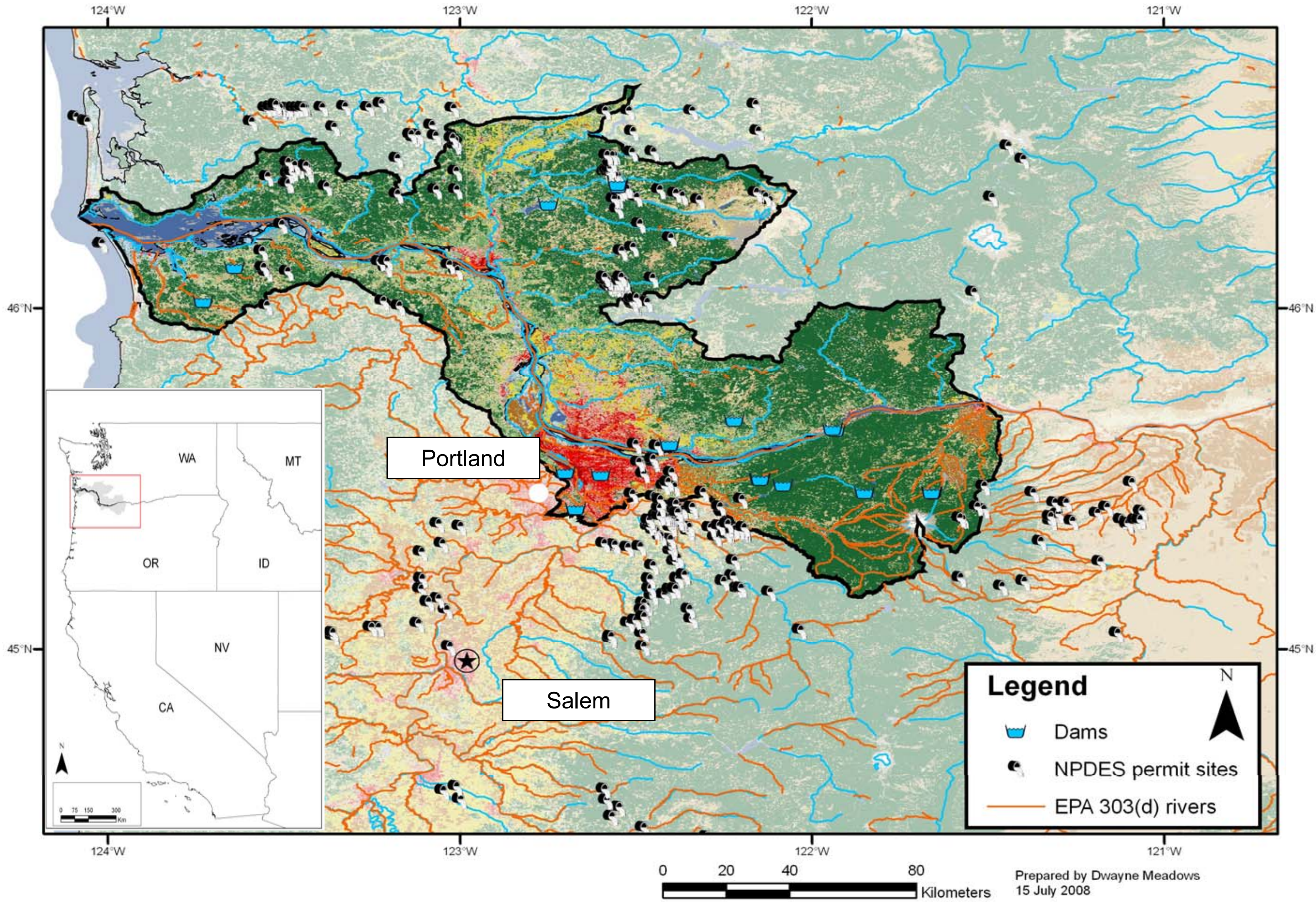

Figure 9. LCR Chinook salmon distribution. Land Cover Class Legend in Figure 7.

Life History

LCR Chinook salmon display three life history types including early fall runs, late fall runs, and spring-runs. Spring and fall runs have been designated as part of a LCR Chinook salmon ESU. The predominant life history type for this species is the fall-run. Fall Chinook salmon enter freshwater typically in August through October to spawn in large river mainstems. The juvenile life history stage emigrates from freshwater as sub-yearling (ocean-type). Spring Chinook salmon enter freshwater in March through June to spawn in upstream tributaries and generally emigrate from freshwater as yearlings (stream-type).

Status and Trends

LCR Chinook salmon were originally listed as threatened on March 24, 1999 (64 FR 14308). This status was reaffirmed on June 28, 2005 (70 FR 37160). 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. Although fall-run Chinook salmon occur throughout much of their historical range, they remain vulnerable to large-scale hatchery production, relatively high harvest, and extensive habitat degradation. The Lewis River late fall Chinook salmon population is the healthiest and has a reasonable probability of being self-sustaining. Abundances largely declined during 1998 to 2000. Trend indicators for most populations are negative, especially if hatchery fish are assumed to have a reproductive success equivalent to that of natural-origin fish.

New data acquired for the Good et al. (2005) report includes spawner abundance estimates through 2001, new estimates of the fraction of hatchery spawners, and harvest estimates. In addition, estimates of historical abundance have been provided by the Washington Department of Fish and Wildlife (WDFW). 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. Almost all of the spring-run Chinook of LCR Chinook are at very high risk of extinction. Near loss of that important life history type remains an important concern. Although some natural production currently occurs in 20 or so populations, only one exceeds 1,000 spawners. Most LCR Chinook salmon populations have not seen increases in recent years as pronounced as those that have occurred in many other geographic areas.

According to Good et al. (2005), the majority of populations for which data are available have a long-term trend of <1; indicating the population is in decline. Currently, the spatial structures of populations in the Coastal and Cascade Fall Run major population groups (MPGs) are similar to their respective historical conditions. The genetic diversity of the Coastal, Cascade, and Gorge Fall Run MPGs (*i.e.*, all except the Late Fall Run Chinook salmon MPG) has been eroded by large hatchery influences and periodically by low effective population sizes. Hatchery programs for spring Chinook salmon are preserving the genetic legacy from populations that were extirpated from blocked areas. High hatchery production also poses genetic and ecological risks to natural populations and masks their performance.

Critical Habitat

Critical habitat was designated 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 Hood Rivers as well as specific stream reaches in a number of tributary subbasins. The critical habitat designation for this ESU identifies PCEs that include sites necessary to support one or more Chinook salmon life stages. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat, and estuarine areas. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity.

Of 52 subbasins 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 (35), were rated as having a high conservation value to LCR Chinook salmon. Factors contributing to the downward trends in this ESU are hydromorphological changes resulting from hydropower development, loss of tidal marsh and swamp habitat, and degraded freshwater and marine habitat from industrial harbor and port development, and urban development. Limiting factors identified for this species include: (1) Habitat degradation and loss due to extensive hydropower development projects, urbanization, logging, and agriculture on Chinook spawning and rearing habitat in the LCR, (2) reduced access to spawning/rearing habitat in tributaries, (3) hatchery impacts, (4) loss of habitat diversity and channel stability in tributaries, (5) excessive fine sediment in spawning gravels, (6) elevated water temperature in tributaries, (7) harvest impacts, and (8) poor water quality.

Upper Columbia River Spring-run Chinook Salmon

Distribution

Endangered Upper Columbia River (UCR) spring-run Chinook salmon includes stream-type Chinook salmon that inhabit tributaries upstream from the Yakima River to Chief Joseph Dam (Figure 10). The UCR spring-run Chinook salmon is composed of three major population groups (MPGs): the Wenatchee River population, the Entiat River population, and the Methow River population. These same populations currently spawn in only three river basins above Rock Island Dam: the Wenatchee, Entiat, and Methow Rivers. Several hatchery populations are also listed including those from the Chiwawa, Methow, Twisp, Chewuch, and White rivers, and Nason Creek (Table 9). Table 9 identifies populations within the UCR Chinook salmon ESU, their abundances, and hatchery input.

Table 9. UCR Chinook salmon - preliminary population structure, abundances, and hatchery contributions (Good, Waples et al. 2005).

Population	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Methow River	~2,100	79-9,904	59%
Twisp River	Unknown	10-369	54%
Chewuch River	Unknown	6-1,105	41%
Lost/Early River	Unknown	3-164	54%
Entiat River	~380	53-444	42%
Wenatchee River	~2,400	119-4,446	42%
Chiwawa River	Unknown	34-1,046	47%
Nason Creek	Unknown	8-374	39%
Upper Wenatchee River	Unknown	0-215	66%
White River	Unknown	1-104	8%
Little Wenatchee River	Unknown	3-74	21%
Total	**~4,880 (min)**		

Life History

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

Status and Trends

UCR spring-run Chinook salmon were listed as endangered on March 24, 1999 (64 FR 14308). This listing was reaffirmed on June 28, 2005 (70 FR 37160) based on a reduction of UCR spring-run Chinook salmon to small populations in three watersheds. Based on redd count data series, spawning escapements for the Wenatchee, Entiat, and Methow rivers have declined an average of 5.6%, 4.8%, and 6.3% per year, respectively, since 1958.

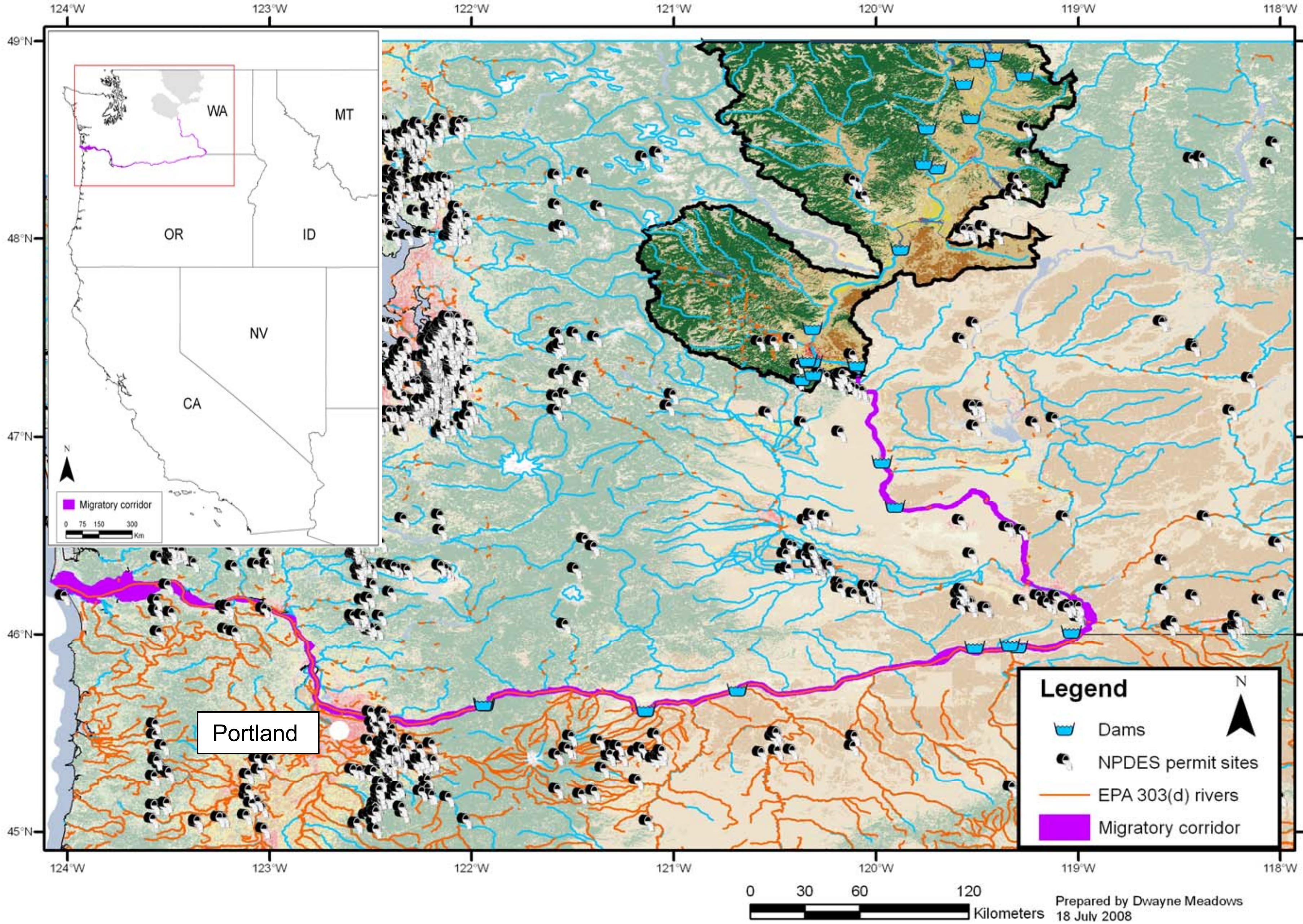

Figure 10. UCR Spring-run Chinook salmon distribution. Land Cover Class Legend in Figure 7.

In the most recent five-year geometric mean (1997 to 2001), spawning escapements 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. However, escapement increased substantially in 2000 and 2001 in all three river systems. Based on 1980-2004 returns, the average annual population growth rate, lambda, for this ESU is estimated at 0.93 (meaning the population is not replacing itself) (Fisher and Hinrichsen 2006). Assuming that population growth rates were to continue at 1980-2004 levels, UCR spring-run Chinook salmon populations are projected to have very high probabilities of decline within 50 years. Population viability analyses for this species suggest that these Chinook salmon face a significant risk of extinction: a 75 to 100% probability of extinction within 100 years (given return rates for 1980 to present). Finally, the Interior Columbia Basin Technical Recovery Team (ICBTRT) characterizes the diversity risk to all UCR spring 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. Straying hatchery fish, and a low proportion of natural-origin fish in some broodstocks and a high proportion of hatchery fish on the spawning grounds have also contributed to the high genetic diversity risk.

Critical Habitat

Critical habitat was designated 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 Chief Joseph Dam and several tributary subbasins. The critical habitat designation for this ESU also identifies PCEs that include sites necessary to support one or more Chinook salmon life stages. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat, and estuarine areas. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity. 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. The Columbia River rearing/migration corridor downstream of the spawning range was rated as a high conservation value. Factors contributing to the downward trends in this ESU include: (1) Mainstem Columbia River hydropower system mortality, (2) tributary riparian degradation and loss of in-river wood, (3) altered tributary floodplain and channel morphology, (4) reduced tributary stream flow and impaired passage, (5) harvest impacts, and (6) degraded water quality.

Puget Sound Chinook Salmon

Distribution

The boundaries of the Puget Sound ESU correspond generally with the boundaries of the Puget Lowland Ecoregion (Figure 11). The Puget Lowland Ecoregion begins in Washington at approximately the Dungeness River near the eastern end of the Strait of Juan de Fuca and extends through Puget Sound to the British Columbia border and up to the Cascade foothills. The Puget Sound ESU includes all runs of Chinook salmon in the Puget Sound region from the North Fork Nooksack River to the Elwha River on the Olympic Peninsula. This ESU is comprised of 31 historical populations. Of these, 22 populations are believed to be extant. Thirty-six hatchery populations were included as part of the ESU and five were considered essential for recovery and listed. They include spring Chinook salmon from Kendall Creek, the North Fork Stillaguamish River, White River, and Dungeness River, and fall run fish from the Elwha River (Table 10). Table 10 identifies populations within the Puget Sound Chinook salmon ESU, their abundances, and hatchery input.

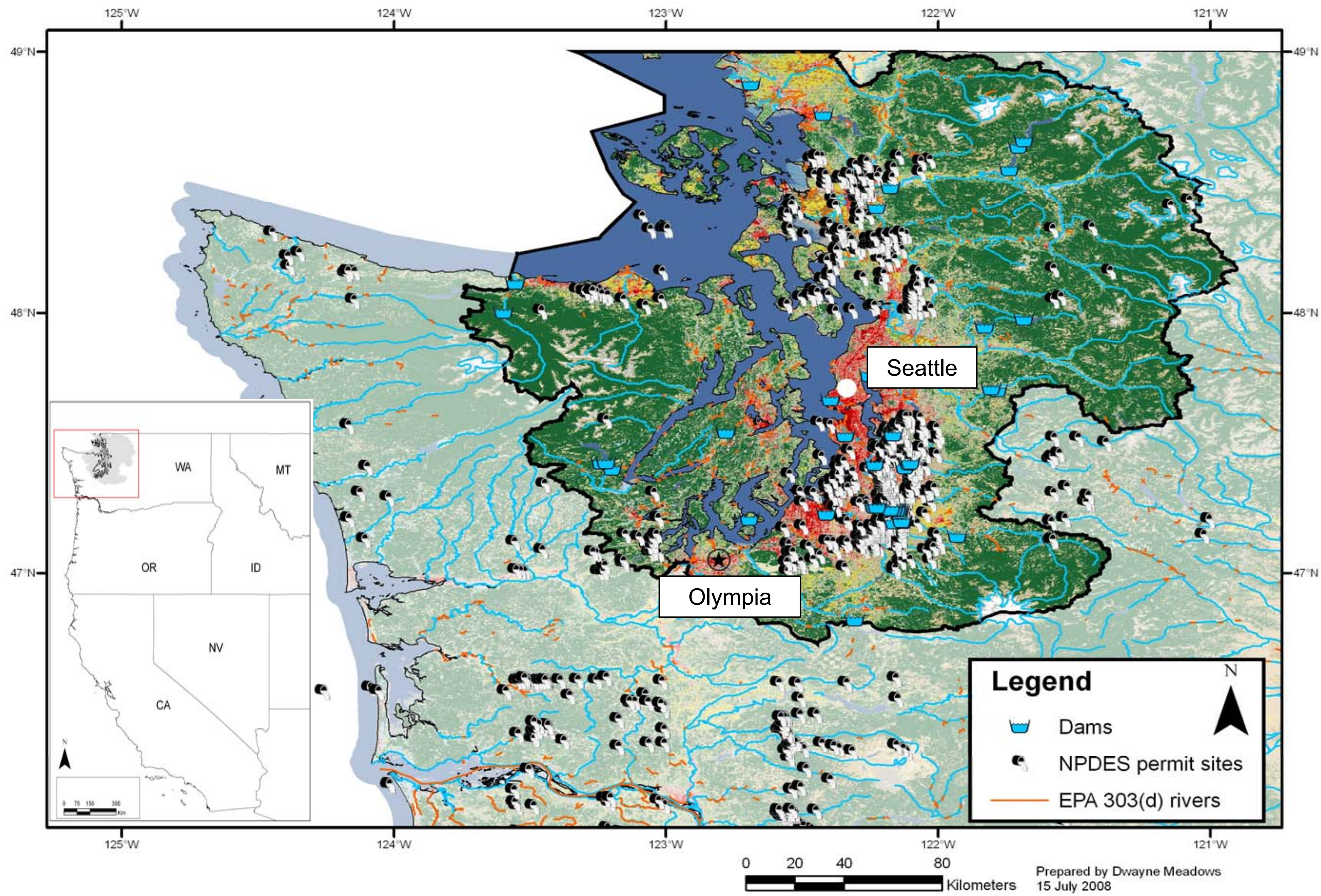

Figure 11. Puget Sound Chinook distribution. Land Cover Class Legend in Figure 7..

Table 10. Puget Sound Chinook salmon - preliminary population structure, abundances, and hatchery contributions (Good, Waples et al. 2005).

Population	Historical Abundance	Most Recent Spawner Abundance	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%
North Lake Washington	Unknown	331	Unknown
Cedar	Unknown	327	Unknown
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
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
Total	**~690,000**	**39,343**	

Life History

Chinook salmon in this area generally have an "ocean-type" life history. Puget Sound populations exhibit both the early-returning and late-returning Chinook salmon spawners described by Healey (1997). However, within these two generalized behavioral forms, substantial variation occurs in juvenile behavior and residence time in fresh water and estuarine environments. Hayman et al. (1996) described three juvenile life histories for Chinook salmon with varying freshwater and estuarine residency times in the Skagit River system in northern Puget Sound. Chinook salmon use the nearshore area of Puget Sound during all seasons of the year and can be found long distances from their natal river systems (Brennan, Higgins et al. 2004).

Status and Trends

Puget Sound Chinook salmon were listed as threatened in 1999 (64 FR 14308). This status was re-affirmed on June 28, 2005 (70 FR 37160). This ESU has lost 15 spawning aggregations that were either demographically independent historical populations or major components of the life history diversity of the remaining 22 existing independent historical populations identified (Good, Waples et al. 2005). Nine of the 15 extinct spawning aggregations were early-run type Chinook salmon (Good, Waples et al. 2005). The disproportionate loss of early-run life history diversity represents a significant loss of the evolutionary legacy of the historical ESU.

The estimated total run size of Chinook salmon in Puget Sound in the early 1990s was 240,000 fish, representing a loss of nearly 450,000 fish from historic numbers. During a recent five-year period, the geometric mean of natural spawners in populations of Puget Sound Chinook salmon ranged from 222 to just over 9,489 fish. Most populations had natural spawners numbering in the hundreds (median recent natural escapement is 766). Of the six populations with greater than 1,000 natural spawners, only two have a low fraction of hatchery fish. Estimates of the historical equilibrium abundance, based on pre-European settlement habitat conditions, range from 1,700 to 51,000 potential Puget Sound Chinook salmon spawners per population. The historical estimates of spawner capacity are several orders of magnitude higher than spawner abundances currently observed throughout the ESU (Good, Waples et al. 2005).

Long-term trends in abundance and median population growth rates for naturally spawning populations of Puget Sound Chinook salmon indicate that approximately half of the populations are declining and the other half are increasing in abundance over the length of available time series. Eight of 22 populations are declining over the short-term, compared to 11 or 12 populations that have long-term declines (Good, Waples et al. 2005). Widespread declines and extirpations of spring- and summer-run Puget Sound Chinook salmon populations represent a significant reduction in the life history diversity of this ESU (Meyers, Kope et al. 1998). The median overall populations of long-term trend in abundance is 1, indicating that most populations are just replacing themselves.

Populations with the greatest long-term population growth rate are the North Fork Nooksack and White rivers.

Regarding spatial structure, the populations (22) presumed to be extinct are mostly early returning fish. Most of these are in the mid- to southern Puget Sound or Hood Canal and the Strait of Juan de Fuca. The ESU populations with the greatest estimated fractions of hatchery fish tend to be in the mid-to southern Puget Sound, Hood Canal, and the Strait of Juan de Fuca. Finally, all but one of the nine extinct Chinook salmon stocks is an early run population (or component of a population).

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52630). The critical habitat designation for this ESU identifies PCEs that include sites necessary to support one or more Chinook salmon life stages. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat, and estuarine areas. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity.

Of 49 subbasins (5th field Hydrological Units) reviewed in NMFS' assessment of critical habitat for the Puget Sound ESUs, nine subbasins were rated as having a medium conservation value, 12 were rated as low, and the remaining subbasins (40), where the bulk of Federal lands occur in this ESU, were rated as having a high conservation value to Puget Sound Chinook salmon. Factors contributing to the downward trends in this ESU are hydromorphological changes (such as diking, revetments, loss of secondary channels in floodplains, widespread blockages of streams, and changes in peak flows), degraded freshwater and marine habitat affected by agricultural activities and urbanization, and upper river tributaries widely affected by poor forest practices, and lower tributaries. Hydroelectric development and flood control also impact Puget Sound Chinook salmon in several basins. Changes in habitat quantity, availability, diversity, flow, temperature, sediment load, water quality, and channel stability are common limiting factors in areas of critical habitat.

Sacramento River Winter-Run Chinook Salmon

Distribution

Sacramento River winter-run Chinook salmon consists of a single spawning population that enters the Sacramento River and its tributaries in California from November to June and spawns from late April to mid-August, with a peak from May to June (Figure 12). Sacramento River winter Chinook salmon historically occupied cold, headwater streams, such as the upper reaches of the Little Sacramento, McCloud, and lower Pit Rivers.

Life History

Winter-run fish spawn mainly in May and June in the upper mainstem of the Sacramento River. Winter-run fish have characteristics of both stream- and ocean-type races. They enter the river and migrate far upstream. Spawning is delayed for some time after river entry. Young winter-run Chinook salmon, however migrate to sea in November and December, after only four to seven months of river life (Burgner 1991).

Status and Trends

Sacramento River winter-run Chinook salmon were listed as endangered on January 4, 1994 (59 FR 440), and were reaffirmed as endangered on June 28, 2005 (70 FR 37160). This was based on restricted access from dams to a small fraction of salmon historic spawning habitat and the degraded conditions of remaining habitat. Sacramento River winter-run Chinook salmon consist of a single self-sustaining population which is entirely dependent upon the provision of suitably cool water from Shasta Reservoir during periods of spawning, incubation, and rearing.

Construction of Shasta Dams in the 1940s eliminated access to historic spawning habitat for winter-run Chinook salmon in the basin. Winter-run Chinook salmon were not expected to survive this habitat alteration (Moffett 1949). However, cold water releases from Shasta Dam have created conditions suitable for winter Chinook salmon for roughly

60 miles downstream from the dam. As a result the ESU has been reduced to a single spawning population confined to the mainstem Sacramento River below Keswick Dam. Some adult winter-run Chinook salmon were recently observed in Battle Creek, a tributary to the upper Sacramento River.

Quantitative estimates of run-size are not available for the period before 1996, the completion of Red Bluff Diversion Dam. However, winter-runs may have been as large as 200,000 fish based upon commercial fishery records from the 1870s (Brown, Moyle et al. 1994).

The CDFG estimated spawning escapement of Sacramento River winter-run Chinook salmon at 61,300 (60,000 mainstem, 1,000 Battle Creek, and 300 in Mill Creek) in the early 1960s. During the first three years of operation of the county facility at the Red Bluff Diversion Dam (1967 to 1969), the spawning run of winter-run Chinook salmon averaged 86,500 fish. From 1967 through the mid-1990s, the population declined at an average rate of 18% per year, or roughly 50% per generation. The population reached critically low levels during the drought of 1987 to 1992. The three-year average run size for the period of 1989 to 1991 was 388 fish.

Based on the Red Bluff Diversion Dam counts, the population has been growing rapidly since the 1990s. Mean run size from 1995-2000 has been 2,191, but have ranged from 364 to 65,683 (Good, Waples et al. 2005). Most recent estimates indicate that the short-term trend is 0.26, and the population growth rate is less than one.

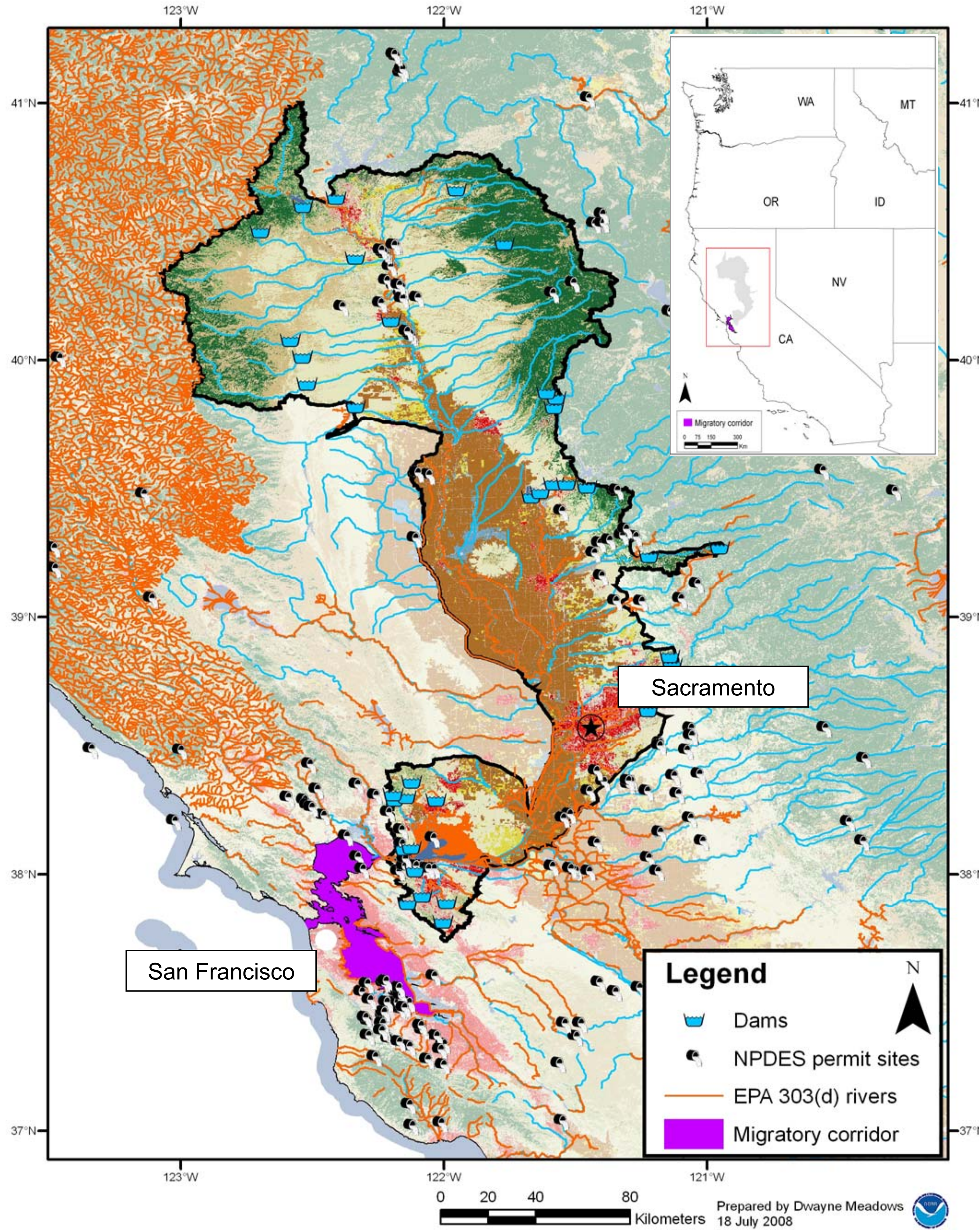

Figure 12. Sacramento River Winter-run Chinook salmon distribution. Land Cover Class Legend in Figure 7.

Critical Habitat

Critical habitat was designated for this species on June 16, 1993 (58 FR 33212). The following areas consist of the water, waterway bottom, and adjacent riparian zones: 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. Factors contributing to the downward trends in this ESU include: (1) Reduced access to spawning/rearing habitat, (2) possible loss of genetic integrity through population bottlenecks, (3) inadequately screened diversions, (4) predation at artificial structures and by nonnative species, (5) pollution from Iron Mountain Mine and other sources, (6) adverse flow conditions, (7) high summer water temperatures, (8) degraded water quality, (9) unsustainable harvest rates, (10) passage problems at various structures, and (11) vulnerability to drought (Good, Waples et al. 2005).

Snake River Fall-Run Chinook Salmon

Distribution

Historically, the primary fall-run Chinook salmon spawning areas occurred on the upper mainstem Snake River (SR) (Connor, Sneva et al. 2005). A series of SR mainstem dams blocks access to the upper SR, which significantly reduced spawning and rearing habitat for SR fall-run Chinook salmon (Figure 13).

The present range of spawning and rearing habitat for naturally-spawned SR fall-run Chinook salmon is limited to the SR below Hells Canyon Dam and the lower reaches of the Clearwater River. SR fall-run Chinook salmon spawn above Lower Granite Dam in the mainstem SR and in the lower reaches of the larger tributaries.

As a consequence of lost access to historic spawning and rearing sites in the Upper SR, fall-run Chinook salmon now reside in waters that are generally cooler than the majority of historic spawning areas. Additionally, alteration of the Lower SR by hydroelectric dams has created a series of low-velocity pools in the SR that did not exist historically.

Life History

Prior to alteration of the SR basin by dams, fall Chinook salmon exhibited a largely ocean-type life history, where they migrated downstream and reared in the mainstem SR during their first year. Today, fall Chinook salmon in the SR Basin exhibit one of two life histories: ocean-type and reservoir-type (Connor, Sneva et al. 2005). The reservoir-type life history is one where juveniles overwinter in the pools created by the dams, prior to migrating out of the SR. The reservoir-type life history is likely a response to early development in cooler temperatures which prevents juveniles from reaching suitable size to migrate out of the SR.

Adult SR fall-run Chinook salmon enter the Columbia River in July and August. Spawning occurs from October through November. Juveniles emerge from gravels in March and April of the following year, moving downstream from natal spawning and early rearing areas from June through early fall.

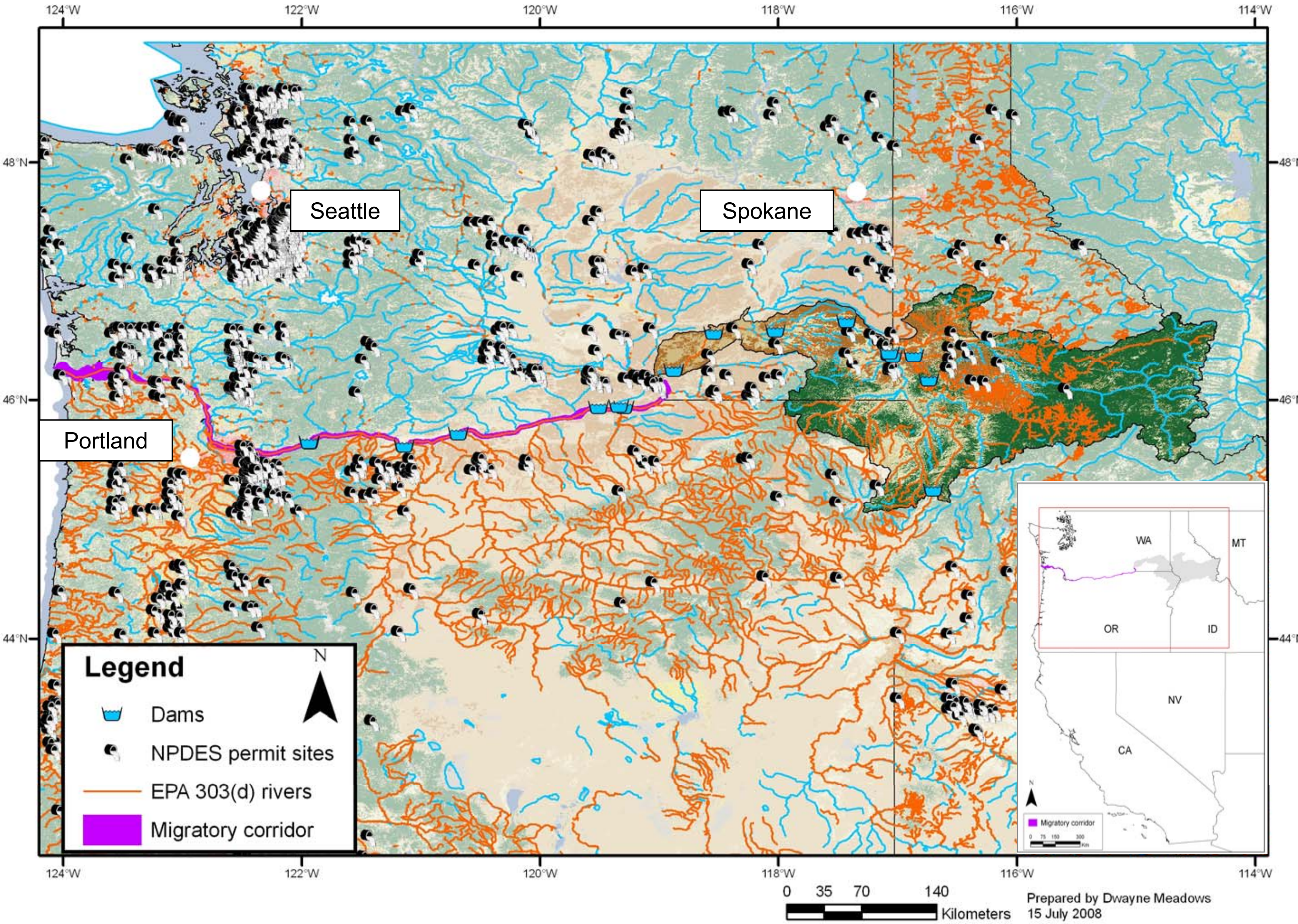

Figure 13. SR fall-run Chinook salmon distribution. Land Cover Class Legend in Figure 7..

Status and Trends

SR fall-run Chinook salmon were originally listed as threatened in 1992 (57 FR 14653). Their classification was reaffirmed following a status review on June 28, 2005 (70 FR 37160). Estimated annual returns for the period 1938 to 1949 was 72,000 fish. By the 1950s, numbers had declined to an annual average of 29,000 fish (Bjornn and Horner 1980). Numbers of SR fall-run Chinook salmon continued to decline during the 1960s and 1970s as approximately 80% of their historic habitat was eliminated or severely degraded by the construction of the Hells Canyon complex (1958 to 1967) and the lower SR dams (1961 to 1975). Counts of natural-origin adult SR fall-run Chinook salmon at Lower Granite Dam were 1,000 fish in 1975, and ranged from 78 to 905 fish (with an average of 489 fish) over the ensuing 25-year period (Good, Waples et al. 2005). 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.

SR fall-run Chinook salmon have exhibited an upward trend in returns over Lower Granite Dam since the mid-1990s. Returns classified as natural-origin exceeded 2,600 fish in 2001, compared to a 1997-2001 geometric mean natural-origin count of 871. Long- and short-term trends in natural returns are positive. Harvest impacts on SR fall-run Chinook salmon declined after listing and have remained relatively constant in recent years. There have been major reductions in fisheries impacting this stock. Mainstem conditions for subyearling Chinook salmon migrants from the SR have generally improved since the early 1990s. The hatchery component, derived from outside the basin, has decreased as a percentage of the run at Lower Granite Dam from the 1998/99 status reviews (five year average of 26.2%) to 2001 (8%). This reflects an increase in the Lyons Ferry component, systematic removal of marked hatchery fish at the Lower Granite trap, and modifications to the Umatilla supplementation program to increase homing of fall Chinook release groups.

Overall abundance for SR fall-run Chinook salmon is relatively low, but has been increasing in the last decade (Good, Waples et al. 2005). 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 8 years. The recent abundance is approaching the delisting criteria. However, hatchery fish are faring better than wild fish.

Regarding productivity [population growth rate (lambda)], the long-term trend in total returns is >1; indicating the population size is growing. Although total abundance has dropped sharply in the past two years, it still remains at levels higher than previous decades. Productivity is likely sustained largely by a system of small artificial rearing facilities in the Lower SR Basin. The growth trend for natural-origin fish is close to 1, and could either be higher or lower, depending on the number of hatchery fish that spawn naturally.

The historic spatial structure has been reduced to one single remnant population. The ESU occupies a relatively small amount of marginal habitat, with the vast majority of historic habitat inaccessible. Genetic diversity is likely reduced from historic levels. Hatcheries affect ESU genetics due to three major components: natural-origin fish (which may be progeny of hatchery fish), returns of SR fall-run fish from the Lyons Ferry Hatchery program, and strays from hatchery programs outside the SR. Nevertheless, the SR fall-run Chinook salmon remains genetically distinct from similar fish in other basins. Phenotypic characteristics have shifted in apparent response to environmental changes from hydroelectric dams (Connor, Sneva et al. 2005).

The ICBTRT has defined only one extant population for the SR fall-run Chinook salmon, the lower SR mainstem population. This population occupies the SR from its confluence with the Columbia River to Hells Canyon Dam, and the lower reaches of the Clearwater, Imnaha, Grande Rhonde, Salmon, and Tucannonh Rivers (ICBTRT 2003).

Critical Habitat

Critical habitat for these salmon was designated on December 28, 1993 (58 FR 68543). This critical habitat encompasses 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 SR fall-run salmon (except reaches above impassable natural falls, and Dworshak and Hells Canyon Dams). Adjacent riparian zones are defined as those areas within a horizontal distance of 300 ft from the normal line of high water of a stream channel or from the shoreline of a standing body of water. Designated critical habitat includes 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 SR, and all SR reaches upstream to Hells Canyon Dam. Critical habitat also includes several river reaches presently or historically accessible to SR fall-run Chinook salmon. Limiting factors identified for SR fall-run Chinook salmon include: (1) Mainstem lower Snake and Columbia hydrosystem mortality, (2) degraded water quality, (3) reduced spawning and rearing habitat due to mainstem lower SR hydropower system, (4) harvest impacts, (5) impaired stream flows, barriers to fish passage in tributaries, excessive sediment, and (6) altered floodplain and channel morphology (NMFS 2005). The above activities and features also introduce sediment, nutrients, biocides, metals, and other pollutants into surface and ground water and degrade water quality in the freshwater, estuarine, and coastal ecosystems throughout the Pacific Northwest.

Snake River Spring/Summer-Run Chinook Salmon

Distribution

SR spring/summer-run Chinook salmon are primarily limited to the Salmon, Grande Ronde, Imnaha, and Tucannon Rivers in the SR basin (Figure 14). The SR basin drains portions of southeastern Washington, northeastern Oregon, and north/central Idaho. Environmental conditions are generally drier and warmer in these areas than in areas occupied by other Chinook salmon species. The ICBTRT has identified 32 populations in five MPGs (Upper Salmon River, South Fork Salmon River, Middle Fork , Salmon

River, Grande Ronde/Imnaha, Lower Snake Mainstem Tributaries) for this species. Historic populations above Hells Canyon Dam are considered extinct (ICBTRT 2003). This ESU includes production areas that are characterized by spring-timed returns, summer-timed returns, and combinations from the two adult timing patterns. Historically, the Salmon River system may have supported more than 40% of the total run of spring and summer Chinook salmon to the Columbia system (Fulton 1968).

Some or all of the fish returning to several of the hatchery programs are also listed, including those returning to the Tucannon River, Imnaha River, and Grande Ronde River hatcheries, and to the Sawtooth, Pahsimeroi, and McCall hatcheries on the Salmon River. The Salmon River system contains a range of habitats used by spring/summer Chinook. The South Fork and Middle Fork Salmon Rivers currently support the bulk of natural production in the drainage. Returns into the upper Salmon River tributaries have reestablished following the opening of passage around Sunbeam Dam on the mainstem Salmon River downstream of Stanley, Idaho. The dam was impassable to anadromous fish from 1910 until the 1930s. Table 11 identifies populations within the SR spring/summer Chinook salmon ESU, their abundances, and hatchery input.

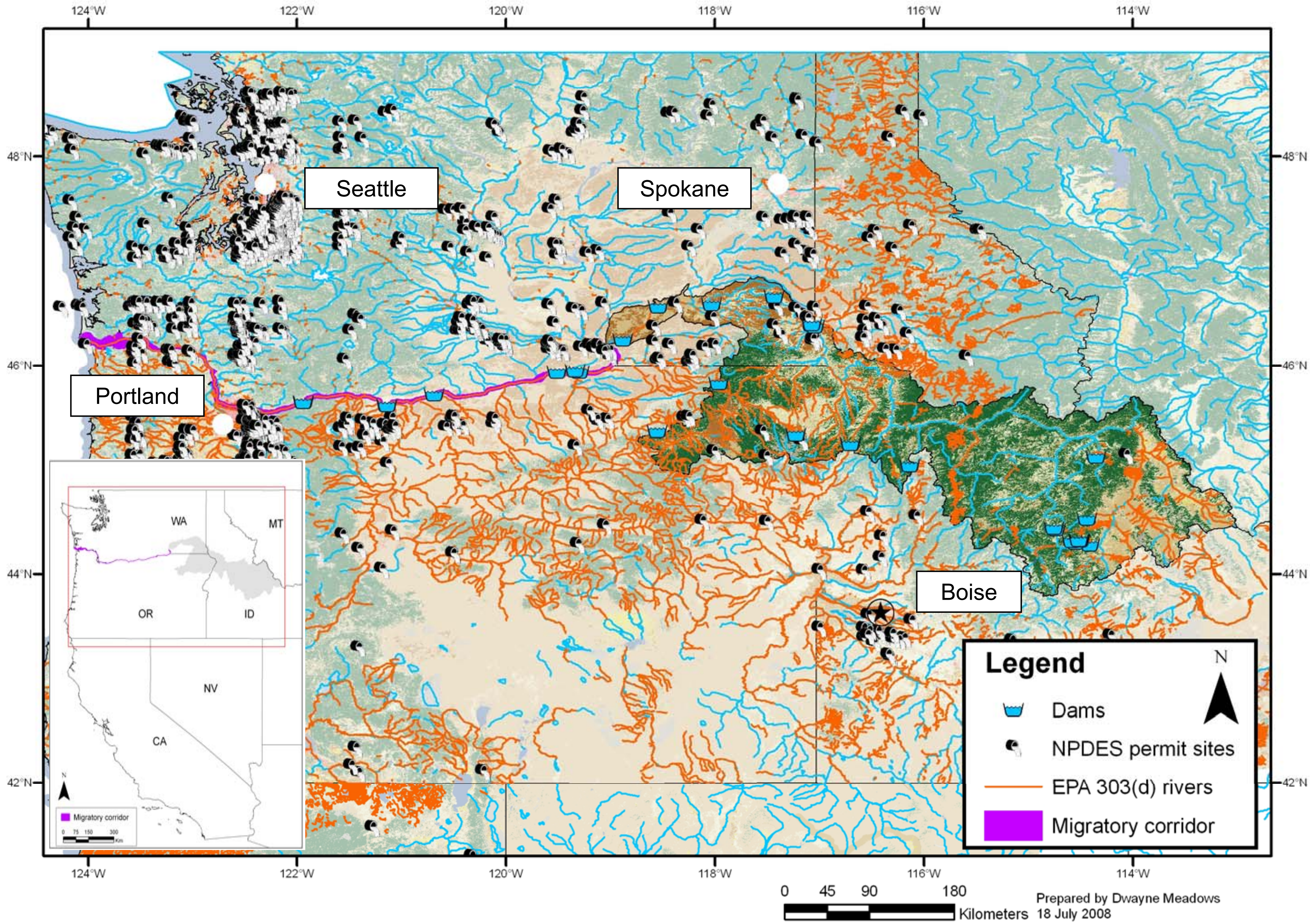

Figure 14. SR Spring/Summer-run Chinook salmon distribution. Land Cover Class Legend in Figure 7.

Table 11. SR Spring/Summer Chinook salmon populations, abundances, and hatchery contributions (Good, Waples et al. 2005). Note: rpm denotes redds per mile.

Current Populations	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Tucannon River	Unknown	128-1,012	76%
Wenaha River	Unknown	67-586	64%
Wallowa River	Unknown	0-29 redds	5%
Lostine River	Unknown	9-131 redds	5%
Minam River	Unknown	96-573	5%
Catherine Creek	Unknown	13-262	56%
Upper Grande Ronde River	Unknown	3-336	58%
South Fork Salmon River	Unknown	277-679 redds	9%
Secesh River	Unknown	38-444 redds	4%
Johnson Creek	Unknown	49-444 redds	0%
Big Creek spring run	Unknown	21-296	0%
Big Creek summer run	Unknown	2-58 redds	Unknown
Loon Creek	Unknown	6-255 redds	0%
Marsh Creek	Unknown	0-164	0%
Bear Valley/Elk Creek	Unknown	72-712	0%
North Fork Salmon River	Unknown	2-19 redds	Unknown
Lemhi River	Unknown	35-216 redds	0%
Pahsimeroi River	Unknown	72-1,097	Unknown
East Fork Salmon spring run	Unknown	0.27 rpm	Unknown
East Fork Salmon summer run	Unknown	1.22 rpm	0%
Yankee Fork spring run	Unknown	0	Unknown
Yankee Fork summer run	Unknown	1-18 redds	0%
Valley Creek spring run	Unknown	2-28 redds	0%
Valley Creek summer run	Unknown	2.14 rpm	Unknown
Upper Salmon spring run	Unknown	25-357 redds	Unknown
Upper Salmon summer run	Unknown	0.24 rpm	Unknown
Alturas Lake Creek	Unknown	0-18 redds	Unknown
Imnaha River	Unknown	194-3,041 redds	62%
Big Sheep Creek	Unknown	0.25 redds	97%
Lick Creek	Unknown	0-29 redds	59%
Total	**~1.5 million**	**~9,700**	

Life History

SR spring/summer-run Chinook salmon exhibit a stream-type life history. Eggs are deposited in late summer and early fall, 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. SR 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. A small fraction of the fish return as three year-old "jacks", heavily predominated by males.

Status and Trends

SR spring/summer-run Chinook salmon were originally listed as threatened on April 22, 1992 (57 FR 14653). Their classification was reaffirmed following a review on June 28, 2005 (70 FR 37160). Although direct estimates of historical annual SR spring/summer Chinook salmon returns are not available, returns may have declined by as much as 97% between the late 1800s and 2000. According to Matthews and Waples (1997), total annual SR spring/summer 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) and were below 10,000 by 1980. 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. The 1997 to 2001 geometric mean total return for the summer run component at Lower Granite Dam was slightly more than 6,000 fish, compared to the geometric mean of 3,076 fish for the years 1987 to 1996. The 2002 to 2006 geometric mean of the combined Chinook salmon runs at Lower Granite Dam was over 18,000 fish. However, over 80% of the 2001 return and over 60% of the 2002 return originated in hatcheries (Good, Waples 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 and the extensive production areas within the SR basin. Year-to-year abundance has high variability and is most pronounced in natural-origin fish. Although the average abundance in the most recent decade is more abundant than the previous decade, there is no obvious long-term trend. 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.

Regarding population growth rate (lambda), long-term trends are <1; indicating the population size is shrinking. However, recent trends, buoyed by last 5 years, are approaching 1. Nevertheless, many spawning aggregates have been extirpated, which has increased the spatial separation of some populations. Populations are widely distributed in a diversity of habitats although roughly one-half of historic habitats are inaccessible. 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 DPS to maintain distinct subpopulations adapted to local environments. Despite the recent increases in total spring/summer-run Chinook salmon returns to the basin, natural-origin abundance and productivity remain below their targets. SR spring/summer Chinook salmon remains likely to become endangered (Good, Waples et al. 2005).

Critical Habitat

Critical habitat for these salmon was designated on October 25, 1999 (64 FR 57399). This critical habitat encompasses 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 SR salmon (except reaches above impassable natural falls, and Dworshak and Hells Canyon Dams). Adjacent riparian zones are defined as those areas within a horizontal distance of 300 ft from the normal line of high water of a stream channel or from the shoreline of a standing body of water. Designated critical habitat includes 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). Critical habitat also includes all river reaches from the estuary upstream to the confluence of the SR, and all SR reaches upstream to Hells Canyon Dam; the Palouse River from its confluence with the SR upstream to Palouse Falls, the Clearwater River from its confluence with the SR upstream to its confluence with Lolo Creek; the North Fork Clearwater River from its confluence with the Clearwater river upstream to Dworshak Dam.

Limiting factors identified for this species include: (1) Hydrosystem mortality, (2) reduced stream flow, (3) altered channel morphology and floodplain, (4) excessive fine sediment, and (5) degraded water quality (Myers, Kope et al. 1998). The above activities and features also introduce sediment, nutrients, biocides, metals, and other pollutants into surface and ground water and degrade water quality in the freshwater, estuarine, and coastal ecosystems throughout the Pacific Northwest.

Upper Willamette River Chinook Salmon

Distribution

Upper Willamette River (UWR) Chinook salmon occupy the Willamette River and tributaries upstream of Willamette Falls (Figure 15). In the past, 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. Historically, access above Willamette Falls was restricted to the spring when flows were high. In autumn, low flows prevented fish from ascending past the falls. The UWR Chinook salmon are one of the most genetically distinct Chinook salmon groups in the Columbia River Basin. Fall-run Chinook salmon spawn in the Upper Willamette but are not considered part of the species because they are not native. None of the hatchery populations in the Willamette River were listed although five spring-run hatchery stocks were included in the species' listing. UWR Chinook salmon migrate far north and are caught incidentally in ocean fisheries, particularly off southeast Alaska and northern Canada, and in spring season fisheries in the mainstem Columbia and Willamette Rivers. Table 12 identifies populations within the UWR Chinook salmon ESU, their abundances, and hatchery input

Table 12. UWR Chinook salmon populations, abundances, and hatchery contributions (Good, Waples et al. 2005). Note: rpm denotes redds per mile

Current 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%
Upper Fork Willamette River	Unknown	Unknown	Unknown
Total	**>70,000**	**~9,700**	Mostly hatchery

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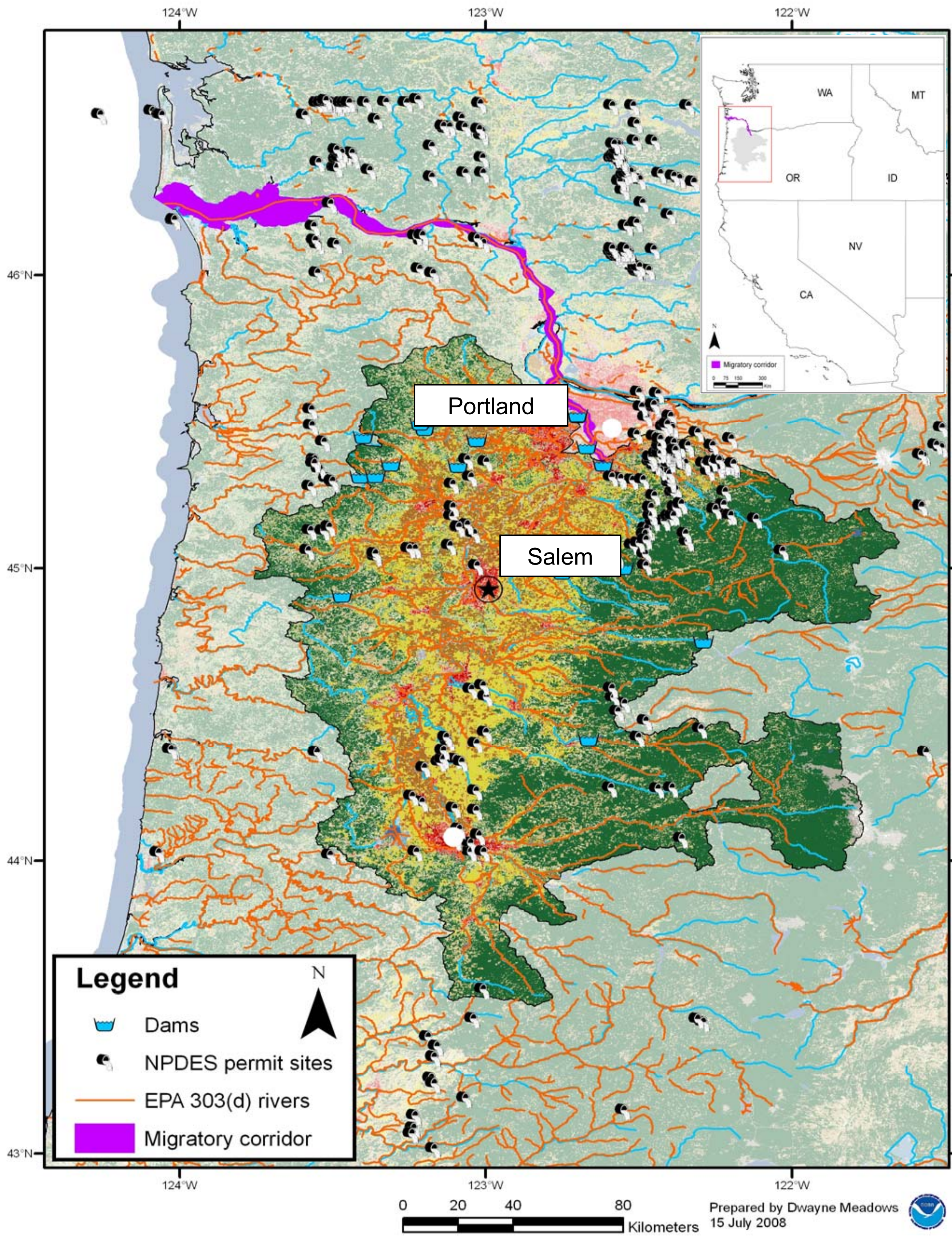


Figure 15. UWR Chinook salmon distribution. Land Cover Class Legend in Figure 7.

Life History

UWR Chinook salmon exhibit an earlier time of entry into the Columbia River and estuary than other spring Chinook salmon ESUs (Meyers, Kope et al. 1998). Although juveniles from interior spring Chinook salmon populations reach the mainstem migration corridor as yearling, some juvenile Chinook salmon in the lower Willamette River are subyearlings (Friesen, Vile et al. 2004).

Status and Trends

UWR Chinook salmon were listed as threatened on March 24, 1999 (64 FR 14308), and reaffirmed as threatened on June 28, 2005 (70 FR 37160). 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, it is an order of magnitude below the peak abundance levels observed in the 1920s (approximately 300,000 adults). Until recent years, interpretation of abundance levels has been confounded by a high but uncertain fraction of hatchery-produced fish.

Most natural spring Chinook salmon populations is likely extirpated or nearly so. Only one remaining naturally reproducing population is identified in this ESU: the spring Chinook salmon in the McKenzie River. Unfortunately, recent short-term declines in abundance suggest that this population may not be self-sustaining (Meyers, Kope et al. 1998; Good, Waples et al. 2005). Most of the natural-origin populations in this ESU have very low current abundances (less than a few hundred fish) and many largely have been replaced by hatchery production. Long- and short-term trends for population growth rate are approximately 1 or are negative, depending on the metric examined (*i.e.*, long-term trend [regression of log-transformed spawner abundance] or lambda [median population growth rate]). Although the population increased substantially in 2000-2003, it was probably due to increased survival in the ocean. Future survival rates in the ocean are unpredictable, and the likelihood of long-term sustainability for this population has not been determined. Although the number of adult spring-run Chinook salmon crossing Willamette Falls is in the same range (about 20,000 to 70,000 adults) it has been for the last 50 years, a large fraction of these are hatchery produced. Of concern is that a majority of the spawning habitat and approximately 30 to 40% of total historical habitat are no longer accessible because of dams (Good, Waples et al. 2005).

Critical Habitat

Critical habitat was designated 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. The critical habitat designation for this ESU also identifies PCEs that include sites necessary to support one or more Chinook salmon life stages. Specific sites include freshwater spawning and rearing sites, freshwater migration corridors. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity. Of 65 subbasins reviewed in NMFS' assessment of critical habitat for the UWR Chinook salmon ESU, 19 subbasins were rated as having a medium conservation value, 19 were rated as low, and the remaining subbasins (27), were rated as having a high conservation value to UWR Chinook salmon. Federal lands were generally rated as having high conservation value to the species' spawning and rearing. Factors contributing to the downward trends in this ESU include: (1) Reduced access to spawning/rearing habitat in tributaries, (2) hatchery impacts, (3) altered water quality and temperature in tributaries, (4) altered stream flow in tributaries, and (5) lost/degraded floodplain connectivity and lowland stream habitat.

Chum Salmon

Description of the Species

Chum salmon has the widest natural geographic and spawning distribution of any Pacific salmonid because its range extends 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 are found only 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 freshwater, 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 distribute throughout the North Pacific Ocean and Bering Sea. North American chum salmon (as opposed to chum salmon originating in Asia) rarely occur west of 175° E longitude.

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 make extended migrations into northern British Columbian and Alaskan waters. Instead, they may travel directly offshore into the north Pacific Ocean.

Chum salmon, like pink salmon, usually spawn in the lower reaches of rivers, with redds usually dug in the mainstem or in side channels of rivers from just above tidal influence to nearly 100 km from the sea. Juveniles outmigrate to seawater 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 at 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.*, coastal cutthroat trout, steelhead, coho salmon, and most types of Chinook and sockeye salmon), which usually migrate to sea at a larger size, after months or years of freshwater rearing. This means that survival and growth in juvenile chum salmon depend less on freshwater conditions (unlike stream-type salmonids which depend heavily on freshwater habitats) than on favorable estuarine conditions. Another behavioral difference between chum salmon and species that rear extensively in freshwater is that chum salmon form schools. Presumably, this behavior reduces predation (Pitcher 1986), especially if fish movements are synchronized to swamp predators (Miller and Brannon 1982).

The duration of estuarine residence for chum salmon juveniles are known for only a few estuaries. Observed residence times range from 4 to 32 days; with a period of about 24 days being the most common (Johnson, Grant et al. 1997). Juvenile salmonids rely on a variety of non-main channel habitats that are critical to rearing. All listed salmonids use shallow, low flow habitats at some point in their life cycle. Examples of off-channel habitat include alcoves, channel edge sloughs, overflow channels, backwaters, terrace tributaries, off-channel dredge ponds, and braids (Anderson 1999; Swift III 1979).

Status and Trends

Chum salmon have been threatened by overharvests in commercial and recreational fisheries, adult and juvenile mortalities associated with hydropower systems, habitat degradation from forestry and urban expansion, and shifts in climatic conditions that changed patterns and intensity of precipitation.

Chum salmon, like the other salmon NMFS has listed, have declined under the combined effects of overharvests in fisheries; competition from fish raised in hatcheries and native and non-native exotic species; dams that block their migrations and alter river hydrology; gravel mining that impedes their migration and alters the dynamics of the rivers and streams that support juveniles; water diversions that deplete water levels in rivers and streams; destruction or degradation of riparian habitat that increase water temperatures in rivers and streams sufficient to reduce the survival of juvenile chum salmon; and land use practices (logging, agriculture, urbanization) that destroy or alter wetland and riparian ecosystems. The above activities and features also introduce sediment, nutrients, biocides, metals, and other pollutants into surface and ground water and degrade water quality in the freshwater, estuarine, and coastal ecosystems throughout the Pacific Northwest.

Columbia River Chum Salmon

Distribution

Columbia River chum salmon includes all natural-origin chum salmon in the Columbia River and its tributaries in Oregon and Washington. The species consists of three populations: Grays River, Hardy, and Hamilton Creek in Washington State (Figure 16).

This ESU also includes three artificial hatchery programs. There were 16 historical populations in three MPGs in Oregon and Washington between the mouth of the Columbia River and the Cascade crest. Significant spawning now occurs for two of the historical populations. About 88% of the historical populations are extirpated. Table 10 identifies populations within the Columbia River Chum salmon ESU, their abundances, and hatchery input.

Table 13. Columbia River Chum salmon populations, abundances, and hatchery contributions (Good, Waples et al. 2005).

Current Populations	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Youngs Bay	Unknown	0	0
Grays River	7,511	331-704	Unknown
Big Creek	Unknown	0	0
Elochoman River	Unknown	0	0
Clatskanie River	Unknown	0	0
Mill, Abernathy, and German Creeks	Unknown	0	0
Scappoose Creek	Unknown	0	0
Cowlitz River	141,582	0	0
Kalama River	9,953	0	0
Lewis River	89,671	0	0
Salmon Creek	Unknown	0	0
Clackamus River	Unknown	0	0
Sandy River	Unknown	0	0
Washougal River	15,140	0	0
Lower gorge tributaries	>3,141	425	0
Upper gorge tributaries	>8,912	0	0
Total	**>283,421**	**756-1,129**	

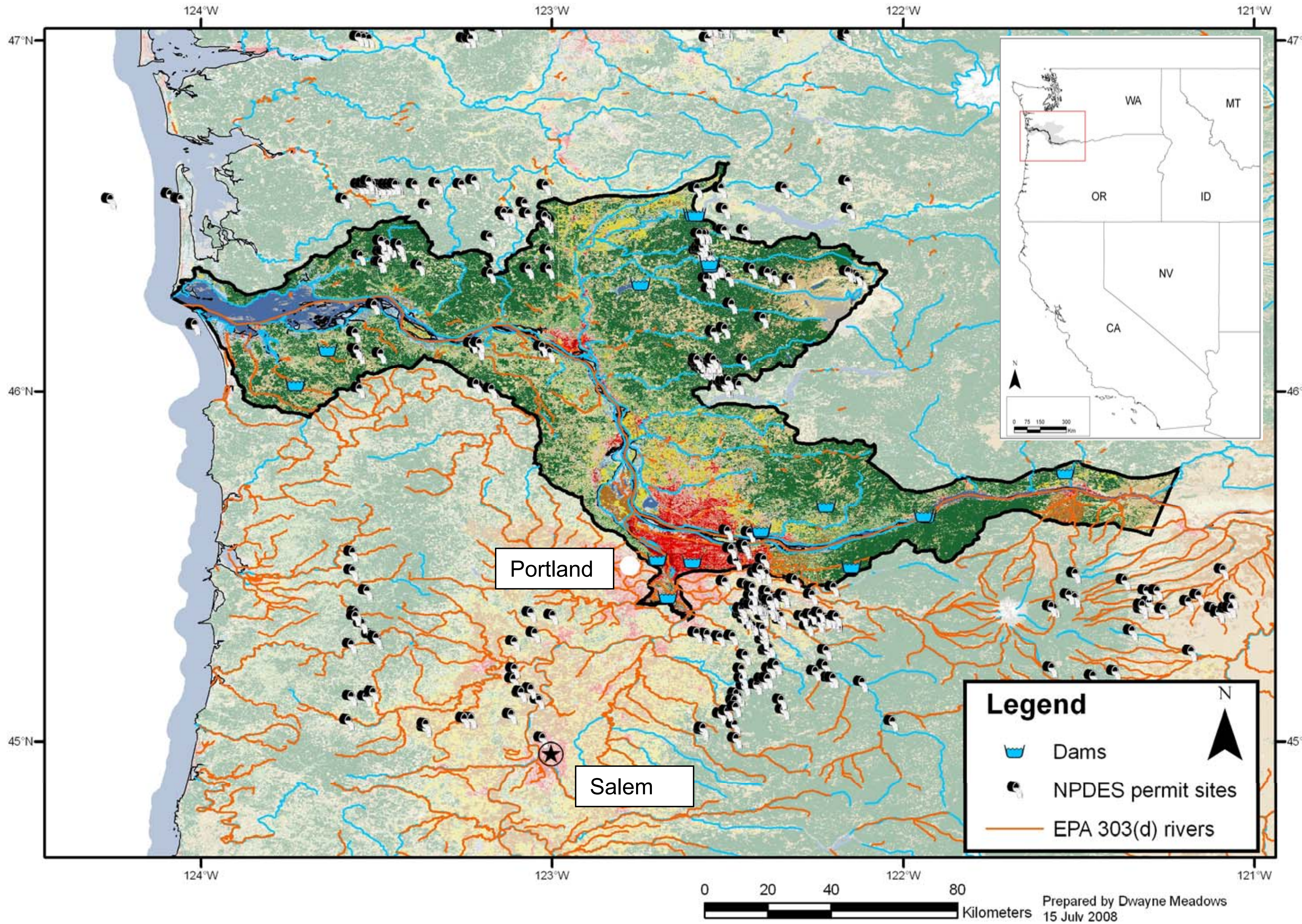


Figure 16. Columbia River Chum salmon distribution. Land Cover Class Legend in Figure 7.

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 in contrast to other salmonids (*e.g.*, steelhead, coho salmon, and most Chinook salmon), which usually migrate to sea at a larger size after months or years of freshwater rearing. Juvenile chum salmon reside in estuaries to feed before beginning a 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 commonly 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 is normally at the time when the fish has grown to a size that allows them to feed upon neritic organisms and and avoid predators (Burgner 1991).

Although most juvenile chum salmon migrate rapidly from freshwater to shallow nearshore marine habitats after emergence from gravel beds, some may remain up to a year in fresh water in large northern rivers. The period of estuarine residence appears to be a critical life history phase and may play a major role in determining the size of the subsequent adult run back to freshwater.

Status and Trends

Columbia River chum salmon were listed as threatened on March 25, 1999, and their threatened status was reaffirmed on June 28, 2005 (71 FR 37160). Chum salmon in the Columbia River once numbered in the hundreds of thousands of adults and were reported in almost every river in the LCR basin. However, by the 1950s most runs disappeared (Rich 1942; Marr 1943; Fulton 1968). The total number of chum salmon returning to the Columbia River in the last 50 years has averaged a few thousand per year, with returns limited to a very restricted portion of the historical range. Significant spawning occurs in only two of the 16 historical populations. Nearly 88% of the historical populations are extirpated. The two remaining populations are the Grays River and the Lower Gorge (Good, Waples et al. 2005). Chum salmon appear to be extirpated from the Oregon portion of this ESU. In 2000, the Oregon Department of Fish and Wildlife (ODFW) conducted surveys to determine the abundance and distribution of chum salmon in the Columbia River. Of 30 sites surveyed, only one chum salmon was observed.

Historically, the Columbia River chum salmon supported a large commercial fishery in the first half of this century which landed more than 500,000 fish per year as recently as 1942. Commercial catches declined beginning in the mid-1950s, and in later years rarely exceeded 2,000 per year. During the 1980s and 1990s, the combined abundance of natural spawners for the Lower Gorge, Washougal, and Grays River populations was below 4,000 adults. In 2002, however, the abundance of natural spawners exhibited a substantial increase at several locations (estimate of natural spawners is approximately 20,000 adults). The cause of this dramatic increase in abundance is unknown. Estimates of abundance and trends are available only for the Grays River and Lower Gorge populations. The 10-year trend was negative for the Grays River population and just over 1.0 for the Lower Gorge. The Upper Gorge population, and all four of the populations on the Oregon side of the river in the Coastal MPG, are extirpated or nearly so (McElhaney, Chilcote et al. 2007). However, long- and short-term productivity trends for populations are at or below replacement. Regarding spatial structure, few Columbia River chum salmon have been observed in tributaries between The Dalles and Bonneville dams. Surveys of the White Salmon River in 2002 found one male and one female carcass and the latter had not spawned (Ehlke and Keller 2003). 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. Finally, most Columbia River chum populations have been functionally extirpated or are presently at very low abundance levels. The loss of off-channel habitat and the extirpation of approximately 17 historical populations increase this species' vulnerability to environmental variability and catastrophic events. Overall, the populations that remain have low abundance, limited distribution, and poor connectivity (Good, Waples et al. 2005).

Critical Habitat

Critical habitat was originally designated for this species on February 16, 2000 (65 FR 7764) and was re-designated on September 2, 2005 (70 FR 52630). The critical habitat designation for this ESU identifies PCEs that include sites necessary to support one or more chum salmon life stages. Columbia River chum salmon have PCEs of: (1) Freshwater spawning, (2) freshwater rearing, (3) freshwater migration, (4) estuarine areas free of obstruction, (5) nearshore marine areas free of obstructions, and (6) offshore marine areas with good water quality. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity.

Of 21 subbasins reviewed in NMFS' assessment of critical habitat for the Columbia River chum salmon ESU, three subbasins were rated as having a medium conservation value, no subbasins were rated as low, and the majority of subbasins (18), were rated as having a high conservation value to Columbia River chum salmon. Washington's federal lands were rated as having high conservation value to the species. The major factors limiting recovery for Columbia River chum salmon are altered channel form and stability in tributaries, excessive sediment in tributary spawning gravels, altered stream flow in tributaries and the mainstem Columbia River, loss of some tributary habitat types, and harassment of spawners in the tributaries and mainstem. The above activities and features also introduce sediment, nutrients, biocides, metals, and other pollutants into surface and ground water and degrade water quality in the freshwater, estuarine, and coastal ecosystems throughout the Pacific Northwest.

Hood Canal Summer-Run Chum Salmon

Distribution

This ESU includes all naturally spawned populations of summer-run chum salmon in Hood Canal and its tributaries as well as populations in Olympic Peninsula rivers between Hood Canal and Dungeness Bay, Washington (64 FR 14508, Figure 17). Eight artificial propagation programs are considered as part of 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. NMFS determined that these artificially propagated stocks are no more divergent relative to the local natural population(s) than what would be expected between closely related natural populations within the species. Table 14 identifies populations within the Hood Canal summer-run Chum salmon ESU, their abundances, and hatchery input.

Table 14. Hood Canal summer-run Chum salmon populations, abundances, and hatchery contributions (Good, Waples et al. 2005).

Current Populations	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Jimmycomelately Creek	Unknown	~60	Unknown
Salmon/Snow creeks	Unknown	~2,200	0-69%
Big/Little Quilcene rivers	Unknown	~4,240	5-51%
Lilliwaup Creek	Unknown	~164	Unknown
Hamma Hamma River	Unknown	~758	Unknown
Duckabush River	Unknown	Unknown	Unknown
Dosewallips River	Unknown	~900	Unknown
Union River	Unknown	~690	Unknown
Chimacum Creek	Unknown	0	100
Big Beef Creek	Unknown	0	100
Dewetto Creek	Unknown	0	Unknown
Total	**Unknown**	**~9,012**	

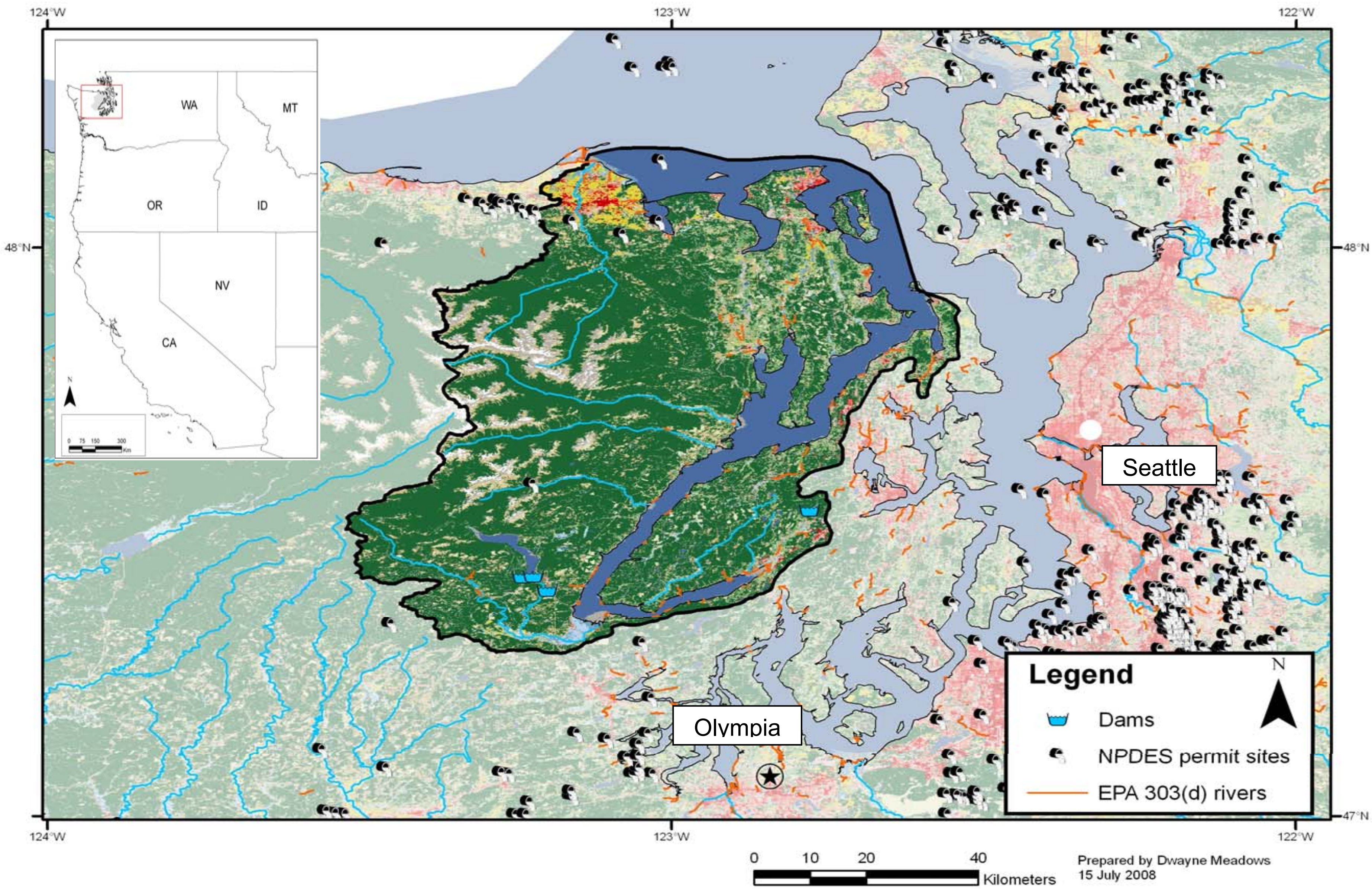


Figure 17. Hood Canal Summer-run Chum salmon distribution. Land Cover Class Legend in Figure 7.

Life History

The Hood Canal summer-run Chum salmon are defined in the Salmon and Steelhead Stock Inventory (WDF, WDW et al. 1993) as fish that spawn from mid-September to mid-October. However, summer chum have been known to enter natal rivers in late August. Fall-run chum salmon are defined as fish that spawn from November through December or January. Run-timing data for as early as 1913 indicated temporal separation between summer and fall chum salmon in Hood Canal (Johnson, Grant et al. 1997). Hood Canal summer Chum salmon are genetically distinct from healthy populations of Hood Canal fall Chum salmon originating within this area. Hood Canal summer Chum salmon return to natal rivers to spawn during the August through early October period. The fall Chum salmon spawn between November and December, when streams are higher and water temperature is lower.

The time to hatching varies among populations and among individuals within a population (Salo 1991). Fry tend to emerge when they had their best chances of surviving in streams and estuaries (Koski 1975). A variety of factors may influence the time to hatching, emergence from the gravel, or both. They include dissolved oxygen, gravel size, salinity, nutritional conditions, behaviour of alevins in the gravel and incubation temperature [reviewed in (Bakkala 1970; Schroder, Koski et al. 1974; Schroder 1977; Salo 1991)]. The average residence time in estuaries for Hood canal chum salmon is 23 days. Fry in Hood Canal have not been observed to display daily tidal migrations (Bax 1983). Fry movement is associated with prey availability. Summer-run chum salmon migrate up the Hood Canal and into the main body of Puget Sound. Fish may emerge from streams over an extended period or juveniles may also remain in Quilcene Bay for several weeks.

Status and Trends

Hood Canal summer-run Chum salmon were listed as threatened on March 25, 1999, and reaffirmed as threatened on June 28, 2005 (70 FR 37160). Adult returns for some populations in the Hood Canal summer-run Chum salmon species showed modest improvements in 2000, with upward trends continuing in 2001 and 2002. The recent five-year mean abundance is variable among populations in the species, ranging from one fish to nearly 4,500 fish. Hood Canal summer-run chum salmon are the focus of an extensive rebuilding program developed and implemented since 1992 by the state and tribal co-managers. Two populations (the combined Quilcene and Union River populations) are above the conservation thresholds established by the rebuilding plan. However, most populations remain depressed. Estimates of the fraction of naturally spawning hatchery fish exceed 60% for some populations. This indicates that reintroduction programs are supplementing the numbers of total fish spawning naturally in streams. Long-term trends in productivity are above replacement for only the Quilcene and Union River populations. Buoyed by recent increases, seven populations are exhibiting short-term productivity trends above replacement.

Of an estimated 16 historical populations in the ESU, seven populations are believed to have been extirpated or nearly extirpated. Most of these extirpations have occurred in populations on the eastern side of Hood Canal, generating additional concern for ESU spatial structure. The widespread loss of estuary and lower floodplain habitat was noted by the BRT as a continuing threat to ESU spatial structure and connectivity. There is some concern that the Quilcene hatchery stock is exhibiting high rates of straying, and may represent a risk to historical population structure and diversity. However, with the extirpation of many local populations, much of this historical structure has been lost, and the use of Quilcene hatchery fish may represent one of a few remaining options for Hood Canal summer-run Chum salmon conservation.

Of the eight programs releasing summer chum salmon that are considered to be part of this ESU, six of the programs are supplementation programs implemented to preserve and increase the abundance of native populations in their natal watersheds. NMFS' assessment of the effects of artificial propagation on ESU extinction risk concluded that these hatchery programs collectively do not substantially reduce the extinction risk of the ESU. The hatchery programs are reducing risks to ESU abundance by increasing total ESU abundance as well as the number of naturally spawning summer-run chum salmon.

Several of the programs have likely prevented further population extirpations in the ESU. The contribution of ESU hatchery programs to the productivity of the ESU in-total is uncertain. The hatchery programs are benefiting ESU spatial structure by increasing the spawning area utilized in several watersheds and by increasing the geographic range of the ESU through reintroductions. These programs also provide benefits to ESU diversity. By bolstering total population sizes, the hatchery programs have likely stemmed adverse genetic effects for populations at critically low levels. Additionally, measures have been implemented to maintain current genetic diversity, including the use of native broodstock and the termination of the programs after 12 years of operation to guard against long-term domestication effects. Collectively, artificial propagation programs in the ESU presently provide a slight beneficial effect to ESU abundance, spatial structure, and diversity. However, artificial propagation programs also provide uncertain effects to ESU productivity.

Critical Habitat

Critical habitat for this species was designated on September 2, 2005 (70 FR 52630). Hood Canal summer-run chum salmon have PCEs of: (1) Freshwater spawning, (2) freshwater rearing, (3) freshwater migration, (4) estuarine areas free of obstruction, (5) nearshore marine areas free of obstructions, and (6) offshore marine areas with good water quality. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity.

Of 17 subbasins reviewed in NMFS' assessment of critical habitat for the Hood Canal chum salmon ESU, 14 subbasins were rated as having a high conservation value, while only three were rated as having a medium value to the conservation. Limiting factors identified for this species include: (1) Degraded floodplain and mainstem river channel structure, (2) degraded estuarine water quality conditions and loss of estuarine habitat, (3) riparian area degradation and loss of in-river wood in mainstem, (4) excessive sediment in spawning gravels, and (5) reduced stream flow in migration areas. These conditions also introduce sediment, nutrients, biocides, metals, and other pollutants into surface and ground water and degrade water quality in the freshwater, estuarine, and coastal ecosystems throughout the Pacific Northwest.

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 et al. 1986). We discuss the distribution, life history diversity, status, and critical habitat of the four endangered and threatened coho species separately.

After entering the ocean, immature coho salmon initially remain in nearshore waters close to the parent stream. Most coho salmon adults are three-year-olds, having spent approximately 18 months rearing in freshwater and 18 months in salt water. Most coho salmon enter rivers between September and February. However, entry is influenced by discharge and other factors. In many systems, coho salmon and other Pacific salmon are unable to enter the rivers until sufficiently strong flows open passages and provide sufficient depth. Wild female coho salmon return to spawn almost exclusively at age three. 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.

Eggs incubate for about 35 to 50 days, and start emerging from the gravel within two to three weeks after hatching. Following emergence, fry move to shallow areas near the stream banks. As fry grow, they disperse upstream and downstream to establish and defend territories. Juvenile rearing usually occurs in tributaries with gradients of 3% or less, although they may move to streams with gradients of 4 to 5%. Juvenile coho salmon are often found in small streams less than five ft wide, and may migrate considerable distances to rear in lakes and off-channel ponds. During the summer, fry prefer pools featuring adequate cover such as large woody debris, undercut banks, and overhanging vegetation. Overwintering tends to occur in larger pools and backwater areas.

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

Status and Trends

Coho salmon survive only in aquatic ecosystems and depend on the quantity and quality of those aquatic systems. Coho salmon, like the other salmon NMFS has listed, have declined under the combined effects of overharvests in fisheries; competition from fish raised in hatcheries and native and non-native exotic species; dams that block their migrations and alter river hydrology; gravel mining that impedes their migration and alters the dynamics of the rivers and streams that support juveniles; water diversions that deplete water levels in rivers and streams; destruction or degradation of riparian habitat that increase water temperatures in rivers and streams sufficient to reduce the survival of juvenile chum salmon; and land use practices (logging, agriculture, urbanization) that destroy wetland and riparian ecosystems. The above activities and features introduce sediment, nutrients, biocides, metals, and other pollutants into surface and ground water and degrade water quality in the freshwater, estuarine, and coastal ecosystems throughout the Pacific Northwest.

Central California Coast Coho Salmon

Distribution

The CCC coho salmon ESU extends from Punta Gorda in northern California south to and including the San Lorenzo River in central California (Weitkamp, Wainwright et al. 1995). Table 15 identifies populations within the CCC Coho salmon ESU, their abundances, and hatchery input (Figure 18).

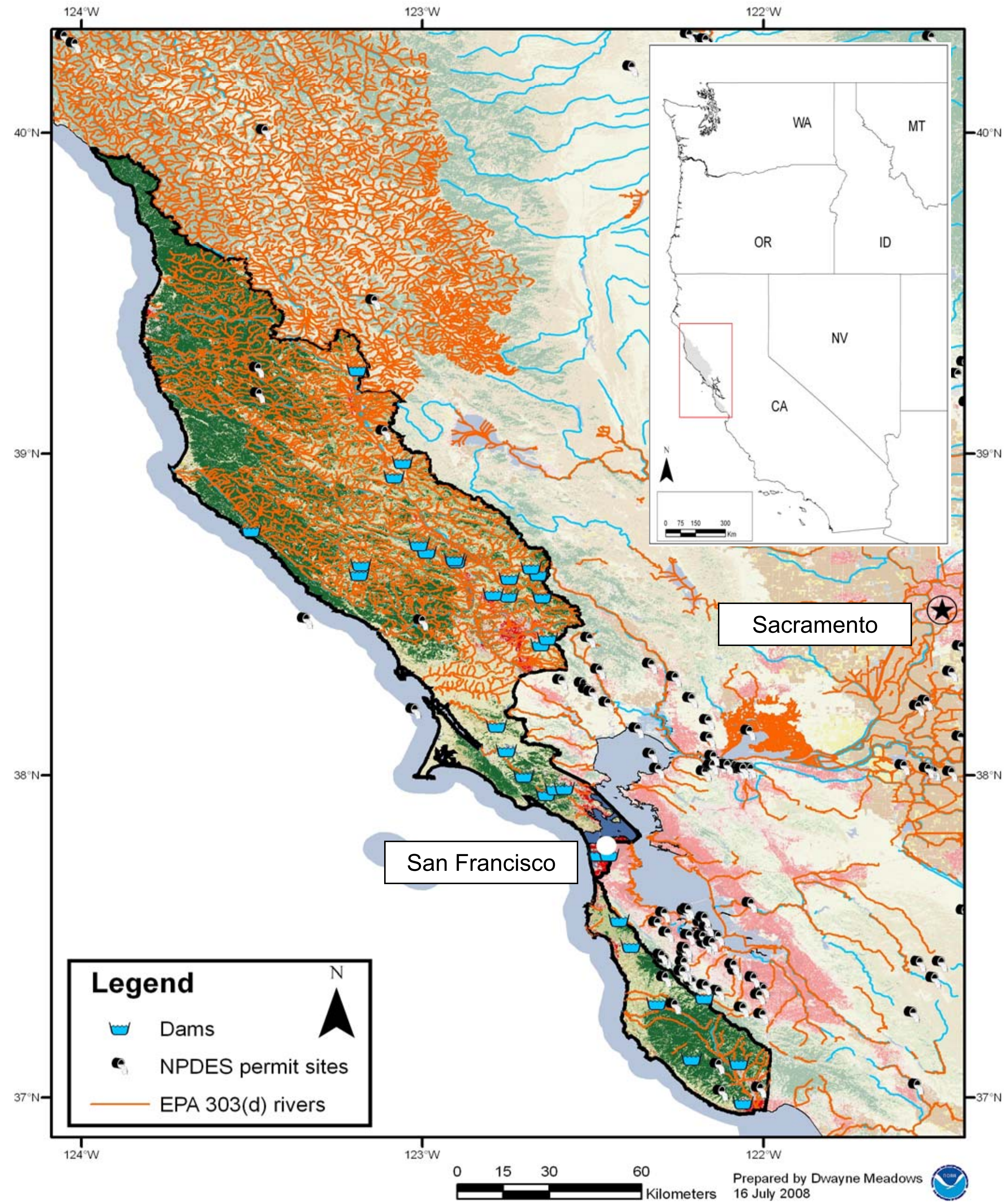


Figure 18. CCC Coho salmon distribution. Land Cover Class Legend in Figure 7.

Table 15. CCC Coho salmon populations, abundances, and hatchery contributions (Good, Waples et al. 2005).

River/Region	Historical Escapement (1963)	1987-1991 Escapement Abundance	Hatchery Abundance Contributions
Ten Mile River	6,000	160	Unknown
Noyo River	6,000	3,740	Unknown
Big River	6,000	280	Unknown
Navarro River	7,000	300	Unknown
Garcia River	2,000	500 (1984-1985)	Unknown
Other Mendacino County rivers	10,000	470	Unknown
Gualala River	4,000	200	Unknown
Russian River	5,000	255	Unknown
Other Sonoma County rivers	1,000	180	Unknown
Marin County	5,000	435	Unknown
San Mateo County	1,000	Unknown	Unknown
Santa Cruz County	1,500	50 (1984-1985)	Unknown
San Lorenzo River	1,600	Unknown	Unknown
Total	**200,000-500,000**	**6,570 (min)**	

Life History

Both run and spawn timing of coho salmon in this region are very late (both peaking in January), with little time spent in freshwater between river entry and spawning. This compressed adult freshwater residency appears to coincide with the single, brief peak of river flow characteristic of this area.

Status and Trends

The CCC coho salmon ESU was originally listed as threatened under the ESA on October 31, 1996 (61 FR 56138) and later revised to endangered status on June 28, 2005 (70 FR 37160). The 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. The ESU also includes four artificial propagation programs: the Don Clausen Fish Hatchery Captive Broodstock Program, Scott Creek/King Fisher Flats Conservation Program, Scott Creek Captive Broodstock Program, and the Noyo River Fish Station egg-take Program coho hatchery programs.

Information on the abundance and productivity trends for the naturally spawning component of the CCC coho salmon ESU is extremely limited. There are no long-term time series of spawner abundance for individual river systems. Analyses of juvenile coho presence-absence information, juvenile density surveys, and irregular adult counts for the South Fork Noyo River indicate low abundance and long-term downward trends for the naturally spawning populations throughout the ESU. Improved ocean conditions coupled with favorable stream flows and harvest restrictions have contributed to increased returns in 2001 in streams in the northern portion of the ESU, as indicated by an increase in the observed presence of fish in historically occupied streams. Data are lacking for many river basins in the southern two thirds of the ESU where naturally spawning populations are considered at the greatest risk. The extirpation or near extirpation of natural coho salmon populations in several major river basins, and across most of the southern historical range of the ESU, represents a significant risk to ESU spatial structure and diversity. Artificial propagation of coho salmon within the CCC ESU has declined since the ESU was listed in 1996 though it continues at the Noyo River and Scott Creek facilities, and two captive broodstock populations have recently been established. Genetic diversity risk associated with out-of-basin transfers appears to be minimal. However, diversity risk from domestication selection and low effective population sizes in the remaining hatchery programs remains a concern. An out-of-ESU artificial propagation program for coho was operated at the Don Clausen hatchery on the Russian River through the mid-1990s. However, the program was terminated in 1996. Termination of this program was considered by the Biological Review Team (BRT) as a positive development for naturally produced coho salmon in this ESU.

CCC coho salmon populations continue to be depressed relative to historical numbers. Strong indications show that breeding groups have been lost from a significant percentage of streams in their historical range. A number of coho salmon populations in the southern portion of the range appear to be either extinct or nearly so. They include those in Gualala, Garcia, and Russian rivers, as well as smaller coastal streams in and south of San Francisco Bay (Good, Waples et al. 2005). For the naturally spawning component of the ESU, the BRT found very high risk (of extinction) for the abundance, productivity, and spatial structure VSP parameters and comparatively moderate risk with respect to the diversity VSP parameter. The lack of direct estimates of the performance of the naturally spawned populations in this ESU, and the associated uncertainty this generates, was of specific concern to the BRT, as the naturally spawned component of the CCC coho salmon ESU was "in danger of extinction".

Critical Habitat

Critical habitat for the CCC coho salmon ESU was designated on May 5, 1999 (64 FR 24049). Designated critical habitat 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.

Lower Columbia River Coho Salmon

Distribution

LCR coho salmon include all naturally spawned populations of coho salmon in the Columbia River and its tributaries in Oregon and Washington, from the mouth of the Columbia up to and including the Big White Salmon and Hood Rivers, and includes the Willamette River to Willamette Falls, Oregon (Figure 19). This ESU also includes 25 artificial propagation programs: the Grays River, Sea Resources Hatchery, Peterson Coho Project, Big Creek Hatchery, Astoria High School Coho Program, Warrenton High School Coho Program, Elochoman Type-S Coho Program, Elochoman Type-N Coho Program, Cathlamet High School FFA Type-N Coho Program, Cowlitz Type-N Coho Program in the Upper and Lower Cowlitz Rivers, Cowlitz Game and Anglers Coho Program, Friends of the Cowlitz Coho Program, North Fork Toutle River Hatchery, Kalama River Type-N Coho Program, Kalama River Type-S Coho Program, Washougal Hatchery Type-N Coho Program, Lewis River Type-N Coho Program, Lewis River Type-S Coho Program, Fish First Wild Coho Program, Fish First Type-N Coho Program, Syverson Project Type-N Coho Program, Eagle Creek National Fish Hatchery, Sandy Hatchery, and the Bonneville/Cascade/Oxbow complex coho hatchery programs.

Table 16 identifies populations within the LCR Coho salmon ESU, their abundances, and hatchery input.

Table 16. LCR Coho salmon populations, abundances, and hatchery contributions (Good, Waples et al. 2005).

River/Region	Historical Abundance	2002 Spawner Abundance	Hatchery Abundance Contributions
Youngs Bay and Big Creek	Unknown	4,473	91%
Grays River	Unknown	Unknown	Unknown
Elochoman River	Unknown	Unknown	Unknown
Clatskanie River	Unknown	229	60%
Mill, Germany, and Abernathy creeks	Unknown	Unknown	Unknown
Scappoose Rivers	Unknown	458	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	1,001	12%
Lower Clackamas River	Unknown	2,402	78%
Salmon Creek	Unknown	Unknown	Unknown
Upper Sandy River	Unknown	310	0%
Lower Sandy River	Unknown	271	97%
Washougal River	Unknown	Unknown	Unknown
LCR gorge tributaries	Unknown	Unknown	Unknown
White Salmon	Unknown	Unknown	Unknown
Upper Columbia River gorge tributaries	Unknown	1,317	>65%
Hood River	Unknown	Unknown	Unknown
Total	**Unknown**	**10,461 (min)**	

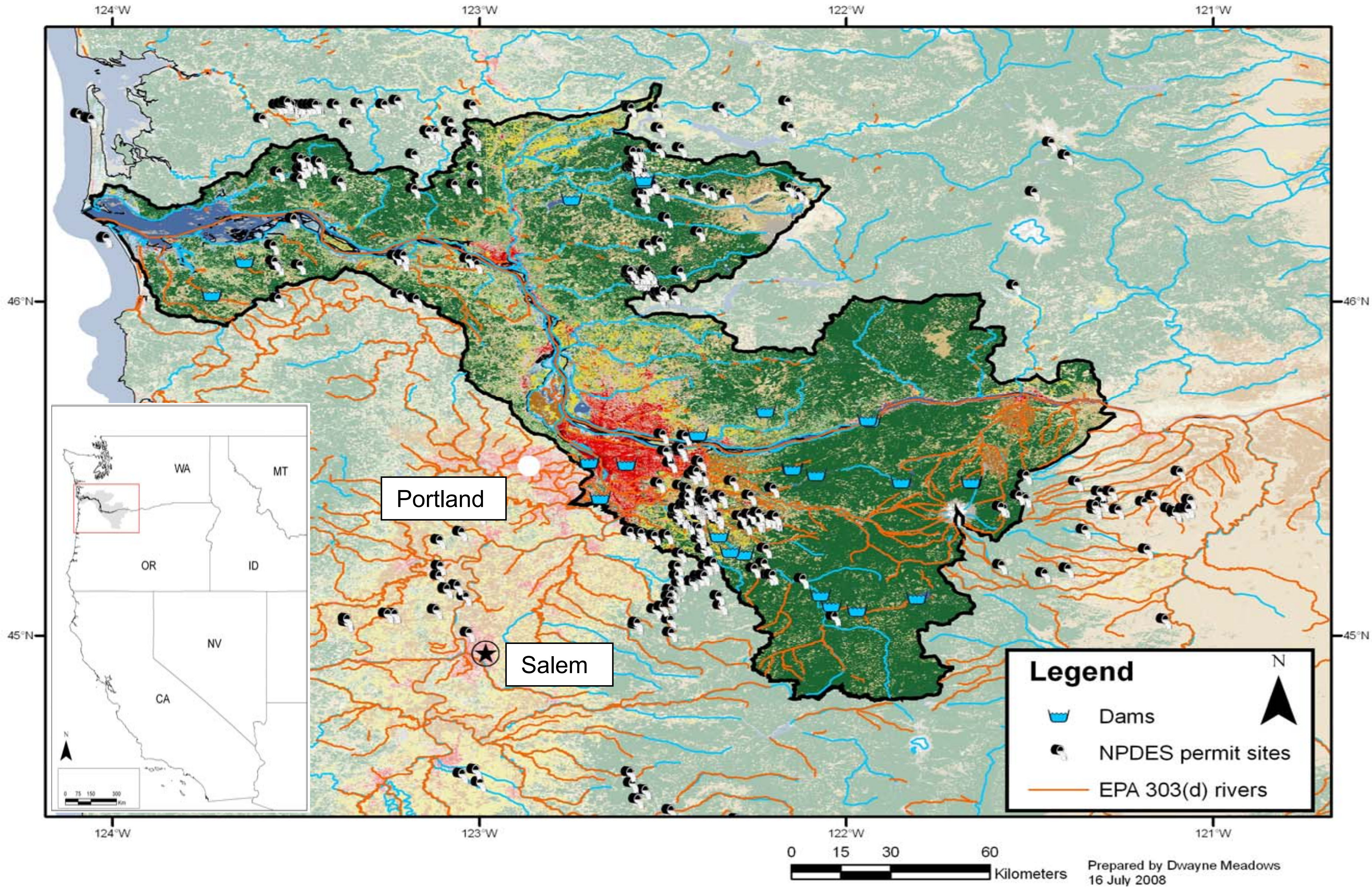


Figure 19 . LCR coho salmon distribution. Land Cover Class Legend in Figure 7.

Life History

Although run time variation is inherent to coho salmon life history, the ESU includes two distinct runs: early returning (Type S) and late returning (Type N). Type S coho salmon generally migrate south of the Columbia once they reach the ocean, returning to freshwater in mid-August and to the spawning tributaries in early September. Spawning peaks from mid-October to early November. Type N coho salmon have a northern distribution in the ocean, 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. However some spawning occurs in February and as late as March (LCFRB 2004). Almost all LCR ESU coho salmon females and most males spawn at three years of age.

Status and Trends

LCR coho salmon were listed as endangered on June 28, 2005 (70 FR 37160). The vast majority (over 90%) of the historic population in the LCR coho salmon ESU appear to be either extirpated or nearly so. The two populations with any significant natural production (Sandy and Clackamas) are at appreciable risk because of low abundance, declining trends, and failure to respond after a dramatic reduction in harvest. Most of the other populations are believed to have very little, if any, natural production.

The Sandy population had a recent mean abundance of 342 spawners and a very low fraction of hatchery-origin spawners. Trends in the Sandy are similar to the Clackamas. The long-term trends and growth rate estimates over the period 1977 to 2001 have been slightly positive and the short-term trends have been slightly negative. Other populations in this ESU are dominated by hatchery production. There is very little, if any, natural production in Oregon beyond the Clackamas and Sandy rivers. The Washington side of the ESU is also dominated by hatchery production. There are no populations with appreciable natural production. The most serious threat facing this ESU is the scarcity of naturally-produced spawners, with attendant risks associated with small population, loss of diversity, and fragmentation and isolation of the remaining naturally-produced fish. In the only two populations with significant natural production (Sandy and Clackamas), short- and long-term trends are negative and productivity (as gauged by pre-harvest recruits) is down sharply from recent (1980s) levels.

The Federal Columbia River Power System Opinion (FCRPS) (2008) describes this ESU as consisting of three MPGs. Each is comprised of three to 14 populations. In many cases, populations have low abundance and natural runs have been extensively replaced by hatchery production. Abundance estimates are available for only five populations and trend estimates for only two. Time series are not available for Washington coho populations. The 100-year risk of extinction was derived qualitatively, based on risk categories and criteria identified by the W/LCTRT in 2004. Most of the population of LCR had high or very high extinction risk probabilities. Spatial structure has been substantially reduced by the loss of access to the upper portions of some basins from tributary hydro development (*i.e.*, Condit Dam on the Big White Salmon River and Powerdale Dam on the Hood River). Finally, the diversity of populations in all three MPGs has been eroded by large hatchery influences and periodically, low effective population sizes. Nevertheless, the genetic legacy of the Lewis and Cowlitz River coho salmon populations is preserved in ongoing hatchery programs.

Critical Habitat

NMFS has not designated critical habitat for LCR coho salmon.

SONCC *Coho Salmon*

Distribution

SONCC coho salmon 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 20).

This ESU also includes three artificial propagation programs: the Cole Rivers Hatchery (ODFW stock #52), Trinity River Hatchery, and Iron Gate Hatchery coho hatchery programs. The three major river systems supporting Southern Oregon / Northern Coastal California coast coho are the Rogue, Klamath (including the Trinity), and Eel rivers.

Life History

SONCC coho salmon enter rivers in September or October. River entry is much 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-Decmeber 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. Coho salmon adults spawn at age three, spending just over a year in freshwater and a year and a half in the ocean.

Status and Trends

SONCC coho salmon were listed as threatened on May 7, 1997 (62 FR 24588). This species retained its original classification when its status was reviewed on June 28, 2005 (70 FR 37160). The status of coho salmon coast wide, including the SONCC coho salmon ESU, was formally assessed in 1995 (Weitkamp, Wainwright et al. 1995). Two subsequent status review updates have been published by NMFS. One review update addressed all West Coast coho salmon ESUs (Busby, Wainwright et al. 1996). The second update specifically addressed the Oregon Coast and SONCC coho salmon ESUs (Gustafson, Wainwright et al. 1997). In the 1997 status update, estimates of natural population abundance were based on very limited information. New data on presence/absence in northern California streams that historically supported coho salmon were even more disturbing than earlier results. Data indicated that a smaller percentage of streams contained coho salmon compared to the percentage presence in an earlier study. However, it was unclear whether these new data represented actual trends in local extinctions, or were biased by sampling effort.

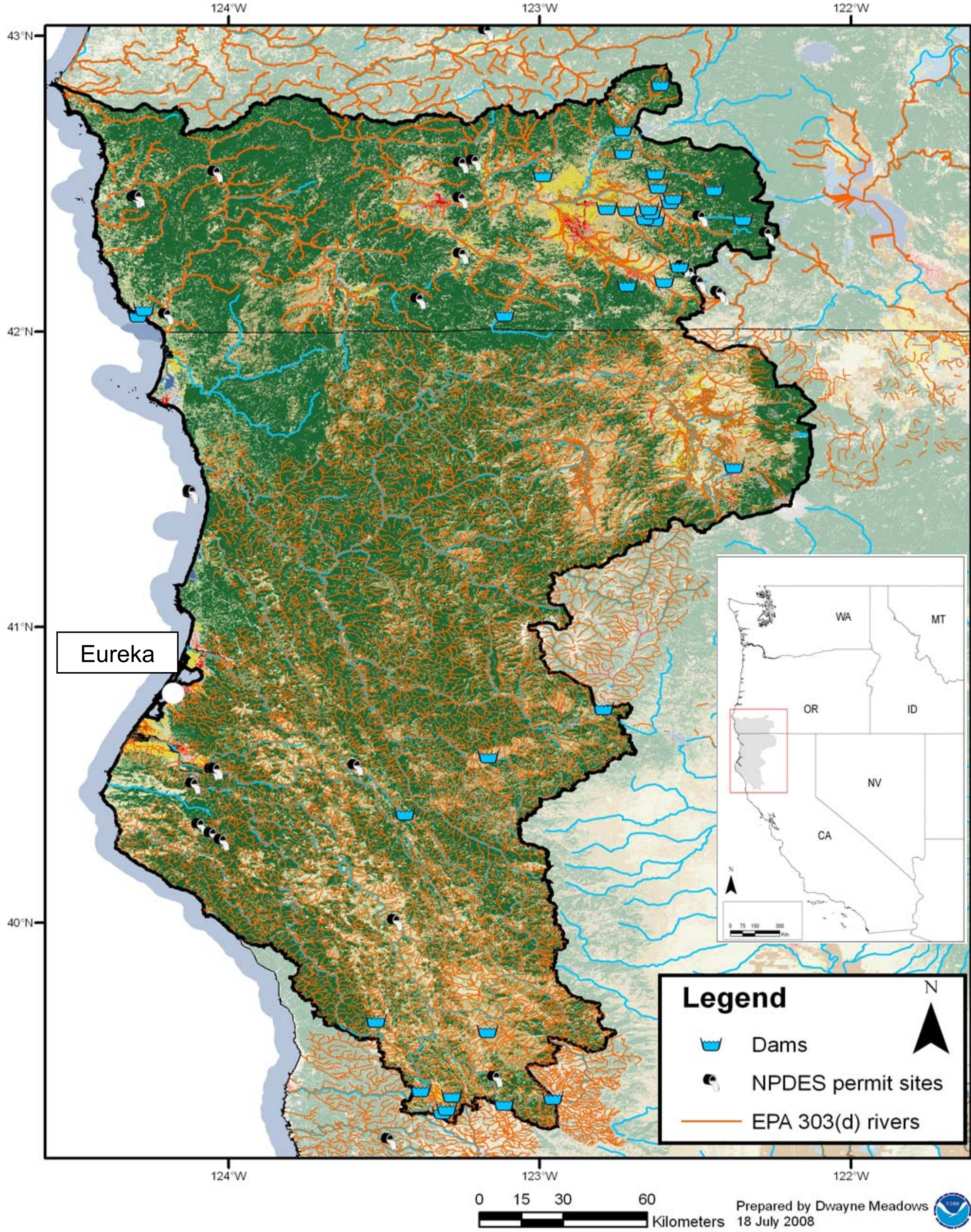


Figure 20. SONCC coho salmon distribution. Legend for Land Cover Class in Figure 7.

Data on population abundance and trends are limited for the California portion of this ESU. No regular estimates of natural spawner escapement are available. Historical point estimates of coho salmon abundance for the early 1960s and mid-1980s suggest that statewide coho spawning escapement in the 1940s ranged between 200,000 and 500,000 fish. Numbers declined to about 100,000 fish by the mid-1960s with about 43% originating from this ESU. Brown et al. (1994) estimated that the California portion of this ESU was represented by about 7,000 wild and naturalized coho salmon (Good, Waples et al. 2005). In the Klamath River, the estimated escapement has dropped from approximately 15,400 in the mid-1960s to about 3,000 in the mid-1980s, and more recently to about 2,000 (Good, Waples et al. 2005). The second largest producing river in this ESU, the Eel River, dropped from 14,000, to 4,000 to about 2,000 during the same period. Historical estimates are considered "best guesses" made using a combination of limited catch statistics, hatchery records, and the personal observations of biologists and managers.

Most recently, Williams et al. (2006) described the structure of historic populations of SONCC coho salmon. They described three categories of populations: functionally independent populations, potentially independent populations, and dependent populations. Functionally independent populations are populations capable of existing in isolation with a minimal risk of extinction. Potentially independent populations are similar but rely on some interchange with adjacent populations to maintain a low probability of extinction. Dependent populations have a high risk of extinction in isolation over a 100-year timeframe and rely on exchange of individuals from adjacent populations to maintain themselves.

Critical Habitat

Critical habitat was designated for the SONCC coho salmon on November 25, 1997, and re-designated on May 5, 1999. 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. Of 155 historical streams for which data are available, 63% likely still support coho salmon. Limiting factors identified for this species include: (1) Loss of channel complexity, connectivity and sinuosity, (2) loss of floodplain and estuarine habitats, (3) loss of riparian habitats and large in-river wood, (4) reduced streamflow, (5) poor water quality, temperature and excessive sedimentation, and (6) unscreened diversions and fish passage structures.

Oregon Coast Coho Salmon

Distribution

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 21). One hatchery stock, the Cow Creek (ODFW stock # 37) hatchery coho, is considered part of the ESU. Table 17 identifies populations within the OC coho salmon ESU, their abundances, and hatchery input.

Table 17. Oregon Coast Coho salmon populations, abundances, and hatchery contributions (Good, Waples et al. 2005).

Basin	Historical Abundance	Recent Spawner Abundance	Hatchery Abundance Contributions
Necanicum	Unknown	1,889	35-40%
Nehalem	Unknown	18,741	40-75%
Tillamook	Unknown	3,949	30-35%
Nestucca	Unknown	3,846	~5%
Siletz	Unknown	2,295	~50%
Yaquima	Unknown	3,665	~25%
Alsea	Unknown	3,621	~40%
Siuslaw	Unknown	16,213	~40%
Umpqua	Unknown	24,351	<10%
Coos	Unknown	20,136	<5%
Coquille	Unknown	8,847	<5%
Total	**924,000**	**107,553**	

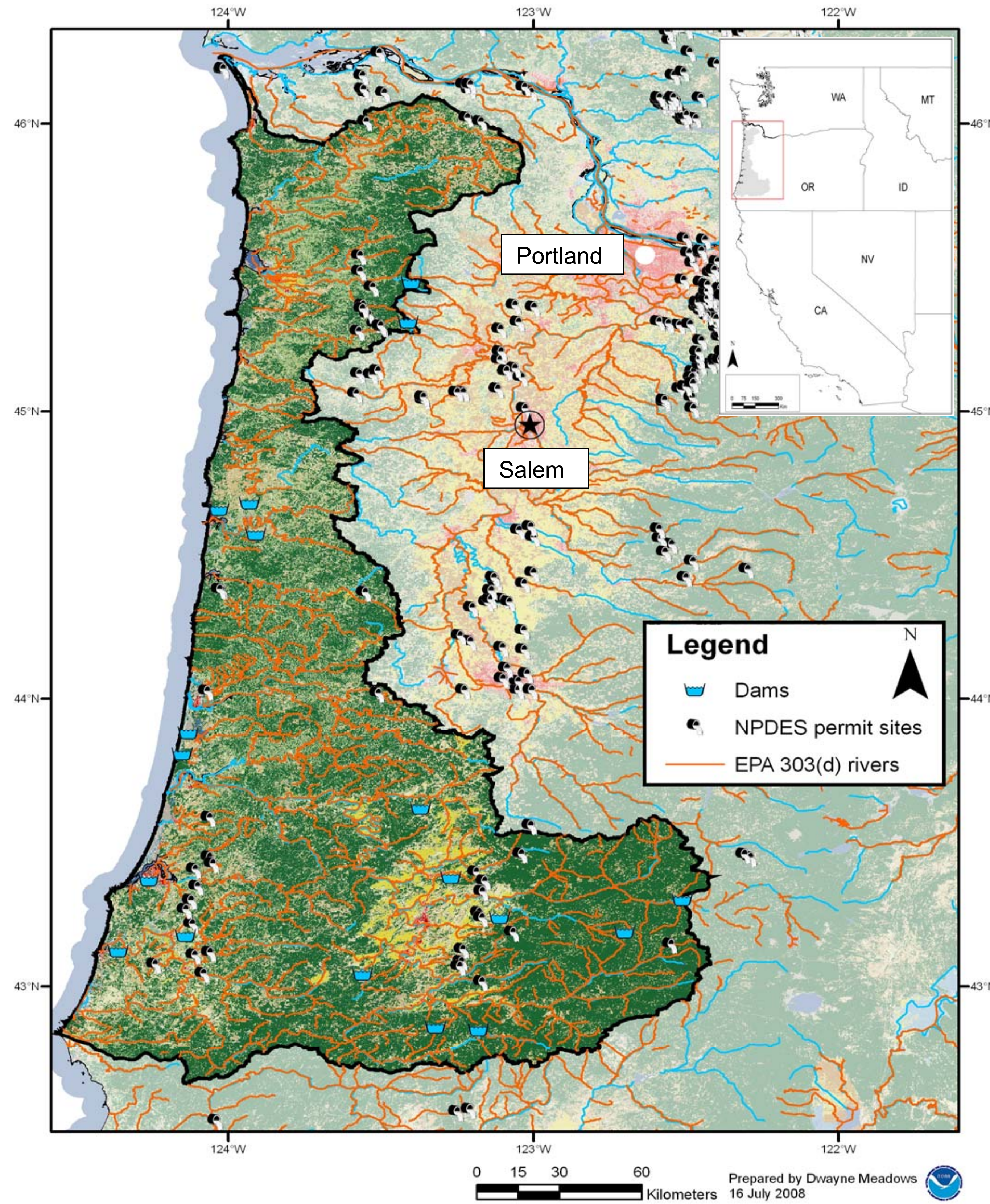


Figure 21. Oregon Coast Coho salmon distribution. Land Cover Class Legend in Figure 7.

Status and Trends

The OC coho salmon ESU was listed as a threatened species on February 11, 2008 (73 FR 7816). The most recent NMFS status review for the OC coho salmon ESU was conducted by the BRT in 2003, which assessed data through 2002. The abundance and productivity of OC coho salmon since the previous status review (Gustafson, Wainwright et al. 1997) represented some of the best and worst years on record. Yearly adult returns for this ESU were in excess of 160,000 natural spawners in 2001 and 2002, far exceeding the abundance observed for the past several decades. These encouraging increases in spawner abundance in 2000–2002 were preceded, however, 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. 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 (although comprehensive population-level survey data have only been available since 1980). The encouraging 2000–2002 increases in natural spawner abundance occurred in many populations in the northern portion of the ESU, which were the most depressed at the time of the last review (Gustafson, Wainwright et al. 1997). Although encouraged by the increase in spawner abundance in 2000–2002, the BRT noted that the long-term trends in ESU productivity were still negative due to the low abundances observed during the 1990s (73 FR 7816). Since the BRT convened, the total abundance of natural spawners in the OC coho salmon ESU has declined each year (*i.e.*, 2003-2006). The abundance of total natural spawners in 2006 (111,025 spawners) was approximately 43% of the recent peak abundance in 2002 (255,372 spawners). In 2003, ESU-level productivity (evaluated in terms of the number of spawning recruits resulting from spawners three years earlier) was above replacement, and in 2004, productivity was approximately at replacement level. However, productivity was below replacement in 2005 and 2006, and dropped to the lowest level since 1991 in 2006.

Preliminary spawner survey data for 2007 (the average peak number of spawners per mile observed during random coho spawning surveys in 41 streams) suggest that the 2007-2008 return of Oregon Coast coho salmon is either: (1) much reduced from abundance levels in 2006, or (2) exhibiting delayed run timing from previous years. As of December 13, 2007, the average peak number of spawners per mile was below 2006 levels in 38 of 41 surveyed streams (ODFW 2007 *in* 73 FR 7816). It is possible that the timing of peak spawner abundance is delayed relative to previous years, and that increased spawner abundance in late December and January 2008 will compensate for the low levels observed thus far.

The recent five year geometric mean abundance (2002-2006) of approximately 152, 960 total natural spawners remains well above that of a decade ago (approximately 52,845 from 1992-1996). However, the decline in productivity from 2003 to 2006, despite generally favorable marine survival conditions and low harvest rates, is of concern. (73 FR 7816). The long-term trends in productivity in this ESU remain strongly negative.

Critical Habitat

Critical habitat was proposed for Oregon Coast coho salmon on December 14, 2004 (69 FR 74578). The final designation of critical habitat is included in the final rule published on February 11, 2008 (73 FR 7816). Approximately 6,568 stream miles (10,570 km) and 15 square miles (38.8 sq km) of lake habitat are designated critical habitat. Refer to the final rule for a detailed description of the watersheds included in the critical habitat, and a map for each subbasin.

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 far 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.

The species exhibits riverine and lake life history strategies, the latter of which may be either freshwater resident forms or anadromous forms. The vast majority of sockeye salmon spawn in outlet streams of lakes or in the lakes themselves. These “lake-type” sockeye use the lake environment for rearing for up to three years and then migrate to sea, returning to their natal lake to spawn after one to four years at sea. Some sockeye spawn in rivers, however, without lake habitat for juvenile rearing. Offspring of these riverine spawners tend to use the lower velocity sections of rivers as the juvenile rearing environment for one to two years, or may migrate to sea in their first year.

Certain populations of *O. nerka* become resident in the lake environment over long periods of time 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.

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 to 2,400 eggs per female to 5,000 eggs, depending upon the population and average age of the female. Fecundity in kokanee is much lower and may range from about 300 to less than 2,000 eggs.

Incubation is a function of water temperatures, but generally lasts between 100 and roughly 200 days (Burgner 1991). After emergence, fry move rapidly downstream or upstream along the banks to the lake rearing area. Fry emerging from lakeshore or island spawning grounds may simply move along the shoreline of the lake (Burgner 1991). Juvenile salmonids rely on a variety of non-main channel habitats that are critical to rearing. All listed salmonids use shallow, low flow habitats at some point in their life cycle. Examples of off-channel habitat include alcoves, channel edge sloughs, overflow channels, backwaters, terrace tributaries, off-channel dredge ponds, and braids (Anderson 1999; Swift III 1979).

Sockeye salmon survive only in aquatic ecosystems and depend on the quantity and quality of those aquatic systems. Sockeye salmon, like the other salmon NMFS has listed, have declined under the combined effects of overharvests in fisheries; competition from fish raised in hatcheries and native and non-native exotic species; dams that block their migrations and alter river hydrology; gravel mining that impedes their migration and alters the hydrogeomorphology of the rivers and streams that support juveniles; water diversions that deplete water levels in rivers and streams; destruction or degradation of riparian habitat that increase water temperatures in rivers and streams sufficient to reduce the survival of juvenile chum salmon; and land use practices (logging, agriculture, urbanization) that destroy wetland and riparian ecosystems. These activities and features introduce sediment, nutrients, biocides, metals, and other pollutants into surface and ground water and degrade water quality in the freshwater, estuarine, and coastal ecosystems throughout the Pacific Northwest.

Ozette Lake Sockeye Salmon

Distribution

This ESU includes all naturally spawned populations of sockeye salmon in Ozette Lake, Ozette River, Coal Creek, and other tributaries flowing into Ozette Lake, Washington. This ESU is composed of one historical population, with substantial substructuring of individuals into multiple spawning aggregations (Figure 22). The primary spawning aggregations occur in two beach locations – Allen's and Olsen's beaches, and in two tributaries Umbrella Creek and Big River (both tributary-spawning groups were initiated through a hatchery introduction program).

Sockeye salmon stock reared at the Makah Tribe's Umbrella Creek Hatchery were considered part of the ESU, but were not considered essential for recovery of the ESU. NMFS determined that it is presently not necessary to consider the progeny of intentional hatchery-wild or wild-wild crosses produced through the Makah Tribal hatchery program as listed under the ESA (March 25, 1999, 64 FR 14528). However, once the hatchery fish return and spawn in the wild, their progeny are considered listed.

Life History

The sockeye salmon life history is one of the most complex of any Pacific salmon species because of its variable freshwater residency (one to three years in freshwater), and because the species has several different forms: fish that go to the ocean and back, fish that remain in freshwater, and fish that do both.

Adult Ozette Lake sockeye salmon enter Ozette Lake through the Ozette River from April to early August. Adults remain in the lake for an extended period of time (return April – August; spawn late October-February) before spawning on beaches or in the tributaries. Sockeye salmon spawn primarily in lakeshore upwelling areas in Ozette Lake (at Allen's Bay and Olsen's Beach). Minor spawning may occur below Ozette Lake in the Ozette River or in Coal Creek, a tributary of the Ozette River. Sockeye salmon do not presently spawn in tributary streams to Ozette Lake. However, they may have spawned there historically. Eggs and alevins remain in gravel redds until the fish emerge as fry in spring. Fry then migrate immediately to the limnetic zone in Ozette Lake, where the fish rear. After one year of rearing, in late spring, Ozette Lake sockeye salmon emigrate seaward as one + smolts. The majority of Ozette Lake sockeye salmon return to spawn as four year old adult fish, having spent one winter in fresh water and two winters at sea (NMFS 2005b). As prespawning mortality is unknown, it is unclear what escapement levels to the spawning aggregations may be.

In Ozette Lake, naturally high water temperatures and low summer flows in the Ozette River may affect migration by altering timing of the runs (La Riviere 1991). Declines in abundance have been attributed to a combination of introduced species, predation, loss of tributary populations, decline in quality of beach spawning habitat, temporarily unfavorable ocean conditions, habitat degradation, and excessive historical harvests (Jacobs, Larson et al. 1996)

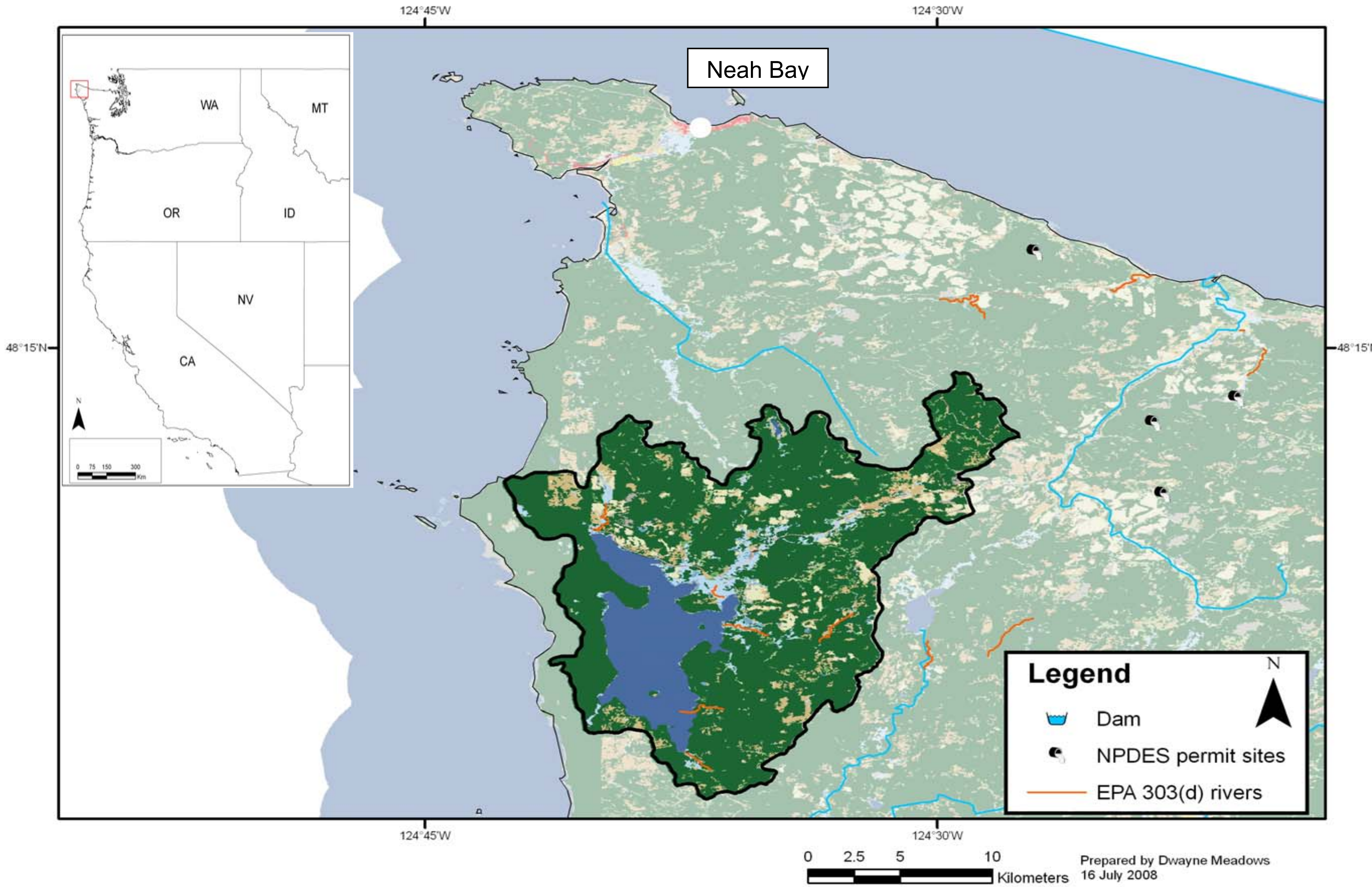


Figure 22. Ozette Lake Sockeye salmon distribution. Land Cover Class Legend in Figure 7.

Status and Trends

The Ozette Lake sockeye salmon ESU was originally listed as a threatened species in 1999 (64 FR 14528). This classification was retained following a species status review on June 28, 2005 (70 FR 37160).

The historical abundance of Ozette Lake sockeye salmon is poorly documented, but may have been as high as 50,000 individuals (Blum 1988). Nevertheless, the overall abundance of naturally–produced Ozette Lake sockeye salmon is believed to have declined substantially from historical levels. In the first study of lake escapement of Ozette Lake sockeye salmon (Kemmerich 1945), the run size entering the lake was estimated at a level of several thousand fish. These counts appear to be roughly double the current mean lake abundance, considering that they were likely conducted upstream from fisheries in or near to the Ozette River. Makah Fisheries Management (2000) concluded that there appears to be a substantial decline in the Tribal catch of Ozette Lake sockeye salmon beginning in the 1950s and a similar decline in the run size since the 1920s weir counts reported by Kemmerich (1945).

An updated NMFS analysis of total annual Ozette Lake sockeye salmon abundance (based on adult run size data presented in Jacobs et al. (1996)) indicates a trend in abundance averaging minus 2% per year over the period 1977 through 1998. The current tributary-based hatchery program was planned and initiated in response to the declining population trend identified for the Ozette Lake sockeye salmon population. The updated analysis also indicated that the most recent ten year (1989-98) trend for the population is plus 2% per year, improving from the minus 9.9% annual trend reported in Gustafson et al. (1999).

Data from the early 1900s indicate the spawning population was as large as 10,000 to 20,000 fish in large run years. Recent information on abundance of Ozette Lake sockeye salmon ESU comes from visual counts at a weir across the lake outlet. Therefore, the counts represent total run size. The estimates of total run size were revised upward after the 1997 status review due to resampling of data using new video counting technology. The Makah Fisheries biologists estimate that previous counts of adult sockeye salmon returning to the lake were underestimates, and they have attempted to correct run-size estimates based on their assessments of human error and variations in interannual run timing (Makah Fisheries Management 2000) *in* (Good, Waples et al. 2005).

The most recent (1996-2003) run-size estimates range from a low of 1,609 in 1997 to a high of 5,075 in 2003, averaging approximately 3,600 sockeye per year (Hard, Jones et al. 1992; Haggerty, Ritchie et al. 2007). For return years 2000 to 2003, the four-year average abundance estimate was slightly over 4,600 sockeye (Haggerty, Ritchie et al. 2007). Because run-size estimates before 1998 are likely to be even more unreliable than recent counts, and new counting technology has resulted in an increase in estimated run sizes, no statistical estimation of trends is reported. The current trends in abundance are unknown for the beach spawning aggregations. Although overall abundance appears to have declined from historical levels, whether this resulted in fewer spawning aggregations, lower abundances at each aggregation, or both, is unknown (Good, Waples et al. 2005). It is estimated that between 35,500 and 121,000 spawners could be normally carried after full recovery (Hard, Jones et al. 1992).

There has been no harvest of Ozette Lake sockeye salmon for the past four brood cycle years (since 1982). Prior to that time, ceremonial and subsistence harvests by the Makah Tribe were low, ranging from 0 to 84 fish per year. Harvest has not been an important mortality factor for the population in over 35 years. In addition, due to the early river entry timing of returning Ozette Lake sockeye salmon (beginning in late April, with the peak returns prior to late-May to mid-June), the fish are not intercepted in Canadian and U.S. marine area fisheries directed at Fraser River sockeye salmon. There are currently no known marine area harvest impacts on Ozette Lake sockeye salmon.

According to Good et al. (2005) it appears that overall abundance is low for this population, which represents an entire ESU, and may be substantially below historical levels. The number of returning adults in the last few years has increased. However, a substantial (but uncertain) fraction of these appear to be of hatchery origin. This condition leads to uncertainty regarding growth rate and productivity of the natural component of the ESU. Genetic integrity may have been compromised due to the artificial supplementation that has occurred in this population. Approximately one million sockeye have been released into the Ozette watershed from the late 1930s to present (Kemmerich 1945; Boomer 1995; Good, Waples et al. 2005).

Critical Habitat

On September 2, 2005, NMFS designated critical habitat for the Ozette Lake sockeye salmon ESU (70 FR 52630), and encompasses areas within the Hoh/Quillayute subbasin. Refer to the final rule for additional information on the watersheds within this subbasin, including a map of the area. Limiting factors for this species include siltation of beach-spawning habitat and logging.

Snake River Sockeye Salmon

Distribution

The SR sockeye salmon ESU includes all anadromous and residual sockeye from the SR basin Idaho, as well as artificially propagated sockeye salmon from the Redfish Lake Captive Broodstock Program (Figure 23).

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, Craddock 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.

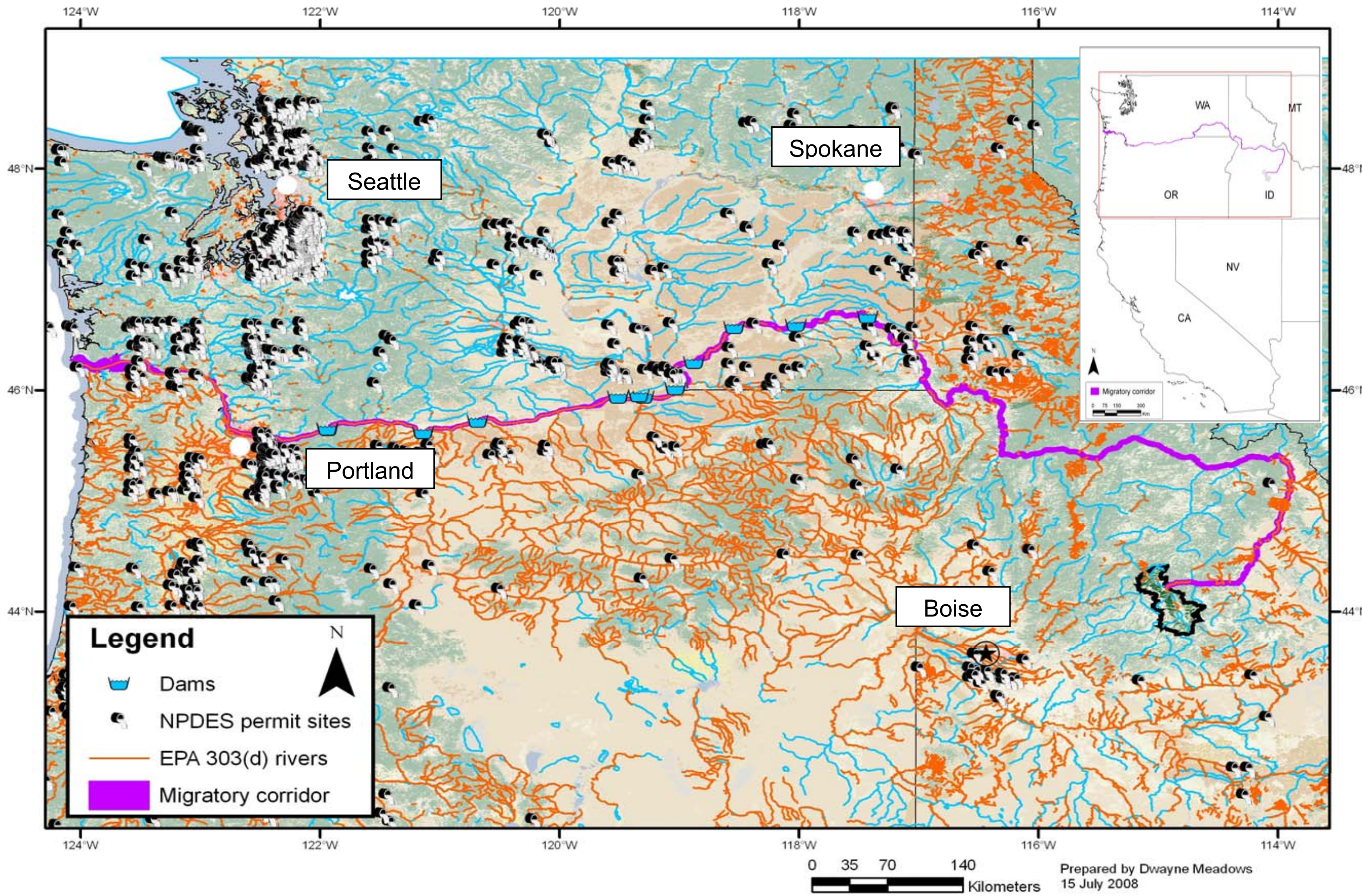


Figure 23. SR Sockeye Salmon distribution. Land Cover Class Legend in Figure 7.

Status and Trends

SR sockeye salmon were originally listed as endangered in 1991. Their classification was retained following a status review on June 28, 2005 (70 FR 37160). The only extant sockeye salmon population in the SR basin at the time of listing was that in Redfish Lake, in the Stanley Basin (upper Salmon River drainage) of Idaho. Other lakes in the SR basin historically supported sockeye salmon populations, including Wallowa Lake (Grande Ronde River drainage, Oregon), Payette Lake (Payette River drainage, Idaho) and Warm Lake (South Fork Salmon River drainage, Idaho) (Gustafson, Wainwright et al. 1997). These populations are now considered extinct. Although kokanee, a resident form of *O. nerka*, occur in numerous lakes in the SR basin, resident *O. nerka* were not considered part of the species at the time of listing in 1991. Subsequent to the 1991 listing, a residual form of sockeye residing in Redfish Lake was identified. The residuals are non-anadromous. They complete their entire life cycle in freshwater, but spawn at the same time and in the same location as anadromous sockeye salmon. In 1993, NMFS determined that residual sockeye salmon in Redfish Lake were part of the SR sockeye salmon. Also, artificially propagated sockeye salmon from the Redfish Lake Captive Propagation program are considered part of this species (June 28, 2005, 70 FR 37160).

NMFS has determined that this artificially propagated stock is genetically no more than moderately divergent from the natural population (Good, Waples et al. 2005). Five lakes in the Stanley Basin historically contained sockeye salmon: Alturas, Pettit, Redfish, Stanley and Yellowbelly (Bjornn, Craddock et al. 1968). It is generally believed that adults were prevented from returning to the Sawtooth Valley from 1910 to 1934 by Sunbeam Dam. Sunbeam Dam was constructed on the Salmon River approximately 20 miles downstream of Redfish Lake. Whether or not Sunbeam Dam was a complete barrier to adult migration remains unknown. It has been hypothesized that some passage occurred while the dam was in place, allowing the Stanley Basin population or populations to persist (Bjornn, Craddock et al. 1968; Matthews and Waples 1991).

Adult returns to Redfish Lake during the period 1954 through 1966 ranged from 11 to 4,361 fish (Bjornn, Craddock et al. 1968). Sockeye salmon in Alturas Lake were extirpated in the early 1900s as a result of irrigation diversions, although residual sockeye may still exist in the lake (Chapman and Witty 1993). From 1955 to 1965, the Idaho Department of Fish and Game eradicated sockeye salmon from Pettit, Stanley, and Yellowbelly lakes, and built permanent structures on each of the lake outlets that prevented re-entry of anadromous sockeye salmon (Chapman and Witty 1993). In 1985, 1986, and 1987, 11, 29, and 16 sockeye, respectively, were counted at the Redfish Lake weir (Good, Waples 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 program adults that had migrated to the ocean returned to the Stanley Basin.

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 entirely supported by adults produced through the captive propagation program at the present time. Current smolt-to-adult survival of sockeye originating from the Stanley Basin lakes is rarely greater than 0.3% (Hebdon, Kline et al. 2004). Based on current abundance and productivity information, the SR sockeye salmon ESU does not meet the ESU-level viability criteria (non-negligible risk of extinction over a 100-year time period).

Critical Habitat

Critical habitat for these salmon was designated on December 28, 1993 (58 FR 68543). Designated habitats encompasses 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 SR salmon (except reaches above impassable natural falls, and Dworshak and Hells Canyon Dams). Adjacent riparian zones are defined as those areas within a horizontal distance of 300 ft from the normal line of high water of a stream channel or from the shoreline of a standing body of water. Designated 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 SR, and all SR 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. Limiting factors identified for SR sockeye include: (1) Reduced tributary stream flow, (2) impaired tributary passage and blocks to migration, (3) degraded water quality; and (4) mainstem Columbia River hydropower system mortality.

Steelhead

Description of the Species

Steelhead are native to Pacific Coast streams extending from Alaska south to northwestern Mexico (Moyle 1976; Gustafson, Wainwright et al. 1997; Good, Waples et al. 2005). We discuss the distribution, life history diversity, status, and critical habitat of the 11 endangered and threatened steelhead species separately.

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 stream-maturing type or summer steelhead enters fresh water in a sexually immature condition. It requires several months in freshwater to mature and spawn. The ocean-maturing type or winter steelhead enters freshwater with well-developed gonads and spawns shortly after river entry. Variations in migration timing exist between populations. Some river basins have both summer and winter steelhead, while others only have one run-type.

Summer steelhead enter freshwater between May and October in the Pacific Northwest (Nickelsen, Nicholas et al. 1992; Busby, Wainwright et al. 1996). They require cool, deep holding pools during summer and fall, prior to spawning (Nickelsen, Nicholas et al. 1992). They migrate inland toward spawning areas, overwinter in the larger rivers, resume migration in early spring to natal streams, and then spawn (Meehan and Bjornn 1991; Nickelsen, Nicholas et al. 1992) in January and February (Barnhart 1986). Winter steelhead enter freshwater between November and April in the Pacific Northwest (Nickelsen, Nicholas et al. 1992; Busby, Wainwright et al. 1996), migrate to spawning areas, and then spawn, generally in April and May (Barnhart 1986). Some adults, however, do not enter some coastal streams until spring, just before spawning (Meehan and Bjornn 1991).

There is a high degree of overlap in spawn timing between populations regardless of run type (Busby, Wainwright et al. 1996). Difficult field conditions at that time of year and the remoteness of spawning grounds contribute to the relative lack of specific information on steelhead spawning. Unlike Pacific salmon, steelhead are iteroparous, or capable of spawning more than once before death (Busby, Wainwright et al. 1996), although steelhead rarely spawn more than twice before dying; most that do so are females (Nickelsen, Nicholas et al. 1992). Iteroparity is more common among southern steelhead populations than northern populations (Busby, Wainwright et al. 1996).

After two to three weeks, in late spring, and following yolk sac absorption, alevins emerge from the gravel and begin actively feeding. After emerging from the gravel, fry usually inhabit shallow water along banks of perennial streams. Fry occupy stream margins (Nickelsen, Nicholas et al. 1992). Summer rearing takes place primarily in the faster parts of pools, although young-of-the-year are abundant in glides and riffles. Winter rearing occurs more uniformly at lower densities across a wide range of fast and slow habitat types. Some older juveniles move downstream to rear in larger tributaries and mainstem rivers (Nickelsen, Nicholas et al. 1992).

Juvenile steelhead migrate little during their first summer and occupy a range of habitats featuring moderate to high water velocity and variable depths (Bisson, Sullivan et al. 1988). Juvenile steelhead feed on a wide variety of aquatic and terrestrial insects (Chapman and Bjornn 1969), and older juveniles sometimes prey on emerging fry. Steelhead hold territories close to the substratum where flows are lower and sometimes counter to the main stream; from these, they can make forays up into surface currents to take drifting food (Kalleberg 1958). Juveniles rear in freshwater from one to four years, then smolt and migrate to the ocean in March and April (Barnhart 1986). Winter steelhead juveniles generally smolt after two years in freshwater (Busby, Wainwright et al. 1996). Juvenile steelhead tend to migrate directly offshore during their first summer from whatever point they enter the ocean rather than migrating along the coastal belt as salmon do. During the fall and winter, juveniles move southward and eastward (Hartt and Dell 1986) *op. cit.* (Nickelsen, Nicholas 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. Juvenile salmonids rely on a variety of non-main channel habitats that are critical to rearing. All listed salmonids use shallow, low flow habitats at some point in their life cycle. Examples of off-channel habitat include alcoves, channel edge sloughs, overflow channels, backwaters, terrace tributaries, off-channel dredge ponds, and braids (Anderson 1999; Swift III 1979).

Status and Trends

Steelhead, like the other salmon discussed previously, survive only in aquatic ecosystems and, therefore, depend on the quantity and quality of those aquatic systems. Steelhead, like the other salmon NMFS has listed, have declined under the combined effects of overharvests in fisheries; competition from fish raised in hatcheries and native and non-native exotic species; dams that block their migrations and alter river hydrology; gravel mining that impedes their migration and alters the hydrogeomorphology of the rivers and streams that support juveniles; water diversions that deplete water levels in rivers and streams; destruction or degradation of riparian habitat that increase water temperatures in rivers and streams sufficient to reduce the survival of juvenile chum salmon; and land use practices (logging, agriculture, urbanization) that destroy wetland and riparian ecosystems. These same activities and features introduce sediment, nutrients, biocides, metals, and other pollutants into surface and ground water and degrade water quality in the freshwater, estuarine, and coastal ecosystems throughout the Pacific Northwest.

Central California Coast Steelhead

Distribution

The Central California Coast (CCC) steelhead DPS includes all naturally spawned anadromous *O. mykiss* (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 24). Tributary streams to Suisun Marsh including Suisun Creek, Green Valley Creek, and an unnamed tributary to Cordelia Slough (commonly referred to as Red Top Creek), excluding the Sacramento-San Joaquin River Basin, as well as two artificial propagation programs: the Don Clausen Fish Hatchery, and Kingfisher Flat Hatchery/ Scott Creek (Monterey Bay Salmon and Trout Project) steelhead hatchery programs. Table 18 identifies populations within the CCC Steelhead salmon ESU, their abundances, and hatchery input.

Table 18. CCC Steelhead salmon populations, abundances, and hatchery contributions (Good, Waples et al. 2005).

Basin	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Russian River	65,000 (1970)	1,750-7,000 (1994)	Unknown
Lagunitas	Unknown	400-500 (1990s)	Unknown
San Gregorio	1,000 (1973)	Unknown	Unknown
Waddell Creek	481	150 (1994)	Unknown
Scott Creek	Unknown	<100 (1991)	Unknown
San Vicente Creek	150 (1982)	50 (1994)	Unknown
San Lorenzo River	20,000	<150 (1994)	Unknown
Soquel Creek	500-800 (1982)	<100 (1991)	Unknown
Aptos Creek	200 (1982)	50-75 (1994)	Unknown
Total	**94,000**	**2,400-8,125**	

Life History

Only winter steelhead are found in this DPS and those to the south. Migration and spawn timing are similar to adjacent steelhead populations. There is little other life history information for steelhead in this DPS.

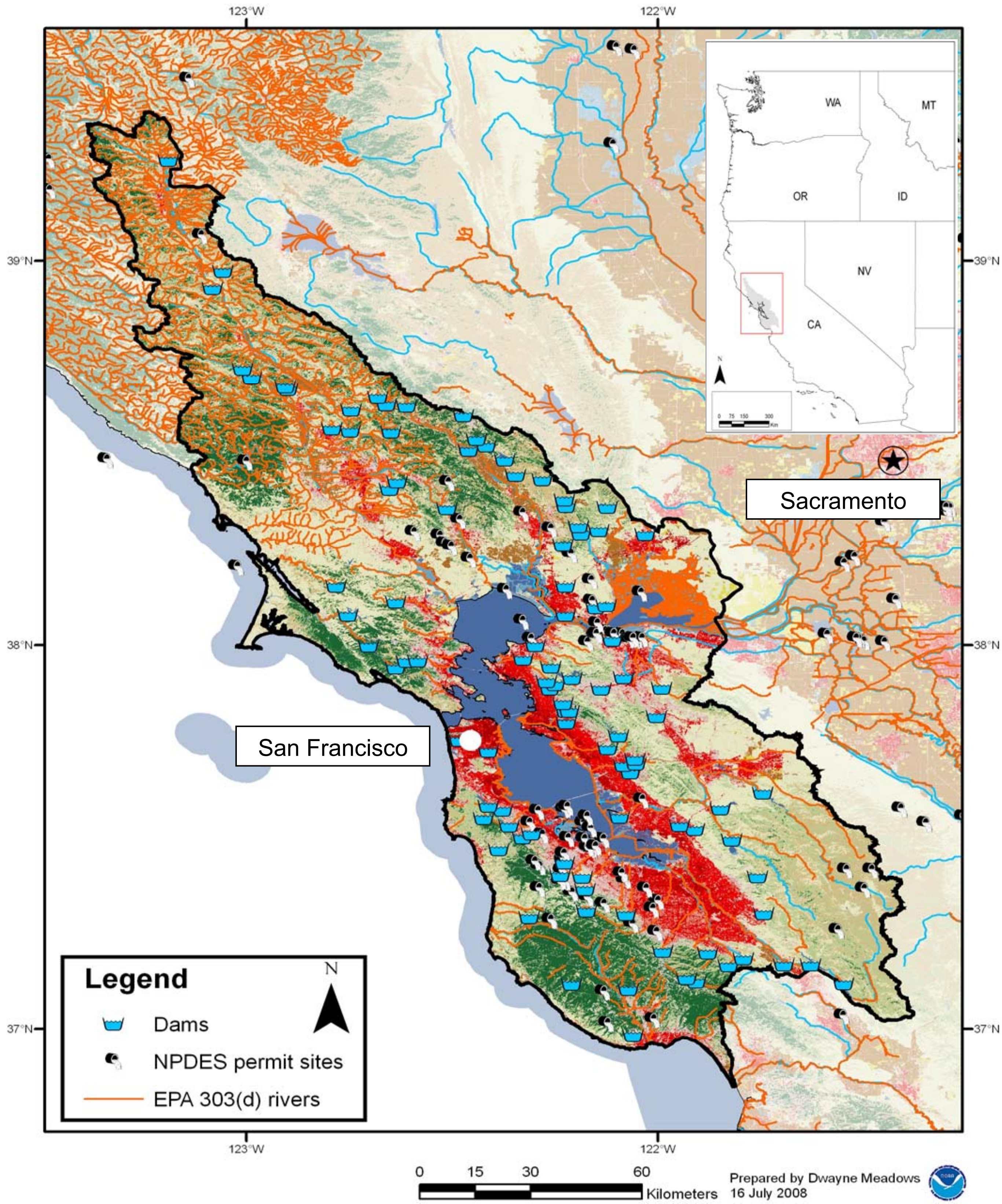


Figure 24. CCC steelhead. Land Cover Class Legend in Figure 7.

Status and Trends

The CCC steelhead DPS was listed as a threatened species on August 18, 1997(62 FR 43937). Its threatened status was reaffirmed on January 5, 2006 (71 FR 834). Busby et al. (1996) reported one estimate of historical (pre-1960s) abundance. Shapovalov and Taft (1954) described an average of about 500 adults in Waddell Creek (Santa Cruz County) for the 1930s and early 1940s. Johnson (1964) estimated a run size of 20,000 steelhead in the San Lorenzo River before 1965. The CDFG (1965) estimated an average run size of 94,000 steelhead for the entire DPS, for the period 1959–1963. The analysis by CDFG (1965) was compromised for many basins, as the data did not exist for the full 5-year analytical period. The authors of CDFG (1965) state that "estimates given here which are based on little or no data should be used only in outlining the major and critical factors of the resource."

Recent data for the Russian and San Lorenzo rivers (Reavis 1991; CDFG 1994; Shumann 1994) suggested that these basins had populations smaller than 15% of their size 30 years earlier. These two basins were thought to have originally contained the two largest steelhead populations in the CCC steelhead ESU.

A status review update in 1997 (Gustafson, Wainwright et al. 1997) concluded that slight increases in abundance occurred in the three years following the status review. However, the analyses on which these conclusions were based had various problems. They include the inability to distinguish hatchery and wild fish, unjustified expansion factors, and variance in sampling efficiency on the San Lorenzo River. Presence-absence data indicated that most (82%) sampled streams (a subset of all historical steelhead streams) had extant populations of juvenile *O. mykiss* (Adams 2000; Good, Waples et al. 2005).

The majority (69%) of BRT votes were for "likely to become endangered," and another 25% were for "in danger of extinction". Abundance and productivity were of relatively high concern (as a contributing factor to risk of extinction), and spatial structure was also of concern. Predation by pinnipeds at river mouths and during the ocean phase was noted as a recent development posing significant risk. There were no time-series data for the CCC steelhead DPS. A variety of evidence suggested the ESU's largest run (the Russian River winter steelhead run) has been, and continues to be, reduced in size. Concern was also expressed about populations in the southern part of the DPS's range—notably those in Santa Cruz County and the South Bay area (Good, Waples et al. 2005).

Critical Habitat

Critical habitat was designated for the CCC steelhead DPS on September 2, 2005 (70 FR 52488), and includes areas within the following hydrologic units: Russian River, Bodega, Marin Coastal, San Mateo, Bay Bridges, Santa Clara, San Pablo, Big Basin. Refer to the final rule for a more detailed description of critical habitat, including a map for each hydrologic unit.

California Central Valley Steelhead

Distribution

California CV steelhead occupy the Sacramento and San Joaquin Rivers and its tributaries (Figure 25).

Life History

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

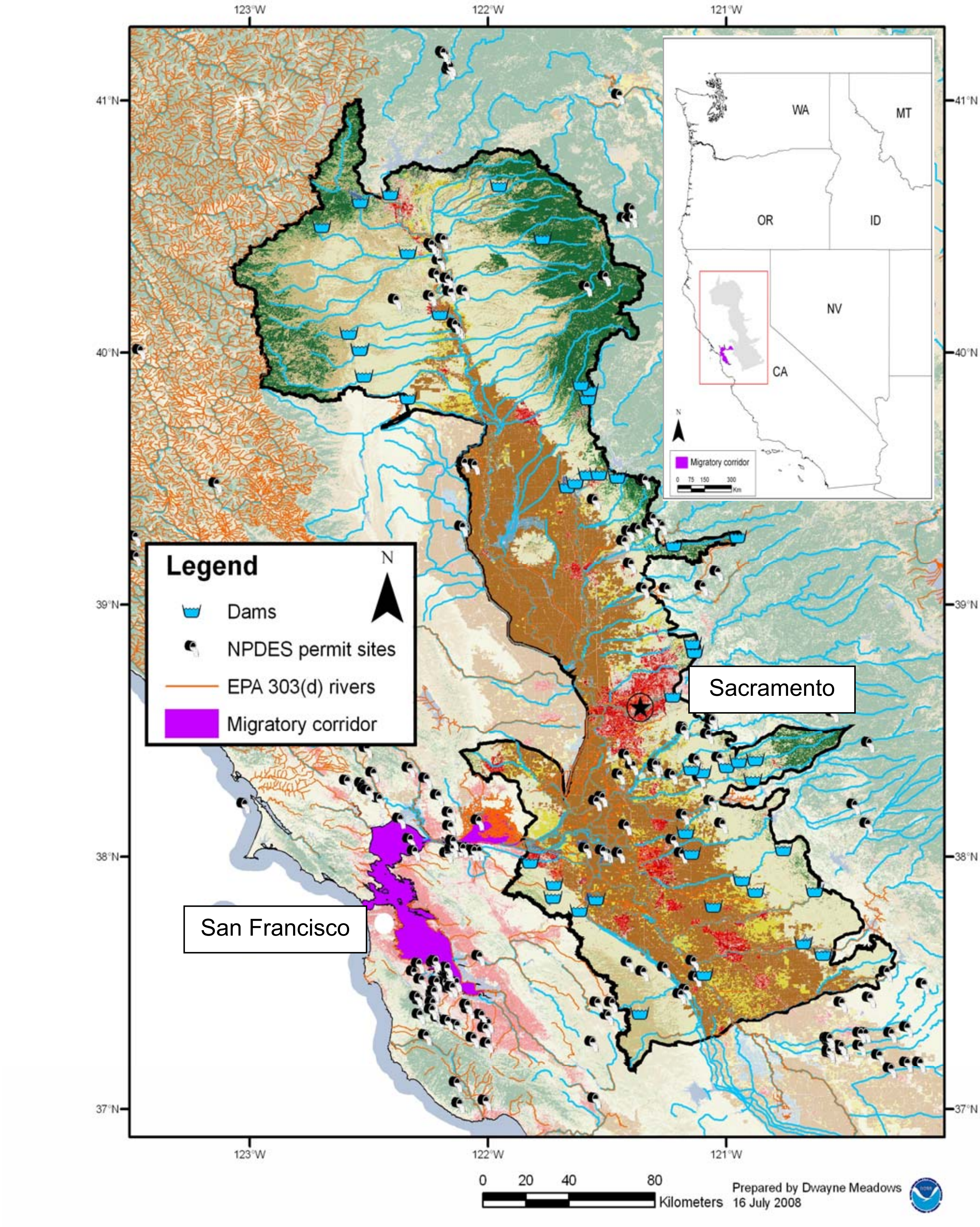


Figure 25. California CV steelhead distribution. Land Cover Class Legend in Figure 7.

Status and Trends

California CV steelhead were listed as threatened on March 19, 1998. Their classification was retained following a status review on January 5, 2006 (71 FR 834). This DPS consists of steelhead populations in the Sacramento and San Joaquin River (inclusive of and downstream of the Merced River) basins in California's CV. Steelhead historically were well distributed throughout the Sacramento and San Joaquin Rivers (Busby, Wainwright et al. 1996). Steelhead were found from the upper Sacramento and Pit River systems (now inaccessible due to Shasta and Keswick Dams), south to the Kings and possibly the Kern River systems (now inaccessible due to extensive alteration from water diversion projects), and in both east- and west-side Sacramento River tributaries (Yoshiyama, Gerstung et al. 1996). The present distribution has been greatly reduced (McEwan and Jackson 1996). The California Advisory Committee on Salmon and Steelhead (1988) reported a reduction of steelhead habitat from 6,000 miles historically to 300 miles today. Historically, steelhead probably ascended Clear Creek past the French Gulch area, but access to the upper basin was blocked by Whiskeytown Dam in 1964 (Yoshiyama, Gerstung et al. 1996). Steelhead also occurred in the upper drainages of the Feather, American, Yuba, and Stanislaus Rivers which are now inaccessible (McEwan and Jackson 1996; Yoshiyama, Gerstung et al. 1996).

Historic CV steelhead run size is difficult to estimate given limited data, but may have approached one to two million adults annually (McEwan 2001). By the early 1960s, the steelhead run size had declined to about 40,000 adults (McEwan 2001). Over the past 30 years, the naturally spawned steelhead populations in the upper Sacramento River have declined substantially. Hallock et al. (1961) estimated an average of 20,540 adult steelhead in the Sacramento River, upstream of the Feather River, through the 1960s. Steelhead counts at Red Bluff Diversion 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, with an estimated total annual run size for the entire Sacramento-San Joaquin system, based on Red Bluff Diversion Dam counts, to be no more than 10,000 adults (McEwan and Jackson 1996; McEwan 2001). Steelhead escapement surveys at Red Bluff Diversion Dam ended in 1993 due to changes in dam operations.

The only consistent data available on steelhead numbers in the San Joaquin River basin come from CDFG mid-water trawling samples collected on the lower San Joaquin River at Mossdale. These data indicate a decline in steelhead numbers in the early 1990s, which have remained low through 2002 (CDFG 2003). In 2004, a total of 12 steelhead smolts were collected at Mossdale (CDFG unpublished data).

Existing wild steelhead stocks in the CV 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 (McEwan and Jackson 1996).

Snorkel surveys from 1999 to 2002 indicate that steelhead are present in Clear Creek (J. Newton, FWS, pers. comm. 2002, as reported *in* Good et al. (2005). Because of the large resident *O. mykiss* population in Clear Creek, steelhead spawner abundance has not been estimated.

Until recently, steelhead were thought to be 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 (McEwan 2001). On the Stanislaus River, steelhead smolts have been captured in rotary screw traps at Caswell State Park and Oakdale each year since 1995 (Demko and Cramer 2000). It is possible that naturally spawning populations exist in many other streams. However, these populations are undetected due to lack of monitoring programs (IEPSPWT 1999).

The majority (66%) of BRT votes was for “in danger of extinction,” and the remainder was for “likely to become endangered”. Abundance, productivity, and spatial structure were of highest concern. Diversity considerations were of significant concern. The BRT was concerned with what little new information was available and indicated that the monotonic decline in total abundance and in the proportion of wild fish in the California CV steelhead ESU was continuing.

Critical Habitat

Critical habitat was designated for this species on September 2, 2005. The critical habitat designation for this DPS identifies PCEs that include sites necessary to support one or more life stages of steelhead. Specific sites include: (1) Freshwater spawning, (2) freshwater rearing, (3) freshwater migration, (4) estuarine areas free of obstruction, (5) nearshore marine areas free of obstructions, and (6) offshore marine areas with good water quality. The physical or biological features that characterize these sites include water quality and quantity, natural cover, and adequate forage.

Lower Columbia River Steelhead

Distribution

LCR steelhead DPS includes 23 historical anadromous populations in four MPGs. This DPS includes naturally-produced steelhead returning to Columbia River tributaries on the Washington side between the Cowlitz and Wind rivers in Washington and on the Oregon side between the Willamette and Hood rivers, inclusive (Figure 26). In the Willamette River, the upstream boundary of this species is at Willamette Falls. This species includes both winter and summer steelhead. Two hatchery populations are included in this species, the Cowlitz Trout Hatchery winter-run stock and the Clackamas River stock. However, neither hatchery population was listed as threatened.

Table 19 identifies populations within the LCR Steelhead salmon DPS, their abundances, and hatchery input.

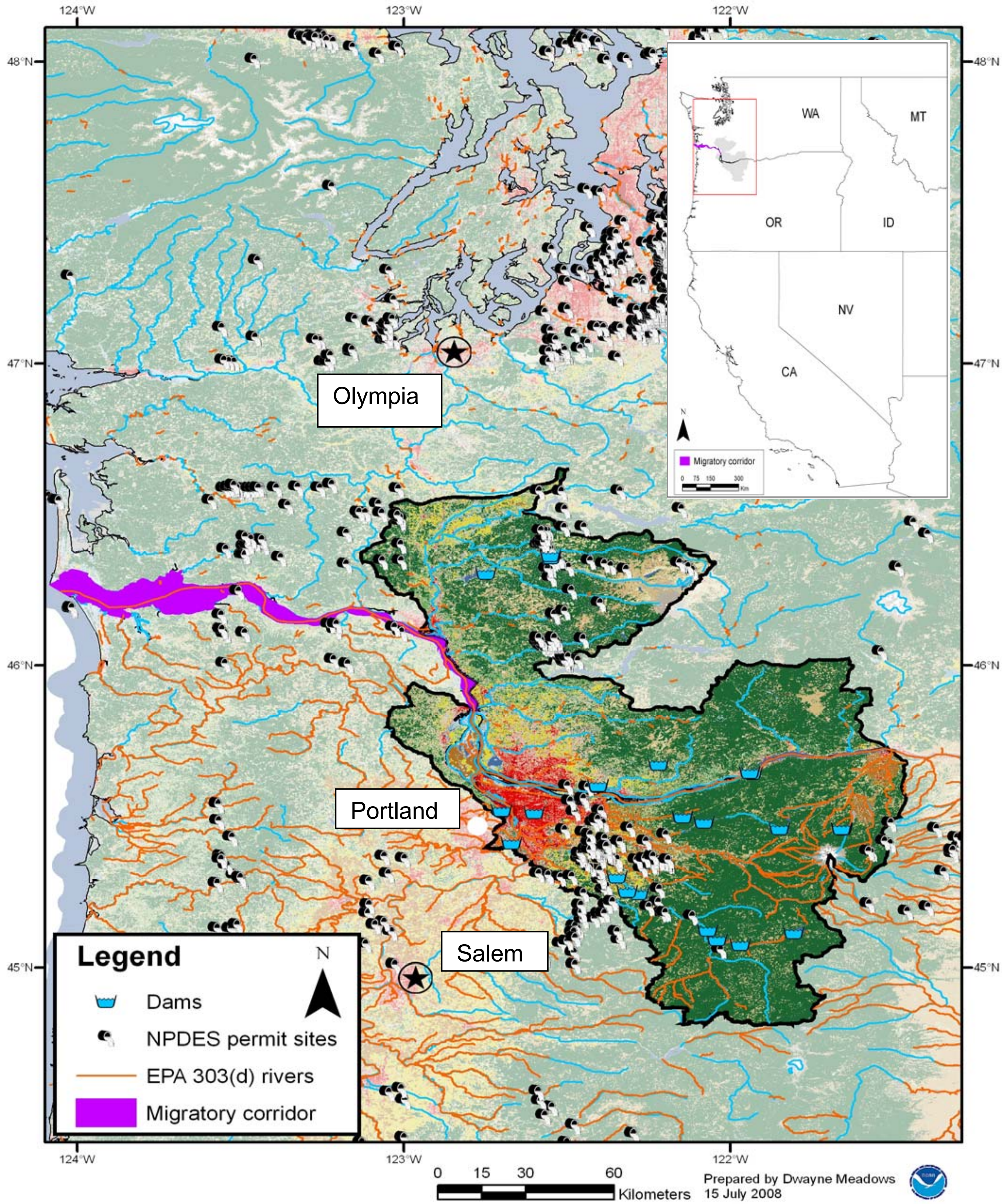


Figure 26. Lower Columbia River Steelhead distribution. Land Cover Class Legend in Figure 7.

Table 19. LCR Steelhead salmon populations, abundances, and hatchery contributions (Good, Waples et al. 2005).

Population	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Cispus River	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	~50%
South Fork Toutle River	2,627	504	~2%
North Fork Toutle River	3,770	196	0%
Kalama River-winter run	554	726	0%
Kalama River-summer run	3,165	474	~32%
North Fork Lewis River-winter run	713	Unknown	Unknown
North Fork Lewis River-summer run	Unknown	Unknown	Unknown
East Fork Lewis River-winter run	3,131	Unknown	Unknown
East Fork Lewis River-summer run	422	434	~25%
Salmon Creek	Unknown	Unknown	Unknown
Washougal River-winter run	2,497	323	0%
Washougal River-summer run	1,419	264	~8%
Clackamas River	Unknown	560	41%
Sandy River	Unknown	977	42%
Lower Columbia gorge tributaries	793	Unknown	Unknown
Upper Columbia gorge tributaries	243	Unknown	Unknown
Hood River-winter run	Unknown	756	~52%
Hood River-summer run	Unknown	931	~83%
Wind River	2,288	472	~5%
Total	**25,537 (min)**	**9,870 (min)**	

Life History

Summer steelhead return to freshwater from May to November, entering the Columbia River in a sexually immature condition and requiring several months in freshwater before spawning. Winter steelhead enter freshwater from November to April. They are close to sexual maturation and spawn shortly after arrival in their natal stream. Where both races spawn in the same stream, summer steelhead tend to spawn at higher elevations than the winter forms. Juveniles rear in freshwater (stream-type life history).

Status and Trends

LCR steelhead were listed as threatened on March 19, 1998 (63 FR 13347), and reaffirmed as threatened on January 5, 2006 (71 FR 834). The 1998 status review noted that this ESU is characterized by populations at low abundance relative to historical levels, significant population declines since the mid-1980s, and widespread occurrence of hatchery fish in naturally-spawning steelhead populations. During this review NMFS was unable to identify any natural populations that would be considered at low risk.

All 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 for the Kalama and Sandy winter-run populations. A number of the populations have a substantial fraction of hatchery-origin spawners in spawning areas. These populations are hypothesized to be sustained largely by hatchery production. Exceptions are the Kalama, the Toutle, and East Fork Lewis winter-run populations. These populations have relatively low recent mean abundance estimates with the largest being the Kalama (geometric mean of 728 spawners).

According to Good et al. (2005), most populations are at relatively low abundance. Those with adequate data for modeling are estimated to have a relatively high extinction probability. Some populations, particularly summer run, have shown higher return in the last two to three years. Many of the long-and short-term trends in abundance of individual populations are negative, some severely so. The trend in natural spawners is <1; indicating the population is not replacing itself and in decline. Spatial structure has been substantially reduced by the loss of access to the upper portions of some basins due to tributary hydro development. Finally, a number of the populations have a substantial fraction of hatchery-origin spawners. Exceptions are the Kalama, North and South Fork Toutle, and East Fork Lewis winter-run populations, which have few hatchery fish spawning in natural spawning areas.

Over 73% of the BRT votes for this species fell in the “likely to become endangered” category. There were small minorities falling in the “danger of extinction” and “not likely to become endangered” categories. The BRT found moderate risks in all VSP categories, with mean risk matrix scores ranging from moderately low for spatial structure to moderately high for abundance and productivity (population growth rate).

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52488). The critical habitat designation for this DPS identifies PCEs that include sites necessary to support one or more steelhead life stages. Specific sites include: (1) Freshwater spawning, (2) freshwater rearing, (3) freshwater migration, (4) estuarine areas free of obstruction, (5) nearshore marine areas free of obstructions, and (6) offshore marine areas with good water quality. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity.

Of 47 subbasins reviewed in NMFS' assessment of critical habitat for the LCR steelhead, 34 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. Limiting factors identified for LCR steelhead include: (1) Degraded floodplain and steam channel structure and function, (2) reduced access to spawning/rearing habitat, (3) altered streamflow in tributaries, (4) excessive sediment and elevated water temperatures in tributaries, and (5) hatchery impacts (NMFS 2005b). The above conditions also introduce sediment, nutrients, biocides, metals, and other pollutants into surface and ground water and degrade water quality in the freshwater, estuarine, and coastal ecosystems throughout the Pacific Northwest.

Middle Columbia River Steelhead

Distribution

Middle Columbia River (MCR) steelhead DPS includes anadromous populations in Oregon and Washington subbasins upstream of the Hood and Wind River systems to and including the Yakima River (Figure 27). There are four MPGs with 17 populations in this DPS. Steelhead from the SR Basin (described elsewhere) are excluded. This species includes the only populations of inland winter steelhead in the U.S., in the Klickitat River and Fifteenmile Creek (Busby, Wainwright et al. 1996).

Two hatchery populations are considered part of this species, the Deschutes River stock and the Umatilla River stock. Listing for neither of these stocks was considered warranted. MCR steelhead occupy the intermontane region which includes some of the driest areas of the Pacific Northwest, generally receiving less than 15.7 inches of rainfall annually. Vegetation is of the shrub-steppe province, reflecting the dry climate and harsh temperature extremes. Because of this habitat, occupied by the species, factors contributing to the decline include agricultural practices, especially grazing, and water diversions and withdrawals. In addition, hydropower development has impacted the species by preventing these steelhead from migrating to habitat above dams, and by killing some of them when they try to migrate through the Columbia River hydroelectric system. Table 20 identifies populations within the MCR Steelhead salmon DPS, their abundances, and hatchery input.

Table 20. MCR Steelhead salmon populations, abundances, and hatchery contributions (Good, Waples et al. 2005).

Population	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Klickitat River	Unknown	97-261 reds	Unknown
Yakima River	Unknown	1,058-4,061	97%
Fifteenmile Creek	Unknown	2.87 rpm	100%
Deschutes River	Unknown	10,026-21,457	38%
John Day upper main stream	Unknown	926-4,168	96%
John Day lower main stream	Unknown	1.4 rpm	0%
John Day upper north fork	Unknown	2.57 rpm	0%
John Day lower north fork	Unknown	.52 rpm	0%
John Day middle fork	Unknown	3.7 rpm	0%
John Day south fork	Unknown	2.52 rpm	0%
Umatilla River	Unknown	1,480-5,157	60%
Touchet River	Unknown	273-527	84%
Total	Unknown		

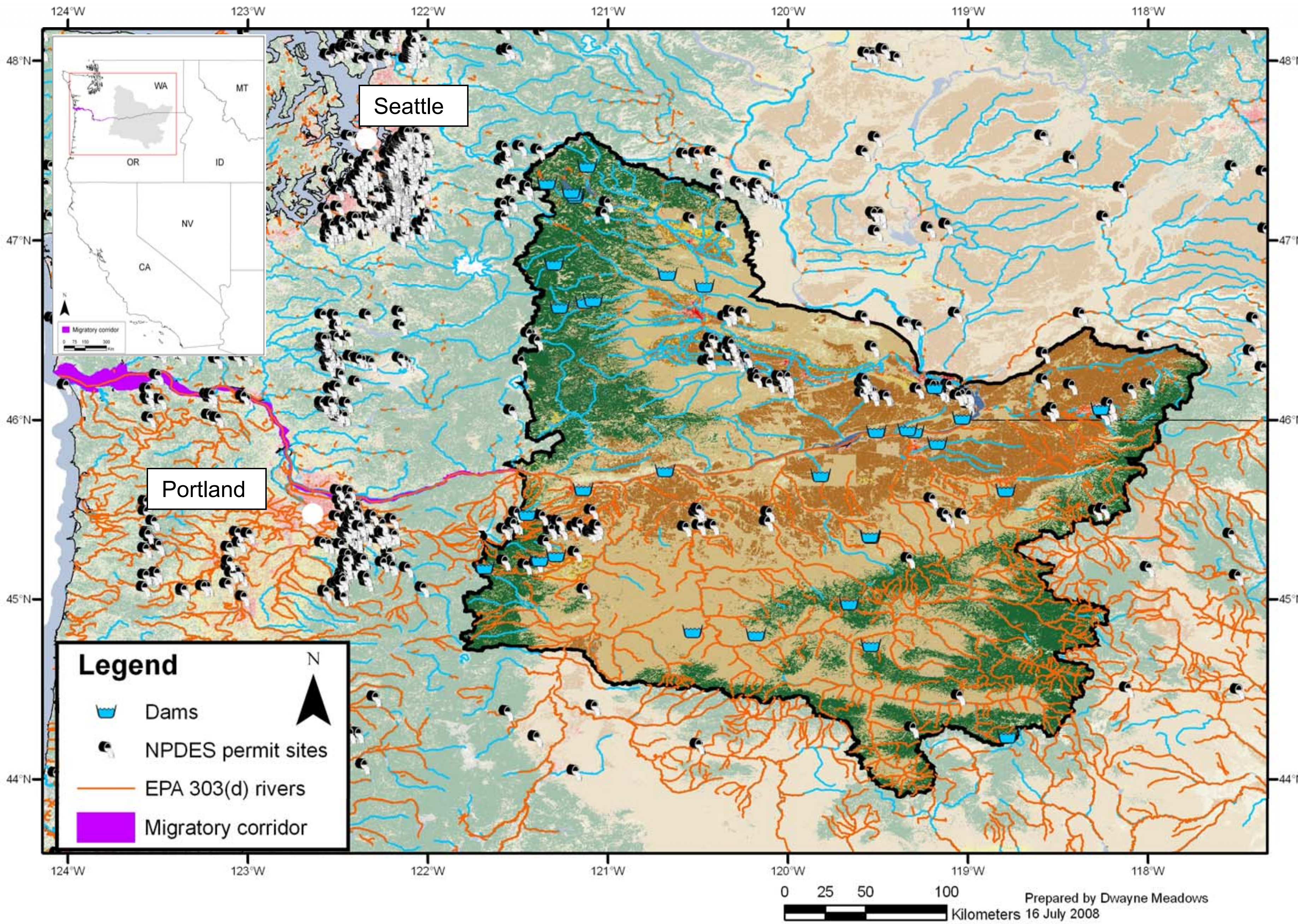


Figure 27. MCR Steelhead distribution. Land Cover Class Legend in Figure 7.

Life History

Most MCR steelhead smolt at two years and spend one to two years in saltwater prior to re-entering freshwater. Here they may remain up to a year prior to spawning (Howell, Jones et al. 1985). Within this ESU, the Klickitat River is unusual as it produces both summer and winter steelhead. The summer steelhead are dominated by age two ocean steelhead. Most other rivers in this region produce about equal numbers of both age one and two ocean steelhead.

Status and Trends

MCR steelhead were listed as threatened in 1999 (64 FR 14517), and their status was reaffirmed on January 5, 2006 (71 FR 834). The ICBTRT (2003) identified 15 populations in four MPGs (Cascades Eastern Slopes Tributaries, John Day River, the Walla Walla and Umatilla Rivers, and the Yakima River) and one unaffiliated independent population (Rock Creek) in this species. There are two extinct populations in the Cascades Eastern Slope MPG: the White Salmon River and Deschutes Crooked River above the Pelton/Round Butte Dam complex.

Seven hatchery steelhead programs are considered part of the MCR steelhead species. These programs propagate steelhead in three of 16 populations and improve kelt survival in one population. No artificial programs produce the winter-run life history in the Klickitat River and Fifteenmile Creek populations. All of the MCR steelhead hatchery programs are designed to produce fish for harvest. However, two hatchery programs are also implemented to augment the naturally spawning populations in the basins where the fish are released. The NMFS assessment of the effects of artificial propagation on MCR steelhead extinction risk concluded that these hatchery programs collectively do not substantially reduce the extinction risk. Artificial propagation increases total species abundance, principally in the Umatilla and Deschutes Rivers. The kelt reconditioning efforts in the Yakima River do not augment natural abundance and benefit the survival of the natural populations. The Touchet River Hatchery program has only recently been established, and its contribution to species viability is uncertain. The hatchery programs affect a small proportion of the species. Collectively, artificial propagation programs provide a slight beneficial effect to species abundance and have neutral or uncertain effects on species productivity, spatial structure, and diversity.

The precise pre-1960 abundance of this species is unknown. However, historic run estimates for the Yakima River imply that annual species abundance may have exceeded 300,000 returning adults (Busby, Wainwright et al. 1996). MCR steelhead run estimates between 1982 and 2004 were calculated by subtracting adult counts for Lower Granite and Priest Rapids Dams from those at Bonneville Dam. The five year average (geometric mean) return of natural MCR steelhead for 1997 to 2001 was up from previous years' basin estimates. Returns to the Yakima River, the Deschutes River, and sections of the John Day River system were substantially higher compared to 1992 to 1997 (Good, Waples et al. 2005). Yakima River returns are still substantially below interim target levels of 8,900 (the current five year average is 1,747 fish) and estimated historical return levels, with the majority of spawning occurring in one tributary, Satus Creek (Berg 2001). The recent five year geometric mean return of the natural-origin component of the Deschutes River run exceeded interim target levels (Good, Waples et al. 2005). Recent five year geometric mean annual returns to the John Day River basin are generally below the corresponding mean returns reported in previous status reviews. However, each major production area in the John Day system has shown upward trends since the 1999 return year (Good, Waples et al. 2005). The Touchet and Umatilla are below their interim abundance targets of 900 and 2,300, respectively. The five year average for these basins is 298 and 1,492 fish, respectively (Good, Waples et al. 2005).

As per the FCRPS (2008), during the most recent 10-year period (for which trends in abundance could be estimated), trends were positive for approximately half of the populations and negative for the remainder. On average, when only natural production is considered, most of the MCR steelhead populations have replaced themselves. The ICBTRT characterizes the diversity risk to all but one MCR steelhead population as "low" to "moderate". The Upper Yakima is rated as having "high" diversity risk because of introgression with resident *O. mykiss* and the loss of presmolt migration pathways.

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52488). The critical habitat designation for this DPS identifies PCEs that include sites necessary to support one or more life stages of steelhead. MCR steelhead have PCEs of: (1) freshwater spawning, (2) freshwater rearing, (3) freshwater migration, (4) estuarine areas free of obstruction, (5) nearshore marine areas free of obstructions, and (6) offshore marine areas with good water quality. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, and adequate passage conditions. Although pristine habitat conditions are still present in some wilderness, roadless, and undeveloped areas, habitat complexity has been greatly reduced in many areas of designated critical habitat for MCR steelhead. Limiting factors identified for MCR steelhead include: (1) Hydropower system mortality; (2) reduced stream flow; (3) impaired passage; (4) excessive sediment; (5) degraded water quality; and (6) altered channel morphology and floodplain.

Northern California Steelhead

Distribution

Northern California steelhead includes steelhead in CC river basins from Redwood Creek south to the Gualala River, inclusive (Figure 28). Table 21 identifies populations within the Northern California Steelhead salmon ESU, their abundances, and hatchery input.

Table 21. Northern California Steelhead salmon populations, abundances, and hatchery contributions (Good, Waples et al. 2005).

River	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Redwood Creek	10,000	Unknown	Unknown
Mad River	6,000	162-384	Unknown
Eel River	82,000	3,127-21,903	Unknown
Mattole River	12,000	Unknown	Unknown
Ten Mile River	9,000	Unknown	Unknown
Noyo River	8,000	Unknown	Unknown
Big River	12,000	Unknown	Unknown
Navarro River	16,000	Unknown	Unknown
Garcia River	4,000	Unknown	Unknown
Gualala River	16,000	Unknown	Unknown
Other Humboldt County streams	3,000	Unknown	Unknown
Other Mendocino County streams	20,000	Unknown	Unknown
Total	**198,000**	**Unknown**	

Life History

Steelhead within this DPS include winter and summer steelhead. Half-pounder juveniles occur in the Mad and Eel Rivers. Half-pounders are immature steelhead that returns to freshwater after only two to four months in the ocean, and generally overwinter in freshwater. These juveniles then outmigrate in the following spring.

Status and Trends

NC steelhead were listed as threatened on June 7, 2000 (65 FR 36074). They retained that classification following a status review on January 5, 2006 (71 FR 834). Long-term data sets are limited for this 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.

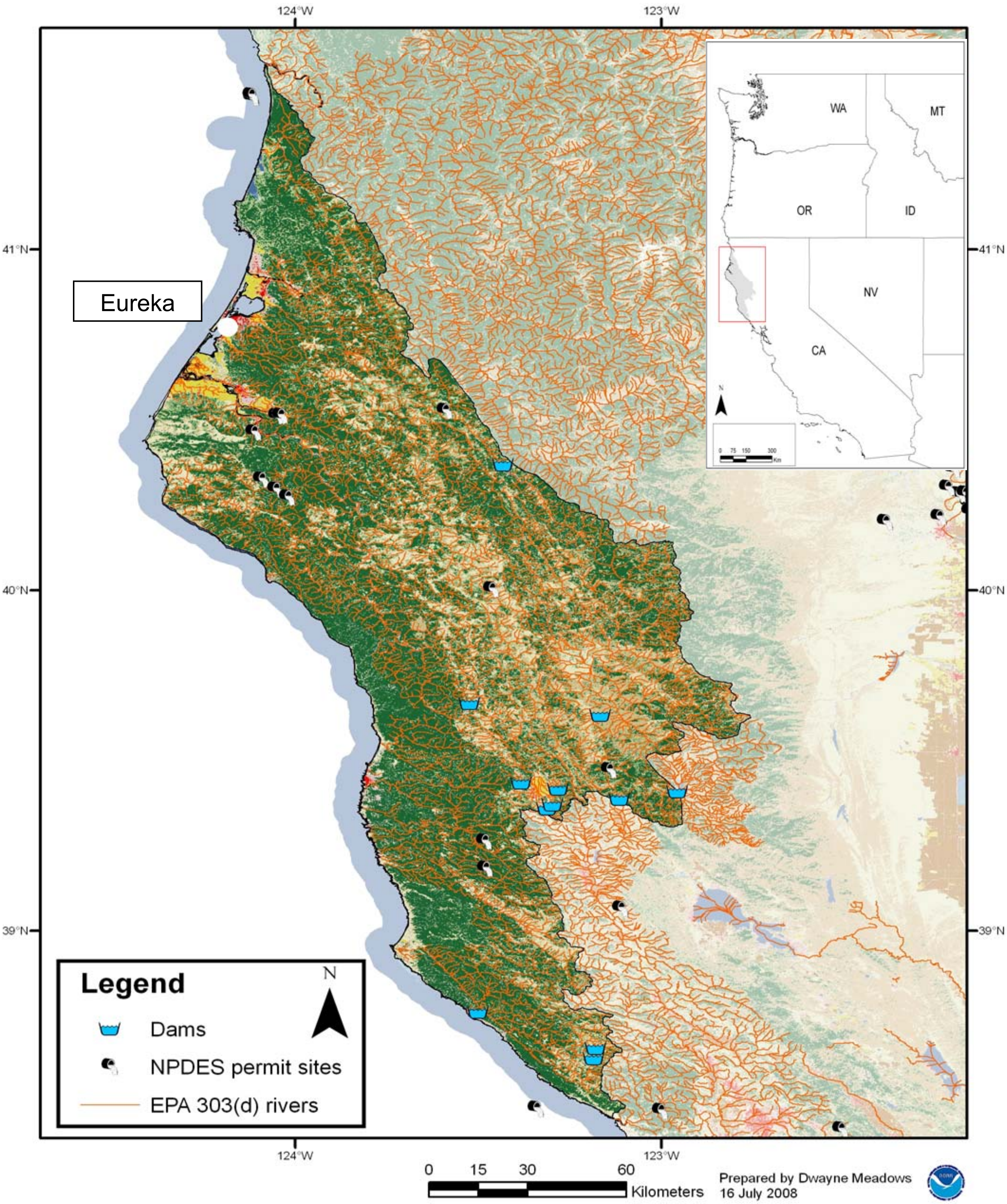


Figure 28. Northern California Steelhead distribution. Land Cover Class Legend in Figure 7.

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

More recent time series data are from snorkel counts conducted on summer-run steelhead in the Middle Fork Eel River. An estimate of lambda over the interval 1966 to 2002 was made and a random-walk with drift model fitted using Bayesian assumptions. Good et al. (2005) estimated lambda at 0.98 with a 95% confidence interval of 0.93 and 1.04. The result is an overall downward trend in both the long- and short- term. Juvenile data were also recently examined. Both upward and downward trends were apparent (Good, Waples et al. 2005). The majority (74%) of BRT votes were for “likely to become endangered,” with the remaining votes split equally between “in danger of extinction” and “not warranted”.

Critical Habitat

Critical habitat was designated for NC steelhead on September 2, 2005 (70 FR 52488). The critical habitat designation for this DPS identifies PCEs that include sites necessary to support one or more life stages of steelhead. Specific sites include: (1) freshwater spawning, (2) freshwater rearing, (3) freshwater migration, (4) estuarine areas free of obstruction, (5) nearshore marine areas free of obstructions, and (6) offshore marine areas with good water quality. The physical or biological features that characterize these sites include water quality and quantity, natural cover, and adequate forage.

Puget Sound Steelhead

Distribution

Puget Sound steelhead occupy river basins of the Strait of Juan de Fuca, Puget Sound, and Hood Canal, Washington. Included are river basins as far west as the Elwha River and as far north as the Nooksack River (Figure 29). Puget Sound's fjord-like structure may affect steelhead migration patterns. For example, some populations of coho and Chinook salmon, at least historically, remained within Puget Sound and did not migrate to the Pacific Ocean. Even when Puget Sound steelhead migrate to the high seas, they may spend considerable time as juveniles or adults in the protected marine environment of Puget Sound. This is a feature not readily accessible to steelhead from other areas of the Pacific Northwest. The species is primarily composed of winter steelhead but includes several stocks of summer steelhead, usually in subbasins of large river systems and above seasonal hydrologic barriers.

Life History

Life history attributes of Puget Sound steelhead (migration and spawn timing, smolt age, ocean age, and total age at first spawning) appear similar to those of other west coast steelhead. Ocean age for Puget Sound summer steelhead varies among populations.

Status and Trends

Puget Sound steelhead were listed as a threatened species on May 11, 2007 (72 FR 26722). Run size for this DPS, was calculated in the early 1980s at about 100,000 winter-run fish and 20,000 summer-run fish. It is unclear what portion were hatchery fish. However, a combined estimate with coastal steelhead suggested that roughly 70% of steelhead in ocean runs were of hatchery origin. The percentage in escapement to spawning grounds would be substantially lower due to differential harvest and hatchery rack returns. By the 1990s, total run size for four major stocks exceeded 45,000, roughly half of which was natural escapement.

Nehlsen et al. (1991) identified nine Puget Sound steelhead stocks at some degree of risk or concern. The WDFW et al. (1993) estimated that 31 of 53 stocks were of native origin and predominantly natural production. The WDFW assessment of the status of these 31 stocks was 11 healthy, three depressed, one critical, and 16 of unknown status. Their assessment of the status of the remaining (not native/natural) stocks was three healthy, 11 depressed, and eight of unknown status.

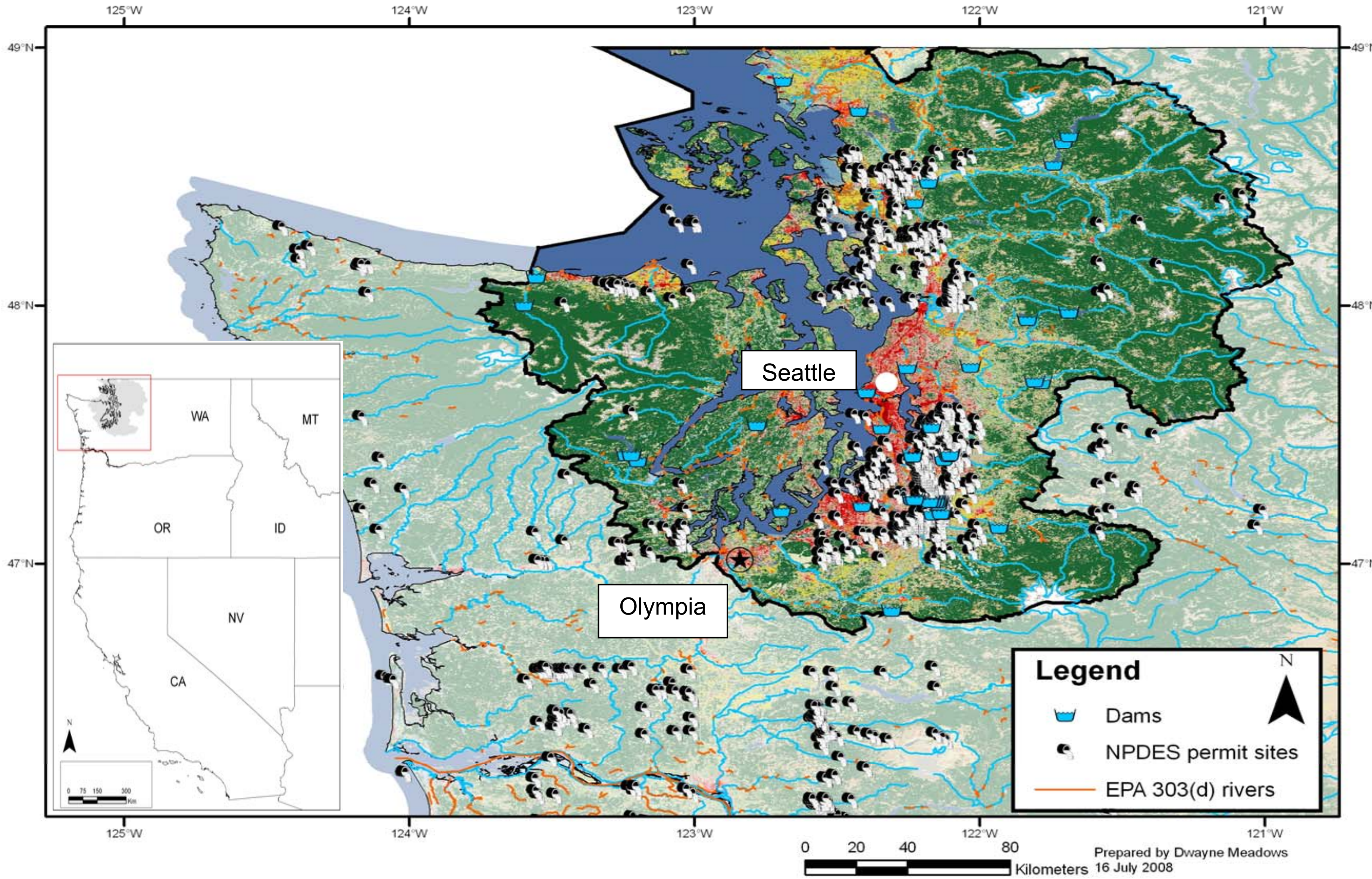


Figure 29. Puget Sound steelhead distribution. Land Cover Class Legend in Figure 7.

Of the 21 populations in the Puget Sound ESU reviewed by Busby et al. (1996), 17 had declining and four had increasing trends, with a range from 18% annual decline (Lake Washington winter-run steelhead) to 7% annual increase (Skykomish River winter-run steelhead). Eleven of these trends (nine negative, two positive) were significantly different from zero. These trends were for the late-run naturally produced component of winter-run steelhead populations. No adult trend data were available for summer-run steelhead. Most of these trends were based on relatively short data series. The Skagit and Snohomish River winter-run populations have been approximately three to five times larger than the other populations in the DPS, with average annual spawning of approximately 5,000 and 3,000 total adult spawners, respectively. These two basins exhibited modest overall upward trends at the time of the Busby et al. (1996) report. Busby et al. (1996) estimated five-year average natural escapements for streams with adequate data range from less than 100 to 7,200, with corresponding total run sizes of 550 to 19,800.

Critical Habitat

Critical habitat is not currently designated for Puget Sound steelhead. However, factors for essential habitat are under evaluation to designate future critical habitat.

Snake River Steelhead

Distribution

SR Basin steelhead is an inland species that occupies the SR basin of Idaho, northeast Oregon, and southeast Washington. The SR Basin steelhead species includes all naturally spawned populations of steelhead (and their progeny) in streams in the SR Basin of Idaho, northeast Oregon, and southeast Washington SR Basin steelhead do not include resident forms of *O. mykiss* (rainbow trout) co-occurring with these steelhead. The historic spawning range of this species included the Salmon, Pahsimeroi, Lemhi, Selway, Clearwater, Wallowa, Grande Ronde, Imnaha, and Tucannon Rivers.

Managers classify up-river summer steelhead runs into two groups based on ocean age and adult size upon return to the Columbia River. A-run steelhead are predominately age-one-ocean fish. B-run steelhead are larger, predominated by age-two-ocean fish. A-run populations are found in the tributaries to the lower Clearwater River, the upper Salmon River and its tributaries, the lower Salmon River and its tributaries, the Grand Ronde River, Imnaha River, and possibly the SR's mainstem tributaries below Hells Canyon Dam. B-run steelhead occupy four major subbasins. They include two on the Clearwater River (Lochsa and Selway) and two on the Salmon River (Middle Fork and South Fork Salmon); areas not occupied by A-run steelhead. Some natural B-run steelhead are also produced in parts of the mainstem Clearwater and its major tributaries. There are alternative escapement objectives of 10,000 (Columbia River Fisheries Management Plan) and 31,400 (Idaho) for B-run steelhead. B-run steelhead represent at least one-third and as much as three-fifths of the production capacity of the DPS. Table 22 identifies populations within the SR Basin Steelhead salmon ESU, their abundances, and hatchery input.

Table 22. SR Basin Steelhead salmon populations, abundances, and hatchery contributions (Good, Waples et al. 2005). Note: rpm denotes redds per mile.

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)**	**?**	

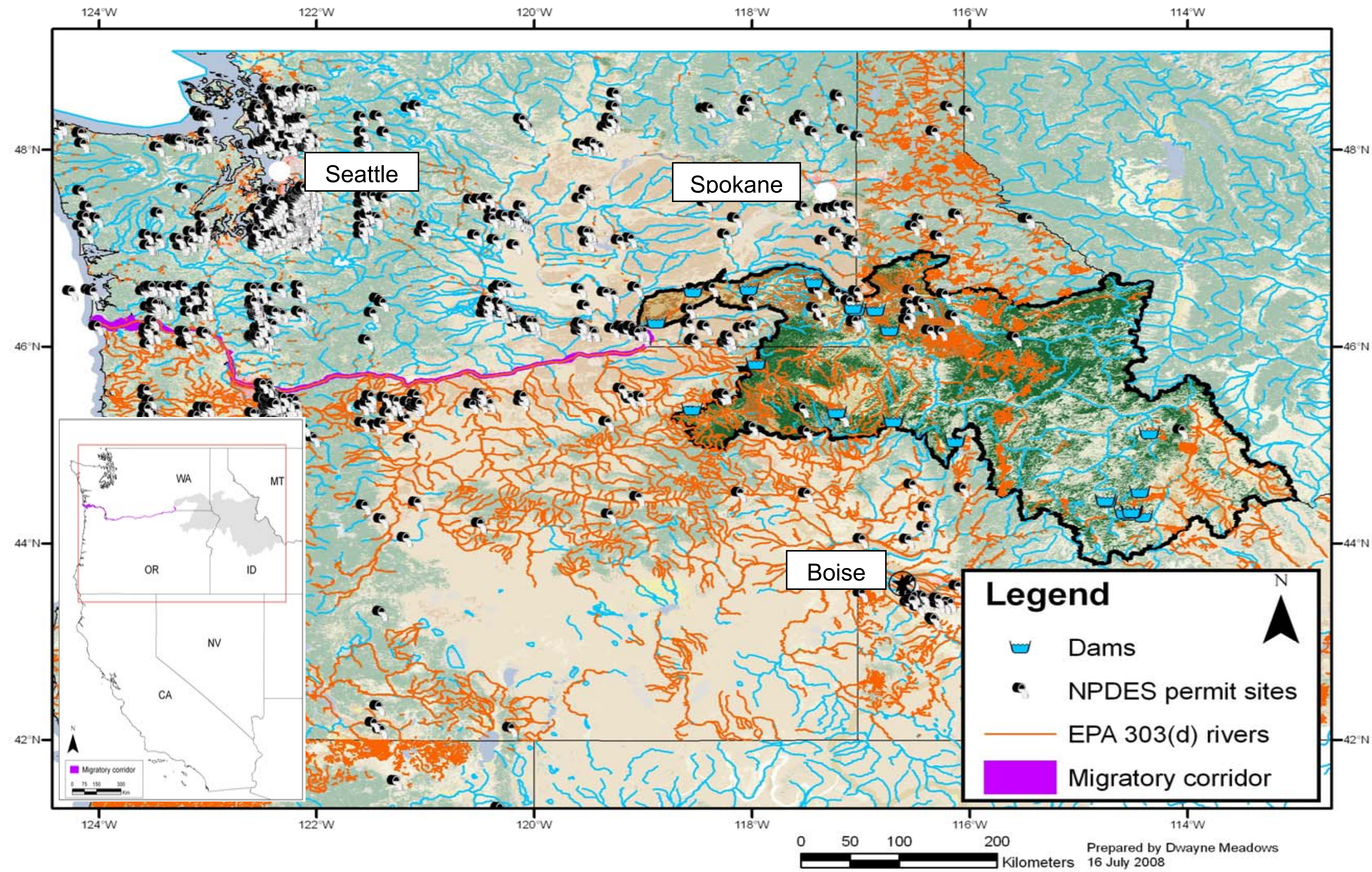


Figure 30. SR Basin Steelhead distribution. Land Cover Class Legend in Figure 7.

Life History

SR Basin steelhead occupy habitat that is considerably warmer and drier (on an annual basis) than other steelhead DPSs. SR Basin steelhead are generally classified as summer run, based on their adult run timing pattern. Sexually immature adult SR Basin summer steelheads enter the Columbia River from late June to October. SR Basin steelhead returns consist of A-run fish that spend one year in the ocean, and larger B-run fish that spend two years at sea. Adults typically migrate upriver until they reach tributaries from 1,000 to 2,000 m above sea level where they spawn between March and May of the following year. Unlike other anadromous members of the *Oncorhynchus* genus, some adult steelhead survive spawning, return to the sea, and later return to spawn a second time. After hatching, juvenile SR Basin steelhead typically spend two to three years in fresh water before they smolt and migrate to the ocean.

Status and Trends

SR Basin steelhead were listed as threatened in 1997 (62 FR 43937). Their classification status was reaffirmed following a status review on January 5, 2006 (71 FR 834). The ICBTRT (2003) identified 23 populations in the following six MPGs: Clearwater River, Grande Ronde River, Hells Canyon, Imnaha River, Lower SR, and Salmon River. SR Basin steelhead remain spatially well distributed in each of the six major geographic areas in the SR basin (Good, Waples et al. 2005). Environmental conditions are generally drier and warmer in these areas than in areas occupied by other steelhead species in the Pacific Northwest. SR Basin steelhead were blocked from portions of the upper SR beginning in the late 1800s and culminating with the construction of Hells Canyon Dam in the 1960s. The SR Basin steelhead "B run" population-levels remain particularly depressed. The ICBTRT has not completed a viability assessment for SR Basin steelhead.

Limited information on adult spawning escapement for specific tributary production areas for SR Basin steelhead made a quantitative assessment of viability difficult. Annual return estimates are limited to counts of the aggregate return over Lower Granite Dam, and spawner estimates for the Tucannon, Grande Ronde, and Imnaha Rivers. The 2001 return over Lower Granite Dam was substantially higher relative to the low levels seen in the 1990s; the recent five-year mean abundance (14,768 natural returns) was approximately 28% of the interim recovery target level. 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 SR supports approximately 63% of the total natural-origin production of steelhead in the Columbia River Basin. The current condition of SR Basin steelhead (Good, Waples et al. 2005) is summarized below:

There is uncertainty for wild populations given limited data for adult spawners in individual populations. Dam counts are currently 28% of interim recovery target for the SR Basin (52,000 natural spawners). Only the Joseph Creek population exceeds the interim recovery target. 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). Homogenization of hatchery stocks occurs within basins, and some stocks exhibit high stray rates.

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52488). The critical habitat designation for this ESU identifies PCEs that include sites necessary to support one or more steelhead life stages. Specific sites include: (1) Freshwater spawning, (2) freshwater rearing, (3) freshwater migration, (4) estuarine areas free of obstruction, (5) nearshore marine areas free of obstructions, and (6) offshore marine areas with good water quality.

Of the 291 fifth order streams reviewed in this DPS, 220 were rated as high, 44 were rated as medium, and 27 were rated as low conservation value. The physical or biological features that characterize these sites include water quality and quantity, natural cover, and adequate forage. Limiting factors identified for SR Basin salmonids include: (1) Hydrosystem mortality, (2) reduced stream flow, (3) altered channel morphology and floodplain, (4) excessive sediment, (5) degraded water quality, (6) harvest impacts, and (7) hatchery impacts (Myers, Kope et al. 1998).

South-Central California Coast Steelhead

Distribution

The South-Central California Coast (S-CCC) steelhead DPS includes all naturally spawned populations of steelhead (and their progeny) in streams from the Pajaro River (inclusive) to, but not including the Santa Maria River, California (Figure 31).

Life History

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

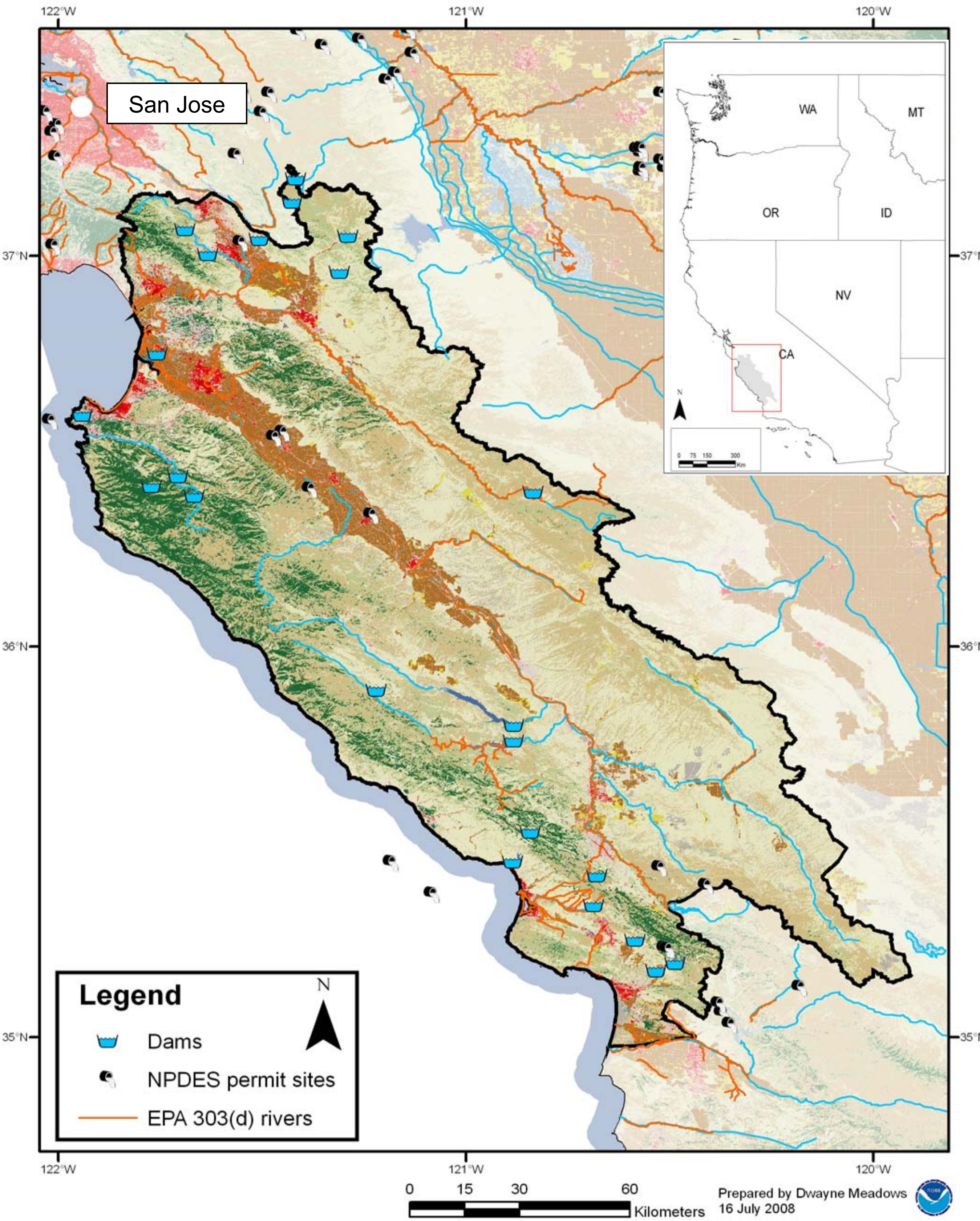


Figure 31. S-CCC steelhead distribution. Land Cover Class Legend in Figure 7.

Status and Trends

S-CCC steelhead were listed as threatened in 1997. Their classification was retained following a status review on January 5, 2006 (71 FR 834). Historical data on the S-C CC steelhead DPS are limited. In the mid-1960s, the CDFG estimated the adult population at about 18,000. We know of no recent estimates of the total DPS. However, five river systems, the Pajaro, Salinas, Carmel, Little Sur, and Big Sur, indicate that runs are currently less than 500 adults. Past estimates for these basins were almost 5,000 fish. Carmel River time series data indicate that the population declined by about 22% per year between 1963 and 1993 (Good, Waples et al. 2005). From 1991 the population increased from one adult, to 775 adults at San Clemente Dam. Good et al. (2005) thought that this recent increase seemed too great to attribute simply to improved reproduction and survival of the local steelhead population. Other possibilities were considered including that the substantial immigration or transplantation occurred, or that resident trout production increased as a result of improved environmental conditions within the basin. Nevertheless, the majority (68%) of BRT votes were for “likely to become endangered,” and another 25% were for “in danger of extinction”.

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52488). The critical habitat designation for this DPS identifies PCEs that include sites necessary to support one or more steelhead life stages. Specific sites include: (1) freshwater spawning, (2) freshwater rearing, (3) freshwater migration, (4) estuarine areas free of obstruction, (5) nearshore marine areas free of obstructions, and (6) offshore marine areas with good water quality. The physical or biological features that characterize these sites include water quality and quantity, natural cover, and adequate forage.

Southern California Steelhead

Distribution

Southern California (SC) steelhead occupy rivers from the Santa Maria River to the U.S. –Mexico border (Figure 32).

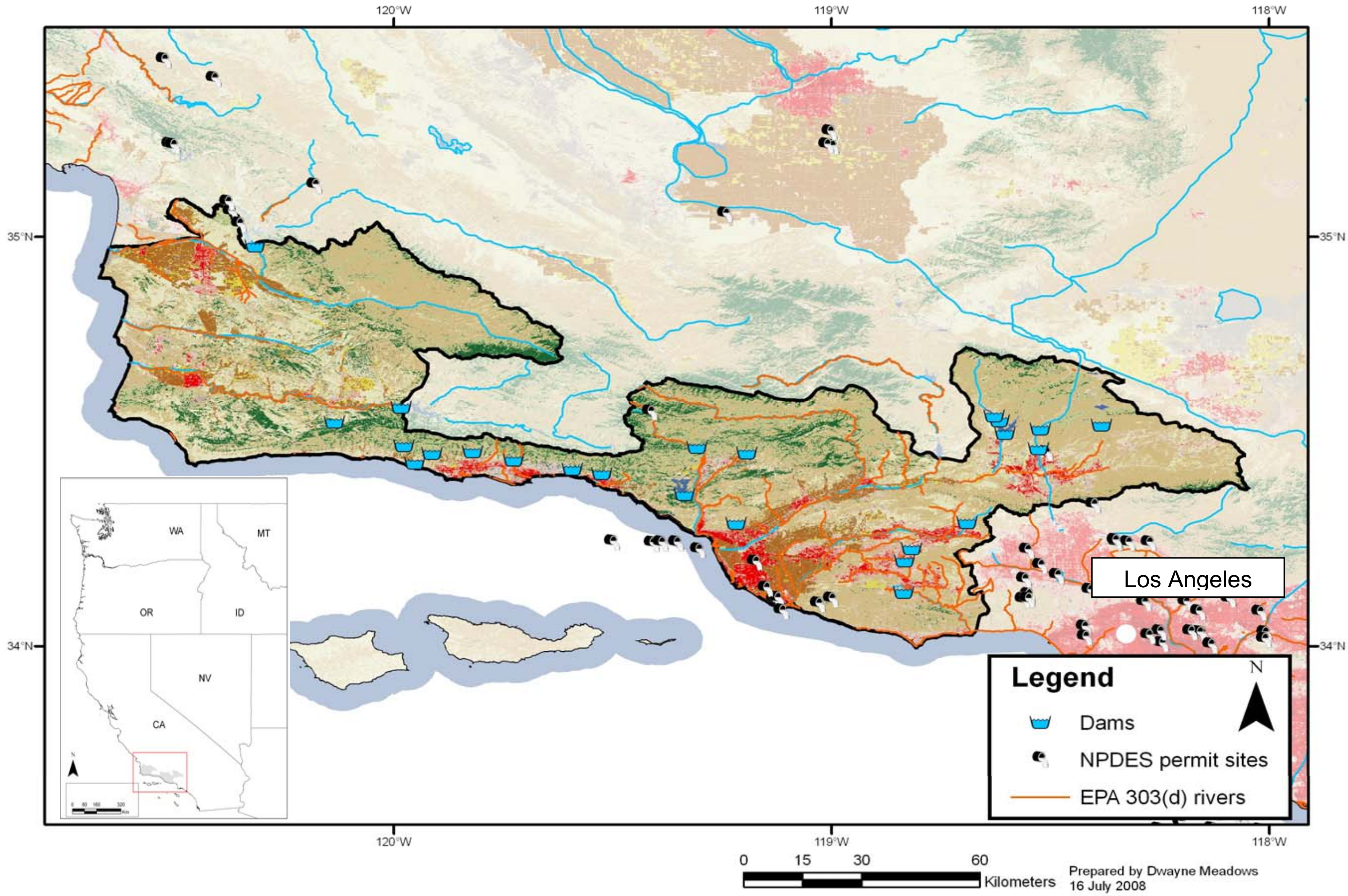


Figure 32. Southern California steelhead distribution. Land Cover Class Legend in Figure 7.

Table 23 identifies populations within the Southern California Steelhead salmon ESU, their abundances, and hatchery input.

Table 23. Southern California Steelhead salmon populations, abundances, and hatchery contributions (Good, Waples 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**	

Life History

Migration and life history patterns of SC steelhead are dependent on rainfall and streamflow (Moore 1980). Steelhead within this DPS can withstand higher temperatures than populations to the north. The relatively warm and productive waters of the Ventura River have resulted in more rapid growth of juvenile steelhead than occurs in more northerly populations (Moore 1980). There is little life history information for steelhead in this DPS.

Status and Trends

SC steelhead were listed as endangered in 1997 (62 FR 43937). Their classification was retained following a status review on January 5, 2006 (71 FR 834). In many watersheds throughout Southern California, dams isolate steelhead from historical spawning and rearing habitats. Dams also alter the hydrology of the basin (*e.g.*, Twitchell Reservoir within the Santa Maria River watershed, Bradbury Dam within the Santa Ynez River watershed, Matilija and Casitas dams within the Ventura River watershed, Rindge Dam within the Malibu Creek watershed). 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. 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, Wainwright et al. 1996). The majority (81%) of the BRT votes were for "in danger of extinction," with the remaining 19% of votes for "likely to become endangered. This was based on extremely strong concern for abundance, productivity, and spatial concern (as per the risk matrix); diversity was also of concern. The BRT also expressed concern about the lack of data on the SC steelhead DPS, including uncertainty on the metapopulation dynamics in the southern part of the DPS's range and the fish's nearly complete extirpation from the southern part of the range.

Critical Habitat

Critical habitat was designated for this species on September 2, 2005. The designation identifies PCEs that include sites necessary to support one or more steelhead life stages. These sites contain the physical or biological features essential for the species conservation. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, and estuarine areas. The physical or biological features that characterize these sites include water quantity, depth, and velocity, shelter, cover, living space and passage conditions.

Upper Columbia River Steelhead

Distribution

UCR steelhead occupy the Columbia River Basin upstream from the Yakima River, Washington, to the border between the U.S. and Canada (Figure 33). This area includes the Wenatchee, Entiat, and Okanogan Rivers. All UCR steelhead are summer steelhead. Steelhead primarily use streams of this region that drain the northern Cascade Mountains of Washington State. This species includes hatchery populations of summer steelhead from the Wells Hatchery because it probably retains the genetic resources of steelhead populations that once occurred above the Grand Coulee Dam. This species does not include the Skamania Hatchery stock because of its non-native genetic heritage.

Abundance estimates of returning naturally produced UCR steelhead have been based on extrapolations from mainstem dam counts and associated sampling information (*e.g.*, hatchery/wild fraction, age composition). 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 (ICBTRT 2003). Table 24 identifies populations within the UCR Steelhead salmon DPS, their abundances, and hatchery input.

Table 24. UCR Steelhead salmon populations, abundances, and hatchery contributions (Good, Waples 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**	

Life History

The life history patterns of UCR steelhead are complex. Adults return to the Columbia River in the late summer and early fall. Most migrate relatively quickly up the mainstem to their natal tributaries. A portion of the returning run overwinters in the mainstem reservoirs, passing over the upper-mid-Columbia dams in April and May of the following year. Spawning occurs in the late spring of the calendar year following entry into the river. Juvenile steelhead spend one to seven years rearing in freshwater before migrating to sea. Smolt outmigrations are predominantly age-two and age-three juveniles. Most adult steelhead return after one or two years at sea, starting the cycle again.

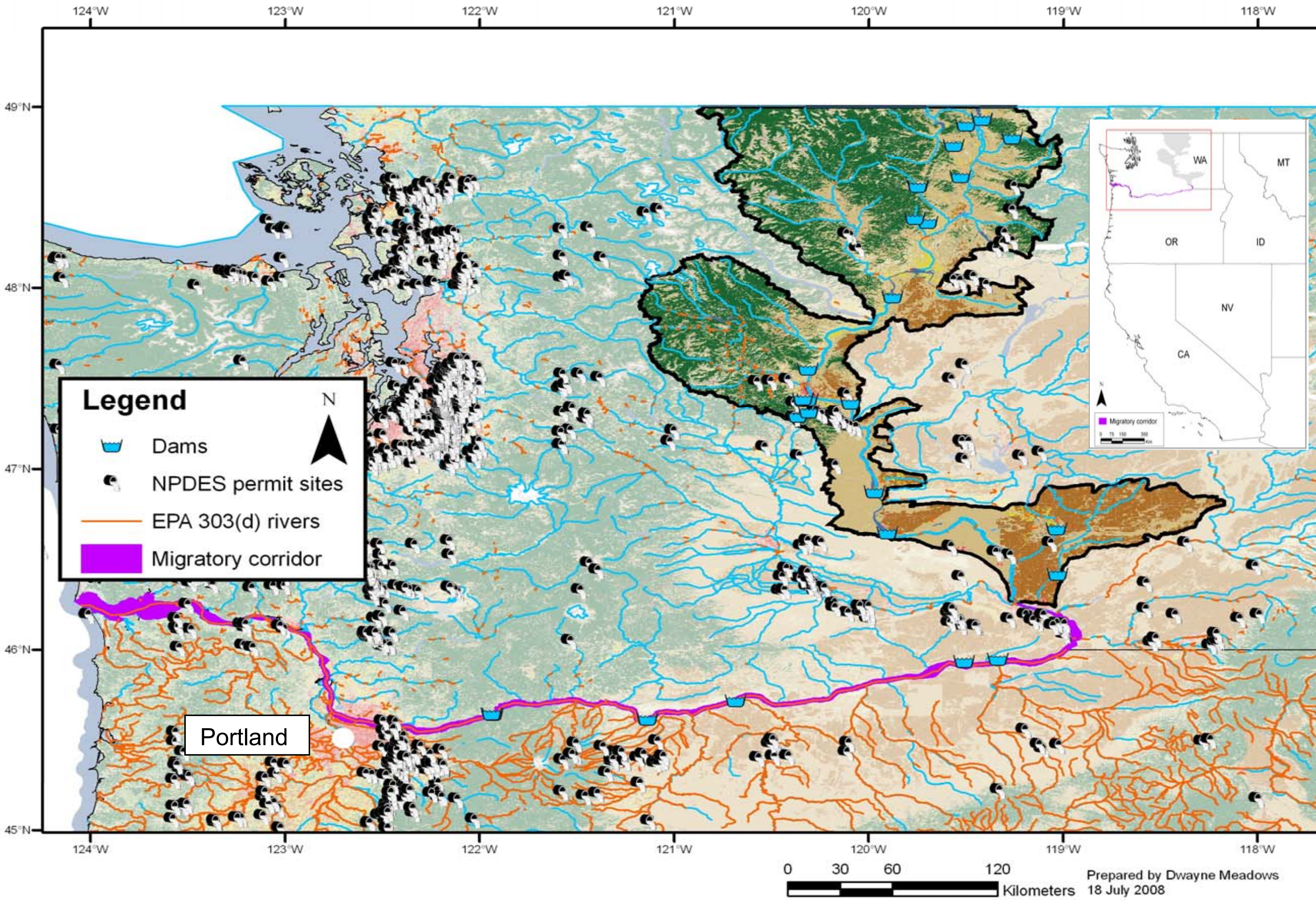


Figure 33. UCR Steelhead distribution. Land Cover Class Legend in Figure 7.

Returns of both hatchery and naturally produced steelhead to the UCR 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 have been based on extrapolations from mainstem dam counts and associated sampling information (*e.g.*, hatchery/wild fraction, age composition). 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 (ICBTRT 2003).

In terms of natural production, recent population abundances for both the Wenatchee and Entiat aggregate population and the Methow population remain well below the minimum abundance thresholds developed for these populations (ICBTRT 2005). A five-year geometric mean (1997 to 2001) of approximately 900 naturally produced steelhead returned to the Wenatchee and Entiat rivers (combined). Although this is well below the minimum abundance thresholds, it represents an improvement over the past (an increasing trend of 3.4% per year). However, the average percentage of natural fish for the recent five-year period dropped from 35% to 29%, compared to the previous status review. For the Methow population, the five-year geometric mean of natural returns over Wells Dam was 358. Although this is well below the minimum abundance thresholds, it is an improvement over the recent past (an increasing trend of 5.9% per year). In addition, the 2001 return (1,380 naturally produced spawners) was the highest single annual return in the 25-year data series. However, the average percentage of wild origin spawners dropped from 19% for the period prior to the 1998 status review to 9% for the 1997 to 2001 returns.

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. The ICBTRT has characterized the spatial structure risk to UCR steelhead populations as “low” for the Wenatchee and Methow, “moderate” for the Entiat, and “high” for the Okanogan. Overall adult returns are dominated by hatchery fish, and detailed information is lacking on the productivity of the natural population. All UCR steelhead populations have reduced genetic diversity from homogenization of populations that occurred during the Grand Coulee Fish Maintenance project from 1939-1943, from 1960, and 1981 (Chapman, Hillman et al. 1994).

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52488). The critical habitat designation for this DPS identifies PCEs that include sites necessary to support one or more steelhead life stages. They include all Columbia River estuarine areas and river reaches upstream to Chief Joseph Dam and several tributary subbasins. Specific sites include freshwater spawning and rearing sites, freshwater migration corridors, estuarine areas free of obstruction, and offshore marine areas. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, and adequate passage conditions.

The UCR steelhead DPS has 42 watersheds within its range. Three watersheds received a low rating, eight received a medium rating, and 31 rated a high conservation value to the DPS. In addition, the Columbia River rearing/migration corridor downstream of the spawning range was rated as a high conservation value. Limiting factors identified for the UCR steelhead include: (1) Mainstem Columbia River hydropower system mortality, (2) reduced tributary streamflow, (3) tributary riparian degradation and loss of in-river wood, (4) altered tributary floodplain and channel morphology, and (5) excessive fine sediment and degraded tributary water quality. The above activities and features also introduce sediment, nutrients, biocides, metals, and other pollutants into surface and ground water and degrade water quality in the freshwater, estuarine, and coastal ecosystems throughout the Pacific Northwest.

Upper Willamette River Steelhead

Distribution

UWR steelhead occupy the Willamette River and its tributaries upstream of Willamette Falls (Figure 34). This is a late-migrating winter group that enters freshwater in March and April (Howell, Jones et al. 1985). Only the late run was included in the listing of this species, which is the largest remaining population in the Santiam River system. Table 25 identifies populations within the UWR Steelhead salmon ESU, their abundances, and hatchery input.

Table 25. UWR Steelhead salmon populations, abundances, and hatchery contributions (Good, Waples et al. 2005). Note: rpm denotes redds per mile.

Population	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**	

Life History

Winter steelhead enter the Willamette River beginning in January and February. They do not ascend to their spawning areas until late March or April (Dimick and Merryfield 1945). Spawning occurs from April to June 1st and redd counts are conducted in May. The smolt migration past Willamette Falls also begins in early April and extends through early June (Howell, Jones et al. 1985) Migration peaks in early- to mid-May. Steelhead smolts generally migrate away from the shoreline and enter the Columbia via Multnomah Channel rather than the mouth of the Willamette. Most spend two years in the ocean before re-entering fresh water to span (Busby, Wainwright et al. 1996). Steelhead in the UWR DPS generally spawn once or twice. A few fish may spawn three times based on patterns found in the LCR steelhead DPS. Repeat spawners are predominantly female and generally account for less than 10% of the total run size (Busby, Wainwright et al. 1996).

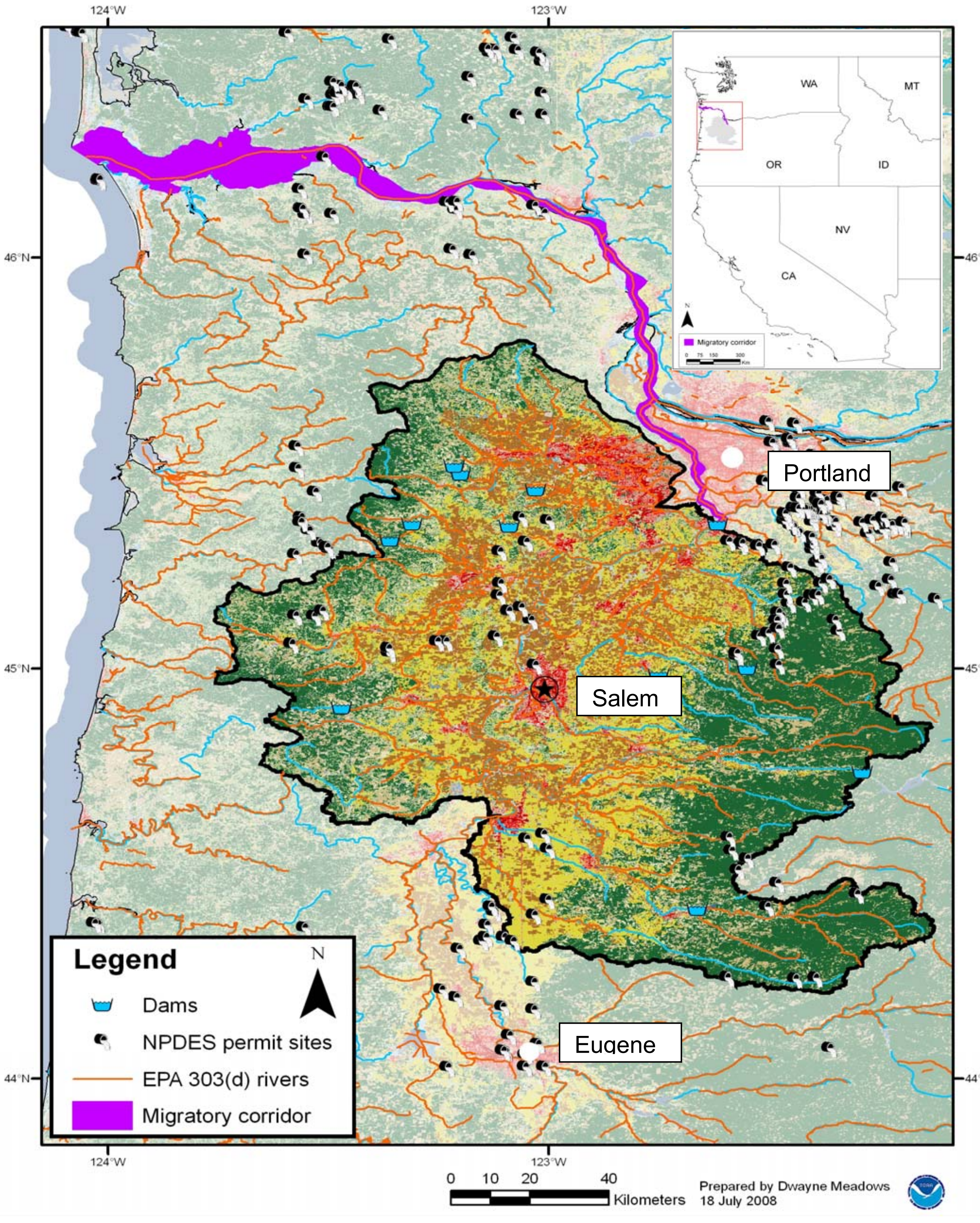


Figure 34. UWR Steelhead distribution. Land Cover Class Legend in Figure 7.

Status and Trends

UWR steelhead were listed as threatened in 1999 (64 FR 14517). Their classification was retained following a status review on January 5, 2006 (71 FR 834). A major threat to Willamette River steelhead results from artificial production practices. Fishways built at Willamette Falls in 1885 have allowed Skamania-stock summer steelhead and early-migrating winter steelhead of Big Creek stock to enter the range of UWR steelhead. The population of summer steelhead is almost entirely maintained by hatchery salmon, although natural-origin, Big Creek-stock winter steelhead occur in the basin (Howell, Jones et al. 1985). In recent years, releases of winter steelhead are primarily of native stock from the Santiam River system.

Steelhead in this DPS are depressed from historical levels, but to a much lesser extent than are spring Chinook in the Willamette basin (McElhaney, Chilcote et al. 2007). All of the historical populations remain extant and moderate numbers of wild steelhead are produced each year. The population growth rate data indicate long-term trends are <1; short-term trends are 1 or higher (McElhaney, Chilcote et al. 2007). Spatial structure for the North and South Santiam populations has been substantially reduced by the loss of access to the upper North Santiam basin and the Quartzville Creek watershed in the South Santiam subbasin due to construction of the dams owned and operated by the U.S. Army Corps of Engineers without passage facilities (McElhaney, Chilcote et al. 2007). Additionally, the spatial structure in the Molalla subbasin has been reduced significantly by habitat degradation and in the Calapooia by habitat degradation and passage barriers. Finally, the diversity of some populations have been eroded by small population size, the loss of access to historical habitat, legacy effects of past winter-run hatchery releases, and the ongoing release of summer steelhead (McElhaney, Chilcote et al. 2007).

Critical Habitat

Critical habitat was designated 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 as well as specific steam reaches in the following subbasins: Upper Willamette, North Santiam, South Santiam, Middle Willamette, Molalla/Pudding, Yamhill, Tualatin, and Lower Willamette (NMFS 2005b). The critical habitat designation for this DPS identifies PCEs that include sites necessary to support one or more steelhead life stages. Specific sites include: (1) Freshwater spawning, (2) freshwater rearing, (3) freshwater migration, (4) estuarine areas free of obstruction, (5) nearshore marine areas free of obstructions, and (6) offshore marine areas with good water quality. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity. Anthropogenic land uses introduce sediment, nutrients, biocides, metals, and other pollutants into surface and ground water and degrade water quality in the freshwater, estuarine, and coastal ecosystems throughout the Pacific Northwest. These human impacts affect the essential feature requirements for this DPS.

Of 43 subbasins reviewed in NMFS' assessment of critical habitat for the UWR steelhead, 20 subbasins were rated as having a high conservation value, while six were rated as having a medium value and 17 were rated as having a low value to the conservation of the DPS.

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 Action*, and *Cumulative Effects* sections of this Opinion. We then evaluate these consequences in combination with the environmental baseline to determine the likelihood of jeopardy or adverse modification of designated critical habitat.

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

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

Natural Mortality Factors

Available data indicate high natural mortality rates for salmonids, especially in the open ocean/marine environment. According to Bradford (1997), 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 1997; Brodeur, Fisher et al. 2004). In freshwater rearing habitats, the natural mortality rate averages about 70% for all salmonid species (Bradford 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 (Marshall and Britton 1990; Bradford 1997). 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 (Kier Associates 1991; Walker and Foott 1993; Foott, Harmon et al. 2003). However, disease outbreaks may occur when water quality is diminished and fish are stressed from crowding and diminished flows (Spence, Lomnicky et al. 1996; Guillen 2003). Young coho salmon or other salmonid species may become stressed and lose their resistance in higher temperatures (Spence, Lomnicky 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. 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.

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 (Hard, Jones et al. 1992; Hanson, Baird et al. 2005; Ford and Ellis 2006). Generally, harbor seals do not feed on salmonids as frequently as California sea lions (Pearcy 1997). California sea lions from the Ballard Locks in Seattle, Washington have been estimated to consume about 40% of the steelhead runs since 1985/1986 (Gustafson, Wainwright 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 LCR winter steelhead in the Bonneville tailrace has increased. This prompted ongoing actions to reduce predation effects. They include the exclusion, hazing, and in some cases, lethal take of marine mammals near Bonneville Dam (FCRPS 2008).

NOAA Fisheries has granted permits to the states of Idaho, Oregon, and Washington for the lethal removal of individual California sea lions that prey on adult spring-run Chinook salmon in the tail race of Bonneville Dam under section 120 of the Marine Mammal Protection Act (NMFS 2006). This action may increase the survival of adult Chinook salmon and steelhead. The Humane Society of the U.S. unsuccessfully challenged the issuance of these permits. The case is now on appeal.

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 et al. 2005). Recent research shows that subyearlings from the LCR Chinook salmon ESU are also subject to tern predation. This may be due to the long estuarine residence time of the LCR Chinook salmon (Ryan, Carper et al. 2006). Caspian terns and cormorants may be responsible for the mortality of up to 6% of the outmigrating stream-type juveniles in the Columbia River basin (Roby, Collis et al. 2006; Collis 2007).

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

Fish Predation

Pikeminnows (*Ptychocheilus oregonensis*) are significant predators of yearling juvenile migrants (Friesen and Ward 1999). Chinook salmon were 29% of the prey of northern 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. Nevertheless, predation of juvenile sockeye likely occurs. 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 subyearling juvenile life history (Friesen and Ward 1999).

The primary fish predators in estuaries are probably adult salmonids or juvenile salmonids which emigrate at older and larger sizes than others. They include cutthroat trout (*O. clarki*) or steelhead smolts preying on chum or pink salmon smolts. Outside estuaries, many large fish population 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 (Beamish, Thomson et al. 1992; Pearcy 1992; Beamish and Neville 1995).

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 et al. 2004; Rinne 2004). Nevertheless, fire is also one of the dominant habitat-forming processes in mountain streams (Bisson, Rieman 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 fishes rely on macroinvertebrates as a food source for at least a portion of their life histories. When small amounts of ash enter the water, there are usually no noticeable changes to the macroinvertebrate community or the water quality (Bowman and Minshall 2000). When significant amounts of ash are deposited into rivers, the macroinvertebrate community density and composition may be moderately to drastically reduced for a full year with long-term effects lasting 10 years or more (Buchwalter, Jenkins et al. 2003), (Minshall, Royer et al. 2001; Buchwalter, Sandahl et al. 2004). 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, Jenkins et al. 2003).

Many species have evolved in the presence of regular fires and have developed population-level mechanisms to withstand even the most intense fires (Greswell 1999). These same species have come to rely on fire's disturbance to provide habitat heterogeneity. In the past century, the human population has increased dramatically, resulting in urban sprawl and the development of formerly remote locations. This condition has increased the urban/wildland interface. As a result, the threat of fires to personal property and people has increased, including the demand for protection of their safety and belongings. We expect listed fish species will be exposed to an increasing number of fires and fire fighting techniques over time. Currently, federal, state, and local resource agencies lack long-term monitoring data on the effects of wildland fire on listed Pacific salmonids and their habitats. Thus, we are unable to quantify the overall effects of wildland fire on the long-term survival of listed Pacific salmonids at this time.

Oceanographic Features and Climatic Variability

Oceanographic features of the action area may influence prey availability and habitat for Pacific salmonids. 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. Ocean conditions and climatic variability may affect salmonids in the action area.

The primary effects of the ocean on salmon productivity involve growth and survival of salmon. All salmon growth is completed in the ocean. According to Welch (1996), fish growth will not reach its maximum potential if food density (food available divided by ocean volume) is insufficient to provide the maximum daily ration. If this critical level of food is not exceeded, then the potential for the ocean to limit salmon growth exists.

The decline in salmon survival in Oregon and Washington since 1977 may be caused by poorly understood processes in the marine (as opposed to freshwater) environment (Welch 1996). Current findings also indicate that the primary control on salmon distribution is temperature. However, the upper thermal limit varies throughout the year (Welch 1996).

Naturally occurring climatic patterns, such as the Pacific Decadal Oscillation and the El Niño and La Niña events, are major causes of changing marine productivity. Recent studies have shown that long-term changes in climate affect oceanic structure and produce abrupt differences in salmon marine survival and returns (Mantua, Hare et al. 1997; Hare, Mantua et al. 1999). A major regime shift in the subarctic and California Current ecosystems during the late 1970s may have been a factor in reducing ocean survival of salmon in the Pacific Northwest and in increasing the marine survival in Alaska (Hare, Mantua et al. 1999). Fluctuations in mortality of salmon in the freshwater and marine environment have been shown to be almost equally significant sources of annual recruitment variability (Bradford 1997). These events and changes in ocean temperature may also influence salmonid abundance in the action area. In years when ocean conditions are cooler than usual, the majority of sockeye salmon returning to the

Fraser River do so via this route. However, when warmer conditions prevail, migration patterns shift to the north through the Johnstone Strait (Groot and Quinn 1987).

Climate Change

Anthropogenic climate change, caused by factors such as the continuing build-up of human-produced atmospheric carbon dioxide, is predicted to have major environmental impacts along the west coast of North America during the 21st century and beyond (IPCC 2001; CIG 2004). Warming trends continue in both water and air temperatures. Projections of the consequences of climate change include disruption of annual cycles of rain and snow, alteration of prevailing patterns of winds and ocean currents, and increases in sea levels (Glick 2005; Snover, Mote et al. 2005). 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 productivity and food webs, including populations of salmon and other salmonid prey (Hard, Jones et al. 1992).

Climate change poses significant hazards to the survival and recovery of salmonids along the west coast. Changes in water temperature can alter migration timing, reduce growth, reduce the supply of available oxygen in the water, reduce insect availability as prey, and increase the susceptibility of fish to toxicants, parasites, and disease (Fresh, Casillas et al. 2005; NMFS 2007). Earlier spring runoff and lower summer flows make it difficult for returning adult salmon to negotiate obstacles (NMFS 2007). Excessively high levels of winter flooding can scour eggs from their nests in the stream beds and increase mortalities among overwintering juvenile salmon. The predicted increased winter flooding, decreased summer and fall stream flows, and elevated warm season temperatures in the streams and estuaries may further degrade conditions for salmon that are already stressed from habitat degradation. 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 (IPCC 2001; CIG 2004).

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

Anthropogenic Mortality Factors

In this section we address anthropogenic threats in the geographic regions across the action area. Among the threats discussed are the "four Hs": hatcheries, harvest, hydropower, and habitat. Prior to discussion of each geographic region, three major issues are highlighted: pesticide contamination, elevated water temperature, and loss of habitat/habitat connectivity. These three factors are the most relevant to the current analysis. To address these issues, 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 carbaryl, carbofuran, and methomyl in the U.S. and its territories. As water temperature plays such a strong role in salmonid distribution, we also provide a general discussion of anthropogenic temperature changes. Finally, we discuss the health of riparian systems and floodplain connectivity, as this habitat is vital to salmonid survival.

Baseline Pesticide Detections in Aquatic Environments

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

In the exposure section of the *Effects of the Proposed Action* we also present more recent unpublished data on the chemicals and degradates addressed in this Opinion from the NAWQA program and state databases maintained by California and Washington. As far as NMFS was able to ascertain, neither Oregon nor Idaho maintain publically available state-wide water quality databases. The California and Washington databases include some data from the NAWQA, but mostly the data are from more localized studies. Overall, data from those databases are relatively consistent in regards to pesticides addressed in this Opinion, with carbaryl generally being the most frequently quantifiable parent compound. Carbaryl and carbofuran were measured in concentrations ranging from 0.0001-33.5 μg/L. Methomyl generally was measured at slightly lower concentrations, ranging from 0.004-5.4 μg/L. Methomyl is also detected less frequently in some monitoring datasets, as it dissipates rapidly in aquatic systems, and non-targeted monitoring does not necessarily coincide with applications. Both 1-napthol (methomyl degradate) and 3-hydroxycarbofuran (carbofuran degradate) were detected in slightly lower concentrations, ranging from 0.0007-0.64 μg/L, than any of the parent compounds.

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.

National Water-Quality Assessment Program.

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

About 40 pesticide compounds accounted for most detections in water, fish, or bed sediment. Twenty-four pesticides and one degradate were each detected in more than 10% of streams in agricultural, urban, or mixed land use settings. These 25 pesticide compounds include 11 herbicides used most heavily in agriculture during the study period (plus the atrazine degradate, deethylatrazine); 7 herbicides used extensively for non-agricultural purposes; and 6 insecticides used in both agricultural and urban settings. Three of those insecticides were chlorpyrifos, diazinon, and malathion. Thirteen organochlorine pesticide compounds, including historically used parent pesticides and their degradates and by-products, were each found in more than 10% of fish or bed-sediment samples from streams draining watersheds with either agricultural, urban, or mixed land use (Gilliom, Barbash et al. 2006).

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

Chlorpyrifos and diazinon were commonly used in agricultural and urban areas from 1992-2001 and prior to the sampling period. About 13 million lbs of chlorpyrifos and about 1 million lbs of diazinon were applied for agricultural use. Non-agricultural uses of chlorpyrifos and diazinon totaled about 5 million and 4 million lbs per year in 2001, respectively (Gilliom, Barbash et al. 2006). For both insecticides, concentrations in most urban streams were higher than in most agricultural streams, and were similar to those found in agricultural areas with the greatest intensities of use. Diazinon and chlorpyrifos were detected about 75% and 30% of the time in urban streams, respectively (Gilliom, Barbash et al. 2006). NMFS (2008) determined that current use of chlorpyrifos, diazinon, and malathion is likely to jeopardize the continued existence of 27 listed salmonid ESUs. NMFS provided EPA with reasonable and prudent alternatives (RPAs), including buffers and vegetative strips, to reduce pesticide exposure to listed salmon. Until the EPA implements the RPAs, we must assume current exposure will continue.

Another dimension of pesticides and their degradates in the aquatic environment is their simultaneous occurrence as mixtures (Gilliom, Barbash 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 (Gilliom, Barbash et al. 2006). Fish experiencing coincident exposure to multiple pesticides may also experience additive and synergistic effects. If the effects on a biological endpoint from concurrent exposure to multiple pesticides can be predicted by adding the potency of the pesticides involved, the effects are said to be additive. If, however, the response to a mixture lead 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. Carbaryl, carbofuran, and methomyl are all AChE inhibitors. In California, there are 61 pesticides that inhibit AChE approved for use (CDPR 2007). According to CDPR, the amount of these chemicals used has decreased (Table 26). However, some AChE a.i.s – such as bensulide and naled – are increasing in use (CDPR 2007). While the trend indicates decreased reliance on these products, we note that their current use remains significant.

Table 26. Use figures for AChE inhibiting pesticides in California (CDPR 2007)

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

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

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

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

More than half of all agricultural streams sampled and more than three-quarters of all urban streams had concentrations of pesticides in water that exceeded one or more benchmarks for aquatic life. Aquatic life criteria are EPA water-quality guidelines for protection of aquatic life. Exceedance of an aquatic life benchmark level indicates a strong probability that aquatic species are being adversely affected. However, aquatic species may also be affected at levels below criteria. Finally, organochlorine pesticides that were discontinued 15 to 30 years ago still exceeded benchmarks for aquatic life and fish-eating wildlife in bed sediment or fish-tissue samples from many streams.

National Pollutant Discharge Elimination System

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

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

California, Idaho, Oregon, and Washington.

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. This rule was vacated by the courts on January 7, 2009 (National Cotton Council v. EPA, 553 F.3d 927 (6^{th} Cir. 2009)).

Baseline Water Temperature- Clean Water Act

Elevated temperature is considered a water pollutant in most states with approved Water Quality Standards under the federal Clean Water Act (CWA) of 1972. As per the CWA, states periodically prepare a list of all surface waters in the state for which beneficial uses - such as drinking, recreation, aquatic habitat, and industrial use – are impaired by pollutants. These are water quality limited estuaries, lakes, and streams that do not meet state surface water quality standards, and are not expected to improve within the next two years. This process is in accordance with section 303(d) of the CWA. There are five categories a waterway can be classified under as per section 303(d):

- Category 1: Meets tested water quality standards;
- Category 2: Some evidence of a water quality problem, action not yet required;
- Category 3: Insufficient data;
- Category 4: Polluted waterway with solution being implemented; and
- Category 5: Polluted waterway, action is required

Water bodies listed under Category 5 are those that are considered impaired or threatened by pollution. The "303(d) list" is generally considered synonymous with the Category 5 waters, and will be treated as such within this Opinion.

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. 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. A TMDL includes a plan for reducing contaminant loading and is required for all impaired waterways. Each state also establishes a priority ranking for the development of TMDLs for such waters, considering the severity of the pollution and the uses to be made of such waters.

Federal non-priority water quality standards have been established for carbaryl and carbofuran, but not for methomyl. The California 303(d) list includes a 49 mile section of the Colusa Basin Drain that exceeds carbofuran standards (Category 5). Several areas in Washington and Oregon have been listed under Category 2 for carbaryl and carbofuran. They include: Willapa Bay, WA (carbaryl, 18 separate listings); Grays Harbor County Drainage Ditch #1, WA (carbaryl); Pacific County Drainage Ditch #1, WA (carbaryl); North River, WA (carbaryl); Palix River, WA (carbaryl); Johnson Creek, OR (carbaryl & carbofuran, 23.7 river miles); Beaverton Creek, OR (carbaryl, 9.8 river miles); Tualatin River, OR (carbaryl, 44.7 river miles); and Mill Creek, OR (carbofuran, 25.7 river miles). In addition to specific compounds, water bodies are listed as impaired due to "pesticides" as a general category. We did not consider these waterways as there was no way to tell what compounds were present.

Temperature is significant for the health of aquatic life. Water temperatures affect the distribution, health, and survival of native cold-blooded salmonids in the Pacific Northwest. These fish will experience adverse health effects when exposed to temperatures outside their optimal range. For listed Pacific salmonids, water temperature tolerance varies between species and life stages. Optimal temperatures for rearing salmonids range from 10ºC and 16ºC. In general, the increased exposure to stressful water temperatures and the reduction of suitable habitat caused by drought conditions reduce the abundance of salmon. Warm temperatures can reduce fecundity, increase egg survival, retard growth of fry and smolts, reduce rearing densities, increase susceptibility to disease, decrease the ability of young salmon and trout to compete with other species for food, and to avoid predation (Spence, Lomnicky et al. 1996; McCullough 1999). Migrating adult salmonids and upstream migration can be delayed by excessively warm stream temperatures. Excessive stream temperatures may also negatively affect incubating and rearing salmonids (Gregory and Bisson 1997).

Sublethal temperatures (above 24ºC) could be detrimental to salmon by increasing susceptibility to disease (Colgrove and Wood 1966) or elevating metabolic demand (Brett 1995). Substantial research demonstrates that many fish diseases become more virulent at temperatures over 15.6ºC (McCullough 1999). Due to the sensitivity of salmonids to temperature, states have established lower temperature thresholds for salmonid habitat as part of their water quality standards. A water body is listed for temperature on the 303(d) list if the 7-day average of the daily maximum temperatures (7-DADMax) exceeds the temperature threshold (Table 27).

Table 27. Washington State water temperature thresholds for salmonid habitat. These temperatures are representative of limits set by California, Idaho, and Oregon (WSDE 2006).

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

Water bodies that are not designated salmonid habitat are also listed if they have a one-day maximum over a given background temperature. Using publicly available GIS layers, we determined the number of km on the 303(d) list for exceeding temperature thresholds within the boundaries of each ESU (Table 28). Because the 303(d) list is limited to the subset of rivers tested, the chart values should be regarded aslower-end estimates.

While some ESU 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. Increased water temperature may result in wastewater discharge, decreased water flow, minimal shading by riparian areas, and climatic variation.

Table 28. Number of kilometers of river, stream and estuaries included in state 303(d) lists due to temperature that are located within each salmonid ESU. 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 Williamette 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 Williamette River	–	1,668.0	–	–	1,668.0

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

Baseline Habitat Condition

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

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

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

Periodic flooding of these areas creates habitat used by salmonids. 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 et al. 2001). Side channels and sloughs are important spawning and rearing habitat for salmonids. The degree to which these off-channel habitats are linked to the main channel via surface water connections is referred to as connectivity (PNERC 2002). As river height increases with heavier flows, more side channels form and connectivity increases.

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

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

Geographic Regions

For a more fine scale analysis, we divided the action area into geographic regions: the Southwest Coast Region (California) and the Pacific Northwest Region (Idaho, Oregon, and Washington). The Pacific Northwest Region was further subdivided according to ecoregions or other natural features important to NMFS trust resources. Use of these geographic regions is consistent with previous NMFS consultations conducted at the national level (NMFS 2007). We summarize the principal anthropogenic factors occurring in the environment that influence the current status of listed species within each region. Table 29 provides a breakdown of these regions and includes the USGS subregions and accounting units for each region. It also provides a list of ESUs found in each accounting unit, as indicated by Federal Register listing notices.

Southwest Coast Region

The basins in this section occur in the State of California and the southern parts of the State of Oregon. Table 30 and Table 31 show land area in km² for each ESU /DPS located in the Southwest Coast Region.

Table 29. USGS Subregions and accounting units within the Northwest and Southwest Regions, along with ESUs present within the area (Seaber, Kapinos et al. 1987).

Region	USGS Subregion	Accounting Unit	State	HUC no.	ESU
Pacific Northwest: Columbia River Basin	Upper Columbia River Basin	—	WA	170200	Upper Columbia Spring-run Chinook; Upper Columbia Steelhead; Middle Columbia Steelhead
	Yakima River Basin	—	WA	170300	Middle Columbia Steelhead
	Lower Snake River Basin	Lower Snake River Basin	ID, OR, WA	170601	Snake River Steelhead; Snake River Spring/Summer-run Chinook; Snake River Fall-run Chinook; Snake River Sockeye
		Salmon River Basin	ID	170602	Snake River Steelhead; Snake River Spring/Summer - Run Chinook; Snake River Fall - Run Chinook; Snake River Sockeye
		Clearwater River Basin	ID, WA	170603	Snake River Steelhead; Snake River Fall - Run Chinook
	Middle Columbia River Basin	Middle Columbia River Basin	OR, WA	170701	Middle Columbia Steelhead; Lower Columbia Chinook; Columbia Chum; Lower Columbia Coho
		John Day River Basin	OR	170702	Middle Columbia Steelhead
		Deschutes River Basin	OR	170703	Middle Columbia Steelhead
	Lower Columbia River Basin	—	OR, WA	170800	Lower Columbia Chinook; Columbia Chum; Lower Columbia Steelhead; Lower Columbia Coho

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

Region	USGS Subregion	Accounting Unit	State	HUC no.	ESU
	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

Table 30. Area of land use categories within the range Chinook and Coho Salmon 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) (National Land Cover Data 2001). Land cover class definitions are available at: http://www.mrlc.gov/nlcd_definitions.php

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

Table 31. 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

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

Select watersheds described herein characterize the past, present, and future human activities and their impacts on the area. The Southwest Coast region encompasses all Pacific Coast rivers south of Cape Blanco, Oregon through southern California. NMFS has identified the Cape Blanco area as an ESU/DPS 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 32).

Table 32. Select rivers in the southwest coast region (Carter and Resh 2005).

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

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

Land Use

Forest and vacant land are the dominant land uses in the northern basins. Grass, shrubland, and urban uses are the dominant land uses in the southern basins (Table 33). 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 (Burton, Izbicki et al. 1998; Belitz, Hamlin et al. 2004). The basin is home to nearly 5 million people. However, this population is projected to increase two-fold in the next 50 years (Burton, Izbicki et al. 1998; Belitz, Hamlin et al. 2004).

Table 33. Land uses and population density in several southwest coast watersheds (Carter and Resh 2005).

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

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 replace septic systems, resulting in point dischages of nutrients and other contaminants not removed in the processing. Additionally, some cities have combined sewer/stormwater overflows (CSOs) and older systems may discharge untreated sewage following heavy rainstorms. Wastewater treatment plant 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. In 1990, nearly 95% of the water diverted from the San Joaquin River was diverted for agriculture. Additionally, 1.5% of the water was diverted for livestock (Carter and Resh 2005). The amount and extent of water withdrawals or diversions for agriculture impact streams and their inhabitants via reduced water flow/velocity and dissolved oxygen levels. For example, adequate water flow is required for migrating salmon along freshwater, estuarine, and marine environments in order to complete their life cycle. Low flow events may delay salmonid migration or lengthen fish presence in a particular water body until favorable flow conditions permit fish migration along the migratory corridor or into the open ocean.

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

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 the presence of specific pollutants, including carbofuran and elevated temperature

(Table 34). See species ESU/DPS maps for NPDES permits and 303(d) waters co-located within listed salmonid ESUs/DPSs in California.

Table 34. California's 2006 Section 303(d) List of Water Quality Limited Segments: segments listed for exceeding temperature and carbofuran limits (CEPA 2007).

Pollutant	Estuary Acres Affected	River / Stream Miles Affected	# Water Bodies
Temperature	-	16,907.2	41
Carbofuran	-	49	1

Estuary systems of the region are consistently exposed to anthropogenic pressures stemming from high human density sources. For example, the largest west coast estuary is the San Francisco Estuary. This water body provides drinking water to 23 million people, irrigates 4.5 million acres of farmland, and drains roughly 40% of California's land area. As a result of high use, many environmental measures of the San Francisco Estuary are poor. Water quality suffers from high phosphorus and nitrogen loads, primarily from agricultural, sewage, and storm water runoff. Water clarity is also compromised. Sediments from urban runoff and historical activities contain high levels of contaminants. They include pesticides,polychlorinated biphenyls (PCBs), nickel, selenium, cadmium, , mercury, copper, and silver. Specific pesticides include pyrethroids, malathion, carbaryl, and diazinon. Other pollutants include DDT and polynuclear aromatic hydrocarbons (PAHs).

Other wastes are also discharged into San Francisco Bay. Approximately 150 industries discharge wastewater into the bay. Discharge of hot water from power plants and industrial sources may elevate temperatures and negatively affect aquatic life. Additionally, about 60 sewage treatment plants discharge treated effluent into the bay and elevate nutrient loads. However, since 1993, many of the point sources of pollution have been greatly reduced. Pollution from oil spills also occur due to refineries in the bay area. As these stressors persist in the marine environment, the estuary system will likely carry loads for future years, even with strict regulation. Gold mining has also reduced estuary depths in much of the region, causing drastic changes to habitat.

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.

Santa Ana Basin: NAWQA assessment

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, Hamlin 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, Hamlin 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. Carbaryl was detected in 42% of urban samples, though it generally did not exceed the standard for protection of aquatic life (Belitz *et al.* 2004). Carbofuran was also detected, but did not exceed any water quality standards. Methomyl was 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 assessment

A study was conducted by the USGS in the mid-1990s on water quality within the San Joaquin-Tulare basins. USGS detected 49 of the 83 pesticides it tested for in the mainstem and three subbasins. Pesticides were detected in all but one of the 143 samples. The most common detections were of the herbicides simazine, dacthal, metolachlor, and EPTC (Eptam), and the insecticides diazinon and chlorpyrifos. Twenty-two pesticides were detected in 20% of the samples (Dubrovsky, Kratzer et al. 1998). Carbaryl and methomyl were detected in all three subbasins, despite land use differences. Carbaryl was detected in roughly 20% of samples from each subbasin, while methomyl detections ranged from 5% to 25%. Further, most samples contained mixtures of between 7 and 22 pesticides. Criteria for the protection of aquatic life were exceeded in 37% of samples of streams (Dubrovsky, Kratzer et al. 1998). Only seven pesticides exceeded their criteria: diuron, trifluralin, azinphos-methyl, carbaryl, chlorpyrifos, diazinon, and malathion. Forty percent of these exceedances were attributed solely to diazinon. However, criteria do not exist yet for over half of the detected compounds (Dubrovsky et al.1998).

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, Kratzer et al. 1998).

Sacramento River Basin: NAWQA analysis

Another study conducted by the USGS from 1996-1998 within the Sacramento River Basin detected up to 24 out of 47 pesticides in surface waters (Domagalski 2000). Pesticides included thiobencarb, carbofuran, molinate, simazine, metolachlor, dacthal, chlorpyrifos, carbaryl, and diazinon. Land use differences between sites are reflected in pesticide detections. Carbofuran was detected in 100% of samples from the agricultural site, but only 6.7% of urban samples (Domagalski 2000). Carbaryl, however, was detected in 100% of urban samples and 42.9% of agricultural samples. Some pesticides were detected at concentrations higher than criteria for the protection of aquatic life in the smaller streams, but were diluted to safer levels in the mainstem river. Intensive agricultural activities also impact water chemistry. In the Salinas River and in areas with intense agriculture use, water hardness, alkalinity, nutrients, and conductivity are also high.

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, 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 some 1,500 abandoned mines. Roughly 1% of these mines are suspected of discharging metal-rich waters into the basins. The Iron Metal Mine in the Sacramento Basin releases more than 1,100 lbs of copper and more than 770 lbs of zinc to the Keswick Reservoir below Shasta Dam. The Iron Metal Mine also released elevated levels of lead (Cain et al. 2000 in Carter and Resh 2005). Metal contamination reduces the biological productivity within a basin. Metal contamination can result in fish kills at high levels or sublethal effects at low levels. Sublethal effects include a reduction in feeding, overall activity levels, and growth. The Sacramento Basin and the San Francisco Bay watershed are two of the most heavily impacted basins within the state from mining activities. The basin drains some of the most productive mineral deposits in the region. Methylmercury 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 et al. 2003).

Hydromodification Projects

Several of the rivers within the area 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. Nine dams occur in this watershed. All major tributaries of the San Joaquin River are impounded at least once and most have multiple dams or diversions. The Stanislaus River, a tributary of the San Joaquin River, has over 40 dams. As a result, the hydrograph of the San Joaquin River is seriously altered from its natural state. Alteration of the temperature and sediment transport regimes had profound influences on the biological community within the basin. These modifications generally result in a reduction of suitable habitat for native species and frequent increases in suitable habitat for nonnative species. The Friant Dam on the San Joaquin River is attributed with the extirpation of spring-run Chinook salmon within the basin. A run of the spring-run Chinook salmon once produced about 300,000 to 500,000 fish (Carter and Resh 2005).

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 (Mokulumne 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.

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 hatchery) 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 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 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. The commercial landings report does not include information on bycatch of listed salmonids (CDFG 2007). The first salmon cannery established along the west coast was located in the Sacramento River watershed in 1864. However, this cannery only operated for about two years because the sediment from hydraulic mining decimated the salmon runs in the basin (NRC 1996).

Alien Species

Plants and animals that are introduced into habitats in which 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 (ANS) Task Force suggests that it is inevitable that cultured species will eventually escape confinement and enter U.S. waterways. Non-native species were cited as a contributing cause in the extinction of 27 species and 13 subspecies of North American fishes over the past 100 years (Miller, Williams et al. 1989). Wilcove, Rothstein et al. (1998) note that 25% of ESA listed fish are threatened by alien 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 alien species are found in California coastal waterways (Foss *et al.* 2007). The majority of these species are representatives of four phyla: annelids (33%), arthropods (22%), chordates (13%), and mollusks (10%). Non-native chordate species are primarily fish and tunicates which inhabit fresh and brackish water habitats such as the Sacramento-San Joaquin Delta (Foss, Ode et al. 2007). The California Aquatic Invasive Species Management Plan (CAISMP) includes goals and strategies for reducing the introduction rate of new invasive species as well as removing those with established populations.

Atmospheric deposition

In 2002, chlopyrifos, diazinon, trifluralin, and other pesticides were detected in air samples collected from Sacramento, California (Majewski and Baston 2002).

Pesticide Reduction Programs

There are several measures in place in California that may reduce the levels of pesticides found in the aquatic environment beyond FIFRA label requirements. Monitoring of water resources is handled by California Environmental Protection Agency's Regional Water Boards. Each Regional Board makes water quality decisions for its region including setting standands 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 et al. 2005). These options included: 1) join a Coalition Group approved by the CVRWQCB, 2) file for an Individual Discharger Conditional Waiver, and 3) comply with zero discharge regulation (Markle, Kalman 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 had to include information on crop patterns and pesticide/nutrient use, as well as mitigation measures that would prevent orchard run-off 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 toxcity to test organisms, stream parameters (*e.g.*, flow, temperature, etc.), nutrient levels, and pesticides used in the region, including diazinon and chlorpyrifos. Sampling diazinon exceedances within the Sacramento and Feather Rivers resulted in the development of a TMDL. The Coalitions were charged with developing and implementing management and monitoring plans to address the TMDL and reduce diazinon run-off.

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 oprganophosphate dormant spray applications. It lists pyrethroids and carbaryl as alternatives, but cautions that these compounds may impact non-target organisms. For example, carbaryl is highly toxic to honeybees, so bees must be removed from the area prior to application

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 organophosphate use, limitations also apply to pyrethroids and carbamates.

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

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

containers at time of use.

- 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 application 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.
- 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.

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. Table 35, Table 36, and Table 37 show the types and areas of land use within each salmonid ESU/DPS.

Table 35. Area of land use categories within Chinook Salmon 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	code	Chinook Salmon					
		Lower Columbia River	Upper Columbia River Spring Run	Puget Sound	Snake River Fall Run	Snake River Spring/ Summer Run	Upper Willamette River
Open Water	11	641	188	6,172	6,172	253	124
Perennial Snow/Ice	12	12	16	313	313	40	7
Developed, Open Space	21	649	203	1,601	1,601	328	632
Developed, Low Intensity	22	517	218	1,694	1,694	113	722
Developed, Medium Intensity	23	290	55	668	668	30	322
Developed, High Intensity	24	118	11	266	266	2	112
Barren Land	31	287	360	1,042	1,042	500	220
Deciduous Forest	41	551	21	999	999	10	248
Evergreen Forest	42	6,497	8,138	14,443	14,443	27,701	9,531
Mixed Forest	43	927	7	2,526	2,526	4	1,130
Shrub/Scrub	52	1,598	6,100	2,415	2,415	13,618	1,940
Herbaceous	71	520	1,737	957	957	11,053	801
Hay/Pasture	81	547	327	1,188	1,188	456	3,617
Cultivated Crops	82	278	636	258	258	3,860	2,355
Woody Wetlands	90	377	92	648	648	96	431
Emergent Herbaceous Wetlands	95	223	59	492	492	92	78
TOTAL (inc. open water)		14,031	18,168	35,683	35,683	58,157	22,269
TOTAL (w/o open water)		13,390	17,981	29,511	29,511	57,904	22,146

Table 36. 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 code		Chum Salmon		Coho Salmon	
		Columbia River	Hood Canal Summer Run	Lower Columbia River	Oregon Coast
Open Water	11	655	704	675	200
Perennial Snow/Ice	12	1	51	12	0
Developed, Open Space	21	605	134	708	1,107
Developed, Low Intensity	22	463	77	563	163
Developed, Medium Intensity	23	258	20	305	49
Developed, High Intensity	24	110	6	124	20
Barren Land	31	247	166	290	467
Deciduous Forest	41	548	97	575	418
Evergreen Forest	42	4,294	2,477	8,487	14,943
Mixed Forest	43	892	200	999	4,126
Shrub/Scrub	52	1,353	299	1,982	3,134
Herbaceous	71	526	133	600	1,478
Hay/Pasture	81	533	64	680	860
Cultivated Crops	82	213	2	348	64
Woody Wetlands	90	363	61	386	263
Emergent Herbaceous Wetlands	95	222	56	225	226
TOTAL (inc. open water)		11,284	4,548	16,959	27,520
TOTAL (w/o open water)		10,628	3,843	16,284	27,320

Table 37. Area of land use categories within sockeye ESUs and steelhead DPSs 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	code	Sockeye Salmon		Steelhead					
		Ozette Lake	Snake River	Lower Columbia River	Middle Columbia River	Puget Sound	Snake River	Upper Columbia River	Upper Willamette River
Open Water	11	30	19	250	575	6,172	285	359	62
Perennial Snow/Ice	12	0	18	12	13	313	42	16	0
Developed, Open Space	21	1	3	518	1,276	1,601	515	343	382
Developed, Low Intensity	22	0	2	506	627	1,694	144	294	513
Developed, Medium Intensity	23	0	0	287	192	668	40	80	231
Developed, High Intensity	24	0	0	116	25	266	3	13	75
Barren Land	31	2	9	174	183	1,042	504	361	77
Deciduous Forest	41	3	0	382	54	999	35	25	171
Evergreen Forest	42	158	755	7,023	18,347	14,443	39,556	8,223	4,133
Mixed Forest	43	3	0	611	41	2,526	17	7	791
Shrub/Scrub	52	14	185	1,589	32,089	2,415	15,644	9,351	994
Herbaceous	71	8	269	398	2,752	957	12,361	1,823	519
Hay/Pasture	81	0	12	605	863	1,188	463	448	2,529
Cultivated Crops	82	0	1	322	11,908	258	6,227	3,236	1,844
Woody Wetlands	90	8	16	244	217	648	116	109	292
Emergent Herbaceous Wetlands	95	1	34	93	291	492	111	81	43
TOTAL (inc. open water)		228	1,323	13,128	69,453	35,683	76,061	24,771	12,655
TOTAL (w/o open water)		199	1,304	12,878	68,878	29,511	75,777	24,411	12,593

Columbia River Basin

The most notable basin within the 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 38 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 (Kammerer 1990; Hinck, Schmitt et al. 2004). 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 38. Select tributaries of the Columbia River (Carter and Resh 2005)

Watershed	Approx Length (mi)	Basin Size (mi^2)	Physiographic Provinces*	Mean Annual Precipitation (in)	Mean Discharge (cfs)	No. Fish Species (native)	No. Endangered Species
Snake/Salmon rivers	870	108,495	CU, NR, MR, B/R	14	55,267	39 (19)	5 fish (4 T, 1 E), 6 (1 T, 5 E) snails, 1 plant (T)
Yakima River	214	6,139	CS, CU	7	3,602	50	2 fish (T)
Willamette River	143	11,478	CS, PB	60	32,384	61 (~31)	5 fish (4 T, 1 E),

* Physiographic Provinces: CU = Columbia-Snake River Plateaus, NR = Northern Rocky Mountains, MR = Middle Rocky Mountains, B/R = Basin & Range, CS = Cascade-Sierra Mountains, PB = Pacific Border

The Columbia river and estuary were once home to more than 200 distinct runs of Pacific salmon and steelhead with unique adaptations to local environments within a tributary (Stanford, Hauer et al. 2005). Salmonids within the basin include Chinook salmon, chum salmon, coho salmon, sockeye salmon, steelhead, redband trout, bull trout, and cutthroat trout.

Land Use

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, Schmitt et al. 2004). See Table 39 for a summary of land uses and population densities in several subbasins within the Columbia River watershed (data from (Stanford, Hauer et al. 2005).

Table 39. Land use and population density in select tributaries of the Columbia River (Stanford, Hauer 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 Department of Ecology (DOE) and 83% exceed temperature standards. 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. In the Yakima River, non-native grasses and other plants are commonly found along the lower reaches of the river (Stanford, Hauer et al. 2005).

Agriculture and Ranching

Agriculture, ranching, 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.

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, Schmitt et al. 2004).

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%).

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, croptype, and basin location are some of the variables that influence the distribution and frequency of pesticides within a tributary. Detection frequencies for a particular pesticide can vary widely. One study conducted by the USGS between May 1999 and January 2000 in the surface waters of Yakima Basin detected 25 pesticide compounds (Ebbert and Embry 2002). Atrazine was the most widely detected herbicide and azinphos-methyl was the most widely detected insecticide. Other detected compounds include simazine, terbacil, trifluralin;

deethylatrazine, carbaryl, diazinon, malathion, and DDE. In addition to current use-chemicals legacy chemicals continue to pose a serious problem to water quality and fish communities despite their ban in the 1970s and 1980s (Hinck, Schmitt et al. 2004).

Fish and macroinvertebrate communities exhibit an almost linear decline in condition as the level of agriculture intensity increases within a basin (Cuffney, Meador et al. 1997; Fuhrer, Morace 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, Morace 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, Morace 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 during the nonirrigation season when ground water is the primary source of streamflow.

The USGS also detected 76 pesticide compounds in the Yakima River Basin. They include 38 herbicides (including metribuzin), 17 insecticides (such as carbaryl, diazinon, and malathion), 15 breakdown products, and 6 others. Ninety-one percent of the samples collected from the small agricultural watersheds contained at least two pesticides or pesticide breakdown products. Carbaryl was detected in 29% of tributary samples and

17% of mainstem Yakima River samples at a screening level of 21 nanograms/liter (Fuhrer *et al.* 2004). Carbofuran was screened for, but not detected. The assessment did not screen for methomyl. The median and maximum number of chemicals in a mixture was 8 and 26, respectively (Fuhrer, Morace 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, Morace et al. 2004). However, the most frequently detected pesticides in the Yakima River Basin are total DDTs ,(DDT and its breakdown products, dichloro-diphenyl-dichloroethylene (DDE), dichloro-diphenyl-dichloroethane (DDD)), and dieldrin (Johnson and Newman 1983; Joy 2002; Fuhrer, Morace et al. 2004). Nevertheless, concentrations of total DDT in water have decreased since 1991. These reductions are attributed to erosion-controlling best management practices (BMPs).

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, of the 86 tested for, 50 pesticides and pesticide degradates were detected in streams. Ten of the pesticides exceeded criteria established by the EPA for the protection of freshwater aquatic life from chronic toxicity. Carbaryl exceeded protective criteria in 17 of its 46 detections, while carbofuran exceeded limits in three of 51 detections (Wentz *et al.* 1998). Atrazine, simazine, metolachlor, deethylatrazine, diuron, and diazinon were detected in more than one-half of stream samples. Methomyl was tested for but not detected. Forty-nine pesticides were detected in streams draining predominantly agricultural land. About 25 pesticides were detected in streams draining mostly urban areas. The highest pesticide concentrations generally occurred in streams draining predominately agricultural land.

Snake River Basin: NAWQA assessment

The USGS conducted a water quality study from 1992-1995 in the upper Snake River basin, Idaho and Wyoming (Clark, Maret et al. 1998). In basin wide stream sampling in May and June 1994, Eptam [EPTC] (used on potatoes, beans, and sugar beets), atrazine and its breakdown product desethylatrazine (used on corn), metolachlor (used on potatoes and beans), and alachlor (used on beans and corn) were the most commonly detected pesticides. These same compounds accounted for 75% of all detections. Seventeen different pesticides were detected downstream from American Falls Reservoir. Carbaryl and carbofuran were each detected in only 1% of samples; methomyl was screened for but not detected (Clark, Maret et al. 1998).

Hood River Basin

The Hood River Basin ranks fourth in the state of Oregon in total agricultural pesticide usage (Jenkins, Jepson et al. 2004). The land in Hood River basin is used to grow five crops: alfalfa, apples, cherries, grapes, and pears. About 61 a.i.s, totaling 1.1 million lbs, are applied annually to roughly 21,000 acres. Of the top nine, three are carbamates and three are organophosphate insecticides (Table 40). These compounds will have a similar mode of action, though different toxicities, as carbaryl, carbofuran, and methomyl.

Table 40. Amount of most common a.i.s applied to crops in Hood River Basin 1990-1996 (Jenkins et al. 2004).

Active Ingredient	Class	Lbs applied
Oil	-	624,392
Lime Sulfur	-	121,703
Mancozeb	Carbamate	86,872
Sulfur	-	60,552
Ziram	Carbamate	45,965
Azinphos-methyl	Organo-phosphate	22,294
Metam-Sodium	Carbamate	17,114
Phosmet	Organo-phosphate	15,919
Chlorpyrifos	Organo-phosphate	14,833

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

Central Columbia Plateau: NAWQA Assessment

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, Munn 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). Carbaryl and carbofuran were detected in 6% and 5% of samples, respectively. Methomyl was screened for, but not detected in any samples (Williamson *et al.* 1998).

Detections at four sites were high, ranging from 12 to 45 pesticides. The two sites with the highest detection frequencies are in the Quincy-Pasco subunit, where irrigation and high chemical use combine to increase transport of pesticides to surface waters. Pesticide detection frequencies at sites in the dryland farming (non-irrigated) areas of the North-Central and Palouse subunits are below the national median for NAWQA sites. All four of the 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 as well as in irrigated areas.

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, Schmitt 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, Schmitt et al. 2004).

Discharges from sewage treatment plants, paper manufacturing, and chemical and metal production represent the top three permitted sources of contaminants within the lower basin according to discharge volumes and concentrations (Rosetta and Borys 1996). Rosetta and Borys (1996) review of 1993 data indicate that 52% of the point source waste water discharge volume is from sewage treatment plants, 39% from paper and allied products, 5% from chemical and allied products, and 3% from primary metals. However, the paper and allied products industry are the primary sources of the suspended sediment load (71%). Additionally, 26% 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 load. 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

Basin wide, critical ecological connectivity (mainstem to tributaries and riparian floodplains) has been disconnected by dams and associated activities such as floodplain deforestation and urbanization. Dams have flooded historical spawning and rearing habitat with the creation of massive water storage reservoirs. More than 55% of the Columbia River Basin that was accessible to salmon and steelhead before 1939 has been blocked by large dams (NWPPC 1986). Construction of the Grand Coulee Dam blocked 1,000 miles (1,609 km) of habitat from migrating salmon and steelhead (Wydoski and

Whitney 1979). Similarly, over one third (2,000 km) of coho salmon habitat is no longer accessible (Good, Waples 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.

Stream habitat degradation in Columbia Central Plateau is relatively high (Williamson, Munn et al. 1998). In the most recent NAWQA survey, a total of 16 sites were evaluated - all of which showed signs of degradation (Williamson, Munn 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, Munn et al. 1998).

Habitat loss has fragmented habitat and human density increase has created additional loads of pollutants and contaminants within the Columbia River Estuary (Anderson, Dugger et al. 2007). About 77% of swamps, 57% of marshes, and over 20% of tree cover have been lost to development and industry. Twenty four threatened and endangered species occur in the estuary, some of which are recovering and others (*i.e.*, Chinook salmon) are not.

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 has been converted to agricultural purposes. These actions, combined with urban development, extensive (96 miles) bank stabilization, and in-river and near-shore 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 subyearling 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 2006, NOAA Fisheries completed consultation on issuance of a 50-year incidental take permit to the State of Washington for its Washington State Forest Practices Habitat Conservation Plan (HCP). The HCP is expected to improve habitat conditions on state forest lands within the action area. Improvements include removing barriers to migration, restoring hydrologic processes, increasing the number of large trees in riparian zones, improving stream bank integrity, and reducing fine sediment inputs (FCRPS 2008).

Mining

Most of the mining in the basin is focused on minerals such as phosphate, limestone, dolomite, perlite, or metals such as gold, silver, copper, iron, and zinc. Mining in the region is conducted in a variety of methods and places within the basin. Alluvial or glacial deposits are often mined for gold or aggregate. Ores are often excavated from the hard bedrocks of the Idaho batholiths. Eleven percent of the nation's output of gold has come from mining operations in Washington, Montana, and Idaho. More than half of the nation's silver output has come from a few select silver deposits.

Many of the streams and river reaches in the basin are impaired from mining. Several abandoned and former mining sites are also designated as superfund cleanup areas (Stanford, Hauer et al. 2005; Anderson, Dugger et al. 2007). According to the U.S. Bureau of Mines, there are about 14,000 inactive or abandoned mines within the Columbia River Basin. Of these, nearly 200 pose a potential hazard to the environment (Quigley, Arbelbide et al. 1997 in Hincke et al. 2004). Contaminants detected in the water include lead and other trace metals.

Hydromodification Projects

More than 400 dams exist in the basin, ranging from mega dams that store large amounts of water to small diversion dams for irrigation. Every major tributary of the Columbia River except the Salmon River is totally or partially regulated by dams and diversions. More than 150 dams are major hydroelectric projects. Of these, 18 dams are located on the mainstem Columbia River and its major tributary, the Snake River. The FCRPS encompasses the operations of 14 major dams and reservoirs on the Columbia and Snake rivers. These dams and reservoirs operate as a coordinated system. The Corps operates 9 of 10 major federal projects on the Columbia and Snake rivers, and the Dworshak, Libby and Albeni Falls dams. The BOR operates the Grand Coulee and Hungry Horse dams. These federal projects are a major source of power in the region. These same projects provide flood control, navigation, recreation, fish and wildlife, municipal and industrial water supply, and irrigation benefits.

BOR has operated irrigation projects within the basin since 1904. The irrigation system delivers water to about 2.9 million acres of agricultural lands. About 1.1 million acres of land are irrigated using water delivered by two structures, the Columbia River Project (Grand Coulee Dam) and the Yakima Project. The Grand Coulee Dam delivers water for the irrigation of over 670,000 acres of croplands and the Yakima Project delivers water to nearly 500,000 acres of croplands (Bouldin, Farris et al. 2007).

The Bonneville Power Administration (BPA), an agency of the U.S. Department of Energy, wholesales electric power produced at 31 federal dams (67% of its production) and non-hydropower facilities in the Columbia-Snake Basin. The BPA sells about half the electric power consumed in the Pacific Northwest. The federal dams were developed over a 37-year period starting in 1938 with Bonneville Dam and Grand Coulee in 1941, and ending with construction of Libby Dam in 1973 and Lower Granite Dam in 1975.

Development of the Pacific Northwest regional hydroelectric power system, dating to the early 20th century, has had profound effects on the ecosystems of the Columbia River Basin (ISG 1996). These effects have been especially adverse to the survival of anadromous salmonids. The construction of the FCRPS modified migratory habitat of adult and juvenile salmonids. In many cases, the FCRPS presented a complete barrier to habitat access for salmonids. Approximately 80% of historical spawning and rearing habitat of Snake River fall-run Chinook salmon is now inaccessible due to dams. The Snake River spring/summer run has been limited to the Salmon, Grande Ronde, Imnaha, and Tuscanon rivers. Damming has cut off access to the majority of Snake River Chinook salmon spawning habitat. The Sunbeam Dam on the Salmon River is believed to have limited the range of Snake River sockeye salmon as well.

Both upstream and downstream migrating fish are impeded by the dams. Additionally, a substantial number of juvenile salmonids are killed and injured during downstream migrations. Physical injury and direct mortality occurs as juveniles pass through turbines, bypasses, and spillways. Indirect effects of passage through all routes may include disorientation, stress, delays 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 Snake River fall Chinook salmon. Improvements include flow augmentation to enhance water flows through the lower Snake and Columbia Rivers

(USBR 1998 *in* (FCRPS 2008); 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 et al. 2007), Appendix 1 *in* (FCRPS 2008)). 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 Snake River 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, BPA 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.

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. The hatcheries are operated by federal, state, and tribal managers. 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 population (*e.g.*, Redfish Lake sockeye salmon). In 1987, 95% of the coho salmon, 70% of the spring Chinook salmon, 80% of the summer Chinook salmon, 50% of the fall-run Chinook salmon, and 70% of the steelhead returning to the Columbia River Basin originated in hatcheries (CBFWA 1990). More recent estimates suggest that almost half of the total number of smolts produced in the basin come from hatcheries (Beechie, Liermann et al. 2005).

The impact of artificial propagation on the total production of Pacific salmon and steelhead has been extensive (Hard, Jones 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 et al. 1995). Past hatchery and stocking practices have resulted in the transplantation of salmon and steelhead from non-native basins. The impacts of these hatchery practices are largely unknown. Adverse effects of these practices likely included: loss of genetic variability within and among populations (Busack 1990; Riggs 1990; Hard, Jones et al. 1992; Reisenbichler 1997), disease transfer, increased competition for food, habitat, or mates, increased predation, altered migration, and the displacement of natural fish (Steward and Bjornn 1990; Hard, Jones et al. 1992; Fresh 1997). Species with extended freshwater residence may face higher risk of domestication, predation, or altered migration than species that spend only a brief time in freshwater (Hard, Jones 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, Jones et al. 1992).

The States of Oregon and Wasington 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 LCR populations and provided its recommendations. A general conclusion is that the current production programs are inconsistent with practices that reduce impacts on naturally-spawning populations, and will have to be modified to reduce adverse effects on key natural populations identified in the Interim Recovery Plan. The adverse effects are caused by hatchery-origin adults spawning with natural-origin fish or competing with natural-origin fish for spawning sites (FCRPS 2008). Oregon and Washington initiated a comprehensive program of hatchery and associated harvest reforms (WDFW 2005; ODFW 2007). The program is designed to achieve HSRG objectives related to controlling the number of hatchery-origin fish on the spawning grounds and in the hatchery broodstock.

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

Commercial, Recreational, and Subsistence Fishing

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

During the mid-1800s, an estimated 10 to 16 million adult salmon of all species entered the Columbia River each year. Large annual harvests of returning adult salmon during the late 1800s ranging from 20 million to 40 million lbs of salmon and steelhead significantly reduced population productivity (Beechie, Liermann et al. 2005). The largest known harvest of Chinook salmon occurred in 1883 when Columbia River canneries processed 43 million lbs of salmon (Lichatowich 1999). Commercial landings declined steadily from the 1920s to a low in 1993. At that time, just over one million lbs of Chinook salmon were harvested (Beechie, Liermann et al. 2005).

Harvested and spawning adults reached 2.8 million in the early 2000s, of which almost half are hatchery produced (Beechie, Liermann et al. 2005). Most of the fish caught in the river are steelhead and spring/summer Chinook salmon. Ocean harvest consists largely of coho and fall Chinook salmon. Most ocean catches are made north of Cape Falcon, Oregon. Over the past five years, the number of spring and fall salmon commercially harvested in tribal fisheries has averaged between 25,000 and 110,000 fish (Beechie, Liermann et al. 2005). Recreational catch in both ocean and in-river fisheries varies from 140,000 to 150,000 individuals (Beechie, Liermann 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* FCRPS (2008).

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. Columbia River 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 Columbia River 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 (FCRPS 2008).

Alien Species

Many non-native species have been introduced to the Columbia River Basin since the 1880s. At least 81 invasive 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 et al. 2004). It is clear that the introduction of non-native species has changed the environment, though whether these changes will impact salmonid populations is uncertain (Sytsma, Cordell 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 (Wydoski and Whitney 1979; Kruckeberg 1991). Important commercial fishes include the five Pacific salmon and several rockfish species. A number of introduced species occur within the region, including brown and brook trout, Atlantic salmon, bass, tunicates (sea squirts), and a saltmarsh grass (*Spartina* spp.). Estimates suggest that over 90 species have been intentionally or accidentally introduced in the region (Ruckelshaus and McClure 2007). At present, over 40 species in the region are listed as threatened and endangered under the ESA.

Puget Sound is unique among the nation's estuaries as it is a deep fjord-like structure that contains many urban areas within its drainage basin (Collier, O'Neill et al. 2006). Because of the several sills that 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, O'Neill et al. 2006).

An indication of this sensitivity occurs in Pacific herring, one of Puget Sound's keystone forage fish species (Collier, O'Neill 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, O'Neill 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, O'Neill et al. 2006). In addition to carrying elevated loads of toxic chemicals in their tissues, Puget Sound's biota also show a wide range of adverse health outcomes associated with exposure to chemical contaminants. They include widespread cancer and reproductive impairment in bottom fish, increased susceptibility to disease in juvenile salmon, acute die-offs of adult salmon returning to spawn in urban watersheds, and egg and larval mortality in a variety of fish. Given current regional projections for population growth and coastal development, the loadings of chemical contaminants into Puget Sound will increase dramatically in future years.

Land Use

The Puget Sound Lowland contains the most densely populated area of Washington. The regional population in 2003 was an estimated 3.8 million people, with 86% residing in King, Pierce, and Snohomish counties (Snohomish, Cedar-Sammamish Basin, Green-Duwamish, and Puyallup River watersheds). The area is expected to attract 4 to 6 million new human residents in the next 20 years (Ruckelshaus and McClure 2007). The Snohomish River watershed, one of the fastest growing watersheds in the region, increased about 16% in the same period.

Land use in the Puget Sound lowland is composed of agricultural areas (including forests for timber production), urban areas (industrial and residential use), and rural areas (low density residential with some agricultural activity). Pesticides are regularly applied to agricultural and non-agricultural lands and are found virtually in every land use area. Pesticides and other contaminants drain into ditches in agricultural areas and eventually to stream systems. Roads bring surface water runoff to stream systems from industrial, residential, and landscaped areas in the urban environment. Pesticides are also typically found in the right-of-ways of infrastructure that connect the major landscape types. Right-of-ways are associated with roads, railways, utility lines, and pipelines.

In the 1930s, all of western Washington contained about 15.5 million acres of "harvestable" forestland. By 2004, the total acreage was nearly half that originally surveyed (PSAT 2007). Forest cover in Puget Sound alone was about 5.4 million acres in the early 1990s. About a decade later, the region had lost another 200,000 acres of forest cover with some watersheds losing more than half the total forested acreage. The most intensive loss of forest cover occurred in the Urban Growth Boundary, which encompasses specific parts of the Puget Lowland. In this area, forest cover declined by 11% between 1991 and 1999 (Ruckelshaus and McClure 2007). Projected land cover changes indicate that trends are likely to continue over the next several decades with population changes (Ruckelshaus and McClure 2007). Coniferous forests are also projected to decline at an alarming rate as urban uses increase.

According to the 2001 State of the Sound report (PSAT 2007), impervious surfaces covered 3.3% of the region, with 7.3% of lowland areas (below 1,000 ft elevation) covered by impervious surfaces. From 1991 to 2001, the amount of impervious surfaces increased 10.4% region wide. Consequently, changes in rainfall delivery to streams alter stream flow regimes. Peak flows are increased and subsequent base flows are decreased and alter in-stream habitat. Stream channels are widened and deepened and riparian vegetation is typically removed which can cause increases in water temperature and will reduce the amounts of woody debris and organic matter to the stream system.

Pollutants carried into streams from urban runoff include pesticides, heavy metals, PCBs, polybrominated diphenyl ethers (PBDEs) compounds, PAHs, nutrients (phosphorus and nitrogen), and sediment (Table 41). Other ions generally elevated in urban streams include calcium, sodium, potassium, magnesium, and chloride ions where sodium chloride is used as the principal road deicing salt (Paul and Meyer 2001). The combined effect of increased concentrations of ions in streams is the elevated conductivity observed in most urban streams.

Table 41. 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 et al. 2005). The concentration, storage, and transport of metals in urban streams are connected to particulate organic matter content and sediment characteristics. Organic matter has a high binding capacity for metals and both bed and suspended sediments with high organic matter content frequently exhibit 50-7,500 times higher concentrations of zinc, lead, chromium, copper, mercury, and cadmium than sediments with lower organic matter content.

Although urban areas occupy only 2% of the Pacific Northwest land base, the impacts of urbanization on aquatic ecosystems are severe and long lasting (Spence, Lomnicky 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 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 (EDCs) 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 2002d). 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, Angermeier 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, Higgins 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 et al. 2004). PAHs tend to adsorb to organic or inorganic matter in sediments, where they can be trapped in long-term reservoirs (Johnson, Collier et al. 2002). Only a portion of sediment-adsorbed PAHs are readily bioavailable to marine organisms, but there is substantial uptake of these compounds by resident benthic fish through the diet, through exposure to contaminated water in the benthic boundary layer, and through direct contact with sediment. Benthic invertebrate prey are a particularly important source of PAH exposure for marine fishes, as PAHs are bioaccumulated in many invertebrate species (Varanasi, Stein et al. 1989; Varanasi, Stein et al. 1992; Meador, Stein et al. 1995).

PAHs and their metabolites in invertebrate prey can be passed on to consuming fish species, PAHs are metabolized extensively in vertebrates, including fishes (Johnson, Collier et al. 2002). Although PAHs do not bioaccumulate in vertebrate tissues, PAHs cause a variety of deleterious effects in exposed animals. Some PAHs are known to be immunotoxic and to have adverse effects on reproduction and development. Studies show that PAHs exhibit many of the same toxic effects in fish as they do in mammals (Johnson, Collier et al. 2002).

Habitat Modification

Much of the region's estuarine wetlands 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 et al. 1980). Tidal wetlands in Puget Sound amount to roughly 18% of their historical extent (Collins and Sheikh 2005). Coastal marshes close to seaports and population centers have been especially vulnerable to conversion with losses of 50-90%. By 1980, an estimated 27,180 acres of intertidal or shore wetlands had been lost at eleven deltas in Puget Sound (Bortleson, Chrzastowski et al. 1980). More recently, tidal wetlands in Puget Sound amount to about 17-19% of their historical extent (Collins and Sheikh 2005). Coastal marshes close to seaports and population centers have been especially vulnerable to conversion with losses of 50-90% common for individual estuaries. Salmon use freshwater and estuarine wetlands for physiological transition to and from saltwater 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, Higgins et al. 2004).

About 800 miles of Puget Sound's shorelines are hardened or dredged (PSAT 2004; Ruckelshaus and McClure 2007). The area most intensely modified is the urban corridor (eastern shores of Puget Sound from Mukilteo to Tacoma). Here, nearly 80% of the shoreline has been altered, mostly from shoreline armoring associated with the Burlington Northern Railroad tracks (Ruckelshaus and McClure 2007). Levee development within the rivers and their deltas has isolated significant portions of former floodplain habitat that was historically used by salmon and trout during rising flood waters.

Urbanization has caused direct loss of riparian vegetation and soils and significantly altered hydrologic and erosion rates. Watershed development and associated urbanization throughout the Puget Sound, Hood Canal, and Strait of Juan de Fuca regions have increased sedimentation, raised water temperatures, decreased LWD recruitment, decreased gravel recruitment, reduced river pools and spawning areas, and dredged and filled estuarine rearing areas (Bishop and Morgan 1996 in (NMFS 2008)). Large areas of the lower rivers have been channelized and diked for flood control and to protect agricultural, industrial, and residential development.

The NMFS' 2005 Report to Congress on implementation of the Pacific Coastal Salmon Recovery Fund listed habitat-related factors as the leading limits to Puget Sound Chinook Salmon and Hood-Canal Summer Run Chum recovery (PCSRF 2006). Similarly, the principal factor for decline of Puget Sound steelhead is the destruction, modification, and curtailment of its habitat and range. Barriers to fish passage and adverse effects on water quality and quantity resulting from dams, the loss of wetland and riparian habitats, and agricultural and urban development activities have contributed and continue to contribute to the loss and degradation of steelhead habitats in Puget Sound (NMFS 2008).

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 42.

Table 42. 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, Embrey et al. 2000) reported that 26 of 47 analyzed pesticides were detected. A total of 74 manmade organic chemicals were detected in streams and rivers, with different mixtures of chemicals linked to agricultural and urban settings. NAWQA results reported that the herbicides atrazine, prometon, simazine and tebuthiuron were the most frequently detected herbicides in surface and ground water (Bortleson and Ebbert 2000). Herbicides were the most common type of pesticide found in an agricultural stream (Fishtrap Creek) and the only type of pesticide found in shallow ground water underlying agricultural land (Bortleson and Ebbert 2000). The most commonly detected VOC in the agricultural land use study area was associated with the application of fumigants to soils prior to planting (Bortleson and Ebbert 2000). One or more fumigant-related compound (1,2-dichloropropane, 1,2,2-trichloropropane, and 1,2,3-trichloropropane) were detected in over half of the samples. Insecticides, in addition to herbicides, were detected frequently in urban streams (Bortleson and Ebbert 2000). Sampled urban streams showed the highest detection rate for the three insecticides: carbaryl, diazinon, and malathion. Carbaryl was detected at over 60% of urban sample sites (Ebbert, Embrey et al. 2000). The insecticide diazinon was also frequently detected in urban streams at concentrations that exceeded EPA guidelines for protecting aquatic life (Bortleson and Ebbert 2000). Insecticides screened for included both carbofuran and methomyl. Carbofuran was detected, while methomyl was not. No insecticides were found in shallow ground water below urban residential land (Bortleson and Ebbert 2000).

Habitat Restoration

Positive changes in water quality in the region are evident. One of the most notable improvements was the elimination of sewage effluent to Lake Washington in the mid-1960s. This significantly reduced problems within the lake from phosphorus pollution and triggered a concomitant reduction in cyanobacteria (Ruckelshaus and McClure 2007). Even so, as the population and industry has risen in the region a number of new and legacy pollutants are of concern.

Mining

Mining has a long history in Washington. In 2004, the state was ranked 13th nationally in total nonfuel mineral production value and 17th in coal production (Palmisano, Ellis et al. 1993; NMA 2007). Metal mining for all metals (zinc, copper, lead, silver, and gold) peaked between 1940 and 1970 (Palmisano, Ellis et al. 1993). Today, construction sand and gravel, Portland cement, and crushed stone are the predominant materials mined. Where sand and gravel is mined from riverbeds (gravel bars and floodplains) it may result in changes in channel elevations and patterns, instream sediment loads, and seriously alter instream habitat. In some cases, instream or floodplain mining has resulted in large scale river avulsions. The effect of mining in a stream or reach depends upon the rate of harvest and the natural rate of replenishment, as well as flood and precipitation conditions during or after the mining operations.

Artificial Propagation

The artificial propagation of late-returning Chinook salmon is widespread throughout Puget Sound (Good, Waples et al. 2005). Summer/fall Chinook salmon transfers between watersheds within and outside the region have been commonplace throughout this century. Therefore, the purity of naturally spawning stocks varies from river to river. Nearly 2 billion Chinook salmon have been released into Puget Sound tributaries since the 1950s. The vast majority of these have been derived from local late-returning adults.

Returns to hatcheries have accounted for 57% of the total spawning escapement. However, the hatchery contribution to spawner escapement is probably much higher than that due to hatchery-derived strays on the spawning grounds. The genetic similarity between Green River late-returning Chinook salmon and several other late-returning Chinook salmon in Puget Sound suggests that there may have been a significant and lasting effect from some hatchery transplants (Marshall, Smith et al. 1995).

Overall, the use of Green River stock throughout much of the extensive hatchery network in this ESU may reduce the genetic diversity and fitness of naturally spawning populations (Good, Waples 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 hydromodified by other means including levees and revetments, bank hardening for erosion control, and agriculture uses. Since the first dike on the Skagit River delta was built in 1863 for agricultural development (Ruckelshaus and McClure 2007), other basins like the Snohomish River are diked and have active drainage systems to drain water after high flows that top the dikes. Dams were also built on the Cedar, Nisqually, White, Elwha, Skokomish, Skagit, and several other rivers in the early 1900s to supply urban areas with water, prevent downstream flooding, allow for floodplain activities (like agriculture or development), and to power local timber mills (Ruckelshaus and McClure 2007).

Over the next few years, however, a highly publicized and long discussed dam removal project is expected to begin in the Elwha River. The removal of two dams in the Elwha River, a short but formerly very productive salmon river, is expected to open up more than 70 miles of high quality salmon habitat (Wunderlich, Winter et al. 1994; Ruckelshaus and McClure 2007). 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

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.

Pesticides are used in commercial oyster-producing areas in Willapa Bay and Grays Harbor. Currently, under a statewide SLN or Section 24 (c) registration, carbaryl can be applied to aerially applied to intertidal areas when exposed at low tide. Pesticide use is intended to control ghost shrimp (*Neotrypaea californiensis)* and mud shrimp *(Upogebia* *pugettensis).* An NPDES permit is required for application, and although the registration allows application anywhere in the state, these locations are the only ones covered under active permits. There has been some discussion of phasing out carbaryl use and replacing it with alternative pesticides.

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

Atmospheric deposition

Pesticides were detected in wet deposition (rain) (Capel, Ma et al. 1998), and snow samples from Mount Rainier National Park, Washington (Hageman, Simonich et al. 2006). Three of the four most frequently detected pesticides were found in the Mount Rainier snow (dacthal, chlorpyrifos, and endosulfan).

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 (Kagan, Hak et al. 1999; Belitz, Hamlin et al. 2004; Carter and Resh 2005).

Land Use

The rugged topography of the western Olympic Peninsula and the Oregon Coastal Range has limited the development of dense population centers. For instance, the Nehalem River and the Umpqua River basins consist of less than 1% urban land uses. Most basins in this region have long been exploited for timber production, and are still dominated by forest lands. In Washington State, roughly 90% of the coastal region is forested (Palmisano, Ellis et al. 1993). Roughly 80% of the Oregon Coastal Range is forested as well (Gregory 2000). Approximately 92% of the Nehalem River basin is forested, with only 4% considered agricultural (Belitz, Hamlin et al. 2004). Similarly, in the Umpqua River basin, about 86% is forested land, 5% agriculture, and 0.5% is considered urban lands. Roughly half the basin is under federal management (Carter and Resh 2005).

Habitat Modification

While much of the coastal region is forested, it has still been impacted by land use practices. Less than 3% of the Oregon coastal forest is old growth conifers (Gregory 2000). The lack of mature conifers indicates high levels of habitat modification. As such, overall salmonid habitat quality is poor, though it varies by watershed. The amount of remaining high quality habitat ranges from 0% in the Sixes to 74% in the Siltcoos (ODFW 2005). Approximately 14% of freshwater winter habitat available to juvenile coho is of high quality. Much of the winter habitat is unsuitable due to high temperatures. For example, 77% of coho salmon habitat in the Umpqua basin exceeds temperature standards.

Reduction in stream complexity is the most significant limiting factor in the Oregon coastal region. An analysis of the Oregon coastal range determined the primary and secondary lifecycle bottlenecks for the 21 populations of coastal coho salmon (Nicholas, McIntosh et al. 2005). Nicholas et al. (2005) determined that stream complexity is either the primary (13) or secondary (7) bottleneck for every population. Stream complexity has been reduced through past practices such as splash damming, removing riparian vegetation, removing LWD, diking tidelands, filling floodplains, and channelizing rivers.

Habitat loss through wetland fills is also a significant factor. Table 43 summarizes the change in area of tidal wetlands for several Oregon estuaries (Good 2000).

Table 43. Change in total area (acres^2) of tidal wetlands (tidal marshes and swamps) due to filling and diking between 1870 and 1970 (Good 2000).

Estuary	Diked or Filled Tidal Wetland	Percent of 1870 Habitat Lost
Necanicum	15	10
Nehalem	1,571	75
Tillamook	3,274	79
Netarts	16	7
Sand Lake	9	2
Nestucca	2,160	91
Salmon	313	57
Siletz	401	59
Yaquina	1,493	71
Alsea	665	59
Siuslaw	1,256	63
Umpqua	1,218	50
Coos Bay	3,360	66
Coquille	4,600	94
Rogue	30	41
Chetco	5	56
Total	20,386	72%

The only listed salmonid population in coastal Washington is the Ozette Lake Sockeye. The range of this ESU is small, including only one lake (31 km^2) and 71 km of stream. Like the Oregon Coastal drainages, the Ozette Lake area has been heavily managed for logging. Logging resulted in road building and the removal of LWD, which affected the nearshore ecosystem (NMFS 2008). LWD along the shore offered both shelter from predators and a barrier to encroaching vegetation (NMFS 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 2008)). The impact of this water level drop is unknown. Possible effects include increased desiccation of sockeye redds and loss of spawning habitat. Loss of LWD has also contributed to an increase in silt deposition, which impairs the quality and quantity of spawning habitat.

Very little is known about the relative health of the Ozette Lake tributaries and their impact on the sockeye salmon population.

Mining

Oregon is ranked 35th nationally in total nonfuel mineral production value in 2004. In that same year, Washington was ranked 13th nationally in total nonfuel mineral production value and 17th in coal production (Palmisano, Ellis et al. 1993; NMA 2007). Metal mining for all metals (*e.g.*, zinc, copper, lead, silver, and gold) peaked in Washington between 1940 and 1970 (Palmisano, Ellis 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, instream sediment loads, may result and alter instream habitat. In some cases, instream or floodplain mining has resulted in large scale river avulsions. The effect of mining in a stream or reach depends upon the rate of harvest and the natural rate of replenishment. Additionally, the severity of the effects is influenced by flood and precipitation conditions during or after the mining operations.

Hydromodification Projects

Compared to other areas in the greater Northwest Region, the coastal region has fewer dams and several rivers remain free flowing (*e.g.*, Clearwater River). The Umpqua River is fragmented by 64 dams, the fewest number of dams on any large river basin in Oregon (Carter and Resh 2005). According to Palmisano *et al.* (1993) dams in the coastal streams of Washington permanently block only about 30 miles of salmon habitat. 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

Most commercial landings in the region are groundfish, Dungeness crab, shrimp, and salmon. Many of the same species are sought by Tribal fisheries, as well as by charter, and recreational anglers. Nets and trolling are used in commercial and Tribal fisheries. Recreational anglers typically use hook and line and may fish from boat, river bank, or docks.

Atmospheric deposition

Pesticides and other chemicals may be transported through the air and later deposited on land and into waterways. For example, orthophosphate insecticides were detected in two Oregon streams, Hood River and Mill Creek (tributaries of the Columbia River). Detection occurred following periods of chemical applications on orchard crops, and may be related to atmospheric drift, mixing operations, or other aspects of pesticide use.

Environmental Protection Programs

When using carbaryl, carbofuran, and methomyl, 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 and Oregon both have Pesticide Use Reporting System (PURS) legislation. California PURS 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 suggested suspending the PURS program for 2009 – 2011 due to budget shortages. A final decision will be made during the summer. If suspension occurs, PURS will resume for the 2012 growing season.

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.

Oregon has also implemented a voluntary program. The Pesticide Stewardship Partnerships (PSP) program began in 1999 through the Oregon Department of Environmental Quality. Like the CIDMP program, the 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.

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. However, their website does not mention carbamate pesticides. The mid-Columbia area is of particular concern, as many orchards are in close proximity to streams.

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.

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 affected by the proposed registration of carbaryl, carbofuran, and methomyl 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

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 predicted on effects to salmonids and designated critical habitats' PCEs. The combined analysis evaluated effects to listed Pacific salmonids and their designated critical habitat (see *Approach to the Assessment)*.

Exposure Analysis

In this section, we identify and evaluate exposure information from the stressors of the action (Figure 1). We begin by presenting a general discussion of the physical and chemical properties of carbaryl, carbofuran, and methomyl that influence the distribution and persistence of action stressors in the environment and exposure of listed species and designated critical habitat (structures shown in Figure 36). Next we present general life history information of Pacific salmon and steelhead and evaluate the likely co-occurrence of action stressors with the listed Pacific salmonids. We then summarize exposure estimates presented in the three BEs and present other sources of information, including other modeling estimates and monitoring data to further characterize exposure to listed species and designated critical habitat. Finally, we conclude with a summary of expected ranges of exposure and the uncertainty contained in the exposure analysis. Because the ESA section 7 consultation process is intended to ensure that the agency action is not likely to jeopardize listed species or destroy or adversely modify critical habitat, NMFS considers a variety of exposure scenarios in addition to those presented in EPA's BEs. These scenarios provide exposure estimates for the range of habitats utilized by listed salmonids.

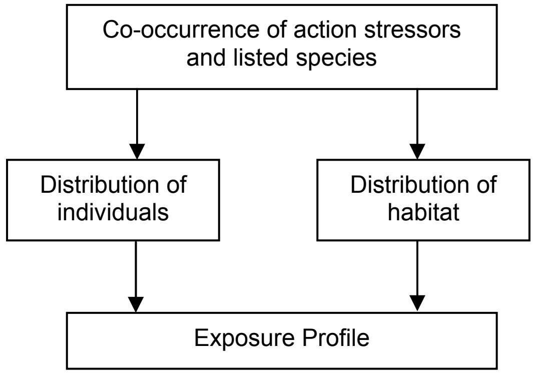
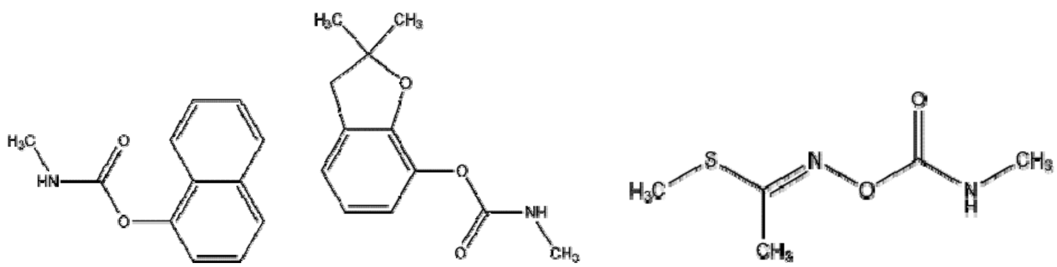


Figure 35. Exposure analysis

Summary of Chemical Fate of A.I.s



Carbaryl Carbofuran Methomyl

Figure 36. Chemical structures of carbaryl, carbofuran, and methomyl.

Carbaryl

"Carbaryl is a widely used pesticide that is commonly detected in the environment from its application in agricultural and non-agricultural settings (EPA 2003)." Carbaryl is primarily applied to terrestrial habitats, although a 24(c) registration in Washington State allows for application to commercial oyster beds to control native ghost shrimp and mud shrimp. Carbaryl can contaminate surface waters via runoff, erosion, leaching, and spray drift from application at terrestrial sites, or direct application to aquatic habitats. Carbaryl and its primary degradate, 1-naphthol, are fairly mobile and slightly persistent in the environment. Although they are not likely to persist or accumulate under most conditions, they may do so under acidic conditions with limited microbial activity.

Carbaryl dissipates in the environment by abiotic and microbially mediated degradation. The environmental fate characteristics for carbaryl are listed below (Table 44).

Table 44. Environmental fate characteristics of carbaryl (EPA 2003).

Parameter	Value
Water solubility	32 mg/L at 20 deg C
Vapor pressure	1.36 10^{-7} torrs
Henry's law constant	1.28 x 10^{-8} atm m^3 /mol
Octanol/Water partition	K_{ow} = 229
Hydrolysis ($t_{1/2}$) pH 5, pH 7, and pH 9	Stable, 12 days, 3.2 hours
Aqueous photolysis ($t_{½}$)	21 days
Soil photolysis($t_{½}$)	assumed stable
Aerobic soil metabolism ($t_{½}$)	4 days - sandy loam soil
Anaerobic soil metabolism ($t_{½}$)	72 days
Aerobic aquatic metabolism ($t_{½}$)	4.9 days
Anaerobic aquatic metabolism ($t_{½}$)	72 days
K_{oc}	177- 249 ml/g

Fish are most likely to be exposed to carbaryl through direct uptake of the chemical from the water column and across the gills, although other routes of exposure may also be important. Potential exposure routes for aquatic organisms include direct uptake from the water column or pore water of sediment, incidental ingestion of the chemical in sediment, or ingestion of the chemical in food items. EPA asserts that "carbaryl is not expected to bioaccumulate" and reports a bioacculation factor of 45 (EPA 2003). Other sources suggest accumulation of carbaryl can occur in fish, invertebrates, algae and plants. Residue levels in fish can be 140 fold greater than the concentration of carbaryl in water (http://pmep.cce.cornell.edu/profiles/extoxnet/carbaryl-dicrotophos/carbaryl-ext.html#14). In general, due to its rapid metabolism and rapid degradation, carbaryl should not pose a significant bioaccumulation risk in alkaline waters. However, under conditions below neutrality accumulation of carbaryl may be significant (http://pmep.cce.cornell.edu/profiles/extoxnet/carbaryl-dicrotophos/carbaryl-ext.html#14).

EPA reports the major degradation products of carbaryl are CO_2 and 1-naphthol, which is further degraded to CO_2 (EPA 2003). Carbaryl is stable to hydrolysis in acidic conditions but hydrolyzes in neutral environments. Hydrolysis rates increase with increasing alkalinity. Carbaryl is degraded by photolysis in water with a half-life of 21 days (d).

Under aerobic conditions, it degrades rapidly by microbial metabolism in soil and aquatic environments. Metabolism is much slower in anaerobic environments, with half-lives on the order of 2 to 3 months. Carbaryl is mobile in the environment. Sorption onto soils is positively correlated with increasing soil organic content. Because of its low octanol/water partition coefficient (K_{ow} values range from 65 to 229), carbaryl is not expected to significantly bioaccumulate (EPA 2003).

The major metabolite of carbaryl degradation by both abiotic and microbially mediated processes is 1-naphthol. This degradate represented up to 67% of the applied carbaryl in degradation studies. It is also formed in the environment by degradation of naphthalene and other PAH compounds. Data suggest 1-naphthol "is less persistent and less mobile than parent carbaryl (EPA 2003).

In a field dissipation study, carbaryl was applied on 3- to 8- ft tall pine trees in an Oregon forest. Maximum measured concentrations were 264 mg/kg on foliage at 2 d post-treatment, 28.7 mg/kg in leaf litter after 92 days, 0.16 mg/kg in the upper 15 cm of litter-covered soil at 62 d, and 1.14 mg/kg in the upper 15 cm of exposed soil at 2 d. Carbaryl was detected in the leaf litter up to 365 d after treatment and in litter-covered soil up to 302 d after treatment. Half-lives were 21 d on foliage, 75 d in leaf litter, and 65 days in soil. Carbaryl was detected at <0.003 mg/L in water and <0.003 mg/kg in sediment from a pond and stream located approximately 50 ft from the treated area (EPA 2003).

Carbofuran

Carbofuran is an *N*-methyl carbamate and is used as a broad spectrum insecticide and nematicide. Carbofuran can contaminate surface waters via runoff, erosion, leaching, and spray drift from application at upland sites.

Potential exposure routes for aquatic organisms include direct uptake of the chemical from the water column or pore water of sediment, incidental ingestion of the chemical in sediment, or ingestion of the chemical in food items. Considering that carbofuran has high water solubility, low K_{ow}, and a relatively low bioconcentration factor (2-12), fish are most likely to be exposed to carbofuran through direct uptake from the water column (Table 45). Exposure via the food chain, pore water, and sediment pathways are also possible but are less likely to be relevant for most life stages of fish (EPA 2004).

Table 45. Environmental fate characteristics of carbofuran (EPA 2004).

Parameter	Value
Water solubility	700 mg/L
Vapor pressure	6 x 10^{-7} torrs
Henry's law constant	No data
Octanol/Water partition	K_{ow} = 0.03
Hydrolysis ($t_{1/2}$) pH 6, pH 7, pH 7, & pH 9	Stable, 28 days, 3 days, and 0.8-15 hrs
Aqueous photolysis (t½)	6 days
Soil photolysis (t½)	78 days
Aerobic soil metabolism (t½)	321 days
Anaerobic soil metabolism (t½)	624 days
Aerobic aquatic metabolism (t½)	No data
Anaerobic aquatic metabolism (t½)	No data
K_{oc}	9 to 62 ml/g

Carbofuran is highly mobile and can leach to ground water in many soils or reach surface waters via runoff. The median K_{oc} of carbofuran is 30 and the Freundlich coefficient (K_f) ranges from 0.10 to 30.3 (EPA 2004). Major factors influencing the fate and persistence of carbofuran are water and soil pH. Carbofuran is very mobile and persistent in acidic environments, but dissipates more rapidly in pHs that are basic. Carbofuran is stable to hydrolysis at pHs < 6, but becomes increasingly susceptible to hydrolysis as the pH increases, hydrolyzing rapidly in alkaline aquatic environments (EPA 2004). A study evaluating its persistence in natural surface waters found it took 3 weeks to degrade by 50% (Sharom, Miles et al. 1980). Carbofuran phenol (7-phenol) was the only degradate detected in hydrolysis studies (EPA 2004). The rate of carbofuran degradation in soils is also pH dependent. In an acidic soil (pH 5.7), carbofuran dissipated with a half-life of 321 d, but when the soil was limed to a pH of 7.7, the half-life dropped to 149 d. The major identified degradate was 3-keto carbofuran, which peaked at 12% of the amount applied after 181 d. The other degradation products formed during photolysis, soil, and aquatic metabolism studies are 3-hydroxycarbofuran, 3-hydroxy-7-phenol, and 3-keto-7-phenol (EPA 2004).

In an aqueous photolysis study carbofuran photodegraded in neutral water (buffered [pH 7] solution) at 25ºC with a half-life of 6 d. In a soil photolysis study, carbofuran photodegraded with a half-life of 78 d on a sandy loam soil. Carbofuran is moderately persistent to microbial degradation, with half-lives on the order of a year. Near-surface photolysis is significant under laboratory conditions in aqueous solution, with a half-life on the order of days (EPA 2004).

Methomyl

Methomyl is a broad spectrum insecticide approved for a variety of terrestrial use sites. Use in aquatic habitats is not permitted. EPA characterized methomyl as moderately persistent and highly mobile suggesting methomyl may contaminate surface waters through runoff, erosion, leaching, and spray drift from application at upland sites (EPA 2003).

Potential exposure routes for fish and other aquatic organisms include direct uptake of the chemical from the water column or pore water of sediment, incidental ingestion of the chemical in sediment, or ingestion of the chemical in food items. Methomyl bioaccumulation has not been studied in fish, although it is expected to be relatively low given low octanol/water partition coefficients (K_{ow} ranges from 1.29 to 1.33, Table 46).

Methomyl degradation occurs through metabolism and photolysis. Site-specific factors affecting the persistence of methomyl include aerobicity, organic matter and soil moisture content, exposure to sunlight, and pH. Climate, crop management, soil type and other site-specific factors also influence leaching and runoff.

Table 46. Environmental fate characteristics of methomyl (EPA 2003).

Parameter	Value
Water solubility	58,000 mg/L
Vapor pressure	1×10^{-5} torrs
Henry's law constant	1.8×10^{-10} atm-m^3/mol
Octanol/Water partition[1]	K_{ow} =1.29 – 1.33
Hydrolysis ($t_{1/2}$) pH 5, pH 7, & pH 9	Stable, stable, & 30 days
Aqueous photolysis ($t_{½}$)	1 day
Soil photolysis ($t_{½}$)	36 days
Aerobic soil metabolism ($t_{½}$)	11 to 45 days
Anaerobic soil metabolism ($t_{½}$)	No data
Aerobic aquatic metabolism ($t_{½}$)	3.5-4.5 days[1]
Anaerobic aquatic metabolism ($t_{½}$)	<7 to 14 days
K_{oc}	24 ml/g

1Reported in (EPA 2007)

Methomyl photolysis occurs relatively quickly in water but slowly in soils. It is moderately stable to aerobic soil metabolism but degrades more rapidly under anaerobic conditions. While methomyl becomes more susceptible to hydrolysis as the pH increases above neutral, this is not expected to be a major route of dissipation under most circumstances. Laboratory studies show that methomyl does not readily adsorb to soil and has the potential to be very mobile. Dissipation from the soil surface occurs by a combination of chemical breakdown and movement. Field studies show that the varying dissipation rates for methomyl were related primarily to differences in soil moisture content, which may affect the microbial activity, and rainfall/irrigation, which could influence leaching (EPA 2003).

Several degradates of methomyl have been identified. CO_2 is a major degradate of methomyl. Methomyl oxime, is a minor degradation product in soil and water that degrades rapidly and has negligible toxicity (DuPont 2009). Another degradate, S-methyl-N-hydroxythioacetamidate, which is highly mobile, appears to be primarily a product of alkaline hydrolysis. In an aquatic metabolism study, methomyl degraded with estimated half-lives of 4-5 d. After 7 d, acetonitrile comprised a maximum of 17% and acetamide up to 14% of the amount of methomyl applied. After 102 d, volatilized acetonitrile totaled up to 27% of the parent methomyl applied and CO_2 up to 46% of the applied material (EPA 2003).

Habitats Occupied by Listed Salmonids

Listed salmonids occupy 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. The temporal and spatial use of habitats by salmonids depends on the species and the individuals' life history and life stage (Table 47). Many migrate hundreds or thousands of miles during their lifetime, increasing the likelihood that they will come in contact with aquatic habitats contaminated with pesticides. Given that all listed Pacific salmonid ESUs/DPSs use watersheds where the use of carbaryl, carbofuran, and methomyl products is authorized, and these compounds are frequently detected in watersheds where they are used (Gilliom, Barbash et al. 2006) we expect all listed Pacific salmonid ESUs/DPSs will continue to be exposed to these compounds and other stressors of the action.

Table 47. General life histories of Pacific salmonids.

Species	General Life History Descriptions		
(number of listed ESUs or DPSs)	Spawning Migration	Spawning Habitat	Juvenile Rearing and Migration
Chinook (9)	Mature adults (usually four to five years old) enter rivers (spring through fall, depending on run). Adults migrate and spawn in river reaches extending from above the tidewater to as far as 1,200 miles from the sea. Chinook salmon migrate and spawn in four distinct runs (spring, fall, summer, and winter). Chinook salmon are semelparous (can spawn only once).	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 swim-up and 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.

Species	General Life History Descriptions		
(number of listed ESUs or DPSs)	Spawning Migration	Spawning Habitat	Juvenile Rearing and Migration
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 (can spawn only once).	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 (can spawn only once).	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.
Sockeye (2)	Mature adults (usually four to five years old) begin entering rivers from May to October. Sockeye are semelparous (can spawn only once).	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.

General life histories of Pacific salmonids (continued)			
(number of listed ESUs)	Spawning Migration	Spawning Habitat	Juvenile Rearing and Migration
Steelhead (11)	Mature adults (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 (can spawn more than once).	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.

Modeling: Estimates of Exposure to Carbaryl, Carbofuran, and Methomyl

Exposure estimates for non-crop pesticide applications

EPA's BEs indicate that pesticides containing carbaryl, carbofuran, and methomyl are registered for over 100 use sites (EPA 2003; EPA 2003; EPA 2004). As previously indicated, many of the uses identified in the BEs have since been modified, or are scheduled to be phased out or cancelled. The BEs provided relatively few estimates of exposure given the number and variety of uses authorized (Table 48). All modeled exposure estimates assumed pesticide use in agricultural crops. No estimates were provided for other uses of carbaryl, carbofuran, or methomyl.

Table 48. Examples of registered uses of carbaryl, carbofuran, methomyl and the exposure method used by EPA in BEs.

a.i.	Examples of Registered Use	Exposure Characterization in BE
Carbaryl	Crops: cranberries, cucumbers, beans, eggplant, grapefruit, grapes, hay, lemons, lettuce, nectarines, olives, onions, oranges, parsley, peaches, peanuts, pears, pecans, peppers, pistachios, plums, potatoes, prunes, pumpkins, rice, sod, spinach, squash, strawberries, sugar beets, sunflowers, sweet corn, sweet potatoes, tangelos, tangerines, tomatoes, walnuts, watermelons, and wheat.	PRZM-EXAMS 9 crops
	Targeted pests: adult mosquitoes, ticks, fleas, fire ants, and grasshoppers.	No estimates provided
	Other use sites: home and commercial lawns, flower beds, around buildings, recreation areas, golf courses, sod farms, parks, rights-of-way, hedgerows, Christmas tree plantations, oyster beds, and rural shelter belts.	No estimates provided
Carbofuran	Crops: All uses will be cancelled. Examples of current uses include Alfalfa, artichoke, banana, barley, coffee, field corn, sweet corn, pop corn, cotton, cucumber, melons, squash, grapes, oats, pepper, plantain, potato, sorghum, soybean, sugar beet, sugarcane, sunflower, wheat, cotton, spinach grown for seed, and tobacco.	PRZM-EXAMS Estimates for 5 crops
	Other use sites: All uses will be cancelled. Examples of current uses include agricultural fallow land, ornamental and/or shade trees, ornamental herbaceous plants, ornamental non-flowering plants, pine, ornamental woody shrubs, and vines.	No estimates provided
Methomyl	Crops: alfalfa, anise, asparagus, barley, beans (succulent and dry), beets, Bermuda grass (pasture), blueberries, broccoli, broccoli raab, brussels sprouts, cabbage, carrot, cauliflower, celery, chicory, Chinese broccoli, Chinese cabbage, collards (fresh market), corn, cotton, cucumber, eggplant, endive, garlic, horseradish, leafy green vegetables, lentils, lettuce (head and leaf), lupine, melons, mint, oats, onions (dry and green), peas, peppers, potato, pumpkin, radishes, rye, sorghum, soybeans, spinach, strawberry, sugar beet, summer squash, sweet potato, tomatillo, tomato, wheat, and orchards including apple, avocado, grapes, grapefruit, lemon, nectarines, oranges, peaches, pomegranates, tangelo, and tangerine.	PRZM-EXAMS Estimates for 4 crops
	Other use sites: sod farms, bakeries, beverage plants, broiler houses, canneries, commercial dumpsters which are enclosed, commercial use sites (unspecified), commissaries, dairies, dumpsters, fast food establishments, feedlots, food processing establishments, hog houses, kennels, livestock barns, meat processing establishments, poultry houses, poultry processing establishments, restaurants, supermarkets, stables, and warehouses.	No estimates provided

Exposure estimates for crop applications

The BEs provide estimated environmental concentrations (EECs) for carbaryl, carbofuran, and methomyl in surface water (Table 49). EECs were generated using the PRZM-EXAMS model and used to estimate exposure of the three a.i.s to listed salmonids and their prey (EPA 2004). PRZM-EXAMS generates pesticide concentrations for a generic "farm pond". The pond is assumed to represent all aquatic habitats including rivers, streams, off-channel habitats, estuaries, and near shore ocean environments. EPA indicated that the PRZM-EXAMS scenarios provide "worst-case" estimates of salmonid exposure and it "believes that the EECs from the farm pond model do represent first order streams, such as those in headwaters areas" used by listed salmon (EPA 2003; EPA 2003; EPA 2004). 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 off-channel habitats that are critical to rearing. All listed salmonids use shallow, low flow habitats at some point in their life cycle (Table 47). 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 that listed Pacific salmonids may be exposed to. Finally, NMFS provides additional modeling estimates to evaluate potential exposure in vulnerable off-channel habitats used by salmonids.

Table 49. PRZM-EXAMS exposure estimates from EPA's BEs.

Scenario: crop, state	Application: rate (lbs a.i./A)/ method/ number of applications	Acute EEC (μg/L)	Chronic EEC 60-d average (μg/L)[1]
Carbaryl			
Sweet corn, OH	2/aerial/8; 3.4/aerial/2	53; 46	19; 13
Field corn, OH	2/aerial/4; 1/aerial/2	47; 13	14; 4
Apples, PA	2/aerial/5; 1.2/spray blast/2	31; 12	7; 2
Sugar Beets, MN	1.5/aerial/2; 1.5/aerial/1	23; 7	6; 2
Citrus, FL	5/aerial/4; 3.4/aerial/2	153; 100	41; 23
Peaches, CA	7/aerial/2; 3.5/air blast/1	57; 14	12; 3
Citrus, CA	5/aerial/4; 3.4/aerial/2	20; 7	11; 2
Tomatoes, CA	2/aerial/4; 0.66/air blast/1	17; 2	7; 1
Apples, OR	2/aerial/5; 1.2/aerial/2	19; 3	6; 1
Blackberries, OR	2/aerial/5; 1.9/air blast/1	12; 8	6; 3
Snap beans, OR	1.5/aerial/4; 0.8/ground/1	10; 1.2	1; 0.3
Carbofuran			
Alfalfa, CA	1/foliar/1	6	3.0
Alfalfa, PA	1/foliar/1	7.9	4.1
Cotton, MS	1/in-furrow/1	11	5.5
Grapes, CA	10/soil surface/1	5.5	2.7
Potatoes, ME	1/not reported/2	26	14
Artichokes, CA	2/ground/1	35	19
Cotton, CA	1/in-furrow/1	0.8	0.4
Potatoes, ID	2/foliar/1; 3/in-furrow/1; 6/chemigation/1	6.2; 0.2; 10.4	4.0; 0.1; 6.2
Methomyl			
Lettuce	0.9/aerial/10; 0.225/aerial/15	88; 30	81; 26
Sweet corn	0.45/aerial/16	60	54
Peaches	1.8/aerial/3	99	85
Cotton	0.6/aerial/3	55	47

[1]The chronic values reported for methomyl are 56-day average concentrations rather than 60-day average concentrations.

Utility of EECs for consultation

As described in the *Approach to the Assessment* section, our exposure analysis begins at the organism (individual) level of biological organization. We consider the number, age (or life stage), gender, 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. Characterization of impacts to individuals provides necessary information 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 may underestimate exposure of some listed organisms 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 2007). EPA has subsequently clarified that rather than providing worst case estimates, PRZM-EXAMS estimates are protective for the vast majority of applications and aquatic habitats (EPA 2007). NMFS agrees that the model is designed to produce estimates of exposure that are protective for a large number of 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 2007; NMFS 2008; USFWS 2008). Concentrations of carbaryl, carbofuran, and methomyl were generally less than concentrations predicted in the BEs using PRZM-EXAMs although there were examples where measured concentrations in surface water exceeded these estimates (*e.g.,* (Hurlburt 1986; Creekman and Hurlburt 1987; Tufts 1989; Tufts 1990; Beyers, Farmer et al. 1995). These findings demonstrate that the EECs generated using PRZM-EXAMS can underestimate peak concentrations that actually occur in some aquatic habitats, and therefore, peak exposure experienced by some individuals of listed species may be underestimated.

Exposure values derived by EPA using PRZM-EXAMS may underestimate or overestimate salmonid exposure to carbaryl, carbofuran, and methomyl. Two assumptions are discussed below that show salmonids may be exposed to higher concentrations than predicted with PRZM-EXAMS modeling:

Assumption 1: Model outputs are 90th percentile time-weighted averages. It is important to recognize that the model predicts concentrations based on site-specific assumptions (*e.g.,* rainfall) and that environmental concentrations provided for the estimate do not represent the highest aquatic concentrations predicted given the assumptions. Rather, the exposure estimates provided in the BEs are time-weighted average concentrations for one day (*i.e.,* peak), 21 d, and 60 d. Although EPA refers to the 1 d-averages as peak concentrations, they do not represent the maximum concentration predicted. Rather, they represent the average concentration over a 24-hour (h) period. Additionally, the concentrations reported represent the 90^{th} percentile of the estimates derived using PRZM-EXAMS (Lin 1998). Although NMFS agrees this is a relatively protective approach for evaluating exposure in most aquatic habitats, it does not represent the possible "worst case" exposure.

Assumption 2: Model inputs used the highest use rates and greatest number of applications. NMFS compiled information on the maximum use rates permitted (single and seasonal), number of applications allowed, and minimum application intervals required through EPA product labeling (Tables 1-3).

Several of the PRZM-EXAMS scenarios did not match up well with current use restrictions and did not account for maximum permitted application rates. For example, PRZM-EXAMS scenarios for carbaryl evaluated a maximum single application rate of 3.5 lb a.i./acre versus higher single application rates authorized by EPA for oyster beds, home lawn, golf course, and sod farms (8-9.1 lb a.i./acre), nut trees (5 lbs a.i./acre), and citrus (7.5 – 12 lbs a.i./acre). Some of the simulations included multiple applications that exceed single application rates. For example, there were simulations for 4 applications of 5 lbs a.i./acre in citrus and 2 applications of 7 lbs a.i./acre in peaches. However, these scenarios do not cover the maximum rate of carbaryl allowed per applications (12 lbs a.i./acre in California citrus) or annually (24 lbs a.i./acre/year in apple, pear, crabapple, and others). The maximum single application rate for methomyl is 0.9 lbs a.i./acre. One scenario assumed 3 applications of 1.8 lbs a.i./acre (California peaches). This scenario may overestimate aquatic concentrations for single applications at comparable use sites. However, considering multiple applications, the maximum application rate of methomyl evaluated in the BE was 9 lbs a.i./acre (lettuce) versus allowable application rates of 10-

32 lbs a.i./acre/year for alfalfa, broccoli, cabbage, corn, lettuce, onions, and spinach (EPA 2007).

Few crop scenarios were assessed relative to the number of approved uses. The BEs provided pesticide exposure estimates from uses in relatively few crops considering the number of registered uses of carbofuran, carbaryl, and methomyl. For example, estimates of carbaryl exposure were provided for nine agricultural crops. An evaluation of currently registered uses of a single carbaryl product label (Sevin brand XLR plus carbaryl insecticide) revealed the product can be applied to more than 75 agricultural crops in California alone (CDPR 2009). Similarly, the product Furadan 4-F Insecticide - Nematicide (carbofuran) can be applied to more than 20 use sites in California while exposure estimates evaluated only 5 agricultural crops. Finally, DuPont Lannate SP Insecticides (methomyl) is authorized for use on more than 80 agricultural crops in California while the BE provided exposure estimates for only 4 crops for all methomyl products. NMFS also reviewed aquatic exposure estimates developed by EPA to assess the risk of carbaryl and methomyl to the red legged frog (EPA 2007). Although these estimates were specific to registered uses in California only, they represent a more thorough evaluation of uses than was provided in the BEs. Estimates for agricultural crops were relatively comparable to estimates in the BEs with most scenarios predicting concentrations of carbaryl and methomyl within an order of magnitude (10-100 μg/L). Exposure estimates associated with carbaryl use in rice (2,579 μg/L) were a notable exception and this use was not evaluated in the BE. The acute EECs for carbaryl ranged from 6 – 167 μg/L for other crops assessed versus 1-153 μg/L in the BE. Acute EECs for methomyl ranged from 3 – 183 μg/L versus 30-99 μg/L in the BE.

Crop scenarios are likely not representative of the entire action area. The regional scale that the modeled scenarios are intended to represent is unclear. Many of the scenarios were conducted for states outside the distribution of listed salmonids. The methomyl BE did not provide information on geographic locations simulated (*e.g.,* county, state, region, etc.). The assumed rainfall and other site-specific input assumptions can have large impacts on predicted exposure. For example, the carbofuran

BE provides a peak EEC of 5.5 μg/L in water following an application rate of 10 lbs a.i./acre. Yet three simulations conducted at a 10-fold lower application rate produced greater estimates of exposure due to differences in site-specific assumptions (Table 49). NMFS also questions whether input assumptions were adequate to represent the range in variability among sites 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 site-specific input assumptions of each scenario nor did they put these assumptions into perspective with regard to the range of conditions throughout the four states. This makes it 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. The pesticide labels NMFS reviewed had few restrictions regarding the co-application (*i.e.,* tank mixture applications) or sequential applications of other pesticide products containing different a.i.s. Also, there were few restrictions for those pesticides containing ingredients that share a common mode of action (*e.g.*, cholinesterase-inhibiting insecticides). For example, we saw no restrictions that would prevent either co-application or sequential application of products containing carbofuran, carbaryl, and methomyl. Examples of fish kill incidents discussed in the *Risk Characterization* section of the Opinion indicate combinations of cholinesterase-inhibiting insecticides are sometimes applied on the same day or over a short interval, increasing the likelihood of salmonid exposure to chemical mixtures that may have additive or synergistic effects. Some labels encourage the use of more than one product. The Sevin Brand XLR Plus Carbaryl Insecticide (EPA registration No. 264-333) advises that 8 applications of carbaryl at 3 d intervals "may not provide adequate levels of protection under conditions of rapid growth or severe pest pressure. The use of an alternative product should be considered in conjunction with this product." Multiple applications of pesticides increase the likelihood of cumulative exposure. We considered cumulative exposure based on generated 60 d time-weighted average concentrations to simulate situations where pesticide products containing the three a.i.s were applied at separate times during the growing season (Table 49). To address potential variability between sites, we generated exposure values for a few labeled uses using the GENEEC model, which is intended to provide screening estimates over large geographic regions (Table 50)[1]. The input parameters used were consistent with recent EPA model inputs (EPA 2006; EPA 2007; EPA 2007).

Table 50. GENEEC estimated concentrations of carbaryl, carbofuran, and methomyl in surface water adjacent to sweet corn, potatoes, sweet corn, potatoes, and citrus.

Chemical use	Rate	No.*	Interval	Buffer	EEC (μg/L)				
Foliar/ ground application	lbs/ acre		days	ft	24-h avg	4-d avg	21-d avg	60-d avg	90-d avg
Sweet Corn									
Carbaryl	2	8	3	0	397	390	348	272	229
Carbofuran	0.5	2	7	0	56	56	55	54	53
Methomyl	0.45 0.225	14 28	1 1	25 25	307 288	276 259	163 153	72 68	49 46
Potatoes									
Carbaryl	2	3	7	0	175	172	153	120	101
Carbofuran	1	2	5	0	112	112	111	108	106
Methomyl	0.9	5	5	25	212	191	113	50	34
Citrus									
Carbaryl	12	1	-	0	485	476	424	332	280
Carbofuran	-	-	-	-	-	-	-	-	-
Methomyl	0.9	3	5	25	133	120	71	31	21

*Number of applications
-Not applicable

The EECs for EPA's effect determinations were derived primarily using the PRZM-EXAMS model. This model predicts runoff to a "farm pond" based on application specifications (rate and method), properties of the a.i. (solubility, soil adsorption coefficient, soil metabolisms rate, etc.,), assumed meteorological conditions (amount of rainfall), and other site-specific assumptions [soil type, slope, etc. (EPA 2004)]. The farm pond scenario is likely a poor surrogate of certain habitats used by salmonids.

In particular, listed salmonids rely extensively upon a variety of off-channel habitats that would be expected to yield higher pesticide concentrations than would be predicted with the "farm pond" based PRZM/EXAMS model. Examples of off-channel habitats include

[1] EPA characterizes GENEEC as a tier-1 screening model (EPA 2004c). GENEEC is a meta-model of the PRZM-EXAMS model that incorporates assumptions that are intended to model exposure estimates on a site vulnerable to runoff. The size of the treated area and aquatic habitat (farm pond) are the same as described for PRZM-EXAMS.

alcoves, channel edge sloughs, overflow channels, backwaters, terrace tributaries, off-channel dredge ponds, off-channel ponds, and braids (Swift III 1979; Anderson 1999). Diverse, abundant communities of invertebrates (many of which are salmonid prey items) also populate these habitats and, in part, are responsible for juvenile salmonids reliance on off-channel habitats. Juvenile coho salmon, stream-type Chinook salmon, and steelhead use off-channel habitats for extended durations (several months). Although these habitats typically vary in surface area, volume, and flow, they are frequently shallow, low to no-flow systems protected from a river's or a stream's primary flow. Thus, rearing and migrating juvenile salmonids use these habitats extensively (Beechie T. and Bolton 1999, Beechie T. J. et al. 2005, Caffrey 1996, Henning 2006, Montgomery 1999, Morley S. A. et al. 2005, Opperman and Merenlender 2004, Roni 2002).

Small streams and some off-channel 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 utilize relatively small aquatic habitats with low dilution capacities.

NMFS estimates of potential exposure in shallow water habitats used by salmonids

Direct over-spray

To estimate potential exposure of salmon to pesticides in off-channel and other shallow-water habitats we first determined the initial average concentrations that will result from a direct overspray of shallow surface water. NMFS is unaware of any circumstances where EPA has authorized direct application of methomyl to aquatic habitats. However, both carbaryl and carbofuran may be applied to aquatic habitats. For example, carbaryl is registered for use in rice crops to control tadpole shrimp and other rice pests. The resulting concentrations in surface water are a function of the amount applied and the volume of the water body when pesticides are applied directly to aquatic habitats (Table 51). Carbaryl can be applied twice in rice at a rate of 1.5 lbs a.i./acre. A single application at that rate would result in an average initial carbaryl concentration of 1,682 μg/L in 10 cm of water. Similarly, EPA estimated aquatic concentrations of 2,579 μg/L associated with 2 applications of carbaryl at 1.5 lbs a.i./acre in a recent assessment for the red legged frog (EPA 2007). Specimen labels for carbaryl do not place any restrictions on rice paddy discharges, yet they warn "discharge from rice fields may kill aquatic and estuarine invertebrates (EPA Reg. No. 264-333 and 264-349)." Although the section 3 registration for rice was canceled in 1997, the BE indicates carbofuran may still be applied in rice from use of existing stock or in connection with emergency exemption requests. The BE indicates it is applied to rice fields prior to flooding at a rate of 0.5 lbs a.i./acre. Assuming this rate in 10 cm of water results in an average initial average concentration of 561 μg/L.

Carbaryl is registered for use at a rate of 2 lbs a.i./acre in cranberries. Five applications can be made at 7 d intervals. The label for Sevin brand XLR Plus warns that the "use in cranberries may kill shrimp and crabs. Do not use in areas where these are important resources (EPA Reg. No. 264-333)." However, there are no restrictions regarding applications to standing water, or label requirements to ensure bog water does not contaminate other surface waters. A single application of 2 lbs a.i./acre to water 10 cm of water results in an initial average concentrations of 2,242 μg/L.

Carbaryl can also be applied at 8 lbs a.i./acre in estuarine areas in Washington state to kill ghost shrimp and mud shrimp in commercial oyster beds. Application of 8 lbs of carbaryl to a static water body of 10 cm deep would result in an initial average concentration of approximately 9 mg/L (8,968 μg/L). This estimate does not consider tidal influences. The 24(c) label for the use of carbaryl on oyster beds does not specify when applications are to be made with relation to incoming or outgoing tides. The label does specify applicators must obtain an NPDES permit for this use. A current NPDES permit specifies that applications to these tidal areas are made when the beds are uncovered by the outgoing tide. Peak concentrations in the water column are expected as the treated area is re-flooded by the incoming tide. These concentrations are expected to decrease as the volume of water over the treated area increases. The area contaminated by the carbaryl application will increase beyond the treatment site, and water column concentrations will simultaneously decrease due to transport and dilution from the incoming tide. This use is discussed in more detail in the *Risk Characterization* portion of the Opinion (see *Field studies in ESA-listed salmonid habitat*).

Table 51. Average initial concentration of any a.i. in surface water resulting from a direct overspray of aquatic habitat.

Application Rate	Water Depth	A.i. Concentration in Surface Water
(lbs a.i. / acre)	(meters)	(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	1121
0.25	0.5	56
0.5	0.5	112
1	0.5	224
3	0.5	673
10	0.5	2242
0.25	0.3	93
0.5	0.3	187
1	0.3	374
3	0.3	1121
10	0.3	3736

Application Rate	Water Depth	A.i. Concentration in Surface Water
(lbs a.i. / acre)	(meters)	(μg/L)
0.25	0.1	280
0.5	0.1	560
1	0.1	1121
3	0.1	3363
10	0.1	11208

Pesticide drift

We also provide estimated pesticide concentrations in shallow off-channel habitats associated with drift from terrestrial applications of pesticides (Table 52). These estimates were derived using the AgDrift model and estimate downwind deposition from pesticide drift (Teske 2001). Additional deposition from runoff was not considered. The drift estimates derived represent mean projected drift. Although AgDrift adequately predicts drift, its field validations studies and other research show drift is highly variable and influenced by site-specific conditions and application equipment (Bird, Perry et al. 2002). No-spray buffer zones (or setbacks) may significantly reduce pesticide exposure to salmonids by reducing runoff and drift inputs.

The methomyl RED requires label statements for methomyl products that specify not to "apply by ground equipment within 25 ft, or by air within 100 ft of lakes, reservoirs, rivers, estuaries, commercial fish ponds and natural, permanent streams, marshes or natural, permanent ponds. Increase the buffer zone to 450 ft from the above aquatic areas when ultra low volume application is made (EPA 1998)." This label requirement appears to address many, but not all aquatic habitats used by listed salmonids. For example, labels specify buffers to natural and permanent streams, marshes, and ponds but there is no mention of important intermittent aquatic habitats or manmade watercourses such as floodplain restoration sites and irrigation systems that either contain listed species or drain to such habitats.

Specific buffer zones that correspond to current label requirements for carbaryl, carbofuran, and methomyl were assessed (Table 52). No buffers to aquatic habitats are required by EPA for carbaryl or carbofuran products. However, we did consider interim "voluntary" buffers recommended by CDPR to protect federally listed endangered species (http://www.cdpr.ca.gov/docs/endspec/prescint.htm). Our simulations assumed the off-channel habitat had a downwind width of 10 m. Pesticide concentrations were predicted for habitats that ranged in depths from 0.1 to 2 m. These dimensions were assumed based on research of salmonid use of off-channel habitats (Beechie et al. 2005, Henning 2006, Montgomery 1999, Morley S. A. et al. 2005, Roni 2002). Average initial concentration estimates derived from the simulations ranged from 0.2-447 μg/L for each lb of a.i. applied. These simulations indicate that applications of several lbs a.i. per acre adjacent to some off-channel habitats could result in aquatic concentrations exceeding 1 mg/L, a value that would result in substantial toxicity to aquatic life, including deaths of exposed salmonids.

Maximum rates of carbaryl permitted for most vegetable crops range from 1-2 lbs a.i./acre. Several tree crops allow much higher application rates (3-8 lbs a.i./acre), with a maximum single application rate of 12 lbs a.i./acre approved for use in California on citrus crops. Carbaryl may be applied by ground boom, chemigation, spray blast, and aerial spray applications. Considering application rates, methods, and no requirements for buffers to aquatic habitat, the estimated initial concentrations of carbaryl in surface waters range from 3 μg/L to 7 mg/L in the modeled habitats. Voluntary buffers recommended by CDPR for protection of federally listed fish would considerably reduce the deposition. For example, drift from aerial applications with the 600 ft buffer to a 10 m wide stream are predicted to be approximately equivalent to 2-6% of the applied rate versus drift equivalent to 30-39% of the applied rate with no buffer. Additionally, drift from ground application with the 120 ft buffer predict drift equivalent to 0.3 – 2% of the applied rate versus 7-23% of the applied rate predicted with no buffer.

Current carbofuran labels reviewed by NMFS specify application rates of 1 lb a.i./acre or less for most crops. Carbofuran can be applied by both ground and aerial methods. Simulations assuming no aquatic buffers at these rates provide estimates of initial average carbofuran concentrations from the low μg/L range to several hundred μg/L. The carbofuran BE indicates several 24(c) uses have previously allowed application of carbofuran at rates as high as 10 lbs a.i./acre. Simulations indicate that application rates in this range, with no buffer restrictions, would result in initial surface water concentrations of 2 μg/L to over 4,000 μg/L.

The maximum single application rate for methomyl is 0.9 lbs a.i./acre. Considering label-required buffer zones, simulations at the maximum labeled application rate predict initial average concentrations of 1-57 μg/L for ground applications and 4-84 μg/L for aerial applications. Incorporating voluntary buffers listed by CDPR predict concentrations ranging from 0.2 – 58 μg/L.

Table 52. Average initial pesticide concentration in 10 m wide off-channel habitat per lb of pesticide applied based on AgDrift simulations.

Depth of aquatic habitat (meters)	Buffer to Aquatic Habitat (ft)	Average Initial Concentration in Surface Water (μg/L)
Aerial Applications, EPA default (ASAE fine-medium droplet size distribution)		
2	0[1]	17
1	0[1]	34
0.5	0[1]	67
0.1	0[1]	333
2	100[2]	5
1	100[2]	9
0.5	100[2]	19
0.1	100[2]	93
2	600[3]	1
1	600[3]	2
0.5	600[3]	4
0.1	600[3]	18
Aerial Applications, (ASAE very fine – fine droplet distribution)		
2	0[1]	22
1	0[1]	45
0.5	0[1]	89
0.1	0[1]	447
2	450[2]	4
1	450[2]	8
0.5	450[2]	16
0.1	450[2]	79
2	600[3]	3
1	600[3]	6
0.5	600[3]	13
0.1	600[3]	64
Air Blast Applications, Dormant Spray		
2	0[1]	11
1	0[1]	21
0.5	0[1]	43
0.1	0[1]	214
2	25[2]	3
1	25[2]	6
0.5	25[2]	12
0.1	25[2]	62
2	120[3]	0.3
1	120[3]	1

Depth of aquatic habitat (meters)	Buffer to Aquatic Habitat (ft)	Average Initial Concentration in Surface Water (μg/L)
0.5	120[3]	1
0.1	120[3]	6
Ground Application, Low Boom, ASAE very fine-fine distribution, 50th percentile		
2	0[1]	4
1	0[1]	8
0.5	0[1]	15
0.1	0[1]	76
2	25[2]	1
1	25[2]	1
0.5	25[2]	2
0.1	25[2]	11
2	120[3]	0.2
1	120[3]	0.4
0.5	120[3]	1
0.1	120[3]	4
Ground Application, High Boom, ASAE very fine-fine distribution, 50th percentile		
2	0[1]	13
1	0[1]	26
0.5	0[1]	52
0.1	0[1]	261
2	25[2]	3
1	25[2]	6
0.5	25[2]	13
0.1	25[2]	64
2	120[3]	1
1	120[3]	2
0.5	120[3]	3
0.1	120[3]	17

[1]Carbaryl and carbofuran labels do not require buffers to aquatic habitats.
[2]Buffer to aquatic habitats specified on current methomyl product labels, as required by the 1998 methomyl RED.
[3]Voluntary "interim bulletin" measures developed by California Department of Pesticide Regulation to protect threatened and endangered species.

Monitoring Data: Measured Concentrations of Carbaryl, Carbofuran, Methomyl, 1-Napthol and 3-Hydroxyfuran

Data Described in USEPA's Biological Evaluations

The BE for carbofuran (EPA 2004) summarized national scale surface water monitoring data available from the USGS NAWQA program and also provided state level summaries based on NAWQA and CDPR data. It should be noted that some USGS data are also reported in the CDPR database. National scale NAWQA data (1991-2001) were presented by land use category, and included information on reporting limits (0.02 μg/L), frequency of detections, and maximum concentrations. Maximum concentrations reported for the various land categories were: agricultural land (7 μg/L), mixed land use (0.678 μg/L), urban land use (0.034 μg/L), and undeveloped land use (0.034 μg/L). An additional table summarizing NAWQA data (1984-2004) by state (California, Idaho, Oregon, and Washington) reported a maximum concentration of 32.2 μg/L. This concentration occurred in surface water sampled from Zollner Creek, near Mt. Angel, Oregon on April 17, 2002, and was also the maximum residue found in the database on a national scale. CDPR data (1990-2003) were presented by county. Zollner Creek is a tributary to the Willamette River and within the distribution of threatened Upper Willamette River Chinook salmon and Upper Willamette River steelhead. The maximum concentration measured in California was 5.5 μg/L, in Imperial County. Date and specific location were not provided.

The BEs for carbaryl (EPA 2003) and methomyl (EPA 2003) provided less detailed information about monitoring data on a national basis. The BE for carbaryl provided a national maximum reported concentration of 5.5 μg/L, and noted that carbaryl was the second most commonly detected insecticide in the NAWQA database. It also noted there were more frequent detections and higher concentrations in urban streams than in streams draining agricultural or mixed land use areas. The BE for methomyl reported that few national level monitoring data were available, and reported data located in EPA's STORET database. The highest reported concentration nationally was 1 μg/L (in Texas).

Table 53. National Maximum Concentrations of Carbaryl, Carbofuran, and Methomyl as Reported in EPA BEs (EPA 2003a, EPA 2003b, and EPA 2004)

Pesticide	Maximum Concentration (μg/L)	Data Source	Years Reported
Carbaryl	5.5	USGS NAWQA	1991-1998
Carbofuran	32.2	USGS NAWQA	1984-2004
Methomyl	1.0	EPA STORET	Not Reported
1-Napthol	Not Reported		

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

We obtained updated data from the USGS NAWQA database to evaluate the occurrence of carbofuran, carbaryl, methomyl, and 1-napthol (a degradate of carbaryl) in surface waters monitored in California, Idaho, Oregon, and Washington. The database query resulted in approximately 5,000 samples in which one or more of the compounds was an analyte Approximately 350 unique sampling locations were represented. Available data covered a range of 15 years, from 1992-2007. Land uses associated with the sampling stations included agriculture, forest, rangeland, urban, and mixed use. Some stations occurred only once in the data set, others appeared multiple times across a span of years. The frequency of detection is a combination of the actual occurrence of pesticides in the water and the sampling intensity. Values reported are for a filtered water sample (0.7 micron glass fiber filter), representing the dissolved phase. Chemicals transported primarily in the particulate phase would be underreported in this data set. No sediment or tissue data were available from USGS for these compounds. Because the USGS monitoring program does not generally coordinate sampling efforts with specific pesticide applications or runoff events, detected concentrations are likely to be lower than actual peak concentrations that occur.

Summary information for quantifiable concentrations of carbaryl, carbofuran, methomyl, and 1-napthol is reported below (Table 54). Non-detects in this data set are reported as less than (“<”) the laboratory reporting level (LRL) for that sample. 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. Quantifiable concentrations ranged from 0.0001 – 33.5 μg/L for carbaryl, 0.0015 – 32.2 μg/L for carbofuran, 0.0039 – 0.8222 μg/L for methomyl, and 0.0007 – 1.6 μg/L for 1-napthol.

Table 54. Summary of Occurrences of Carbaryl, Carbofuran, Methomyl, and 1-Napthol in USGS NAWQA Database (1992-2007) for California, Idaho, Oregon, and Washington

Statistic	Carbaryl	Carbofuran	Methomyl	1-Napthol
Samples	4938	4545	1297	1256
Percent Detections	30%	8.5%	2.3%	8.5%
LRL range (μg/L)	0.0005-5.2	0.002-0.150	0.0044-1.22	0.007-0.09
Minimum concentration (μg/L)	0.0001	0.0015	0.0039	0.0007
Maximum concentration (μg/L)	33.5	32.2	0.8222	1.6
Arithmetic mean concentration (μg/L)	0.0943	0.3846	0.1171	0.0258
Standard deviation (μg/L)	1.11	2.13	0.19	0.15
Median concentration	0.0314	0.0318	0.0440	0.0072

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 (CDPR 2008). Data were available for carbaryl, carbofuran, methomyl and 3-hydroxycarborfuran (a degradate of carbofuran), but not 1-napthol. Data in the database (http://www.cdpr.ca.gov/docs/emon/surfwtr/surfdata.htm) are from multiple sources, including monitoring conducted by CDPR, USGS (data from the NAWQA program as well as other studies), state, city, and county water resource agencies; and some non-governmental or inter-governmental groups such as Deltakeeper. The CDPR requires a formal QA/QC protocol for data submitted or does a separate QA/QC review, thus only data subject to appropriate QA/QC procedures are included in the surface water database. Unlike the USGS NAWQA data set, the CDPR database may contain whole water samples as well as filtered samples. If whole water concentrations are reported for compounds that sorb significantly to the particulate phase, concentrations would appear higher than in a filtered sample, which represents only the dissolved phase. The majority of the studies, which are described in metadata available from CDPR, are not targeted at correlating water concentrations with specific application practices, with the exception of some studies evaluating rice pesticides.

Summary information for carbaryl, carbofuran, methomyl, and 3-hydroxycarbofuran is reported below (Table 55). No monitoring for 1-napthol was reported in the database. 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.

Carbaryl (5,265 records) and carbofuran (5,864 records) were the subject of greater monitoring intensity than either methomyl (1,982 records) or 3-hydroxycarbofuran (1,149 records). Quantifiable amounts of methomyl appeared in 11.0% of the samples. Carbaryl (3.6%) and carbofuran (5.9%) were quantifiable less often, and 3-hydroxycarbofuran (0.3%) was only rarely quantifiable. Quantifiable concentrations ranged from 0.0030 – 8.4 µg/L for carbaryl, 0.0066 –

5.2 μg/L for carbofuran, 0.0150 – 5.4 μg/L for methomyl, and 0.0600 – 0.1800 μg/L for 3-hydroxycarbofuran.

Table 55. Summary of Occurrences of Carbofuran, Carbaryl, Methomyl, and 3-Hydroxycarbofuran in CDPR Database (1991-2006)

Statistic	Carbaryl	Carbofuran	Methomyl	3-Hydroxycarbofuran
Samples in data set	5265	5864	1982	1149
Quantifiable samples	191	347	218	3
% of quantifications in data set	3.6%	5.9%	11.0%	0.3%
LOQ range (μg/L)	0.003-1.0	0.002	0.002-1.0	0.0058-0.69
Minimum concentration (μg/L)	0.0030	0.0066	0.0150	0.0600
Maximum concentration (μg/L)	8.4	5.2	5.4	0.18
Arithmetic mean concentration (μg/L)	0.3733	0.3197	0.2984	0.1367
Standard deviation (μg/L)	0.8999	0.5521	0.5223	0.0666
Median concentration	0.1200	0.1100	0.1550	0.1700

Monitoring Data from Washington State

Data from ~30 pesticide monitoring studies conducted in the state of Washington are included in Department of Ecology 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. 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 EIM requires a formal QA/QC protocol for data submitted or does a separate QA/QC review, thus only data subject to appropriate QA/QC procedures are included. Some of the studies contained in this database are targeted with respect to specific pesticide uses, while others are more generalized water quality surveys. Results from a database query on carbaryl, carbofuran, methomyl, 1-napthol, and 3-hydroxycarbofuran, conducted by NMFS, are provided below (Table 56).

Included in the EIM database are monitoring efforts conducted jointly by the Washington Departments of Agriculture and of Ecology in some salmon-bearing streams. Final reports for 2003-2007 seasons are publically available on their website (http://agr.wa.gov/PestFert/natresources/SWM/default.htm). A separate summary of data from those investigations is provided below (Table 57). Water samples are not filtered, and thus concentrations reported include pesticides in both dissolved and particulate phases, although the sampling protocol specifies an attempt to avoid collection of excessive particulates (Johnson and Cowles 2003). Whole water concentrations for compounds that sorb significantly to the particulate phase will appear higher than those for a filtered sample, which represents only the dissolved phase.

The 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 (Table 56) 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").

Summary information for carbaryl, carbofuran, methomyl, 1-napthol, and 3-hydroxycarbofuran based on all data in the database is reported below (Table 56). Sampling intensity was approximately the same for the three parent compounds considered in this Opinion (~1,400 samples each), with slightly fewer sampling events for the degradates. Based on the method of reporting in the database, we cannot determine how many samples may have contained detection of any of the compounds that were below LOQs. Carbaryl appeared most often in the samples (8.5%), followed by carbofuran (4.0%), and methomyl (3.5%). The degradates were detected slightly less often when they were analytes, with 1-napthol quantifiable in 1.8% of the samples, and 3-hydroxycarbofuran quantifiable in 3.3%. Quantifiable concentrations ranged from 0.002 – 10.0 μg/L for carbaryl, 0.01 – 2.3 μg/L for carbofuran, 0.0150 – 0.17 μg/L for methomyl, 0.01-0.64 for 1-napthol, and 0.05 – 0.42 μg/L for 3-hydroxycarbofuran.

Table 56. Summary of Occurrences of Carbofuran, Carbaryl, Methomyl, 1-Napthol and 3-Hydroxycarbofuran in Surface Water - Washington EIM Database (1988-2007)

Statistic	Carbaryl	Carbofuran	Methomyl	1-Napthol	3-Hydroxy carbofuran
Samples in data set	1477	1410	1402	1173	1396
Quantifiable samples	126	57	49	23	46
% of quantifications in data set	8.5%	4.0%	3.5%	2.0%	3.3%
LOQ range (μg/L)	0.002-25.0	0.01-25.0	0.01-5.0	0.01-5.0	0.01-10.0
Minimum concentration (μg/L)	0.01	0.028	0.02	0.01	0.10
Maximum concentration (μg/L)	10.0	0.16	0.17	0.64	0.15
Arithmetic mean concentration (μg/L)	0.30	0.09	0.04	0.13	0.12
Standard deviation (μg/L)	1.40	0.07	0.05	0.19	0.04
Median concentration	0.03	0.08	0.02	0.08	0.12

Summary information for carbaryl, carbofuran, methomyl, 1-napthol, and 3-hydroxycarbofuran based on the recent studies (2003-2007) conducted by Washington Departments of Agriculture and Ecology are presented below (Table 57). All sampling was done in streams that contain listed Pacific salmonids. These data are a subset of the data listed in Table 56. Sampling intensity was approximately the same for the three parent compounds considered in this Opinion (~1,200 samples each), with slightly fewer sampling events for the degradates. Based on the method of reporting in the database, we cannot determine how many samples may have contained detection of any of the compounds that were below LOQs. Carbaryl appeared most often in the samples (4.3%), followed by 1-napthol (1.0%), and methomyl (0.7%). Carbofuran and 3-hydroxycarbofuran appeared less often (0.2% and 0.2% respectively). Quantifiable concentrations ranged from 0.01 – 10.0 μg/L for carbaryl, 0.028 – 0.16 μg/L for carbofuran, 0.015 – 0.17 μg/L for methomyl, 0.01 – 0.641 for 1-napthol, and 0.095 – 0.15 μg/L for 3-hydroxycarbofuran.

Table 57. Summary of Occurrences of Carbofuran, Carbaryl, Methomyl, 1-Napthol and 3-Hydroxycarbofuran in recent studies by Washington Department of Ecology (2003-2007)

Statistic	Carbaryl	Carbofuran	Methomyl	1-Napthol	3-Hydroxy carbofuran
Samples in data set	1,223	1,208	1,207	1,003	102
Quantifiable samples	52	3	9	10	2
% of quantifications in data set	4.3%	0.2%	0.7%	1.0%	0.2%
LOQ range (μg/L)	0.01-0.25	0.01-0.25	0.01-0.25	0.03-0.25	0.02-0.25
Minimum concentration (μg/L)	0.0100	0.0280	0.015	0.01	0.0950
Maximum concentration (μg/L)	10.0	0.16	0.17	0.641	0.15
Arithmetic mean concentration (μg/L)	0.30	0.09	0.043	0.1338	0.12
Standard deviation (μg/L)	1.40	0.07	0.050	0.189	0.04
Median concentration	0.03	0.08	0.019	0.078	0.12

Three studies in the EIM database reported sediment concentrations. One was conducted to determine the concentrations of carbaryl and 1-napthol in the sediment of Willapa Bay, and two were sediment toxicity assessments. The sediment toxicity assessments, which provided the reported values (Table 58) for carbofuran, methomyl, and 3-hydoxycarbofuran as well as some data on carbaryl and 1-napthol, had an LOQ of 100 μg/kg. The LOQs for the Willapa Bay analysis were lower, ranging from 22-37 μg/kg. Neither of the sediment toxicity studies reported detectable concentration of any of the chemicals. In the Willapa Bay study, both carbaryl (range 29-3,400 μg/kg) and 1-napthol (range 34-280 μg/kg) were detected. These data represent a very small subset of potentially contaminated sediments, and the negative results should not necessarily be interpreted to mean that these chemicals are not in the sediment.

Table 58. Summary of Occurrences of Carbofuran, Carbaryl, Methomyl, 1-Napthol and 3-Hydroxycarbofuran in Sediments in Washington EIM Database (1988-2007)

Statistic	Carbaryl	Carbofuran	Methomyl	1-Napthol	3-Hydroxy carbofuran
Samples in data set	26	26	26	69	26
Quantifiable samples	17	0	0	9	0
% of quantifications in data set	65.32%	0%	0%	13.0%	0%
LOQ range (□g/kg)	22-160	100	100	22-58	100
Minimum concentration (□g/kg)	29	NA	NA	34	NA
Maximum concentration (□g/kg)	3,400	NA	NA	280	NA
Arithmetic mean concentration (□g/kg)	762	NA	NA	122	NA
Standard deviation (□g/kg)	1,166	NA	NA	75	NA
Median concentration (□g/kg)	200	NA	NA	120	NA

Summary of National and State Monitoring Databases

Overall, data from the three sets of monitoring data examined by NMFS are relatively consistent, with carbaryl generally being the most frequently quantifiable parent compound. No state monitoring data were available from Oregon or Idaho. Carbaryl and carbofuran were measured in concentrations ranging from 0.0001-33.5 µg/L. Methomyl generally was measured at slightly lower concentrations, ranging from 0.004-5.4 µg/L. Both 1-napthol and 3-hydroxycarbofuran were detected in slightly lower concentrations, ranging from 0.0007-0.64 µg/L, than any of the parent compounds. Water concentrations based on these monitoring data, as anticipated, were generally lower than those based on modeling estimates.

Targeted Monitoring Studies

Numerous studies have been conducted in coordination with applications of carbaryl in Washington State to control burrowing shrimp in commercial oyster beds. Several investigators have documented water column concentrations exceeding several mg/L, with a peak detection of 27,800 µg/L (27.8 mg/L) in one estuary. Those monitoring studies are discussed in greater detail in the *Risk Characterization* portion of the Opinion.

The BEs for carbofuran (EPA 2004) and methomyl (EPA 2003) report some targeted studies evaluating pesticide concentrations in field runoff and receiving water bodies. The carbofuran BE reports concentrations of 16-28 µg/L of carbofuran in runoff from treated rice fields 26 d after flooding (Nicosia, Carr et al. 1991) as cited in (EPA 2004)). Concentrations declined to ≤5 μg/L by 37 d after flooding. The discussion does not describe when or how the carbofuran was applied prior to flooding. Other studies described merely note that carbofuran does move into surface water, and that concentrations decline over time. Ground water monitoring data for carbofuran reported in the BE included concentrations of up 4.3 μg/L in wells on Long Island, NY 20 years after use was prohibited, and a maximum reported concentration of 176 μg/L in Suffolk County, NY.

The methomyl BE reports data from monitoring studies for specific crops that appear to have been designed to approximate PRZM-EXAMS scenarios. These studies evaluated application rates ranging from 0.3 – 1.35 lb ai/acre on crops such as cantaloupe, sweet corn, apples, lettuce, and tomatoes. All of the studies included multiple applications (from 5 to 29) and short re-application intervals (1-5 d). Some studies measured concentrations in field runoff (96-1,320 μg/L), and all measured concentrations in various receiving waterbodies (4.6 – 175 μg/L).

Carbaryl is one of the pesticides used by the U.S. Department of Agriculture's Animal and Plant Health Inspection Service (APHIS) to control grasshopper infestations. Research was conducted to evaluate the effects of rangeland aerial applications of a carbaryl formulation known as Sevin-4-Oil (Beyers, Farmer et al. 1995). Fixed-wing aircraft were used to apply the carbaryl to rangeland on both sides of the Little Missouri River. Pesticide applicators were instructed to observe a 152 m no-spray buffer around the Little Missouri River. Carbaryl was applied at a rate of 0.5 lbs a.i./acre in 1991 and 0.4 lbs per acre in 1993. Surface water concentrations were monitored during a drought and non-drought year. During the study, discharge in the river ranged from 0.026 to 0.057 m^3/s in 1991 and 37.7 to 49.8 m^3/s in 1993. Concentrations of carbaryl in the river were monitored for 3 d following application (Table 59). Maximum values in surface water were measured on the day of application and peaked at 85.1 and 12.6 μg/L in 1991 and 1993. Concentrations declined but were detectable when the study was terminated 96 h post treatment. Peak values were approximately 7 fold during the drought year when flows were low, demonstrating the influence of dilution capacity on receiving water concentrations. These values represent concentrations that might be observed in similar habitats used by listed salmonids. The 85.1 μg/L represents one of the higher values observed in the monitoring data despite the low use rate (0.5 lbs a.i./acre) and no-spray buffer of almost 500 ft.

Table 59. Mean concentration (μg/L) of carbaryl in the Little Missouri River following rangeland application of Sevin-4-Oil

	Hours after application						
	1	**2**	**4**	**8**	**12**	**24**	**96**
1991	85.1	-	12.3	3.08	5.30	10.3	0.100
1993	12.0	12.6	3.84	4.01	-	4.51	5.14

-No Data

The CDPR database contained data from several studies conducted by CDPR in the 1990s evaluating concentrations of rice chemicals in the Colusa Basin Drain, Butte Slough, and the Sacramento River (referenced in their metadata as studies 17, 30, 34, and 40). Carbofuran, along with molinate, malathion, methyl parathion, and thiobencarb, was one of the analytes in all years. Some years, analytes also included other pesticides such as 2,4-D, triclopyr, and propanil (Norberg-King, Durhan et al. 1991). Sampling was conducted in March-July. Carbofuran was quantifiable in 100% of the samples for which it was an analyte. Concentrations across all years ranged from 0.087 – 2.97 μg/L (Table 60). Another study located in open literature, conducted in 1991-1992 (Crepeau and Kuivila 2000), measured peak concentrations in the Colusa River Basin of 0.6-1.1 μg/L. This study also measured concentrations in the Sacramento River at Sacramento and the Sacramento River at Rio Vista. Carbofuran concentrations in the river were typically an order of magnitude lower than the concentrations in the Colusa Basin Drain. The CDPR studies included acute biotoxicity studies (test organism not specified in metadata) and generally found no significant toxicity. However, a study on water from the Colusa Basin Drain using *Ceriodaphnia dubia* as the test organism in a Toxicity Indicator Evaluation (TIE) procedure found that an extract of Colusa Basin Drain water caused toxicity in laboratory tests (Norberg-King, Durhan et al. 1991) The procedure indicated that carbofuran and methyl parathion accounted for the toxicity of the sample, although other chemicals such as molinate and thiobencarb were present as well. The concentrations observed in surface water monitoring are, as expected; much lower than those predicted for the flooded rice paddies themselves. It should be noted that California requires rice growers to obtain discharge permits or get conditional waivers before discharging water from treated rice fields. Obtaining the permits or waivers typically requires holding periods following pesticide applications, which helps reduce pesticide concentrations in the discharge. Those conditions are not a requirement of the federal label.

Table 60. Carbofuran Concentrations in CDPR Studies of Rice Effluents (1995-1998)

Statistic	All years	1995	1996	1997	1998
Samples in data set	97	22	29	25	21
Quantifiable samples	97	22	29	25	21
% of quantifications in data set	100%	100%	100%	100%	100%
LOQ range (μg/L)	0.1-0.35	0.05-0.35	0.05-0.35	0.05-0.35	0.05-0.35
Minimum concentration (μg/L)	0.0872	0.1240	0.1620	0.1400	0.0872
Maximum concentration (μg/L)	2.97	0.7000	2.97	0.6600	1.35
Arithmetic mean concentration (μg/L)	0.5661	0.3907	0.8995	0.3998	0.4800
Standard deviation (μg/L)	0.5098	0.1465	0.7862	0.1472	0.3276
Median concentration	0.4125	0.3700	0.6100	0.4080	0.3800

Based on these studies, other targeted monitoring studies described in EPA BEs, and open literature evaluated for the Opinion rendered on chlorpyrifos, diazinon, and malathion (NMFS 2008), and the general state of knowledge regarding field runoff from pesticide applications, we 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 may contain the highest concentrations (approaching or exceeding modeled concentrations), and
- measured concentrations are likely to be lower than peak runoff concentrations, as sampling may not coincide with initial application and/or runoff events.

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. None of the datasets discussed above were designed to capture peak concentrations of exposure for salmon, or define exposure distributions.

Of the monitoring programs discussed, only the studies conducted by the Washington State DOE were designed to evaluate exposure to listed Pacific salmonid habitats in several Washington State watersheds. This sampling program was intended to evaluate pesticide occurrence in some salmonid-bearing streams during the pesticide application seasons (Johnson and Cowles 2003).

Sample sites for this study are best characterized as integration sites selected based on the presence of the listed Yakima salmonid population (one of 17 independent populations that comprise the Middle Columbia River steelhead DPS) and high diversity and intensity of agriculture. The study design included sampling during the pesticide application season but did not target specific applications of pesticides nor did it target salmonid habitats that would be expected to produce the highest concentrations of pesticides (*e.g.,* shallow off-channel habitat in close proximity to pesticide application sites). Sampling was generally conducted on a weekly basis, so it is likely peak concentrations associated with drift and runoff events were not captured. Sampling stations included both agricultural- and urban-dominated watersheds, and some storm events are captured in the sampling. Sampling favored the detection of multiple pesticides, rather than peak concentrations in some habitats used by Middle Columbia River steelhead.

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. For example, 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 carbaryl, carbofuran, and methomyl. 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 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 cholinesterase inhibiting insecticides has declined in California over recent decades. However, pesticide use patterns change annually and may result in either increases or decreases in use of pesticide products for specific uses. There is considerable uncertainty regarding the representativeness of monitoring conditions to forecast future use of products containing the three a.i.s.

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

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 carbaryl, carbofuran, and methomyl

Available monitoring data for some metabolites and degrades of the three a.i.s in surface water are presented above. The BE indicates that degradation studies show 1-naphthol is found at up to 67% of the applied carbaryl. It is also formed in the environment by degradation of naphthalene and other polyaromatic hydrocarbon compounds. EPA suggests that salmonid exposure to 1-naphthol is expected, particularly in alkaline waters, but indicates exposure to aquatic organisms cannot be calculated due to lack of environmental fate and transport data for this degradate.

The major transformation product of carbofuran in water is 7-phenol (EPA 2004). The BE indicates other transformation products have the potential to reach the aquatic environment, including 3-hydroxycarbofuran and 3-ketocarbofuran, and that these typically occur in small amounts and are relatively short lived as compared to the parent. The major degradate in a soil metabolism study was 3-ketocarbofuran, which peaked at 12% of the amount applied after 181 days.

Several transformation products were also identified in the methomyl BE (EPA 2003). CO_2 was identified as the major degradate in most metabolism studies. S-methyl-N-hydroxythioacetamidate was identified as a highly mobile product of alkaline hydrolysis. In an aquatic metabolism study, after 7 d, acetonitrile comprised a maximum of 17% and acetamide up to 14% of the amount of methomyl applied. After 102 d, volatilized acetonitrile totaled up to 27% of the applied methomyl.

The BEs recognized that listed salmonids are likely exposed to several metabolites and degradates of the a.i.s. However, estimates quantifying exposure to these transformation products were not provided and remain a considerable source of uncertainty.

Other ingredients in formulated products

Registered pesticide products containing carbaryl, carbofuran, and methomyl generally include other ingredients such as carriers and surfactants. NMFS searched NPIRS and located several pesticide products that contain multiple a.i.s. NMFS located several labels of currently registered products that contain more than one a.i. (Table 61). Carbaryl is a common component in several fertilizer products used on turf. It is also commonly formulated with other pesticidal ingredients including rotenone, a botanical extract used to eradicate fish and as an insecticide. Carbaryl is formulated with malathion (another cholinesterase-inhibiting insecticide), bifenthrin (a neurotoxic synthetic pyrethroid) and copper sulfate (registered for use as an insecticide, algicide, and fungicide). Methomyl is formulated with muscalure, a fly attractant used in fly bait formulations. NMFS is not aware of any carbofuran products that contain other a.i.s.

Table 61. Examples of listed a.i.s on pesticide products containing carbaryl and methomyl.

EPA Product Registration Number	A.i.s	Other Ingredients
4-29	Basic cupric sulfate 7%, carbaryl 1.25%, rotenone 0.5%, cube resins other than rotenone 1%	90.25%
4-59	Carbaryl 0.5%, malathion 3%, captan 5.87%	90. 63%
4-122	Carbaryl 0.3%, malathion 6%, captan 11.7%	82%
4-458	Basic cupric sulfate 7%, carbaryl 2%	19%
4-333, 239-2514, 8119-5, 71096-15	Metaldehyde 2%, carbaryl 5%	93%
9198-233, 9198-234, 9198-235	Carbaryl 2.3%, Bifenthrin 0.058%	97.642%
270-255	Methomyl 1%, component of muscalure 0.025%	98.975%
2724-274	Methomyl 1%, component of muscalure 0.049%	98.951%
7319-6	Methomyl 1%, component of muscalure 0.026%	98.974%
53871-3	Methomyl 1%, component of muscalure 0.04%	98.96%

Nonylphenol (NP) and nonylphenol polyethoxylates are inert ingredients that may be part of a pesticide product formulation and are common adjuvant ingredients added during pesticide applications. NP and nonylphenol polyethoxylates are also ingredients in detergents, cosmetics, and other industrial products and are a common wastewater contaminant from industrial and municipal sources. NP has been linked to endocrine disrupting effects in aquatic systems. A national survey of streams found that NP was among the most ubiquitous organic wastewater contaminants in the U.S., detected in more than 50% of the samples tested. The median concentration of NP in streams surveyed was 0.8 μg/L and the maximum concentration detected was 40.0 μg/L (Table 62). Related compounds were also detected at a relatively high frequency (Koplin, Furlong et al. 2002).

Table 62. Detection of nonionic detergent degradates in streams of the U.S. (Koplin, Furlong 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 carbaryl, carbofuran, and methomyl product formulations 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 (EPA 2008). 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 lbs of these products are applied to the landscape each year (CDPR 2007). 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. For example, Sevin XLR Plus (EPA Reg. No. 264-333), which contains carbaryl, specifies the product is compatible with a wide range of pesticides. The label indicates that when used as a fruit thinner, Sevin XLR Plus may be mixed with other fruit thinners. Although mixtures with other pesticides products are not specifically recommended on the Furadan LFR label (EPA Reg. No. 279-3310, containing carbofuran), they are not prohibited. Furadan LFR does specifically indicate that the product may be mixed with liquid fertilizer. Lannate SP (EPA Reg. No. 352-342, containing methomyl) also provides instructions for tank mixing with other products and identifies some tank mixes that are not compatible. 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, and consequently risk associated with potential ingredients in tank mixtures were not addressed in EPA's BEs and remain a significant source of uncertainty.

Environmental Mixtures

As described in the *Approach to the Assessment*, we analyze the status of listed species, in conjuction 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 is most likely in freshwater habitats and nearshore environments adjacent to areas where pesticides are used. As of 1997, about 900 a.i.s were registered in the U.S. for use in more than 20,000 different pesticide products (Aspelin and Grube 1999). Typically 10 to 20 new a.i.s are registered each year (Aspelin and Grube 1999). In a typical year in the U.S., pesticides are applied at a rate of approximately five billion lbs of a.i. per year (Kiely, Donaldson et al. 2004). Pesticide contamination in the nation's freshwater habitats is ubiquitous and pesticides usually occur in the environment as mixtures (Gilliom, Barbash et al. 2006). "More than 90% of the time, water from streams with agricultural, urban, or mixed-land-use watersheds had detections of two or more pesticides or degradates, and about 20% of the time they had detections of 10 or more (Gilliom, Barbash 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 et al. 2006). An average of three pesticides was found in each sample collected on urban sampling sites with as many as nine pesticides found in a single sample. Agricultural sites averaged three to five pesticides per sample with as many as 14 pesticides being detected in a single sample (Burke, Anderson et al. 2006). Mixtures of chemicals that share a common mode or mechanism of action are of particular concern. Six to 11 million lbs of cholinesterase-inhibiting insecticides are used annually in California (CDPR 2007). One a.i., thiodicarb, degrades into methomyl. Potential effects of thiodicarb and other pesticide mixtures containing carbaryl, carbofuran, and methomyl were not addressed in the BEs.

Cholinesterase inhibiting insecticides, including carbaryl, diazinon, chlorpyrifos, and malathion are the most frequently detected mixtures in urban streams across the U.S. (Gilliom, Barbash et al. 2006).

Gilliom and others (2006) suggested that assessment of pesticide mixture toxicity to aquatic life is needed given the widespread and common occurrence of pesticide mixtures, particularly in streams, because the total combined toxicity of pesticides in water is often greater than that of any single pesticide compound. Exposure to multiple pesticide ingredients can result in additive and synergistic responses as described below in the *Risk Characterization* section. It is reasonable to conclude that compounds sharing a common mode of action cause additive effects and in some cases synergistic effects. CDPR's most recent pesticide use report indicates 6,857,530 lbs of cholinesterase-inhibiting insecticides were applied in California during 2006. Over 60 cholinesterase-inhibiting a.i.s are currently registered in California (CDPR 2007). Exposure to these compounds and other baseline stressors (*e.g.,* thermal stress) was not a consideration in the BEs. Therefore, risk to listed species may be underestimated.

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. Carbaryl, carbofuran, and methomyl are commonly detected in freshwater habitats within the four western states where listed Pacific salmonids are distributed. Because the proposed action of registration of the three a.i.s for the next 15 years authorizes many of the same uses, these three a.i.s will continue to be present in the action area. Therefore, we expect some individuals within all the listed Pacific salmon and steelhead ESUs/DPSs will be exposed to these chemicals and other stressors of the action. Carbaryl can exceed several mg/L in coastal estuaries based on measured environmental concentrations. Carbaryl and carbofuran concentrations in off-channel habitat can also exceed several mg/L where buffers are lacking. Peak concentrations of methomyl are expected to be significantly lower given label restrictions that require buffers to aquatic habitats and a relatively low maximum application rate. However, model estimates indicate methomyl can reach concentrations of several hundred μg/L in surface waters given some uses that allow repeated applications at short re-application intervals. Given variable use of these pesticides across the landscape, and variable temporal and spatial distributions of listed salmonids, we expect exposure is also highly variable among individuals and populations of listed salmon. However, defining exposure and distributions of exposure among differing life stages of each independent population is complicated by several factors. Paramount among these is the uncertainty associated with the use of pesticide products containing these a.i.s. More specifically:

- Although the BEs and RED documents provide information on EPA regulatory decisions, they lack a full characterization of label-specific information needed to assess exposure (*e.g.,* application restrictions including application methods, rates, and intervals are lacking for many non-agricultural uses);

- EPA-authorized labels contain language that frequently does not provide clear boundaries on product use (*e.g.,* the maximum number of applications is commonly not specified and labels often instruct applicators to repeat applications "as necessary");

- Product labels authorize the application of chemical mixtures that are not specified or not clearly defined (*e.g.,* the ingredients of pesticide formulations are not fully disclosed, labels recommend tank mixture applications with other pesticides and adjuvants and tank mixtures with other pesticides are permitted unless specifically stated otherwise);

- Defining use of these products is highly uncertain because products are not likely to be used to the full extent permitted on the labels and historical use information is limited and may not reflect future use.

A major limitation of these assessments is that the majority of monitoring data used were not designed to determine exposure to listed salmonids, with the exception of specific studies conducted in Washington. Therefore, caution should be exercised in using these data for that purpose. Additionally, the assessments lack uncertainty analyses of the monitoring and toxicity data used, which limit the confidence in the given estimates (Warren-Hicks and Moore 1998). Given the complexity and scale of this action we are unable to accurately define exposure distributions for the chemical stressors. We assume the highest probability of exposure occurs in freshwater, and nearshore estuarine/marine environments with close proximity to areas where pesticide products containing carbaryl, carbofuran, and methomyl are applied. We considered several sources of information to define the range of potential exposure to action stressors. EPA provided a number of exposure estimates with maximum concentrations of 153, 35, and 99 μg/L predicted for registered uses of carbaryl, carbofuran, and methomyl, respectively. We generated additional exposure estimates for shallow off-channel habitats with predicted concentrations exceeding 1,000 μg/L for carbaryl and carbofuran, and a peak concentration of 58 μg/L for methomyl. Additionally, we considered monitoring data presented by EPA and from other sources which indicate comparable concentrations of carbaryl, carbofuran, and methomyl have been detected in surface waters within the four states where the listed salmon and steelhead are distributed (85, 32, and 175 μg/L, respectively).

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 parent 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 6 inches of water. Therefore, chum salmon fry are less likely to be exposed to high concentrations of pesticides than other salmonids given the duration of time spent in shallow water habitats and the habitat they occupy. 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 distribute to shallow beach areas in the littoral zones of lakes. 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 off-channel habitats associated with their natal rivers and streams. These species are most likely to experience higher pesticide exposures given their utilization of shallow freshwater habitats as juveniles for rearing. Coho salmon and steelhead have a greater preference for the shallow habitats and rear in freshwater for more than a year.

Substantial data gaps in EPA's exposure characterization include exposure estimates associated with product uses on many crops and particularly, on non-crop uses. The highest concentrations detected in surface waters were those associated with applications to aquatic habitats. Those types of applications although mentioned, were not evaluated in EPA's BEs. Additionally, exposure estimates for other chemical stressors including other ingredients in pesticide formulations, other pesticide products authorized for co-application, adjuvants, degradates, and metabolites are not available or are non-existent. 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 exposure based on all stressors that could result from all uses authorized by EPA's action.

Response Analysis

In this section, we identify and evaluate toxicity information from the stressors of the action and organize the information under assessment endpoints (Figure 37). The endpoints target potential effects from the stressors of the action to individual salmonids and their supporting habitats. The assessment endpoints represent biological attributes that, when adversely affected, lead to reduced fitness of individual salmonids or degrade PCEs (*e.g.,* prey abundance and water quality). We constructed a visual conceptual model to guide development of risk hypotheses and assessment endpoints to highlight potential uncertainties uncovered by our analysis of the available information. We begin the response analysis by describing the toxic mode and mechanism of action of carabaryl, carbofuran, and methomyl. Next we summarize the toxicity data presented in the three BEs and organize the information to applicable assessment endpoints (*e.g.*, survival, growth, etc.). The information provided by EPA addressed aspects of survival, growth and reproduction of aquatic species (freshwater and saltwater), as well as providing some discussion on other information found in the open literature, such as results from some field experiments and experiments that evaluated sublethal effects. NMFS is charged under the ESA to evaluate all direct and indirect effects of an action. We therefore evaluate all aspects of an action that may reduce fitness of individuals or reduce primary constituent elements of designated critical habitat. The evaluation includes information that EPA provided on survival, growth, or reproduction, but also encompasses a broader range of endpoints including behaviors, endocrine disruption, and other physiological alterations. The information we assessed is derived from published, scientific journals and information from government agency reports, theses, books, information and data provided by the registrants identified as applicants, and independent reports. Typically, the most relevant study results are those that directly measure effects to an identified assessment endpoint derived from experiments with salmonids, preferably listed Pacific salmonids or hatchery surrogates, exposed to the stressors of the action.

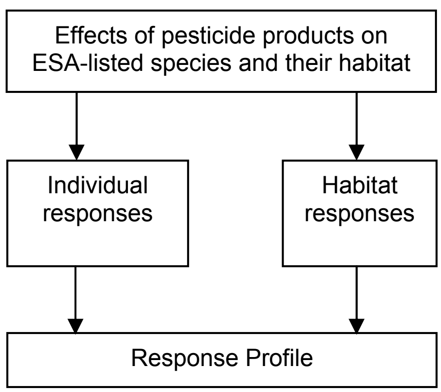


Figure 37. Response analysis

Mode and Mechanism of Action

Carbaryl, carbofuran, and methomyl share a similar mode and mechanism of toxic action and are a part of a group known as *N*-methyl carbamates. All three have similar chemical structures and act as neurotoxicants by impairing nerve cell transmission in vertebrates and invertebrates. They inhibit the enzyme AChE, which is present in cholinergic synapses. The normal function of AChE is to break down (hydrolyze) the neurotransmitter, acetylcholine, thereby serving as an "off-switch" for the electrochemical signal transmissions along nerve cells and neuromuscular junctions. AChE is prevalent in a variety of cell and organ types throughout the body of vertebrates and invertebrates (Walker and Thompson 1991). Interference of normal nerve transmission by *N*-methyl carbamates may affect a wide array of physiological systems in fish (Figure 4). Organophosphates (OPs) share this mode of action and physiological responses are similar.

The mechanism of action of *N*-methyl carbamates, inhibition of AChE, involves a series of enzyme-mediated reactions. Briefly, in a reversible reaction carbamates bind to AChE, thereby inhibiting AChE's normal activity to hydrolyze the neurotransmitter acetylcholine at nerve synapses. This reaction is similar to organophosphorus insecticides with the main exception being a carbamylation of AChE instead of a phosphorylation. Carbamate inhibition of AChE is "reversible" in cases of sublethal exposure and recovery of *N*-methyl carbamate-inhibited AChE is typically rapid compared to OP-inhibited AChE. The key result of AChE inhibition by carbamate and OP insecticides is accumulation of acetylcholine in a cholinergic synapse. The buildup of acetylcholine causes continuous nerve firing and eventual failure of nerve impulse propagation. A variety of adverse effects to organisms can result ranging from sublethal behavioral effects to death (Mineau 1991).

Incidences of acute poisoning from AChE inhibitors are prevalent for wildlife, particularly for birds and fish (Mineau 1991). The following passage describes the classic signs of AChE-inhibiting insecticide poisonings of fish:

> "Fish initially change normal swimming behavior to rapid darting about with loss of balance. This hyper excitability is accompanied by sharp tremors which shake the entire fish. The pectoral fins are extended stiffly at right angles from the body instead of showing the usual slow back and forth motion normally used to maintain balance. The gill covers open wide, and opercular movements become more rapid. With death the mouth is open and the gill covers are extended. Hemorrhaging appears around the pectoral girdle and base of the fins (Weiss and Botts 1957)."

Numerous reports, peer-reviewed journal articles (Williams and Sova 1966; Holland, Coppage et al. 1967; Coppage and Matthews 1974; Rabeni and Stanley 1975; Haines 1981; Antwi 1985) as well as multiple reviews, text books (Mineau 1991; Smith 1993; Geisy, Solomon et al. 1999), and wildlife poisoning cases document inhibition of AChE activity in exposed invertebrates

(Detra and Collins 1986; Detra and Collins 1991) and vertebrates including salmonids following exposures to carbamates and OPs (Beyers and Sikoski 1994, Eder et al. 2007, Grange 2002, Hoy et al. 1991, Laetz et al. 2009, Li H. X. et al. 2008, Li S. N. and Fan 1996, Liu et al. 2007, Sandahl J. F. et al. 2004, Sandahl J.F. et al. 2005, Scholz et al. 2006, St. Aubin 2004, Tierney et al. 2007, Yi et al. 2006, Zinkl et al. 1987).

One highly relevant study measured inhibition of brain AChE, duration of recovery, survival at 24 h, and tissue concentrations in juvenile rainbow trout (*O. mykiss*) following exposure to carbaryl at 0, 250, 500, 1,000, 2,000, and 4,000 μg/L (Zinkl, Shea et al. 1987). Rainbow trout showed dose-dependent AChE inhibition from 61 to 91 % when exposed to 250 – 4,000 μg/L for 24 h. Most trout that died had 85% or greater inhibition. Trout recovered AChE activity following 24 h in uncontaminated water, indicating that fish recover if given the opportunity following carbamate exposures (Zinkl, Shea et al. 1987). This study showed that carbaryl is acutely toxic to rainbow trout in a matter of hours (incidences of death at 1.5 – 4 h) at concentrations at or above 1,000 μg/L.

pH and toxicity

We located several data sets on toxicity to freshwater fish and aquatic invertebrates exposed to carbaryl and carbofuran at different pHs (Mayer and Ellersieck 1986). Acute toxicity of carbaryl and carbofuran increases as pH increased, based on the available freshwater fish assays (Mayer and Ellersieck 1986). This largely is considered a result of the influence pH has on the creation of hydrolysis products (Mayer and Ellersieck 1986). However, the persistence of carbaryl and carbofuran is reduced as pH increases. As noted in the exposure section, degradation half-lives can vary from hours (in alkaline waters) to weeks (in acidic waters) depending on pH. Within the Pacific Northwest and California pH varies seasonally and typically may range from 6 – 9. For methomyl, pH seems to have less of an influence on hydrolysis rates. Due to the influence of pH on persistence of carbaryl and carbofuran, we evaluated reported toxic effects within the context of pH if provided and discuss pH in relationship to salmonid habitat utilization.

Temperature and toxicity

We found no consistent correlation with temperature and toxicity of the three *N*-methyl carbamates.

Studies with mixtures of AChE inhibiting insecticides

Because the three carbamates share a common mechanism of action, are registered for use and applied in the same watersheds, and have demonstrated additive and synergistic effects in aquatic organisms, we evaluate the response of salmonids and their habitat not just from exposure to single carbamates, but also to common mixtures of *N*-methyl carbamates. We therefore include an analysis of combinations of carbaryl, carbofuran, and methomyl based on additive toxicity observed in recent publications with Chinook and coho salmon (discussed in the *Risk Characterization* section) (Scholz, Truelove et al. 2006; Laetz, Baldwin et al. 2009). Because NMFS identified mixture effects to listed salmonids as a critical data gap in understanding the occurrences of multiple insecticides in salmonid habitats, a number of the experiments described below were conducted by researchers at or associated with NOAA's Northwest Fisheries Science Center.

One of the earliest mixture studies available evaluated bluegill survival following a range of exposure durations (24, 48, 72, or 96 h) to binary combinations of 19 insecticide mixtures (Macek 1975). Carbaryl and several of the OP pesticides were tested. Macek (1975) used the equation $AB/(A+B) = X$ to calculate mixture toxicity; where AB was the number of dead fish from a mixture of pesticides A and B, and A + B was the sum of dead fish from A and B alone. The resulting ratios, X, were designated by the author as less than additive for a ratio of less than 0.5, additive when the ratio fell between 0.5 and 1.5, and synergistic for a ratio of more than 1.5. Carbaryl containing mixtures resulted in additive toxicity for some compounds (DDT, methoxychlor, parathion, methyl parathion) and synergistic toxicity for other compounds (malathion, copper sulfate). Antagonism is when the cumulative toxicity of a mixture is less than additive. In this study, mixtures containing carbaryl were not antagonistic. Differences in classification between additive and synergistic combinations should be interpreted cautiously, as the threshold for synergism was arbitrarily set at 1.5 by the authors. Mixture results with DDT and toxaphene were 1.31 and 1.14, respectively. The binary combination of diazinon and parathion was classified as synergistic toxicants, (*i.e.,* more fish died than predicted based on an additive response). Validation of chemical concentrations with analytical chemistry was not conducted. Although the lack of raw data makes it difficult to determine exact concentrations tested and the lack of analytical confirmation precludes determination of precise concentrations, the study shows that binary combinations containing carbaryl exhibit greater toxicity than exposure to carbaryl alone. Additionally, the majority of pesticide-containing mixtures tested resulted in either additive or synergistic responses including mixtures containing carbamate and OP insecticides.

Additive toxicity of binary combinations of carbamates and OPs at a cellular level was demonstrated from *in vitro* experiments with Chinook salmon (Scholz, Truelove et al. 2006). Carbaryl and carbofuran, in addition to the oxons of diazinon, chlorpyrifos, and malathion caused additive toxicity as measured by AChE inhibition in salmonid brain tissue (Scholz, Truelove et al. 2006). Further, the joint toxicity of the mixtures could be accurately predicted from each insecticide's toxic potency, simply by adding the two potencies together at a given concentration. Since the experiments were conducted using *in vitro* exposures with the oxon degradates and not with the parent compounds, the authors conducted subsequent experiments to investigate whether additive toxicity as measured by AChE inhibition also occurred when live, juvenile coho salmon were exposed for 96 h to the parent compounds, *i.e., in vivo* exposures.

The results of the second set of experiments were unexpected by the authors (Laetz, Baldwin et al. 2009). Measured AChE inhibition from some of the binary combinations was significantly greater than the expected additive toxicity, *i.e.,* synergistic toxic responses were found (Laetz, Baldwin et al. 2009). As with the *in vitro* study, brain AChE inhibition in juvenile coho salmon (*O. kisutch*) exposed to sublethal concentrations of the carbamates carbaryl and carbofuran, as well as the OPs chlorpyrifos, diazinon, and malathion, was measured (Laetz, Baldwin et al. 2009). Dose-response data for individual chemicals were normalized to their respective EC_{50} concentrations (AChE activity compared to control) and collectively fit to a non-linear regression. The regression line was used to determine whether toxicological responses to binary mixtures were antagonistic, additive, or synergistic. No binary mixtures resulted in antagonism. Additivity and synergism were both observed, with a greater degree of synergism at higher exposure concentrations. Moreover, certain combinations of OPs were lethal at concentrations that were sublethal in single chemical trials. Concentrations of each insecticide are listed in Table 63. Based on a default assumption of dose-addition, the five pesticides were combined in all possible pairings to yield target levels of AChE inhibitions in the brains of exposed coho salmon.

Table 63. Concentrations (μg/L) of insecticides used in mixture exposures. EC50s were calculated from dose-response data of AChE activity using non-linear regression. Coho salmon exposed to 1.0, 0.4, or 0.1 EC50 treatments had an equipotent amount of each carbamate or OP within the treatment *e.g.,* to attain the 1.0 EC50 treatment for diazinon and chlorpyrifos, 1.0 μg/L of chlorpyrifos (0.5 the EC50) was combined with 72.5 μg/L of diazinon (0.5 of the EC50).

Insecticide	Measured EC50	Concentration of each ingredient in binary combination to achieve treatment level		
		1.0 EC50 units	0.40 EC50 units	0.10 EC50 units
Carbaryl	145.8	72.9	29.2	7.3
Carbofuran	58.4	29.2	11.7	2.9
Chlorpyrifos	2.0	1.0	0.4	0.1
Diazinon	145.0	72.5	29.0	7.3
Malathion	74.5	37.3	14.9	3.7

As determined by the regression, these levels of enzyme inhibition would result from exposure to 0.1, 0.4, and 1.0 EC50 units, respectively. Two thirds (20/30) of pesticide pairs yielded AChE levels that were significantly lower, *i.e.,* indicative of synergism, than would be expected based on additivity *i.e.,* dose-addition (t-test with Bonferroni correction, $p < 0.005$). The number of combinations that were statistically synergistic increased with increasing exposure concentrations. All pairings analyzed at 1.0 EC50 showed synergism in AChE inhibition.

The combination of carbaryl at 79.9 μg/L and carbofuran at 29.2 μg/L to achieve 1.0 EC50 produced synergistic toxicity, however no incidences of coho mortality were observed. At 0.1 and 0.4 EC50 levels with the two carbamates, synergistic toxicity was observed, although the deviation from the predicted additive response was not statistically significant. In binary mixtures containing an OP and a carbamate synergistic toxicity occurred with some of the combinations at 0.1 and 0.4 EC50 treatments. The mechanism for synergistic toxicity in salmonids is unknown.

Additionally, pairings of two OPs produced a greater degree of synergism than mixtures containing one or two carbamates. This was particularly true for mixtures containing malathion coupled with either diazinon or chlorpyrifos. At the highest exposure treatment, 1.0 EC50 (malathion at 37.3, chlorpyrifos at 2, diazinon at 72.5 μg/L), binary combinations produced synergistic toxicity. Many fish species die following high rates of acute brain AChE inhibition, between 70-90% (Fulton and Key 2001).

Coho salmon exposed to combinations of diazinon and malathion (1.0 and 0.4 EC_{50}) as well as chlorpyrifos and malathion (1.0 EC_{50}) all died (Laetz, Baldwin et al. 2009). This result was in contrast to the predicted AChE inhibition from *in vitro* tests with Chinook salmon. Coho exposed to these OP mixtures showed toxic symptoms of inhibition of AChE, including loss of equilibrium, rapid gilling, altered startle response, and increased mucus production before dying. OP combinations were also synergistic at the lowest concentrations tested. Diazinon and chlorpyrifos were synergistic when combined at 7.3 μg/L and 0.1 μg/L, respectively. The pairing of diazinon (7.3 μg/L) with malathion (3.7 μg/L) produced severe (> 90%) AChE inhibition, including classical signs of poisoning as well as death. We expect that juvenile salmonids exposed to these effect concentrations in the environment will respond similarly.

Multiple studies indicate compounds that share a common mode of action frequently result in additive and at times synergistic responses in aquatic organisms (Anderson T. D. and Lydy 2002, Belden et al. 2007, Bocquene et al. 1995, Jin-Clark et al. 2002, Laetz et al. 2009, Lydy and Austin 2005, Macek 1975, Monserrat et al. 2001, Scholz et al. 2006). Unfortunately, we are unable to create a predictive model of synergistic toxicity as dose-response relationships with multiple ratios of pesticides are not available at the present time and the mechanism of synergism remains to be determined. That said, we conducted a mixture analysis with carbaryl, carbofuran, and methomyl based on additive toxicity with the caveat that synergism is likely where circumstances mirror the experimental conditions of this study, *i.e.,* similar exposure durations and pesticide concentrations. This is a reasonable approach based on the current state of the science. We used the mixture work of Laetz, Baldwin et al. (2009) and Scholz, Truelove et al. (2006) to construct mixture dose-response relationships predicated on additivity (see mixture analyses in the *Risk Characterization* section).

Summary of Toxicity Information Presented in the Biological Evaluations

Each BE primarily summarized acute and chronic toxicity data from "standardized toxicity tests" submitted by pesticide registrants during the registration process, tests from government laboratories available in EPA databases, or from published, peer-reviewed scientific publications (books and journals). The assessment endpoints from these tests for an individual organism generally included aspects of survival (death), reproduction, and growth measured in laboratory dose-response experiments (EPA 2004). Survival is measured in both acute and chronic tests. Reproduction and growth are generally measured and reported in the chronic tests. Population-level endpoints and analyses were generally absent in the BEs, other than a few measurements of fish and aquatic invertebrate reproduction. Adverse effects to organisms were not translated into consequences to populations. The BEs also presented some information on multispecies microcosm and mesocosm studies. For this Opinion, NMFS translates effects to individual salmonids into potential population-level consequences as explained in the *Risk Characterization* portion of the *Effects of the Proposed Action* section, and ultimately draws a conclusion on the likely risk to listed salmonids based on exposure and anticipated individual and population-level effects.

Survival of individual fish is typically measured by incidences of death following 96 h exposures (acute test) and incidences of death following 21 d, 30 d, 32 d, and "full life cycle" exposures (chronic tests) to a subset of freshwater and marine fish species reared in laboratories under controlled conditions (temperature, pH, light, salinity, dissolved oxygen, etc.,) (EPA 2004). Lethality of the pesticide is usually reported as the median lethal concentration (LC50), the statistically-derived concentration sufficient to kill 50% of the test population. For aquatic invertebrates it may be reported as an EC50, because death of these organisms may be difficult to detect and immobilization is considered a terminal endpoint. An LC50 is derived from the number of surviving individuals at each concentration tested following a 96 h exposure and is typically estimated by probit or logit analysis and recently by statistical curve fitting techniques. In FIFRA guideline tests, LC50s are typically calculated by probit analysis. If the data are not normally distributed for a probit analysis, than either a moving average or binomial is used, resulting in no slope being reported. Ideally, to maximize the utility of a given LC50 study, a slope, variability around the LC50, and a description of the experimental design- such as experimental concentrations tested, number of treatments and replicates used, solvent controls, etc.- are needed. The slope of the observed dose-response relationship is particularly useful in interpolating incidences of death at concentrations below or above an estimated LC50. The variability of an LC50 is usually depicted by a confidence interval (95% CI) or standard deviation/error and is illustrative of the degree of confidence associated with a given LC50 estimate *i.e.,* the smaller the range of uncertainty the higher the confidence in the estimate. Without an estimate of variability, it is difficult to infer the precision of the estimate. Furthermore, survival experiments are of most utility when conducted with the most sensitive life stage of the listed species or a representative surrogate. In the case of ESA-listed Pacific salmonids, there are several surrogates that are available for toxicity testing including hatchery reared coho salmon, Chinook salmon, steelhead, and chum salmon, as well as rainbow trout[2]. The available toxicity data include a varitety of salmonids. Unfortunately, slopes, estimates of variability for an LC50, and experimental concentrations frequently are not reported. In our review of the 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. Consequently, we must err on the side of the species in the face of these uncertainties and select LC50s from the lower range of available salmonid studies. We selected LC50s and associated slopes as input in the population modeling exercises discussed later. We evaluate the likelihood of concentrations that are expected to kill fish and apply qualitative and quantitative methods to infer population-level responses of ESA-listed salmonids within the *Risk Characterization* section (Figure 2).

Growth of individual organisms is an assessment endpoint derived from standard chronic fish and invertebrate toxicity tests summarized in the BEs. It is difficult to translate the significance of impacted growth derived from a guideline study to fish growth in aquatic ecosystems. The health of the fish, availability and abundance of prey items, and the ability of the fish to adequately feed are not assessed in standard chronic fish tests. These are important factors affecting the survival of wild fish. What is generally assessed is size or weight of fish measured

[2] 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.

at several times during an experiment. The test fish are usually fed twice daily, *ad libitum, i.e.,* an over abundance of food is available to the fish. Therefore, any reductions in size are a result of fish being affected to such an extent that they are not feeding even when presented with an abundance of food. Subtle changes in feeding behaviors or availability of food would not be detected from these types of experiments. If growth is affected in these experiments, it is highly probable that growth of fish in natural aquatic systems would be severely affected. If effects to growth are likely, we assess salmonid population-level consequences based on reductions in juvenile growth and subsequent reduction in size prior to ocean entry.

Reproduction, 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 juvenile and larval survival are measured. NMFS considers many other assessment measures of reproduction, including egg size, spawning success, sperm and egg viability, gonadal development, reproductive behaviors, and hormone levels. These endpoints are not generally measured in standardized toxicity assays used in pesticide registration.

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 for carabaryl, carbofuran, and methomyl, and when noted, pertained to unusual swimming behaviors. None of these behaviors were rigorously measured and therefore are of limited value in assessing the effects of the three insecticides on Pacific salmonids. We do, however note a few of the observations when they pertained to a relevant assessment endpoint, such as impaired swimming. Some BEs presented toxicity information on degradates, metabolites, and formulations. However, toxicity information on other or "inert" ingredients found in pesticide formulations was usually not presented.

Results from multiple species tests, called microcosm and mesocosm studies, were also discussed in the BEs to a varying degree. These types of experiments are likely closer approximations of potential ecosystem-level responses such as interactions among species (predator-prey dynamics), recovery of species, and indirect effects to fish. However, the interpretation of results is complicated by how well the results represent natural aquatic ecosystems and how well the studies apply to salmonid-specific assessment endpoints and risk hypotheses. These studies typically measured individual responses of aquatic organisms to contaminants in the presence of other species. Some are applicable to questions of trophic effects and invertebrate recovery, as well as providing pesticide fate information. The most useful mesocosm study results for this Opinion are those that directly pertain to identified assessment endpoints and risk hypotheses. We discuss study results in the context of salmonid prey responses, emphasizing survival and recovery of prey taxa as well as shifts from preferred taxa to other taxa if measured. One of the notable limitations of these study types is they do not take represent real world aquatic ecosystems that are degraded from various stressors including contaminants and elevated water temperature..

Results from aquatic field studies were generally not discussed in great detail within the BEs. We discuss field studies that evaluated identified assessment endpoints, particularly those which address salmonid prey responses in systems with ESA-listed salmonids.

Ranges in toxicity values presented in the BEs for each a.i. are summarized in Table 64. Ranges in toxicity values (μg/L) are organized by assessment endpoint and associated assessment measures. The BEs provided toxicity information from EPA's EFED Pesticide Toxicity Database and from the ACQUIRE database.

Table 64. Assessment endpoint toxicity values (μg/L) presented in BEs and REDs for carbaryl, carbofuran, and methomyl

Assessment Endpoint		carbaryl (μg/L)			carbofuran (μg/L)			methomyl (μg/L)		
	Assessment measure	> 95% a.i.	< 95% a.i.	degradate (s)	> 95% a.i.	< 95% a.i.	degradate (s)	> 95% a.i.	< 95% a.i.	degradate (s)
Survival	Freshwater fish LC_{50}	250 – 20,000 n=17	14,000-290,000 n=8	1,400-1,800 n=6	88 – 1,990 n=20	240 – 3,100 n=6	--	480 – 6,800 n=11	300 – 7,700 n=16	462,000 n=1
Survival	Salmonid LC_{50}	250 – 3,000 n=7	1,400 – 4,500 n=4	1,400, 1,600 n=2	164 - 600 n=10	610 n=1	--	560 – 6,800 n=6	1,200 – 3,200 n=7	--
Reproduction or larval survival	NOEC/LOEC	fathead minnows 210/ 680 n=1	--	--	rainbow trout 24.8/ 56.7 n=1 (larval survival[1])	--	--	fathead minnow 57/117 n=1 (larval survival[1])	--	--
Fish growth	NOEC/LOEC	--	--	--	rainbow trout[2] 24.8/ 56.7 n=1	--	--	fathead minnow 76,117/ 142,243 n=2	--	--
Habitat-salmonid prey	invertebrate survival	1.7 - 26 n=5	4.3 - 13.0 n=5	200-2,100 n=5	2.2 – 2,700 n=3	41 n=1	--	8.8 - 920 n=8	7.6 - 720 n=6	--
	invertebrate fecundity/ emergence/ developmental rate	500-1,000 n=1	--	--	--	--	--	0.4-0.8 n=1	--	--
	invertebrate reproduction	1.5-3.3 n=1	--	--	9.8/ 27 n=1	--	--	1.6- 3.1 n=1	--	--

1. Larval survival derived from 28 day fish assays
2. Juvenile growth measured at 60, 75, 90 d exposure
n is the number of studies

Assessment endpoint: Fish survival

Assessment measure: 96 h survival from laboratory bioassays reported as an LC50.

Carbofuran is the most toxic of three insecticides based on fish survival values (LC50s) followed by carbaryl and methomyl, respectively (Table 64). All three carbamates have a range of acute freshwater fish LC50s spanning 1 to 2 orders of magnitude. EPA reported the following ranges of LC50s: carbofuran LC50s ranged from 88-3,100 μg/L; carbaryl LC50s ranged from 250-290,000 μg/L; and methomyl LC50s ranged from 300-7,700 μg/L. Based on these LC50 ranges, EPA classified these insecticides as "highly toxic" to "moderately toxic". Salmonids were well represented in the data set, with 11 results for carbaryl, 11 for carbofuran, and 13 for methomyl. A cumulative frequency distribution of carbaryl LC50s for freshwater fish indicated that Atlantic salmon were the most sensitive of the species tested, and salmonids as a group were much more sensitive than fathead minnow and bluegill sunfish (EPA 2003).

EPA classified the three carbamates as very highly toxic to moderately toxic to estuarine and salt water species depending on the chemical and the fish species tested. Carbofuran LC50s for marine and estuarine fish ranged from 33 μg/L to more than 100 μg/L (n=5), indicating that saltwater/estuarine species were more sensitive to carbofuran than salmonids tested in freshwater. Two LC50s (2,200 and 2,600 μg/L, both sheepshead minnow) were reported for carbaryl and one LC50 (1,160 μg/L, sheepshead minnow) was reported for methomyl. Based on available data presented in EPA documents, it is uncertain whether marine and estuarine species are more sensitive, less sensitive, or equally sensitive to carbaryl and methomyl compared to freshwater fish. No data were presented on salmonids exposed in saline environments.

Assessment endpoint: Reproduction

Assessment measure: Number of offspring, hatchability, number of fish that attained sexual maturity by 136 d, and number of spawns per spawning pair

For carbaryl, one chronic study was listed in the BE (EPA 2003) and briefly discussed in the Science Chapter (EPA 2003), which evaluated a variety of assessment endpoints of fathead minnows including reproductive endpoints (referenced in (EPA 2003) as TOUCAR05 Carlson 1972, although this is an erroneous citation of (Carlson 1971)). Reproductive endpoints measured included number of mature males, number of mature females, number of immature fish, number of eggs per mature female, number of eggs spawned, and hatchability of eggs (Carlson 1971). Fathead minnows were exposed to five treatments (8, 17, 62, 210, and 680 μg/L [analytically verified]) of carbaryl in a flow through system for nine months; capturing the life cycle of the fathead minnow. Fathead minnows showed reduced number of eggs per female and reduced number of eggs spawned when exposed to 680 μg/L and of the eggs spawned, none hatched (Carlson 1971).

The carbofuran BE (EPA 2004) stated that no full life cycle fish tests were available, and as it did not report any early-life stage test results, presumably none of those were available either, as the only data reported were for a partial life cycle test for rainbow trout (*O. mykiss*). Reported NOAECs were based on growth effects, and no reproductive endpoints were discussed. A definitive reference to the study was not provided in the BE. The carbofuran Science Chapter (EPA 2005) reports on an early life stage test for rainbow trout, referring to it as Acc.# GEOCAR08. We were unable to locate a specific citation for this study in the references but the reported NOAEC (24.8 μg/L) and LOAEC (56.7 μg/L) were the same as the unreferenced study discussed in the BE. However, the Science Chapter stated that larval survival was the most sensitive endpoint of those evaluated, and that scoliosis was observed in the larval fish at carbofuran concentrations ≥56.7 μg/L. The Science Chapter also references a sheepshead minnow (*Cyprinodon varigatus*) early life stage study (MRID 432505-01) that resulted in a NOAEC of 2.6 μg/L and LOAEC of 6.0 μg/L based on reduced embryo hatching. Specific extent of reduced hatching was not reported. No sublethal effects, such as the scoliosis in the rainbow trout, were mentioned in this study or two other studies on sheepshead minnow (MRIDs 408184-01 and 426974-01) that were submitted but did not produce definitive NOAECs or LOAECs.

The Science Chapter for methomyl (EPA 1998) reports a NOAEC (57 μg/L) and LOAEC (117 μg/L) for fathead minnow (*Pimephales promelus*) based on larval survival. Specific reductions in larval survival were not reported. In the Science Chapter for methomyl (EPA 1998), the study is referred to as MRID 00131255 and Driscoll 1982 and appears to reference the same study as Acc. 251424. The same data are reported in the BE (EPA 2003), but without any citation. We located the data report and confirmed that embryo hatchability was not affected by the dose regime tested: 0, 57, 113, 254, 395, and 972 μg/L (measured concentrations).

Assessment endpoint: Fish growth

Assessment Measure: Growth rate, weight, length, or biomass of second generation as measured in chronic toxicity tests

The carbaryl BE (EPA 2003) did not report any effects on growth for either freshwater or estuarine/marine species, although reduced growth is listed as the affected endpoint in a fathead minnow (*Pimephales promelas*) study referenced in the Science Chapter (EPA 2003). For this study, the NOAEC is given as 210 μg/L and the LOAEC as 680 μg/L However, the fathead minnow study discussed within the reproductive endpoints summary measured growth of fathead minnow larvae (length) at 30 and 60 days (Carlson 1971). Although statistical tests were not used, no differences in growth were apparent between exposed and control fish at the concentrations tested: 8, 17, 62, 210, and 680 μg/L (Carlson 1971).

The carbofuran BE (EPA 2004) reports larval rainbow trout (*O. mykiss*) exposed to 56.7 and 88.7 μg/L carbofuran had significantly reduced lengths and reduced survival at 60, 75, and 90 d compared to unexposed trout in a partial life cycle test. Additionally, trout weight was significantly reduced after 75 d of exposure to 56.7 and 88.7 μg/L. Although, no definitive reference was provided for these data, NMFS obtained the study report to verify study results. Upon review, several additional adverse effects to fry were noted throughout the experiment. At day 20, trout exposed to 57.8 and 88.7 μg/L respired rapidly compared to control fish and were hyperactive at 88.7 μg/L. These effects continued for the remainder of the experiment (~ 70 days). Following 75 d of exposure to carbofuran at 56.7 and 88.7 μg/L, 23 and 35% of the fish, respectively, had curved spines. The Science Chapter (EPA 2005) reports growth was the most sensitive endpoint in two sheepshed minnow studies (*Cyprinodon varigatus*, MRIDs 40818401 and 42697401), but notes that neither of these tests produced a definitive NOAEC or LOAEC.

The methomyl BE (EPA 2003) lists a NOAEC of 76 μg/L and a LOAEC of 142 μg/L based on growth endpoints for fathead minnow (*Pimephales promelas*), a freshwater species. It also lists a NOAEC of 260 μg/L and a LOAEC of 490 μg/L for sheepshead minnow (*Cyprinodon* *varigatus*), an estuarine species, but the endpoint affected is described as reproduction and/or growth, so whether or not growth was affected is uncertain. The Science Chapter (EPA 1998) does not describe any data for estuarine/marine species.

Assessment endpoint: Habitat- salmonid prey

Assessment measure: Aquatic invertebrate survival, growth, reproduction from acute and chronic laboratory toxicity tests

The carbaryl BE (EPA 2003) lists a range of acute EC50 values for freshwater aquatic invertebrate survival. Six EC50 values are given for *Daphnia magna*, ranging from 4.3 -13.0 μg/L (mean 7.3 μg/L). Although not specified as such, given the test organism, the fact that they are noted as 48 h tests, and the range of percent a.i., we assume that these are guideline tests conducted by the registrant or an acceptable government lab. EC50s were also given for three species of stoneflies, ranging from 1.7-5.6 μg/L (mean 3.6 μg/L). An EC50 for an amphipod (*Gammarus fasciatus*) of 26 μg/L was also given. There were also EC50 values for a number of estuarine species. There were five EC50 values given for mysid shrimp (*Mysidopsis bahia*), a common guideline test organism, ranging from 5.7-20.2 μg/L (mean 10.3 μg/L. EC50s for various other shrimp species were given, ranging from 1.5-170 μg/L. All of these tests appear to have been conducted on the a.i. An LC50 for blue crab of 320 μg/L was given. The BE also reported a number of EC50/LC50 values for aquatic insects and other invertebrates derived from the USEPA ECOTOX database.

The carbaryl Science Chapter (EPA 2003) used an acute value of 5.1 μg/L (stonefly, *Chlorperla grammatica*) as the survival assessment endpoint for freshwater aquatic invertebrates, and 5.7 μg/L (mysid shrimp) as the survival assessment endpoint for estuarine/marine aquatic invertebrates. The chapter also notes:

> "Studies have indicated that acute exposure to carbaryl impacts predator avoidance mechanisms in invertebrates, reduces overall zooplankton abundance (Hanazato and Yasuno 1989; Havens 1995), and may actually promote phytoplankton growth through reduced predation by zooplankton."

Acute toxicity data for 1-napthol presented in the BE (EPA 2003) listed 48 h EC50s of 700-730 μg/L for *D. magna*, and EC50s of 200-210 μg/L for *M. bahia*. No additional information regarding acute toxicity of 1-napthol to aquatic invertebrates was provided in the Science Chapter (EPA 2003).

Overall, results presented show that carbaryl and formulations of carbaryl are acutely toxic to a wide array of aquatic invertebrates in the low µg/L range, frequently with EC50s/LC50s of less than 10 µg/L. The degradate, 1-napthol, appears less toxic with respect to comparable invertebrates; acute survival EC50s ranged from 200-730 µg/L. However, no data for the genera more sensitive to the parent carbaryl, such as the stoneflies, caddisflies, or mayflies, are available. Thus the lower end of the toxicity range is not well established.

The carbaryl BE (EPA 2003) lists a NOAEC of 1.5 µg/L and a LOAEC of 3.3 µg/L for *D. magna* based on reproduction and a NOAEC of 500 µg/L and a LOAEC of 1,000 µg/L for the midge fly (*Chironomous riparius*) based on emergence/developmental rate. Specific percentages of inhibition are not noted. The Science Chapter (EPA 2003) does not add any additional detail, but does note:

> "midge larvae are benthic macroinvertebrates and exposure may have been better characterized had it been based on sediment pore water concentrations as opposed to carbaryl concentrations in overlying water"

as a potential explanation for the difference in sensitivity between *D. magna* and *C. riparius*. No data regarding reproductive endpoints were presented for 1-napthol.

The carbofuran BE (EPA 2004) included acute EC50s for the *D. magna* (29-38.6 µg/L) and pink shrimp (*Penaeus duorarum,* 4.6-7.3 µg/L). All tests appear to have been conducted with the a.i. EC50s of 2.2-2.6 µg/L were reported in the Science Chapter (EPA 2005) for the freshwater water flea *Ceriodaphnia dubia*. The Science Chapter also provided toxicity values for the freshwater red crayfish (*Procambarus clarkii,* LC50 2,700 µg/L) and the eastern oyster (*Crassostrea virginica*, EC50s of >1,000->5,000 µg/L). Probit slopes were included in Appendix H of Appendix 1 (EPA 2005) for studies where they were reported or data were available to calculate them. Probit slopes were not available for *D. magna* or *C. dubia*, but were available for the red crayfish (slope 2.91) and the pink shrimp (slope 2.25). Data provided from chronic toxicity tests included a NOAEC of 9.8 µg/L and a LOAEC of 27 µg/L for the freshwater aquatic invertebrate *D. magna*, and NOAEC of 0.4 µg/L and a LOAEC of 0.98 µg/L for the estuarine invertebrate *M. bahia*. In both cases, the most sensitive endpoints appeared to be survival of the adults and/or growth rather than decreased production of offspring.

The BE (EPA 2004) and Science Chapter (EPA 2005) reference several field studies from the open literature that examined effects on aquatic invertebrates following application of carbofuran either to fields or directly to water. One study (Matthiessen, Shearan et al. 1995) noted complete mortality of caged amphipods (*Gammarus pulex)* in a stream draining a field treated with granular carbofuran at 2.7 lbs a.i./acre. EPA documents reported a pond enclosure study (Wayland 1991) noting decreases in amphipod *(Hyallela azteca)* abundance and biomass and Chironominae larvae biomass at concentrations of 25 µg/L in a pond enclosure study. EPA also reported several studies in which carbofuran applied directly to water caused mortality in aquatic invertebrates at concentrations ranging from 5-25 µg/L (Flickinger et al. 1986, Mullie et al. 1991, Wayland and Boag 1990).

The methomyl BE (EPA 2003) reported acute survival EC50s for the freshwater invertebrates water flea (*D. magna*), amphipod (*Gammarus pseudolimnaeus*), midge (*Chironomous plumosus*) and three genera of stoneflies (*Skwala* sp., *Pteronarcella badia*, and *Isogenus* sp.). Most values appeared to be derived from 48-or 96 h standard laboratory tests. Many of the tests were with a.i. (95-99% a.i.), but others were conducted with a 24% formulation (EPA 2003). The product name was not specified. Information in the BE is noted as having come from the EFED Pesticide Ecotoxicity Database, not the RED Science Chapter. There are some inconsistencies between these two documents regarding which EC50 is associated with the formulation versus the technical a.i. Based on information in the BE, which lists data for a formulation test on all species, the 24% formulation appears to be more toxic than the technical. EC50s for the technical range from 8.8-31.8 µg/L for *D. magna*, and are reported as 920 µg/L for *G. psuedolimnaeus*, 88 µg/L for *C .plumosus,* 34 µg/L for *Skwala* sp, 69 µg/L for *P. badia*, and 343 for *Isogenus* sp. EC50s for the 24% formulation are reported as 7.6 µg/L for *D. magna*, 720 µg/L for *G. psuedolimnaeus*, 32 µg/L for *C. plumosus* 29 µg/L for *Skwala* sp, 60 µg/L for *P. badia*, and 29 for *Isogenus* sp.

Acute EC50s are also reported for several species of estuarine/marine shrimp in both the BE (EPA 2003) and the RED Science Chapter (EPA 1998). Again, there are inconsistencies between the documents, with the same EC50s from formulation tests reported in the BE as 30% a.i. and in the Science Chapter as 24% a.i. Some EC50s were also reported for the a.i. Only one species, grass shrimp (*Palmonetes vulgaris*) appears to have been tested with both the a.i. and the formulation (EPA 2003). In this case, the formulation (EC50 130 μg/L, reported as a 30% a.i.), appears less toxic than the technical (EC50 49 μg/L). Other EC50s listed in the BE were for pink shrimp (*P. duorarum*, 19 μg/L) and mysid shrimp (*M. bahia*, 230 μg/L), both of which appear to be technical a.i. (90-98.4%).

The methomyl BE (EPA 2003) reports NOAECs and LOAECs for reproductive endpoints for two studies conducted on *D. magna*, both with the technical a.i. In one study, with a NOAEC of 0.4 μg/L and a LOAEC of 0.8 μg/L, the number of young per female was reduced. A second resulted in a NOAEC of 1.6 μg/L and a LOAEC of 3.1 μg/L, based on an unspecified reproductive endpoint. The BE also reports a NOAEC of 29 μg/L and a LOAEC of 59 μg/L for the estuarine mysid shrimp, based on "reproduction and/or growth."

The BE and Science Chapter both report on an outdoor microcosom study conducted with methomyl (MRID 437444-02). The Science Chapter, Appendix C (EPA 1998), describes the study design as: "Methomyl was applied to seven treatment groups, at two application rates, at three different application intervals, over a period of 22 days (pg 51)." Specific application rates and intervals were not provided, nor was use of a control specifically mentioned. Zooplankton (Cladocera, Copepodia and Rotifera) abundance and community composition were altered in at least some treatments, and Ephemeroptera abundance decreased in the two highest treatments.

Toxicity of Carbaryl, Carbofuran, and Methomyl Degradates

The BEs briefly addressed the issues of degradates. The carbaryl BE (EPA 2003c) provides acute fish and invertebrate toxicity data the degradate, 1-napthol. The LC50s for three species of fish tested (freshwater (*O. mykiss*, *Lepomis macrochirus*)) and marine/estuarine (*Cyprinodon varigatus*) range from 750 – 1,800 μg/L. LC50s presented for aquatic invertebrates include 700-

730 µg/L for *D. magna* (freshwater organism), 200-210 µg/L for *M. bahia* (estuarine organism) and 2,100 µg/L for *C. virginica* (estuarine organism).

During our open literature review, we located a study that compared acute lethalities between carbaryl and 1-naphthol in two species of fish (Shea and Berry 1983). In goldfish (*Carassius auratus*) and killifish (*Fundulus heteroclitus*), 1-naphtol was significantly more toxic than carbaryl based on 10 d acute lethality tests (Shea and Berry 1983). The degradate 1-naphthol was approximately five times more toxic than carbaryl in goldfish, and in killifish twice as toxic as carbaryl (Shea and Berry 1983). Additionally, fish exposed to 1-naphthol showed neurological trauma including pronounced erratic swimming behaviors and increased opercula beats following exposure to 5 and 10 mg/L after 4 and 24 h exposures, respectively. None of these symptoms were observed in the carbaryl treatments (Shea and Berry 1983).

Other degradates identified in fate studies submitted to EPA included 5-hydroxy-1-napthylmethylcarbamate (aerobic soil metabolism and anaerobic aquatic); 1-naphthyl (hydroxymethyl) carbamate (aerobic soil metabolism and anaerobic aquatic); 1,4-napththoquinones (degradates of 1-napthol); 4-hydroxy-1-napthyl methylcarbamate (anaerobic aquatic); 1,5-naphthalenediol (anaerobic aquatic); and 1,4-naphthalenediol (anaerobic aquatic). No toxicity information was presented for these degradates.

The carbofuran BE (EPA 2005) describes three degradates: 3-hydroxycarbofuran, 3-ketocarbofuran, and carbofuran 7-phenol in the toxicity section, and refers to two additional degradates: 3-hydroxy-7-phenol, and 3-keto-7-phenol in the section on environmental fate and transport. No toxicity data are included, and the document noted that "inclusion of environmental transformation products in the risk analysis of carbofuran would not be expected to result in substantive changes to conclusions drawn using the parent alone (pg 30)" (EPA 2005).

Based on the more detailed fate information in the Science Chapter for the Carbofuran RED (EPA 2005), carbofuran 7-phenol is a hydrolysis product (up to 75% of applied). Carbofuran 7-phenol is expected to be less toxic than parent carbofuran, as the toxic moiety (the carbamylating radical) has already dissociated (EPA 2005). No toxicity data for this compound are provided in either the BE or the Science Chapter. The degradates 3-hydroxycarbofuran and 3-ketocarbofuran are detected in soil photolysis and aerobic soil metabolism studies. These compounds were each approximately 3-5% of applied (EPA 2004a), and are structurally more similar to the parent carbofuran. The Science Chapter cites an open literature study using Microtox (Kross, et. al., as cited in EPA 2005) noting “3-ketocarbofuran appears as toxic or slightly more toxic than the parent, but 3-hydroxycarbofuran is much less toxic.” A second study evaluated (Gupta 1994, as cited in EPA 2005) indicates 3-hydroxycarbofuran is “equally as toxic as the parent.” No specific toxicity values were presented. Other degradates mentioned in the fate summary are 3-hydroxycarbofuran phenol (3-hydrocy-7-pheonl) and 3-ketocarbofuran phenol (3-keto-7-phenol), which are less than 5% of applied (EPA 2005).

The methomyl BE notes “a degradate (thiolacetohyroxamic acid 5-methyl ester)… was tested and found to be practically nontoxic to bluegill” (EPA 2003a). This study is also mentioned in the Environmental Risk Assessment for the RED (EPA 1998c). The bluegill LC50 for this degradate is 462,000 μg/L. No other degradate toxicity data were presented. The fate portion of the BE (EPA 2003a) and the Environmental Risk Assessment (EPA 1998c) both reference S-methyl-N-hydroxythioacetamidate as a product of hydrolysis in both water and soil. This may be the same compound expressed under different naming conventions, but no structures were provided to confirm this identification. No toxicity data were presented for acetonitrile or acetamide.

Formulations and other (inert) ingredients found in carbaryl’s, carbofuran’s, and methomyl’s formulations

Assessment endpoint: Fish survival, aquatic invertebrate survival, and primary production

Assessment measure: Aquatic invertebrate survival, growth, and reproduction from acute and chronic laboratory toxicity tests

The carabaryl BE (EPA 2003c) and the carbaryl Science Chapter (EPA 2003b) provide some toxicity data on formulations containing 5-81.5% carbaryl for both aquatic invertebrates and fish. We did not receive information on whether the formulations tested are currently registered. Data for formulations (referred to in the Science Chapter (EPA 2003b) as technical end-product or

TEP) are contained in Appendix D-1. The technical a.i. EC50 for water flea (*D. magna*), based on a single test, is 5.6 μg/L. No 95% confidence interval is given for the test. Data for several formulations with 43.7-81.5% carbaryl are presented. EC50s for the formulations range from 4.3-13.0 μg/L. Data are also available to compare toxicity of the technical grade to formulations (43.7-81.5%) for the estuarine mysid shrimp, *M. bahia*. No 95% confidence intervals are given. Two technical a.i. LC50s were provided in the data, 5.7 μg/L and 6.7 μg/L. Tests with formulations (n=3) resulted in LC50s of 9.3-20.2 μg/L. No chronic tests were described for any formulations.

For carbaryl, acute LC50 formulation data are also given for two species of fish, rainbow trout (*O. mykiss*, 44-81.5 % carbaryl) and bluegill sunfish (*Lepomis macrochirus*, 5-50% carbaryl). No 95% confidence intervals are given. Although other fish data for the technical carbofuran are provided, a same-species comparison of toxicity data is better than evaluating against a range of species, as the various species will exhibit a range of sensitivity. A single a.i. LC50 (1,200 μg/L) is given for rainbow trout. A total of four LC50s are given for formulations, ranging from 1,400-4,500μg/L. Three a.i. LC50s for bluegill are given, ranging from 5,000-14,000 μg/L. The bluegill LC50s (n=4) for formulations range from 9,800-290,000 μg/L. Based on these data, it appears the formulations tested are relatively similar in toxicity to the a.i. on an acute lethality basis (survival endpoint). No chronic tests, which evaluate reproduction and growth, were described for any formulations.

The carbofuran BE (EPA 2004a) lists some formulation toxicity data for aquatic invertebrates, referenced to the EFED Science Chapter and other data referenced to the EPA Acquire Database. In some cases, the formulation data referenced to the Science Chapter provides percent a.i. and in others it contains designations such as “5G” or “50 WP”. In cases where the percent a.i. is reported as a number, we presume the toxicity value has been corrected to percent a.i. It is unclear whether the others have been corrected, whether the data from the Acquire database has been corrected, or the specific products with which the toxicity data are associated.

Toxicity data presented in the carbofuran Science Chapter (EPA 2005) were found in Appendix H of Appendix 1 to the main document, titled Animal Toxicity Tests with DERs. Study data on formulations were presented for bluegill sunfish (*L. macrochirus*) and rainbow trout (*O. mykiss*). A single test on rainbow trout was conducted using the formulation Furadan 75WP with a reported LC50 of 458 µg/L. LC50s for rainbow trout tested with technical carbofuran ranged from 362-600 µg /L. A number of different formulations (Furadan 4F, Furadan 10G, Furadan 50WP and Furadan 75WP) were tested with bluegill sunfish. LC50s ranged from 240-488 µg carbofuran/L. LC50s for bluegill sunfish tested with technical carbofuran ranged from 88-126 µg a.i/L. Only one formulation test for freshwater invertebrates was reported. *D. magna* exposed to "5G" had a survival EC50 of 41 µg /L. Tests with *D. magna* on technical carbofuran produced survival EC50s of 29-38.6 µg /L. The estuarine fish Atlantic silverside (*Mendia menidia*) was tested in two formulations, Furadan 4F and Furadan 15G. LC50s from these tests were 36 µg a.i./L and 64 µg a.i./L, respectively. A test with technical grade on the same species produced an LC50 of 33 µg a.i./L. (95% CI 27-41 µg a.i./L). The estuarine/marine invertebrate pink shrimp was tested both in a formulation (Furadan 15G) EC50 13.3 µg a.i./L and with technical (EC50s 4.6-7.3 µg a.i./L, n=2). Based on these data, there is no indication that the carbofuran formulations tested are more toxic than the a.i. on survival as an endpoint. No long term studies, such as life cycle studies, with formulations were reported for either fish or invertebrates.

The methomyl BE (EPA 2003a) and Science Chapter for the RED (EPA 1998c) both reported toxicity data for a 24% formulation and a 29% formulation for both fish and aquatic invertebrates. In addition, the BE reported some toxicity data for a 30% formulation. While it is difficult to make a definitive comparison given the structure of the data, the 24% formulation appears more toxic to both fish and invertebrates. Neither the toxicity section of the BE nor the use characterization in the Science Chapter list specific products. Thus, we cannot cross-reference the data to a product or determine if it is a currently registered product.

Identified data gaps and uncertainties of carbofuran's, carbaryl's and methomyl's toxicity information presented in BEs and REDs:

Overall, data provided in the BEs and their Science Chapters were insufficient to allow a thorough evaluation of all identified assessment endpoints and measures considered by NMFS. The Data Evaluation Reviews (DERs) or the original studies may have helped reduce some of the uncertainties related to experimental design, dose-response slopes, and confidence estimates of the data, but we did not receive summaries of this information from EPA when preparing this Opinion. However, even if the DER data were submitted, other aspects of EPA's assessment endpoints (survival, reproduction, and growth) were not presented. When this missing information is combined with the absence of information presented on other non-assessed endpoints such as AChE inhibition in fish and invertebrates, swimming behaviors, and olfactory-mediated impairments, an incomplete picture emerges on the potential effects of EPA's action to listed salmonid and habitat endpoints.

Toxicity information presented in the BEs (EPA 2003c, EPA 2004a, and EPA 2003a) lacks details NMFS requires for analyses in support of this consultation. More complete information is presented in the Science Chapters for carbaryl (EPA 2003c) and carbofuran (EPA 2005), yet the information was not applied by EPA in its characterization of risk to listed salmonids or salmonid habitats. Specific data gaps identified include the following, although not all gaps apply to all three carbamates.

- Reported LC50s not accompanied by slopes, experimental design (number of treatments and replicates, life stage of organism, concentrations tested), measures of variability such as confidence intervals or standard deviations/errors;
- No analysis of the degree or magnitude of inhibition of acetylcholinesterase by the three carbamates and expected response by listed salmonids or aquatic, salmonid prey communities;
- Summary and discussion of fish sublethal data absent from BEs including effects to swimming and chemoreception;
- Limited or no toxicity data on current formulations,
- Limited or no toxicity data presented for identified surface water degradates of the three carbamates;
- Sensitivity of surrogate lab strains compared to wild, listed fish, particularly comparisons between warm and cold water fish species used in chronic guideline tests;
- No data summarized for mixture toxicity including tank mixtures and environmental mixtures to assessment endpoints;
- No toxicity data presented on "inert" or other ingredients present in formulations containing each of the a.i.s;
- No analysis of influence of environmental factors (pH, temperature) on exposure and toxicity.

Summary of Toxicity Information from Open Literature

To organize the available toxicity information on listed salmonids and habitat, we developed risk hypotheses with associated assessment endpoints as described in the *Approach to the Assessment* section. Recall that assessment endpoints are biological attributes of salmonids and their habitat potentially susceptible to the stressors of the action. In addition to toxicity data presented in the BEs, we also considered information from other sources to evaluate both individual and population-level endpoints. The results of those studies are summarized below under corresponding assessment endpoints. We qualitatively assigned the most significance to study results that were: 1) derived from experiments using salmonids (preferably listed Pacific salmonids or hatchery surrogates); 2) measured an assessment endpoint of concern *e.g.,* survival, growth, behavior, reproduction, abundance etc., identified in a risk hypothesis; 3) resulted from exposure to stressors of the action or relevant chemical surrogates (*i.e.,* other AChE inhibitors); and 4) had no substantial flaws in the experimental design. When a study did not meet these criteria, we highlighted the issue(s) and discussed how the information was used or why the information could not be used.

Assessment endpoint: Swimming

Assessment measures: Burst swimming speed, distance swam, rate of turning, baseline speed, tortuosity of path, acceleration, swimming stamina, and spontaneous swimming activity

Swimming is a critical function for anadromous salmonids that is necessary to complete their life cycle. Impairment of swimming may affect feeding, migrating, predator avoidance, and spawning (Little and Finger 1990). It is the most frequently assessed behavioral response of toxicity investigations with fish (Little and Finger 1990). Swimming activity and swimming capacity of salmonids have been measured following exposures to a variety of AChE-inhibiting insecticides including the carbamates carbaryl and carbofuran and a variety of OP insecticides. Swimming capacity is a measure of orientation to flow as well as the physical capacity to swim against it (Howard 1975; Dodson and Mayfield 1979). Swimming activity includes measurements of frequency and duration of movements, speed and distance traveled, frequency and angle of turns, position in the water column, and form and pattern of swimming. A review paper summarized many of the experimental swimming behavioral studies and concluded that effects to swimming activity generally occur at lower concentrations than effects to swimming capacity (Little and Finger 1990). Therefore, measurements of swimming activity are usually more sensitive than measurements of swimming capacity. A likely reason is that fishes having impaired swimming to the degree that they cannot orient to flow or maintain position in the water column are moribund (*i.e.,* death is imminent). The authors of the review also concluded that swimming-mediated behaviors are frequently adversely affected at 0.3 – 5.0% of reported fish LC50s[3], and that 75% of reported adverse effects to swimming occurred at concentrations lower than reported LC50s (Little and Finger 1990). Both swimming activity and swimming capacity are adversely affected by AChE-inhibiting insecticides.

We located studies that measured impacts to salmonid swimming behaviors from exposure to carbaryl. Several studies also measured AChE inhibition from OPs and provided correlations between AChE activity and swimming behaviors. We did not locate any studies that tested mixtures of AChE-inhibiting insecticides on swimming behaviors of any aquatic species.

Carbaryl

Experiments with carbaryl have shown that cutthroat trout's (*Oncorhynchus clarki clarki*) swimming abilities are compromised by 6 h exposures to 750 and 1,000 μg/L, resulting in increased susceptibility to predation (Labenia, Baldwin et al. 2007). Cutthroat trout swimming capacity was not significantly affected by carbaryl at 500 μg/L, although muscle AChE activity was 29% relative to unexposed cutthroat. More sensitive swimming activity measurements were not evaluated. At 750 and 1,000 μg/L, AChE activity was 24 and 23% relative to unexposed fish, respectively. A known predator of juvenile cutthroat, lingcod (*Ophiodon elongates*), consumed on average more cutthroat that were exposed to carbaryl compared to those that were not exposed to carbaryl. The experimental design ensured that the predator was exposed to minimal concentrations when carbaryl-exposed fish were transferred in a large experimental chamber along with unexposed cuthroat. In the predation experiment with lingcod, cutthroat were exposed for 2 h to 200, 500, and 1,000 μg/L carbaryl. Results indicated a dose-dependent decrease in the ability of carbaryl-exposed trout to avoid being eaten by the lingcod predator. At

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[3] The current hazard quotient-derived threshold for effects to threatened and endangered species used by EPA is 5 % (1/20th) of the lowest fish LC50 reported. If the exposure concentration is less than 5 % of the LC50 a no effect determination is made which likely underestimates risk to listed salmonids based on swimming behaviors.

200 μg/L, an increase in predation was evident, although not statistically significant. At 500 and 1,000 μg/L carbaryl, cutthroat trout were consumed at significantly higher rates than unexposed fish. Cutthroat trout's AChE activity was also reduced in a dose-dependent fashion showing greater reductions with increasing carbaryl concentrations. Six hour exposures significantly reduced AChE activity in brain and muscle, where 50% reductions (IC50) in brain AChE occurred at 213 μg/L carbaryl. The onset of inhibition occurred within 2 hours, at which point AChE activity was near its lowest value. Benchmark concentrations corresponding to 20% inhibition were 32 μg/L in brain and 23 μg/L in muscle tissues. Recovery of AChE activity (to pre-exposure levels) took 42 hours at 500 μg/L carbaryl. We ranked these sets of experiments as highly relevant to understanding the effects of carbaryl on inhibition of AChE and subsequent salmonid swimming behaviors. The test also provided information on lack of predator avoidance behaviors by salmonids.

Catfish (*Mystus vittatus*) showed increased swimming activity following 72 h of exposure to 12,500 μg/L and no mortalities were noted (Arunachalam, Jeyalakshmi et al. 1980). The 72 h LC50 for catfish was 17,000 μg/L, indicating this species of catfish is much less sensitive to acute concentrations of carbaryl than salmonids. Other sublethal endpoints were also affected by carbaryl, including food intake, growth, metabolism, and rate of opercular beats. Catfish increased their rates of opercular beats from 74 per minute exposed to freshwater alone to 124 per minute when exposed to 12,500 μg/L carbaryl, which the authors attributed to acute stress from carbaryl. We ranked this study as relevant because swimming was measured, but the high concentrations used, lack of chemical verification, and lack of compatibility to listed salmonids introduces uncertainty.

Multiple behavioral responses related to swimming were assessed in rainbow trout fry (0.5-1 g) following 96 h exposures to carbaryl at 10, 100, and 1,000 μg/L; nominal concentrations (Little, Archeski et al. 1990). These included swimming capacity (cm/sec), swimming activity (sec), prey strike frequency, daphnids consumed, percent consuming daphnids, and percent survival from predation. All endpoints were significantly affected at 1,000 μg/L carbaryl relative to unexposed fry. At 10, 100, and 1,000 μg/L carbaryl, significantly more rainbow trout were consumed relative to unexposed fish. By using very young fry, the studies also provide information on a sensitive early lifestage where swimming behaviors are critical to survival (*i.e.,* feeding and predator avoidance). We ranked these studies as highly relevant to a variety of essential swimming-associated behaviors.

One study tested whether schooling of fish in estuarine waters was affected by a single short-term exposure of carbaryl at 100 µg/L (Weis and Weis 1974). Carbaryl impaired schooling of Atlantic silversides (*M. menidia*) by increasing the school area by up to twice that of control groups. Recovery of schooling behavior took three days (Weis and Weis 1974). The authors suggested that increased schooling areas would increase energy expenditures of individual fish and also increase rates of predation. The study lacked analytical verification of exposure concentrations; only one concentration was tested; and the study was conducted with a non-salmonid fish. The study is relevant because it addressed an important swimming behavior (*i.e.,* schooling), which juvenile salmonids use at times, and presented data on time to recovery following an adverse behavioral affect. However, it remains uncertain at which concentrations juvenile salmonids that school would be affected.

Neurological effects on the startle response and ability to avoid predation by juvenile medaka (*Oryzias latipes*) were evaluated following exposures of 2.5, 5.1, 7.0, and 9.4 mg/L carbaryl (Carlson, Bradbury et al. 1998). At 5.1 mg/L and higher, carbaryl increased the time between motor neuron peak and initiation of muscle activity (*i.e.,* swimming response) and at 7.0 mg/L and higher response to stimuli ratios were also higher. Predation trials showed no differences between medaka exposed to carbaryl and unexposed fish. While these concentrations are extremely high compared to concentrations affecting salmonids, the results indicate that neurological-associated swimming behaviors are affected by carbaryl. Given that the 48 h LC50 of juvenile medaka is estimated at 9.4 mg/L and that detectable sublethal neurological effects occurred at 5.1 mg/L, roughly half the LC50, it is uncertain at what concentrations juvenile salmonids neurological endpoints would be affected. We ranked this experiment as relevant as it provided evidence that neurological effects manifest at lower concentrations than 48 h LC50s.

Carbofuran

Carbofuran adversely affected swimming behaviors in goldfish (*C. auratus*) following 24 and 48 h exposures to the lowest concentration tested, 5 µg/L (Bretaud, Saglio et al. 2002). Swimming activity (fish swimming from one zone to another), the least sensitive endpoint, was significantly affected at 500 µg/L carbofuran, while burst swimming, the most sensitive endpoint, was significantly affected at 5 µg/L following 24 h exposure (Bretaud, Saglio et al. 2002). Burst swimming behavior, sheltering, and nipping in goldfish were significantly increased by a 4 h exposure to 1 µg/L carbofuran (Saglio, Trijasse et al. 1996). At 12 h exposures, significant effects were observed at 100 µg/L for sheltering, nipping, and burst swimming. Grouping of goldfish showed non dose-dependent responses as significant effects ($p<0.05$) appeared at 10 µg/L, but were absent at 1 and 100 µg/L (Saglio, Trijasse et al. 1996). We ranked both of these studies as relevant, but due to difficulties inherent in translating the observed behavioral effects in goldfish to salmonids, combined with the lack of analytical verification of concentrations, the study did not receive a highly relevant ranking. The results support that ecologically relevant swimming-related behaviors are impacted at lower concentrations than more coarse measures of swimming such as swimming stamina or swimming capacity.

Methomyl

We were unable to locate any studies that evaluated effects to swimming related behaviors in fish. Although this is a data gap, we assume methomyl does inhibit swimming in the same fashion as the other carbamates because swimming behaviors are typically affected when AChE is sufficiently inhibited (*e.g.* by 30% or more). A review also concluded that swimming-mediated behaviors are frequently adversely affected at 0.3 – 5.0% of reported fish LC50s, and that 75% of reported adverse effects to swimming occurred at concentrations lower than reported LC50s (Little and Finger 1990).

Other AChE-inhibiting insecticides effects on swimming and related behaviors

We also reviewed study results conducted with other carbamate and OP insecticides because both classes of compounds share the same mode of action. Recovery of swimming to pre-exposure levels is much more rapid in carbamate-affected fish compared to OP-affected fish due to the reversible binding of carbamates with AChE (Mineau 1991). In one study with carbaryl and rainbow trout, AChE activity returned to control levels after 24 h in clean water following

24 hr exposure (Zinkl, Shea et al. 1987). In another study with cutthroat trout within 42 h recovery occurred following 6 h exposure(Labenia, Baldwin et al. 2007). In the latter study, cutthroat trout recovered approximately half of their pre-exposure AChE activity within 6 h, following peak AChE inhibition within 2 h. Both of these studies indicate that recovery of AChE activity following exposure to carbamates occurs quickly and is highly dependent on environmental exposure conditions found in aquatic habitats.

We did not locate additional studies with other carbamates, but located multiple studies with OP insecticides. The OPs chlorpyrifos, diazinon, and malathion showed significant and persistent effects to a suite of swimming related behaviors in salmonids at concentrations expected in salmonid habitats; as reviewed in NMFS' November 18, 2008 Opinion on these three OP insecticides (NMFS 2008). Robust evidence of the three OPs showed reductions in swimming speed (Brewer, Little et al. 2001), distance swam (Brewer, Little et al. 2001), acceleration (Tierney, Casselman et al. 2007), food strikes (Sandahl, Baldwin et al. 2005) and significant correlations with AChE activity (Brewer, Little et al. 2001; Sandahl, Baldwin et al. 2005). Additionally, other OPs including fenitrothion, parathion, and methyl parathion, adversely affected a suite of swimming behaviors reviewed by (Little and Finger 1990). One noteworthy study investigated the effects of six pesticides including methyl parathion (OP) and tribufos (OP) on rainbow trout swimming behavior (Little, Archeski et al. 1990). All insecticides adversely affected spontaneous swimming activity, while DEF also reduced swimming capacity in juvenile rainbow trout (Little, Archeski et al. 1990). In bluegill, methyl parathion adversely affected burst swimming behavior at 300 μg/L (Henry and Atchison 1984). Respiratory disruptions, comfort movements, and aggression behaviors in bluegill were all adversely affected by 24 h exposures to methyl parathion at 3.5 μg/L. This suggests that these social behaviors are very sensitive to AChE inhibition (Henry and Atchison 1984).

Two month old juvenile rainbow trout, brook trout, and coho were exposed to malathion (Phillaps Malathion 55%) for 7-10 days depending on species (Post and Leasure 1974). Swimming performance, brain AChE activity, and recovery time were measured following exposure to malathion concentrations of 0, 40, 90, 120 μg/L in brook trout; 0, 55, 112, 175 μg/L in rainbow trout; and 0, 100, 200, 300 μg/L in coho. Additionally, once fish recovered AChE activity, they were subjected to a second exposure to determine if prior exposure altered susceptibility to malathion. Swimming performance and AChE activity did not differ from values of the initial exposure *i.e.,* a second exposure resulted in no evidence of increased susceptibility. Brook trout were the most sensitive based on AChE inhibition followed by rainbow trout and coho salmon, respectively. AChE inhibition of 25% relative to control fish occurred at 40 µg/L (brook trout), 55 µg/L (rainbow trout), and 100 µg/L (coho). Swimming performance was affected at the lowest concentrations tested in each salmonid species and showed a dose-dependent decrease in swimming performance as malathion concentration increased. The data indicated that AChE inhibition of approximately 20-30% resulted in a 5% or less reduction in swimming performance and as inhibition increased, swimming performance decreased. Note, however, that the swimming test conducted in the study is a coarse measure of swimming capacity. Thus, other non-measured swimming activity endpoints would likely be affected at lower concentrations (Little and Finger 1990; Little, Archeski et al. 1990).

In summary, the information presented on swimming behaviors from AChE-inhibiting insecticides provide a weight of evidence that carbamates (and OPs) adversely affect swimming behaviors which can reduce the fitness of affected salmonids.

Assessment endpoints: Olfaction and olfactory-mediated behaviors: Predator avoidance, prey detection and subsequent growth, imprinting of juvenile fish to natal waters, homing of adults returning from the ocean, and spawning/reproduction

Assessment measures: Olfactory recordings (electro-olfactogram), behavioral measurements such as detection of predator cues and alarm response, adult homing success, AChE activity in olfactory rosettes and bulbs, and avoidance/preference

The olfactory sensory system in salmonids is particularly sensitive to toxic effects of metals and organic contaminants. This is likely a result of the direct contact of olfactory neurons and dissolved contaminants in surface waters. Olfactory-mediated behaviors play an essential role in the successful completion of anadromous salmonid life cycles, and include detecting and avoiding predators, recognizing kin, imprinting and homing in natal waters, and reproducing. It is well established that Pacific salmon lose navigation skills when olfactory function is lost and consequently are unable to return to natal streams (Wisby and Hasler 1954).

We located studies that measured olfactory responses of fish to carbofuran and carbaryl. We found no studies with methomyl. Several studies with other carbamates were found, but most were with thiocarbamates which have a different mode of action than *N*-methyl carbamates, *i.e.,* they do not inhibit acetylcholineterase. We do not discuss or use studies with OPs as surrogates for effect to olfaction in salmonids because AChE inhibition does not appear to be the putative mode of action affecting olfaction, although more empirical data are needed to confirm this. Below we discuss the available literature of carbamate effects on fish olfaction.

The olfactory activity of juvenile cutthroat trout appeared unresponsive to carbaryl following 10 second pulses across the olfactory epithelia, and juveniles showed no preference/avoidance to carbaryl (Labenia, Baldwin et al. 2007). No departures relative to unexposed fish were observed at 5, 50, or 500 μg/L carbaryl from neurophysiological recordings *i.e.,* electro-olfactograms. Additionally, in a behavioral avoidance assay, cutthroat trout did not avoid seawater containing carbaryl at 500 μg/L. These results suggest that cutthroat trout do not actively avoid carbaryl and that short term exposures do not affect olfaction at the concentrations tested. We found no other studies that evaluated other salmonids in estuarine conditions or in freshwaters. We ranked these study results as highly relevant to the effects of carbaryl on salmonid olfaction and an olfactory-mediated behavior, avoidance.

In one set of experiments, coho salmon exposed for 30 minutes to three carbamates (carbofuran, antisapstain IPBC, mancozeb) separately, had reduced olfactory ability as well as disruption of normal AChE activity (Jarrard, Delaney et al. 2004). Carbofuran reduced olfaction by 50% (EC50) at 10.4 μg/L; IPBC reduced olfaction at 1.28 μg/L (EC50); and mancozeb reduced olfaction at 2.05 mg/L (EC50). In addition, carbofuran reduced AChE activity in the olfactory receptor at at 200 μg/L (68%), but none of the treatments (0.1, 10, or 200 μg/L) reduced brain or olfactory bulb AChE activity. This could be a result of the limited exposure duration. The authors concluded that too little information exists to develop a causal relationship between AChE inhibition and olfaction (Jarrard, Delaney et al. 2004). The data do show that carbofuran inhibited both olfaction and AChE activity at 10 and 200 μg/L, respectively. We ranked these study results as highly relevant to carbofuran's effect on salmonid olfaction.

Goldfish showed no avoidance of carbofuran up to a concentration of 10 mg/L, but at the onset of the contaminated flow goldfish showed increased and immediate burst swimming responses relative to unexposed fish (Saglio, Trijasse et al. 1996). In the same study, olfactometric tests were performed to assess the influence of 1, 10, and 100 μg/L carbofuran on a suite of behavioral responses to a food extract (a chironomid–containing solution). Sheltering, grouping, nipping, burst swimming, and attraction behaviors were assessed for ten minutes at 4, 8, and 12 h time points. Significant effects to goldfish behavior in the presence of food stimuli were observed as low as 1 μg/L carbofuran. Sheltering activity increased, while attraction decreased in a dose-dependent manner at 4 h across treatment range (1-100 μg/L) relative to unexposed fish. At 10 and 100 μg/L, goldfish nipped more, relative to controls; which may be representative of increased aggression and stress. The authors concluded that olfactory-mediated behaviors in goldfish were adversely affected by sublethal concentrations of carbofuran. We ranked this study as relevant to the assessment endpoint of olfaction, as ecologically relevant olfactory-mediated behaviors were tested with sublethal concentrations of carbofuran. We note that concentrations were not analytically verified and direct correlation of olfactory-affected goldfish behaviors to salmonid behaviors remains uncertain.

The olfactory ability of male Atlantic salmon to detect a female priming hormone was measured following 5 d exposures to carbofuran (1.1, 2.7, 6.5, 13.9, 22.7 μg/L) (Waring and Moore 1997). The response to the female pheromone prostaglandin F2α, by male olfactory epithelium was reduced at carbofuran concentrations as low as 1.0 μg/L., while the threshold for detection was reduced 10-fold. Furthermore, at 2.7 μg/L carbofuran, male fish completely lost priming ability induced by the female pheromone resulting in no increase in milt or plasma steroids at 2.7 μg/L. These results suggest that Atlantic salmon exposed to these concentrations would have difficulty preparing for spawning due to their impaired priming. Reductions in productivity are possible if male fish miss spawning opportunities. We ranked this study as highly relevant to the effects of carbofuran on olfactory–mediated behaviors such as spawning synchronization.

Mixtures containing carbaryl, carbofuran, and methomyl

We located no experiments that tested mixtures of the three a.i.s, nor did we find mixture studies with other *N*-methyl carbamates. Therefore, the potential for mixture toxicity to olfactory endpoints of salmonids remains a recognized data gap. Given the differences in olfactory toxicity in salmonids for carbaryl (no observed toxicity) and carbofuran (inhibited olfaction at 10 μg/L) combined with no available information on methomyl's affect on olfaction, no apparent dose-response pattern of toxicity emerges for the three a.i.s.

Assessment endpoints: Toxic effects in salmonids from consuming contaminated prey
Assessment measures: Survival, swimming performance

A current uncertainty is the degree to which secondary poisoning of juvenile salmonids may occur from feeding on contaminated dead and dying drifting insects. Secondary poisoning is a frequent occurrence with OPs and carbamates in bird deaths (Mineau 1991), yet is much less studied in fish. Uptake, metabolism, and accumulation of carbaryl by a salmonid prey item, *Chironomus riparius* (midge), exposed for 24 h indicated significant uptake over the first 8 h, significant metabolism (more than 85-99%) of parent carbaryl to metabolites and low bioconcentration factors (5-10) (Lohner and Fisher 1990). These results suggest that contaminated prey items, such as aquatic invertebrates, do not accumulate significant carbaryl, and what they do accumulate is likely rapidly metabolized. That said, juvenile salmonids could still get a dose of carbaryl from feeding on drifting, contaminated insects that have not had time to metabolize carbaryl. Juvenile brook trout gorged on drifting insects following applications of carbaryl and AChE activity was reduced (15-34%) in the trout (Haines 1981). However, it is not possible to differentiate the contribution to AChE inhibition from the aqueous and dietary routes because concentrations were not measured in the water, prey, or fish. In another study, resident trout feeding on dying and dead drifting invertebrates (from the pyrethroid cypermethrin) caused a range of physiological symptoms in brook trout: loss of self-righting ability and startle response; lethargy; hardening and haemolysis of muscular tissue similar to muscle tetany; and anemic appearance of blood and gills (Davies and Cook 1993). The possibility that the adverse effects in the trout manifested from exposure to the water column instead of from feeding on contaminated prey was ruled out by the authors as measured field concentrations of pesticides did not produce known toxic responses. In a laboratory feeding study with the OP fenitrothion, brook trout (*S. fontinalis*) were fed contaminated pellets (1 or 10 mg/g fenitrothion for four wks) (Wildish and Lister 1973). Growth was reduced in both treatments. AChE inhibition was measured at 2, 12, and 27 d following termination of contaminated diet treatments. Trout had lower AChE activity than unexposed fish at both treatments, and by 27 d following termination, AChE levels recovered slightly. The treatment concentrations used in this study are very high and indicate that brook trout are not sensitive to diet-induced toxicity of fenitrothion. The experiment did show that AChE inhibition from the diet is possible, yet it is difficult to determine the relative toxicity of carbaryl, carbofuran, and methomyl found in contaminated insects consumed by Pacific salmonids.

Habitat assessment endpoints:

Prey survival, prey drift, nutritional quality of prey, abundance of prey, health of aquatic prey community, and recovery of aquatic communities following N-methyl carbamate exposures

Assessment measures: 24, 48, and 96 h survival of prey items from laboratory bioassays reported as EC/LC50s; sublethal effects to prey items; field studies on community abundance; indices of biological integrity (IBI); community richness; and community diversity.

Death of aquatic invertebrates in laboratory toxicity tests were reported in each of the BEs. Salmonid aquatic and terrestrial prey are highly sensitive to the three carbamate insecticides. Death of individuals and reductions in individual taxa and prey communities have been documented and are expected following applications of *N*-methyl carbamates that achieve effect concentrations-several of which are at the low μg/L levels (Schulz 2004). Complete or partial elimination of aquatic invertebrates from streams contaminated by insecticides has been documented for carbaryl (Muirhead-Thomson 1987) as well as many other insecticides. A review of field studies published from 1982-2003 on insecticide contamination concluded that "about 15 of the 42 studies revealed a clear relationship between quantified, non-experimental exposure and observed effects in situ, on abundance [aquatic invertebrate], drift, community structure, or dynamics" (Schulz 2004). Although the top three insecticides most frequently detected at levels expected to result in toxicity were chlorpyrifos (OP), azinphos-methyl (OP), and endosulfan, the *N*-methyl carbamates carbaryl, carbofuran, oxymyl, and fenobucarb all showed clear or assumed relationships between exposure and effect (Schulz 2004).

Drift, feeding behavior, swimming activity, and growth are sublethal endpoints of aquatic prey negatively affected by exposure to AChE inhibitors (Courtemanch and Gibbs 1980; Haines 1981; Hatakeyama, Shiraishi et al. 1990; Davies and Cook 1993; Beyers, Farmer et al. 1995; Schulz 2004). Drift of aquatic invertebrates is an evolutionary response to aquatic stressors. However, insecticides, particularly carbamates and OPs, can trigger catastrophic drift of salmonid prey items (Courtemanch and Gibbs 1980; Haines 1981; Hatakeyama, Shiraishi et al. 1990; Davies and Cook 1993; Schulz 2004). Some invertebrates may drift actively to avoid pesticides and settle further downstream, which can provide temporary spikes in available food items for feeding salmonids. Catastrophic drift can also deplete benthic populations resulting in long-term prey reduction that may affect salmonid growth at critical time periods. We located no studies that address this line of reasoning directly with Pacific salmonids. Davies and Cook (1993) did show aquatic invertebrate community changes, mortality of invertebrates, drift of dying and dead invertebrates, and affected trout following spraying of a pyrethroid pesticide, cypermethrin, an invertebrate and fish neurotoxicant (Davies and Cook 1993). Effect concentrations were estimated at 0.1-0.5 μg/L cypermethrin. It is difficult to compare these effect concentrations to carbamate insecticides. However, it is illustrative of how insecticides can damage multiple endpoints of an aquatic community including reducing abundance of prey (Davies and Cook 1993).

Several scientific peer-reviewed publications and EPA documents have reviewed aspects of the available information on multi-organism microcosm, mesocosm, and field test results for the AChE insecticides (Leeuwangh 1994; Barron and Woodburn 1995; Schulz 2004; Van Wijngaarden, Brock et al. 2005). Van Wijngaarden et al. (2005) conducted a literature review that listed ecological threshold values (*e.g.,* $NOEC_{eco}$ and $LOEC_{eco}$) for carbaryl, carbofuran, and bendiocarb (another *N*-methyl carbamate) from model ecosystems or "adequate" field studies. A $NOEC_{eco}$ represented "the highest tested concentration at which no, or hardly any, effects on the structure and functioning of the studied model ecosystem were observed. The $LOEC_{eco}$ is the lowest tested concentration at which significant treatment-related effects occurred" (Van Wijngaarden, Brock et al. 2005). Below we discuss some of this information in relation to effects on salmonid prey. The majority of studies were conducted in littoral systems, *i.e.,* ponds, and other static systems, but one study with carbaryl was conducted in a running water (lotic) system (Courtemanch and Gibbs 1980). Population densities of salmonid prey items (*i.e.*, Ephemeroptera, Diptera, Amphipoda, Cladocera, Copepoda, Isopoda, Ostracada, Trichoptera) decline following exposures to AChE-inhibiting insecticide concentrations, including carbaryl and carbofuran (Van Wijngaarden, Brock et al. 2005). Adverse effects to these groups occurred at or below "1 toxic unit"-where a toxic unit equals field concentrations normalized by dividing them by the 48 h EC50 of *Daphnia magna* for a given AChE inhibitor.

We did not locate any microcosm, mesocosm, or field experiments that measured responses of aquatic communities that contained salmonids and salmonid prey simultaneously; a recognized data gap. Several studies evaluated aquatic invertebrate responses to *N*-methyl carbamates (carbaryl, carbofuran, and bendiocarb) in static and running water systems. We found no field studies with methomyl. We summarize open literature studies with aquatic invertebrates organized by insecticide in Table 65. We found no studies that addressed effects of methomyl on aquatic invertebrates in the field or from multispecies microcosms or mesocosms.

The available literature from field experiments indicates that populations of aquatic insects and crustaceans are likely the first aquatic organisms damaged by exposures to carbaryl, carbofuran, and methomyl contamination. Benthic community shifts from sensitive mayfly, stonefly and caddisfly taxa, the preferred prey of salmonids, to worms and midges occur in areas with degraded water quality including from contaminants such as pesticides (Cuffney, Meador et al. 1997; Hall, Killen et al. 2006). Reduced salmonid prey availability correlated to OP use in salmonid bearing watersheds (Hall, Killen et al. 2006). We found no studies that evaluated the effects on carbamates and that made correlations to salmonid bearing watersheds. Subsequent effects to salmonid's growth from reduced prey availability and quality remain untested and are a current data gap.

We located a study addressing impacts on fish growth due to prey reduction. Although this study was conducted on chlorpyrifos, an insecticide not considered in this Opinion, we deemed it highly relevant due to the ecological context it provided. The study indicated that native fathead minnows exposed to chlorpyrifos had reduced growth due to reductions in prey item abundance in littoral enclosures (pond compartments) (Brazner and Kline 1990). The experiment tested the hypothesis that, "addition of chlorpyrifos would reduce the abundance of invertebrates and cause diet changes that would result in reduced growth rates." Nominal, chlorpyrifos treatment concentrations of 0.5, 5.0, and 20 µg/L (chemical analysis of water concentrations provided at 0, 12, 24, 96, 384, 768 h) all resulted in statistically significant reductions in growth at 31 days. A single pulse of chlorpyrifos was introduced into each enclosure at day 0. Invertebrate abundance was determined in each replicate on days -3, 4, 16, and 32. Fathead minnows were sampled from enclosures on days -2, 7, 15, and 31 where fish were weighed, measured, and dissected to determine gut content (dietary items identified). By day 7, significant differences in mean numbers of rotifers, cladocerans, protozoans, chironomids, mean total number of prey being eaten per fish, and mean species richness were greater in fish from the control enclosures than in some of the treatments. By day 15, control minnows were significantly larger than fish from treated levels. These experimental results support the conclusion that reductions in abundance of prey to juvenile fish can result in significant growth effects. It is reasonable to assume that reductions in prey from carbamate insecticides can also result in reduced juvenile salmonid growth and ultimately reduced survival and productivity. The precise levels of prey reduction necessary to cause subsequent reductions in salmonid growth remain a recognized data gap.

Recent declines in aquatic species in the Sacramento-San Joaquin River Delta in California have been attributed in part to toxic pollutants, including pesticides (Werner, Deanovic et al. 2000). Significant mortality or reproductive toxicity in *C. dubia* was detected in water samples collected at 24 sites in the Sacramento-San Joaquin River Delta in California. Ecologically important back sloughs had the largest percentage of toxic samples (14 - 19%). The TIEs identified carbofuran and carbaryl as well as the OPs chlorpyrifos, diazinon, and malathion as the primary toxicants in these samples responsible for the adverse effects (Werner, Deanovic et al. 2000).

Recovery of salmonid prey communities following acute and chronic exposures from carbaryl, carbofuran, and methomyl depends on the organism's sensitivity, life stage, length of life cycle, among other characteristics. Univoltine species will take longer than multivoltine species to recover (Liess and Schulz 1999). Recovery of salmonid prey items such as caddisflies, stoneflies, and mayflies will be slow, considering their long life cycles and infrequent reproduction. Additionally, these species also require clean, cool waters to both recover and maintain self-sustaining populations. In several salmonid-supporting systems these habitats are continually exposed to anthropogenic disturbances, including pesticide contamination, which limits their recovery and can also limit recovery of multivoltine species. For example, urban environments are seasonally affected by stormwater runoff that introduces toxic levels of contaminants and scours stream bottoms with high flows. Consequently, urban environments do not typically support diverse communities of aquatic invertebrates (Paul and Meyer 2001; Morley and Karr 2002).

Similarly, yet due to a different set of circumstances, watersheds with intensive agriculture land uses show compromised invertebrate communities (Cuffney, Meador et al. 1997). Indices of biological integrity (IBI) and other invertebrate community metrics are useful measures of the health of an aquatic community because cumulative impacts of aquatic stressors are integrated over time. The IBI is also valuable because it converts relative abundance data of a species assemblage into a single index of biological integrity (Allan 1995). Salmonid-inhabited watersheds have been assessed using IBIs and other metrics of aquatic community health.

A study on the condition of Yakima River Basin's aquatic benthic community found that invertebrate taxa richness was directly related to the intensity of agriculture *i.e.,* at higher agriculture intensities taxa richness declined significantly both for invertebrates as well as for fish (Cuffney, Meador et al. 1997). Locations with high levels of impairment were associated with high levels of pesticides and other agricultural activities, which together with habitat degradation were likely responsible for poor aquatic conditions in the Yakima (Cuffney, Meador et al. 1997). Salmonid ESUs and DPSs occur in the Yakima River Basin as well as other watersheds where invertebrate community measurements indicate severely compromised aquatic invertebrate communities such as the Willamette River Basin, Puget Sound Basin, and the Sacramento-San Joaquin River Basin.

Table 65. Study designs and results with freshwater aquatic invertebrates

Chemical	Taxa/species	Assessment measures	Concentrations tested	Exposure duration	Effects	Data source
Carbaryl	multiple n= 25; including predators, herbivores, zooplankton	species richness, biomass, abundance, survival	510 μg/L; single application; simulating direct overspray	13 d	Reduced richness of community (by 15%), predators, zoo plankton; reduced biomass of predators; increased abundance of large herbivores; No effect on zooplankton abundance; elimination of diving beetle larvae, *Daphnia pulex*, *D. ambigua*	(Relyea 2005)
Carbaryl	*C. californica* (stonefly) *C. sp.* (mayfly) *A. sp* (mayfly) *B. americanus* (caddisfly) *P. sp.* (caddisfly) early instar *P. sp.* (caddisfly) late instar *L. unicolor* (caddisfly)	Survival; LC1,LC50	1-30 μg/L 4-100,000 μg/L 10-28 μg/L 5-55 μg/L 5-80 μg/L 5-80 μg/L 5- 80 μg/L	96 h	9.0 (LC1); 17.3 (LC50) μg/L 3.0; 11.1 7.5; 20.4 28.8; 41.2 14.8; 30.3 33.8; 61.0 9.5; 29.0	(Peterson, Jepson et al. 2001)
Carbaryl	Coleopteran (beetles), Diptera (flies), Trichoptera (caddisflies), terrestrial insects, Emphemeroptera (mayflies)	salmonid stomach contents	0.5 lbs/acre, 2 applications, 7 days interval	na	Increased numbers of Diptera in stomach after first spray; increased numbers of Emphemeroptera, Trichoptera, Diptera, and terrestrial insects indicating drift; condition factor of fish increased following applications due to increased feeding; fish AChE inhibited 15-34%fish AChE inhibited 15-34%	(Haines 1981)
Carbaryl	*Chironomus riparius* (midge)	Survival (EC50); Uptake rate; Bioconcentration factors (BCFs)	Multiple concentration ranges at pH 4,6,8; temps 12, 20, 30 °C	24 h	EC50s (61-133 μg/L); Lowest EC50 at 30 °C, pH 4 and 6; Highest EC50s at 10°C, pH 4 and 6. Uptake rates increased with temp. BCFs = 5.36-10.12	(Lohner and Fisher 1990)

Chemical	Taxa/species	Assessment measures	Concentrations tested	Exposure duration	Effects	Data source
Carbaryl	*C. californica* (stonefly) *Cinygma sp.* (mayfly)	Survival (EC50)	17.3, 173, 1730 µg/L 10.2, 102, 204, 408, 1020 µg/L	15 min 30 60 15 min 30 60 96 h recovery period	EC50= na EC50= na EC50 = 1139 95% CI (370-15400) µg/L EC50 = 848 no CI EC50 = 220 no CI EC50 = 165 95% CI (124-232)	(Peterson, Jepson et al. 2001)
carbaryl	Stream invertebrate aquatic community	Aquatic invertebrate drift, survival; presence and abundance of riffle invertebrates; recovery of benthic invertebrates	0.75 lbs/ acre; 1 lb/acre	2, 30, 60, 365 day drift sampling post spray; 1,2, 3, 30, 60 day benthic samples	170-fold increase in drift at 2 d. All invertebrates in drift samples dead at 2 d (Plecoptera, Ephemeroptera Diptera were common), at 30 and 60 d drift below prey-spray levels. Within hrs larger stream inverts found dead (Plecoptera, Trichoptera, Emphemerptera), Tricoptera abandoned their cases; at 30 and 60 d significant population declines of salmonid-prey invertebrates compared to pre-spray levels. Total number of organisms not significantly reduced.	(Courtemanch and Gibbs 1980)
carbaryl	*Xathocnemis zealandica* (damselfly)	Emergence of damselflies, % emerged	1, 10, 100 µg/L (concentrations replenished every 12 days at ½ the nominal treatment level)	67 days	90% reduction in emergence at 100 µg/L no significant differences at 1 and 10 µg/L.	(Hardersen and Wratten 1998)

Chemical	Taxa/species	Assessment measures	Concentrations tested	Exposure duration	Effects	Data source
carbaryl	*Daphnia magna* (daphnids)	Population abundance at different phases; Life stage sensitivity (survival, 48 h LC50)	15 µg/L applied as a single pulse at different population phases (treatments): growing, density peak, stable	1 pulse at day 8 to growing treatment (3 replicates), day 11 to density peak treatment , day 25 stable treatment; 48 h	70% decline in abundance during growth phase, 92.6% decline in peak phase, 34.8 % decline in stable phase , compare to a 61% decline in control population; Survival (48h LC50s)	(Takahashi and Hanazato 2007)
carbaryl	Zooplankton pond community	Abundance of zooplankton species	500 µg/L; 1-3 applications to pond mesocosms; 0.1 mg/L	0 – 3 months	At 500 µg/L sustained reductions in daphnid and copepod populations. No observable reductions in rotifer populations. At 100 µg/L, reductions in daphnid population observed, but not with copepods or rotifers	(Hanazato and Yasuno 1990)
carbaryl formulation (43.3% a.i.)	Bluegill sunfish, multiple aquatic species including phytoplankton, macrophytes, zooplankton, and macroinvertebrates	Abundance and richness of species groups; Fish survival and growth	2, 6, 20, 60, 200 µg/L; applications at weekly intervals for 6 weeks	0 – 12 weeks, with 6 weekly pulses beginning at week 10	no long term reductions in major invertebrate groups were observed; no detectable differences in bluegill survival or growth between control and pesticide treatments, carbaryl half life was 30 min, mean pH of treatment groups in application events ranged from 9.2-10.3; mean temperatures ranged from 17 – 27 ◦C, with the majority between 20-24 ◦C.	Registrant-submitted study (1993)

Chemical	Taxa/species	Assessment measures	Concentrations tested	Exposure duration	Effects	Data source
carbaryl formulation (Sevin-4-Oil)	Aquatic macroinvertebrate community; *Platygobio gracilis* (Flathead chub)	Invertebrate drift; AChE inhibition in resident fish	Aerial application in 1991 and 1993 applied at 0.5 lbs carabaryl/acre and 0.4 lbs/acre, respectively	River water sampled at 1, 2, 4, 8, 12, 24, 48, 96 h; AChE inhibition measured at 24 h; Drift sampled 4 times daily for 4 days	1 h average 85 μg/L and declined to 0.100 μg/L by 96 h in 1991; 2 h average 12 μg/L L and declined to 5.14 μg/L by 96 h in 1993; No detectable differences in AChE inhibition at 24 h, low statistical power (0.82 and 1); Note: AChE is expected to recover by 24 h; Increase in coefficient of variation of drift measure from day 1and 2 post application compared to reference site drift in 1991; Similar increase on day 2 in 1993; drift composed primarily of live mayflies (Emphemeroptera)	(Beyers, Farmer et al. 1995)
carbofuran	*Gammarus pulex* (fw amphipod)	*in situ* survival, feeding rates; laboratory bioassays measured 24 h survival (LC50) and feeding rates	Field study: measured stream concentrations in stormwater runoff following an application of 3 kg a.i./ ha; 27 μg/L stream, 256 μg/L in agricultural field ditch Lab study: Feeding rate- 0.75,1.25,2.25,4.0,7.0, 12.0 μg/L	Field study: 5 day; 2 rain events captured Lab study: Survival- 24-, 48- 96-hr Feeding rate- 7 day exposures	Field study: All amphipod chambers showed reduced feeding rates at time during rain events and all amphipods died in 10 experimental chambers Lab study: Survival- μg/L (95% CI) 24 h LC50 = 21 (14.7-30) 48 h LC50 = 12.5 (5.7-27.5) 96 h LC50 = 9 (5.8-13.9) Feeding rate- >3 μg/L feeding rates reduced to 0	(Matthiessen, Shearan et al. 1995)

Chemical	Taxa/species	Assessment measures	Concentrations tested	Exposure duration	Effects	Data source
carbofuran	1 h average 85 μg/L and declined to 0.100 μg/L by 96 h in 1991; 2 h average 12 μg/L L and declined to 5.14 μg/L by 96 h in 1993; No detectable differences in AChE inhibition at 24 h, low statistical power (0.82 and 1); Note: AChE is expected to recover by 24 h; Increase in coefficient of variation of drift measure from day 1and 2 post application compared to reference site drift in 1991; Similar increase on day 2 in 1993; drift composed primarily of live mayflies (Emphemeroptera)	Taxa abundance and biomass	5 μg/L 25 μg/L Note: pH of 9 in experimental pond reduced persistence of carbofuran by increasing the rate of hydrolysis (Chapman and Cole 1982)	Taxa assessed at 5, 11, 32, 55 days post application	No detectable reductions to abundance or biomass of taxa from 5 μg/L; *H azteca*: 10% reduction in abundance; 6% reduction in biomass Chironominae: 77% dead in 48-72 h dip net samples compared to 0% dead in control and 5 μg/L treatment; biomass reduced by 17% at 55 d; evidence that younger larvae were less sensitive than older larvae; No detectable reductions in remaining taxa	(Wayland 1991)
carbofuran	Aquatic invertebrate community in rice fields: ostracada, copepod, cladocera, mosquito larvae, chironomid larvae	Abundance of ostracada, copepod, cladocera, mosquito larvae, chironomid larvae	0.1 kg carbofuran/ha-applied once 0.3 kg carbofuran/ha applied in two regimes: 3, 52, 85, 97 days; 16, 57, 69 days	variable application intervals and frequencies	Ostracada: timing and magnitude of peak abundances not affected by carbofuran, multiple application regimes of 0.3 k/ha significantly reduced abundance compared to one application of 0.1 kg/ha Copepods: significant reductions at 45 and 57 d at 0.3 kg/ha. Cladoceran: rate of population growth significantly affected from 52 and 85 days at 0.3 kg/ha. Abundance lower at 0.1 kg/ha compared to 0.3 kg/ha. Chironomid larvae: isolated significant differences in abundances were found (*P*<0.05), but showed no dose-response relationship Mosquito larvae: No change in abundances	(Simpson, Roger et al. 1994)

Chemical	Taxa/species	Assessment measures	Concentrations tested	Exposure duration	Effects	Data source
carbofuran	*Cerodaphnia dubia* (daphnids)	Survival (24 and 48 h LC50s); survival from mixture of carbofuran and methylpararthion (48 h LC50); Mean young per female and percent survival from 7 day chronic assays	Irrigation field-collected water samples (toxicity identification evaluations employed) Mixture toxicity (1:1, 3:1, 1:3 ratios tested) 0.16, 0.33, 0.65, 1.3, 2.6 μg/L	24, 48 hours 7 days	Carbofuran and methyl-parathion putative agents of toxicity in field-collected samples from CA rice drain. Carbofuran acute toxicity: 24 h LC50 3.4 μg/L (95% CI 2.8-4.3; 48 h LC50 2.6 μg/L (95% CI NA); chronic study- NOEC 1.3 μg/L; LOEC at 2.6 μg/L Strict additive toxicity of combinations tested of carbofuran and methyl-parathion.	(Norberg-King, Durhan et al. 1991)
carbofuran	*Chironomus tentans* (midge)	Survival in spiked-sediments, mixed with atrazine, and field-collected runoff	Spiked sediment concentrations: 25, 45, 83, 150 μg/kg; Mixture concentrations: 25, 45, 80 μg/kg carbofuran + 5, 10, 20 mg/kg atrazine; Field collected sediments in runoff: collected sediments in runoff: 186 μg/kg sediment-associated 73 μg/L pore water	10 days	Spiked sediment results: Nominal [carbofuran] LC50 47.9 μg/kg (95% CI 43.9-52.1); Sediment bound LC50 20.9 μg/kg (95% CI 18.4-33.2); Interstitial water LC50 11.8 μg/L (95% CI 10.9-17.1) Mixture results: Additive toxicity present with atrazine + carbofuran, no synergistic or antagonistic toxicity responses. Atrazine affected survival in the low mg/kg range. Field collected sediments in runoff: No chironomids survived exposure, note control survival was low (57%)	(Douglas, McIntosh et al. 1993)
carbofuran	*Chorophium volutator* (estuarine amphipod)	Survival and avoidance behavior in contaminated spiked sediments	2.5, 25, 250 ng/g (dry weight sediment)	48 and 96 h exposures	48 h LC 50 = 73 ng/g (95% CI 0-128); 48 h LC20 = 41 (95% CI 2-82) ng/g; no effect on avoidance behavior of sediment	(Hellou, Leonard et al. 2009)

Chemical	Taxa/species	Assessment measures	Concentrations tested	Exposure duration	Effects	Data source
carbofuran	*Daphnia magna* (daphnid) *Hydra attenuate* (fw jellyfish) *Macrobrachium rosenbergii* (fw shrimp) Aquatic macroinvertebrate community	Survival (acute LC50); macroinvertebrate community structure	Average concentrations detected 5 d post application 0.7 and 0.16 µg/L maximum carbofuran detection 2.1 µg/L	24 , 48, 72, 96 h exposures to field collected water samples	No acute toxicity reported to tested species. Significant reduction of % EPT taxa and increase in community loss indices at treated sites. Mayflies were primary affected species from carbofuran applications	(Castillo, Martinez et al. 2006)
carbofuran	*Ceriodaphnia dubia (daphnid)* *Neomysis mercedi(* mysid*)* *Pimephales promelas (*fathead minnow*)*	Survival (% mortality)	Not reported; conducted toxicity identification evaluations (TIE)	96 h exposures to field collected water samples	3/47 irrigation samples had 100% toxicity to *C. dubia* attributed to carbofuran	(de Vlaming, DiGiorgio et al. 2004)
carbofuran	Wetland invertebrate aquatic community*:* *Daphnia magna,* *Chironomus riparius*	Survival (EC50) Primary productivity	10, 100, 1,000 µg/L water only 48 h test; 10, 100, 1,000 µg/L mixed with soil and added to microcosms	48- hr; 30 day microcosm test	*D. magna* EC50 = 48 µg/L *C. riparius* EC50 = 56 µg/L Populations of *D. magna* were viable at day 1 in 10 and 100 µg/L, it took 4 d for a viable population (*i.e.,* 5 or more live daphnids) to establish at 1000 µg/L . No dead daphnids or chironomids observed at 14 d and 30 d. No effects to microbial community enzyme activity were found.	(Johnson 1986)

Studies with other AChE inhibitors on salmonid prey items:

Robust evidence shows that salmonid prey taxa and communities can be substantially reduced following exposures to the OP insecticides chlorpyrifos, diazinon, and malathion. NMFS reviewed these data and presented its findings in a biological opinion (NMFS 2007d). We use these findings to show the types of aquatic community responses following exposures to AChE inhibiting insecticides. The toxic potency of a pesticide is a function of concentration and duration of exposure, which in turn is a function of a pesticide's physical properties and interactions of the pesticide with environmental variables such as temperature, pH, sunlight, soil micro-organisms, etc. With this in mind, if carbaryl, carbofuran, and methomyl are at concentrations individually (or together in mixtures) expected to reduce salmonid prey communities, we infer a similar magnitude of response (reductions in abundance from death and catastrophic drift) and similar recovery period for those observed in affected aquatic communities treated with OP insecticides.

We note that individual aquatic invertebrates exhibiting adverse sublethal responses from carbamates will likely recover much more quickly than those exposed to OPs (Kallander, Fisher et al. 1997). The midge, *Chironomus tentans*, showed complete recovery of AChE activity in 24 h following two 1 h pulses of carbaryl separated by 24 h. Additionally, two, 1 h pulses of carbamates (carbaryl, carbofuran, aldicarb, propoxur) caused significantly fewer symptoms of intoxication than 2 h of continuous exposure when chironomids were given 2 to6 h of recovery in clean water between doses (Kallander, Fisher et al. 1997). In contrast, sublethal exposures to OPs were equally toxic when exposures are either pulsed or continuous.

Reviews of field, mesocosm, and microcosm studies with the three OPs document reductions in aquatic invertebrate populations and lengthy recovery times for populations of some taxa (Giesy, Solomon et al. 1998; Van Wijngaarden, Brock et al. 2005). A recent study found significant changes to macroinvertebrate assemblages of artificial stream systems following a 6 h exposure to chlorpyrifos at 1.2 μg/L (Colville, Jones et al. 2008). The addition of chlorpyrifos to the artificial streams resulted in a rapid (6 h) change in the macroinvertebrate assemblages of the streams, which persisted for at least 124 d after dosing (Colville, Jones et al. 2008). The chlorpyrifos dissipated from the system within 48 h (Pablo, Krassoi et al. 2008), however the macroinvertebrate community did not recover rapidly. Several species similar to salmonid prey items were significantly affected.

Zooplankton and insect taxa appeared the most sensitive in studies with diazinon. In particular, the salmonid prey taxa Trichoptera, Diptera, and Cladocera were highly sensitive (Giddings, Hall et al. 2000). Field studies in salmonid habitat also show reductions in salmonid prey abundances. For example, in listed steelhead habitat in the Salinas River, California, abundances of the salmonid prey items including mayfly taxa, daphnids, and an amphipod (*Hyalella azteca*) were significantly reduced downstream of an irrigation return drain compared to upstream (Anderson, Hunt et al. 2003; Anderson, Hunt et al. 2003; Anderson, Phillips et al. 2006). Diazinon and chlorpyrifos were detected above acute toxicity thresholds in surface waters and sediments. Combined toxicity of the two OPs using a toxic unit approach correlated strongly with mortality of daphnids. For *H. azteca*, acute toxicity was attributed to sediment pore-water concentrations of chlorpyrifos (Anderson, Hunt et al. 2003). Other pesticides, including carbaryl, were likely present and responsible for some of the toxicity in the Salinas River,. In a subsequent study on the Salinas River, TIE demonstrated that chlorpyrifos and diazinon were responsible for the observed death of the daphnid *C. dubia* (Hunt, Anderson et al. 2003). These data support the line of evidence that field concentrations of OPs can and do adversely affect aquatic invertebrates in salmonid habitats. It is reasonable to assume that the same situation occurs with carbamates, given the toxicity of the compounds and the similar mode of action.

Adjuvant toxicity

Assessment endpoints: Survival of fish and aquatic prey items, endocrine disruption in fish

Assessment measures: 24, 48, 96 h LC50s, and vitellogenin levels in fish plasma

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

We queried EPA's ECOTOX online database and retrieved 707 records of nonylphenol's (NP) acute toxicity to freshwater and saltwater species. The lowest reported LC50 for a salmonid was 130 μg/L for Atlantic salmon. Aquatic invertebrates, particularly crustaceans, were killed at low concentrations of NP, with the lowest reported LC50 = 1 μg/L for *H. azteca*. These data indicate that a wide array of aquatic species are killed by NP at μg/L concentrations. We also queried EPA's ECOTOX database for sublethal toxicity and retrieved 689 records of freshwater and saltwater species tested in chronic experiments. The lowest fish LOEC reported was 0.15 μg/L for fathead minnow reproduction. Numerous fish studies reported LOECs at or below 10 μg/L. Additionally, salmonid prey species are sensitive to sublethal effects of NP at low μg/L concentrations. The amphipod, *Corophium volutator*, grew less and had disrupted sexual differentiation (Brown, Conradi et al. 1999). Multiple studies with fish indicated that NP disrupts fish endocrine systems by mimicking the female hormone 17β-estradiol (Brown and Fairchild 2003; Arsenault, Fairchild et al. 2004; Madsen, Skovbolling et al. 2004; Jardine, MacLatchy et al. 2005; Luo, Ban et al. 2005; McCormick, O'Dea et al. 2005; Segner 2005; Hutchinson, Ankley et al. 2006; Lerner, Bjornsson et al. 2007; Lerner, Bjornsson et al. 2007). NP induced the production of vitellogenin in fish at concentrations ranging from 5-100 μg/L (Hemmer, Bowman et al. 2002; Ishibashi, Hirano et al. 2006; Arukwe and Roe 2008; Schoenfuss, Bartell et al. 2008). Vitellogenin is an egg yolk protein produced by mature females in response to 17-β estradiol, however immature male fish have the capacity to produce vitellogenin if exposed to estrogenic compounds. As such, vitellogenin is a robust biomarker of exposure. A retrospective analysis of an Atlantic salmon population crash suggested the crash was due to NP applied as an adjuvant in a series of pesticide applications in Canada (Fairchild, Swansburg et al. 1999; Brown and Fairchild 2003). Additionally, processes involved in sea water adaptation of salmonid smolts are impaired by NP (Madsen, Skovbolling et al. 2004;

Jardine, MacLatchy et al. 2005; Luo, Ban et al. 2005; McCormick, O'Dea et al. 2005; Lerner, Bjornsson et al. 2007; Lerner, Bjornsson et al. 2007).

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

Summary of Response Analysis:

We summarize the available toxicity information by assessment endpoint in Table 66. Data and information reviewed for each assessment endpoint was assigned a general qualitative ranking of either "low", "medium", or "high." To achieve a high confidence ranking, the information stemmed from direct measurements of an assessment endpoint, conducted with a listed species or appropriate surrogate, and was from a well-conducted experiment with stressors of the action or relevant chemical surrogates. A medium ranking was assigned if one of these three general criteria was absent and low ranking was assigned if two criteria were absent. Evidence of adverse effects to assessment endpoints for salmonids and their habitat from the three a.i.s was prevalent. However, much less information was available for other ingredients, in part, due to the lack of formulation information provided in the BEs as well as the statutory mandate under FIFRA for toxicity data on the a.i.s to support registration. We did locate a substantial amount of data on one group of adjuvants/surfactants, the NP ethoxylates. However, we received and located minimal information for the majority of tank mixes and other ingredients within formulations.

Table 66. 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, medium, high)
Carbaryl			
Fish:			
-survival (LC50)	yes	250-4,500	high
-growth	no	8,17,62,210, 680	medium
-reproduction	yes	680	medium
-swimming	yes	10-12,500	high
-olfactory-mediated behaviors	no	5, 50, 500	high
Habitat: -prey survival (LC50)	yes	1.5-35[1]	high
Carbofuran			
Fish:			
-survival (LC50)	yes	88-3,100	high
-growth	yes	56.7	medium
-reproduction	yes	6	low
-swimming	yes	5, 100	medium
-olfactory-mediated behaviors	yes	1-200	high
Habitat: -prey survival (LC50)	yes	1.6[2]-2,700	high
Methomyl			
Fish:			
-survival (LC50)	yes	300-7,700	high
-growth	yes	142, 243	medium
-reproduction	no	31-500	low
-swimming	-	-	-
-olfactory-mediated behaviors	-	-	-
Habitat: -prey survival (LC50)	yes	8.8-920	high
Other ingredient:			
Nonylphenol (NP)			
Fish:			
-survival (LC50)	yes	130 - >1,000	high
-reproduction	yes	0.15 - 10	high
-smoltification	yes	5 - 100	medium
-endocrine disruption	yes	5.0 – 100	high
Habitat: -prey survival (LC50)	yes	1- >1,000	high
Additive toxicity of *N*-methyl carbamates	yes	multiple	high
Synergistic toxicity of *N*-methyl carbamates	yes	multiple	high

[1] reported 48 h EC50s of *D. magna* are low and high values of the range (Takahashi and Hanazato 2007)
[2] 48 h survival EC50 of *Chironomus tentans* (Karnak and Collins 1974)

Risk Characterization

In this section we integrate our exposure and response analyses to evaluate the likelihood of adverse effects to individuals, populations, species, and designated critical habitat (Figure 38). We combined the exposure analysis with the response analysis to: 1) determine the likelihood of salmonid and habitat effects occurring from the stressors of the action; 2) evaluate the evidence presented in the exposure and response analyses to support or refute risk hypotheses; 3) translate fitness level consequences of individual salmonids to population-level effects; and 4) translate habitat-associated effects to potential impacts on PCEs of critical habitat. The risk characterization section concludes with a general summary of species responses from population-level effects. We then evaluate the effects to specific ESUs and designated critical habitat in the *Integration and Synthesis* section.

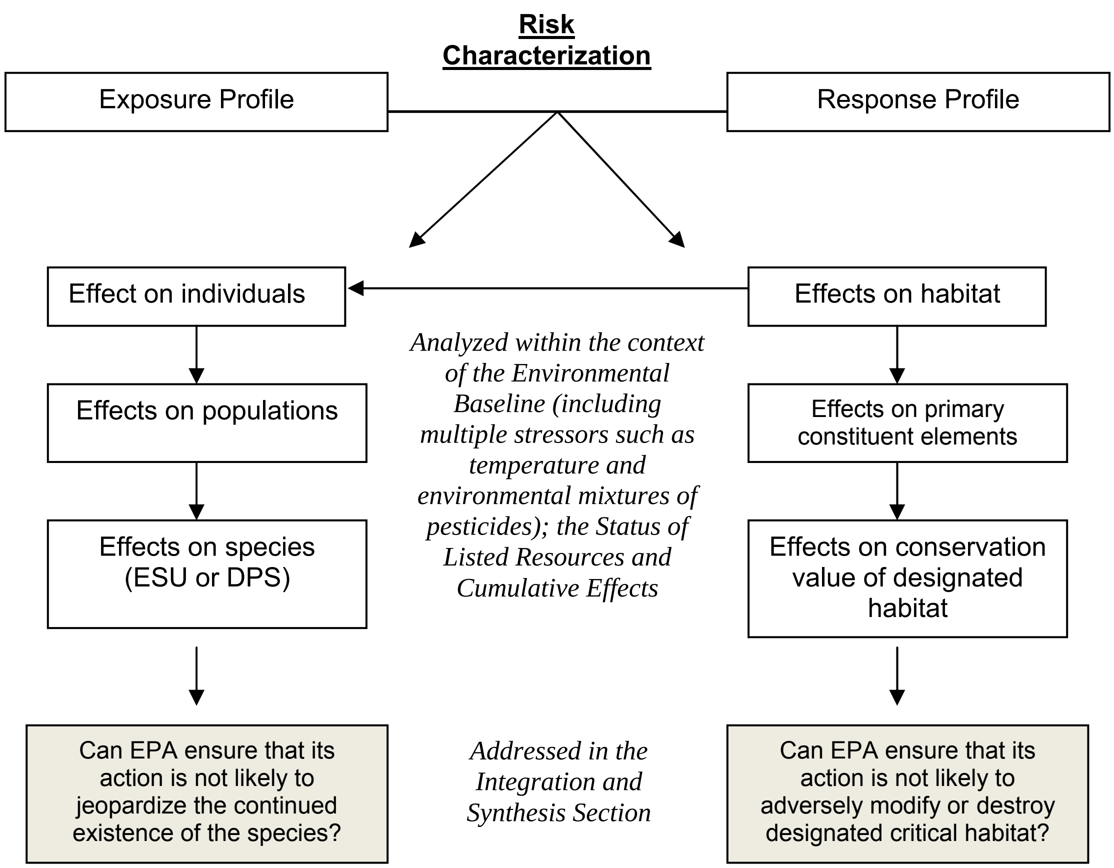

Figure 38. Schematic of the Risk Characterization Phase

Exposure and Response Integration

In Figures 39-41, we show the overlap between exposure estimates for the three carbamates and concentrations that affect assessment endpoints. The figures show the exposure concentration ranges (minimum – maximum values) gleaned from the three predominant sources of exposure data we analyzed: monitoring data; EPA's estimates presented in the BEs that represent crop uses; and NMFS' modeling estimates for off-channel habitats. None of the modeled exposure estimates were derived for non-crop use. This is a major data gap as carbaryl is used extensively in urban and residential areas. The effect concentrations are values taken from the toxicity data reviewed in the *Response Analysis Section*. With respect to the assessment endpoint survival, recall that the effect concentrations are LC50s, thus death of sensitive individuals is not represented by this metric and can occur at concentrations well below LC50s. Additionally, we cannot accurately predict at what concentrations death first occurs because dose-response slope information was generally not provided for most of the acute lethality studies. Although we are unable to determine at what concentration an individual organism might die, we do incorporate survival endpoints from acute 96 h studies using a default slope in a population modeling exercise discussed below. This slope is recommended by EPA when more relevant information is unavailable (EPA 2004). Where overlap occurs between exposure concentrations and effect concentrations NMFS explores the likelihood of adverse effects. If data suggest exposure exceeds adverse effects thresholds, we discuss the likelihood and expected frequency of effects based on species information and results of the exposure and response analyses.

This is a coarse analysis because it does not present temporal aspects of exposure nor does it show the distribution of toxicity values. However, it does allow us to systematically address which assessment endpoints are affected from carbaryl, carbofuran, and methomyl exposure. Where significant uncertainty arises, NMFS highlights the information and discusses its influence on our inferences and conclusions. We discuss the uncertainties related to these endpoints under associated risk hypotheses later in this section.

Carbaryl

Concentration ranges overlap with most of the assessment endpoint ranges indicating that adverse effects are expected in salmonids if exposed for sufficient durations (Figure 39). Prey survival appears to be the most sensitive endpoint as all three ranges encompass this endpoint and the maximum concentration values far exceed the prey survival range. Swimming is likely impaired at the higher end of the concentration range from monitoring data, at the middle and higher range from EPA estimates, and throughout the range predicted in the off-channel habitat estimates. Concentrations occurring in the off-channel habitats that NMFS modeled will kill juvenile salmonids. Furthermore, given the LC50 values for salmonids following 96 h exposures, we expect that fewer deaths of juveniles will occur in many of the freshwater habitats exposed to carbaryl. Fish reproduction (based on a single fathead minnow study) would be affected by concentrations in the off-channel habitat. We also note minimal data exist on effects to fish growth: only a single study with fathead minnows. We expect that carbaryl will impair swimming of salmonids, kill salmonid prey, and in certain circumstances kill salmonids when exposed for sufficient durations. The effect concentrations shown in the figure do not account for the potential enhanced toxicity of carbaryl to salmonids or their prey items in aquatic habitats where other AChE inhibitors are present. We also note that pH is a major factor in carbaryl's persistence in aquatic habitats. At pHs above 8, carbaryl breaks down fairly rapidly (half-life of 24 h) while at pHs less than 8 carbaryl is much more resistant to hydrolysis (half-life of 1- 30 d for pH of 7.9 – 5.7). The pH of natural surface waters commonly ranges from 7 to 9, thus pH is an important consideration when evaluating toxicity of carbaryl.

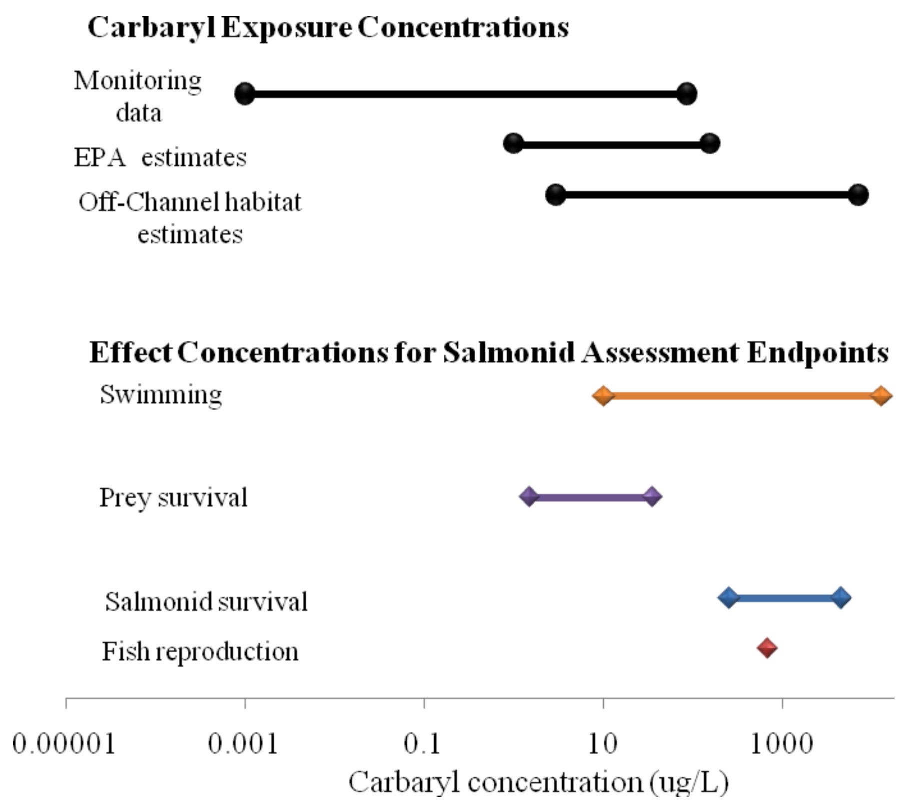

Figure 39. Carbaryl exposure concentrations and salmonid assessment endpoints' effect concentrations in µg/L

Carbofuran

Concentration ranges overlap with most of the assessment endpoint ranges, indicating that adverse effects are expected in salmonids if exposed for a sufficient duration (Figure 40). Salmonid prey items appear just as sensitive to carbofuran as to carbarylalthough significant variation among prey species was observed in reported acute survival EC/LC 50s. Fish reproduction, swimming and salmonid olfactory-mediated behaviors are encompassed or exceeded by all three exposure ranges. The fish growth value (56 µg/L LOEC) was exceeded by the off-channel habitat concentrations. Salmonid survival was the least sensitive response reviewed. As with carbaryl, concentrations of carbofuran occurring in the off-channel habitats that NMFS modeled will kill juvenile salmonids. Furthermore, given the LC50 values for salmonids following 96 h exposures, we expect that fewer deaths of juveniles will occur in many of the freshwater habitats exposed to carbofuran. As with carbaryl, we note that pH is a major factor in the degree of toxicity of carbofuran. At pHs above 9, carbofuran breaks down fairly rapidly (half-life of 0.8-15 h) while at pH 7 reported half-lives range from 2- 28 d. Therefore, pH is an important consideration when evaluating toxicity of carbofuran in the aquatic environment. We discuss these effects and other considerations in more detail under the risk hypotheses.

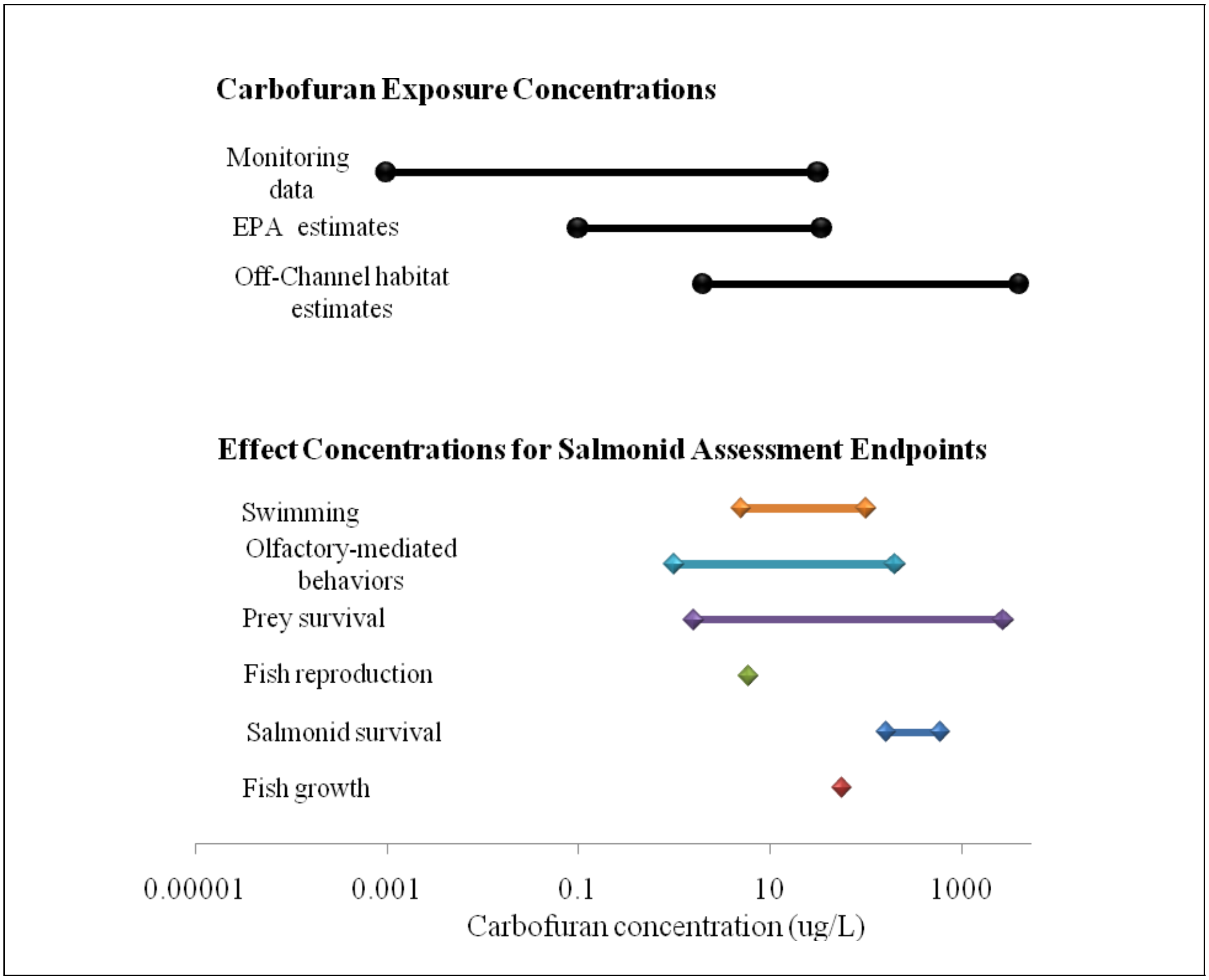

Figure 40. Carbofuran exposure concentrations and salmonid assessment endpoints' effect concentrations in µg/L

Methomyl

We expect a portion of fish to be exposed to upper end concentrations which would overlap with most of the effect thresholds. (Figure 41). As with carbaryl and carbofuran, prey survival was the most sensitive endpoint assessed. More sensitive prey, with EC50s at the lower end of the distribution, are expected to die based on all three exposure estimate ranges, with the highest rates of mortality based on EPA estimates and NMFS' off-channel habitat estimates. The magnitude of reduction in prey abundance will depend on which taxa are present in the exposed water body and the actual concentrations and exposure durations. If reductions in prey abundance, especially of the smaller organisms, coincide with fry feeding for the first time following yolk sac absorption, starvation is likely. We discuss this in greater detail in the risk hypotheses below. The lower value of the fish reproduction endpoint, 31 µg/L, overlapped with both EPA's and NMFS' off-channel habitat estimates, and the highest value in the monitoring data was greater. A notable data gap is the absence of information on methomyl's toxicity to olfactory-mediated behaviors. Although we found no studies that evaluated swimming, we expect swimming impairment at concentrations less than LC50s because swimming is impaired with other AChE inhibitors and AChE inhibition correlates to impaired swimming (Little and Finger 1990; Little, Archeski et al. 1990).

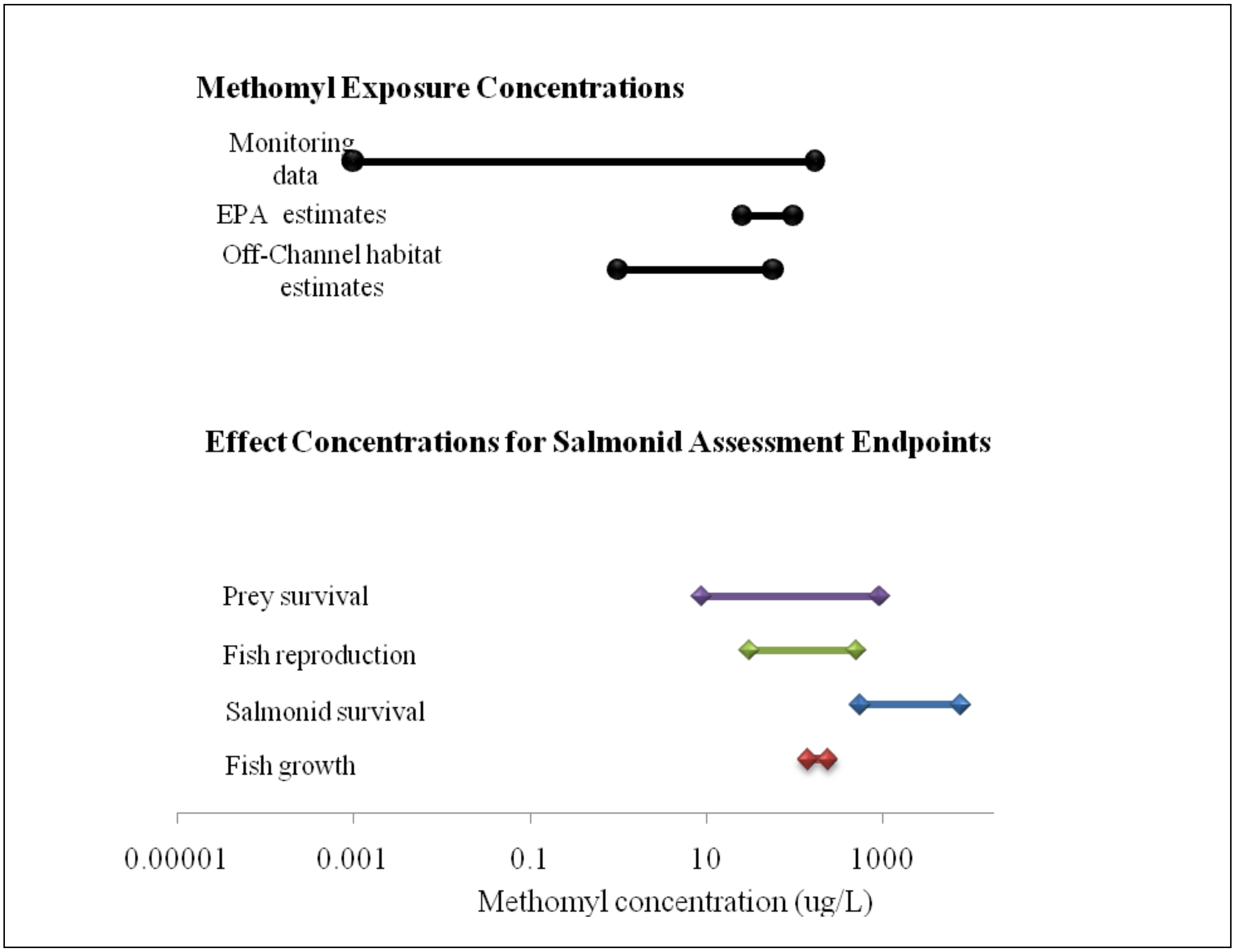

Figure 41. Methomyl exposure concentrations and salmonid assessment endpoints' effect concentrations in µg/L

Relationship of pesticide use to effects in the field

Schulz (2004) reviewed 45 field and *in situ* studies published in peer-reviewed journals (from 1982-2003) that evaluated relationships between insecticide contamination and biological effects in freshwater aquatic ecosystems and included invertebrates and fishes. For each study, the authors classified the relationship of exposure to effect in one of four categories: no relation, assumed relation, likely relation, and clear relation based on the cited authors' judgment of their own results. A relationship was classified as clear only if the exposure was quantified and the effects were linked to exposure temporally and spatially. The review concluded that "about 15 of the 42 studies revealed a clear relationship between quantified, non-experimental exposure and observed effects *in situ*, on abundance [aquatic invertebrate], drift, community structure, or dynamics" (Schulz 2004). Although the top three insecticides most frequently detected at levels expected to result in toxicity were chlorpyrifos (OP), azinphos-methyl (OP), and endosulfan, the *N*-methyl carbamates carbaryl, carbofuran, oxymyl, and fenobucarb all showed clear, likely or assumed relationships between exposure and effect (Schulz 2004). No studies involving methomyl were evaluated. It should be noted that these studies were not designed to establish effect thresholds and in our review, are not sufficient to define thresholds. However, the data do provide information on concentrations of insecticides known to cause biological and ecological effects under field conditions.

Schulz (2004) noted that for all of the studies "that seem to establish a clear link between exposure and effect, the pesticide concentrations measured in the field were not high enough to support an explanation of the observed effects simply based on [laboratory bioassays] acute toxicity." Some authors have suggested differences in field measured exposure and actual organismal exposure (aquatic, sediment, dietary; environmental variables that affect exposure) as a reason for higher mortalities *in situ* than predicted by laboratory toxicity data. Schulz concluded that on the basis of present knowledge, it cannot be determined whether the measured concentration in the field regularly underestimates the actual exposure or if a general difference between the field and laboratory reactions of aquatic invertebrates is responsible; both are reasonable assertions. The review by Schulz shows a body of evidence that natural aquatic ecosystems can be adversely affected by AChE inhibitors including carbaryl and carbofuran. We expect a similar relationship for methomyl because it shares the same mechanism of action as carbaryl and carbofuran (AChE inhibition) and concentrations that lead to death of aquatic invertbrates are similar to carbaryl and carbofuran.

We reviewed multiple studies that showed direct, adverse effects to salmonid prey species following exposures to *N*-methyl carbamates (Table 67). These included reduction in individual aquatic species, changes in community abundance, richness, and diversity with salmonid prey items.

Table 67. . Examples of published field studies designed to establish a relationship between *N*-methyl carbamate contamination of aquatic habitats and agricultural practices (adapted from Table 2 in Schulz 2004).

Source	Concentration µg/L	Duration	Endpoint	Species	Relationship of exposure and effect
Carbaryl					
Aerial application	0.1-85.1	24	aquatic invertebrate Drift	Various invertebrates	Likely
Carbofuran					
	0.05-26.8	Few hours	mortality	Amphipod (*Gammarus pulex*)	Clear

Field studies in ESA-listed salmonid habitats with other AChE inhibitors

A group of field studies evaluated macroinvertebrate community responses in the orchard-dominated Hood River Basin, Oregon and correlated results with chlorpyrifos and azinphos-methyl use and detections (Grange 2002, St. Aubin 2004, Van der Linde 2005). Hood River Basin contains several listed anadromous salmonids, including lower Columbia River steelhead. The goals of the studies were to determine whether in-stream OPs affected steelhead AChE activity and changed the aquatic macroinvertebrate community. An additional objective addressed how changes in macroinvertebrate community might affect salmonid growth. A suite of reference and orchard-dominated sampling sites within the Hood River Basin were sampled pre and post the two primary application seasons, spring (chlorpyrifos) and summer (azinphos-methyl). Significant differences in macroinvertebrate community assemblages were found between upstream reference sites and downstream agricultural sites (St. Aubin 2004), similar to the results described in a California stream (Hall, Killen et al. 2006). However, no significant differences were found at each individual site, before and after summer spraying (St. Aubin 2004). Therefore, the second Hood River study investigated the spring spray events as well as the summer spray events to determine seasonal effects (Van der Linde 2005). Sharp declines in species abundance between reference sites and downstream sites during the spring-spray period correlated to chlorpyrifos applications and subsequent aquatic detections (one site over an 8 d period showed chlorpyrifos ranging from 0.032 -0.183 µg/L). There were more pollutant tolerant taxa and less intolerant taxa at the agricultural sites (Van der Linde 2005). Collector-gatherer species, many of which are salmonid prey items, declined rapidly at agricultural sites compared to abundances at the reference sites following applications. Interestingly, reductions in biodiversity in 2001 agricultural sites compared to reference sites was not seen in 2002 (Van der Linde 2005). The authors commented that diversity metrics do not always behave consistently or predictably in response to environmental stress. More than two years of data are likely needed to more sufficiently address community variability at this site.

Two sets of field experiments directly investigated juvenile steelhead (hatchery- reared) AChE activity from caged-fish studies in an agricultural basin in Hood River Basin, OR (Grange 2002; St. Aubin 2004). The studies analyzed water samples for chlorpyrifos, azinphos-methyl, and malathion before, during, and after orchard spray periods. One of the studies also monitored the aquatic invertebrate community's response (discussed later under prey effects) in conjunction with the AChE inhibition (St. Aubin 2004). Steelhead from reference sites had statistically significantly greater AChE activity than steelhead from orchard-dominated areas. The reductions in AChE activity corresponded to the application seasons and detections of chlorpyrifos and azinphos-methyl insecticides.

The data from one study indicated that OP-insecticides inhibited AChE activity in steelhead held in cages in the Hood River Basin and this inhibition correlated to chlorpyrifos and azinphos-methyl detections and to a lesser degree with malathion detections (Grange 2002). None of the pesticides were detected at reference sites and both chlorpyrifos (range in maxima of (0.077-0.196 µg/L) and azinphos-methyl were frequently detected at orchard stream and river sites. AChE activity was inhibited up to 21% in smolts, and 33% in juveniles relative to reference locations. Temperature was a confounding factor as lower temperatures showed lower AChE activity while higher temperatures showed higher AChE activity at reference sites. The authors normalized data to temperature and found a greater number of statistically significant reductions in AChE in steelhead. Study results show that steelhead in these systems exposed to OP insecticides lose AChE activity (up to 33%) which, depending on the percentage of inhibition, can manifest into fitness level consequences (Grange 2002; St. Aubin 2004).

The field studies conducted in Hood River Basin, Oregon show that salmonids' AChE activity was reduced in streams near orchards during chlorpyrifos and azinphos-methyl applications.

Additionally, the macroinvertebrate communities in these systems were compromised to such an extent that there was a reduction in salmonid prey abundance. These findings highlight the importance of characterizing the presence of AChE insecticides, *N*-methyl carbamates and OPs, within aquatic habitats because the compounds share a mode of action and adverse biological responses to mixtures containing multiple AChE insecticides are likely cumulative. These data can serve as a surrogate for expected effects of the aquatic community when concentrations of carbaryl, carbofuran, and methomyl attain effect thresholds.

Field studies in ESA-listed salmonid habitats: Willapa Bay and Grays Harbor Washington

History of Carbaryl use in Washington estuaries

Willapa Bay and Grays Harbor support large commercial oyster-producing areas (approximately 600 acres in Willapa and 200 acres in Grays Harbor) and are located north of the Columbia River along the southwestern coast of Washington. The Pacific oyster (*Crassostrea gigas*), introduced from Japan in the 1920s, is the principal species cultivated in those estuaries (Feldman, Armstrong et al. 2000). Two species of endemic shrimp, ghost shrimp (*Neotrypaea californiensis*) and mud shrimp (*Upogebia pugettensis*), are abundant in the oyster growing areas of these estuaries and create burrows in the sediment that are unfavorable for optimal oyster cultivation. The Washington State oyster industry has used aerial applications of carbaryl to kill the burrowing shrimp in oyster beds since the early 1960s. EPA authorizes the use of carbaryl on oyster beds in Washington through a SLN label, or Section 24(c) (EPA Reg. No. 264-316). Carbaryl is applied directly to intertidal areas when they are exposed at low tide, primarily by helicopter (Dumbauld, Brooks et al. 2001). This use has been controversial and there have been numerous studies dating back several decades that evaluated the ecological impact of carbaryl use in these two estuaries (Feldman, Armstrong et al. 2000). Although the EPA label does not limit applications to Willapa Bay and Grays Harbor, historically these have been the only areas treated with carbaryl. Additionally, Washington State requires applicators to obtain NPDES permits for applications to aquatic habitats.

ESA-Listed Pacific salmonids that occur in Willapa Bay and Grays Harbor

Juvenile and adult Chinook salmon from ESA-listed populations are expected to be present in Willapa Bay and Grays Harbor at the time of carbaryl applications. It is expected that juvenile

Chinook salmon from the Lower Columbia River use these estuaries during the period when carbaryl is applied based on their behaviors (Personal communication Eduardo Casillas NMFS 2/18/2009, Anne Shaffer WDFW 3/04/2009, and Thom Hooper NMFS, 2/12/2009). Specifically, their use of dendritic channels to access inundated mud flats suggests juvenile Chinook salmon will be exposed to peak carbaryl concentrations in the water column and also from feeding on live and dead contaminated prey. Also, recent information (six years of coastline surveys) shows that juvenile Columbia River Chinook salmon "bounce" north along the Washington coastline and south along the Oregon coastline in the nearshore surf zones (Personal communication Eduardo Casillas NMFS 2/18/2009) and have been found occupying nearshore habitats as far North as the Strait of Juan de Fuca (Shaffer, Crain et al. submitted). In one study, juvenile Chinook salmon captured along the coast Washington State were genetically analyzed to determine natal river origins. Of the fish collected for genetic analysis, 45% of juvenile Chinook salmon collected in Crescent Bay, 75% of those in Freshwater Bay, and 60% of those in Pysht Bay were from ESA-listed stocks of the Columbia River (Shaffer, Crain et al. submitted).

DNA analysis revealed that adult salmonids harvested in Willapa Bay in July and August, when carbaryl is typically applied, include several ESA-listed species: Lower Columbia River Chinook salmon, California Coastal Chinook salmon, Puget Sound Chinook salmon, and Snake River Fall-run Chinook salmon (Kassler and Marshall 2004). Given this information we also expect these same species of Chinook salmon will be present in Grays Harbor as Grays Harbor is located just north of Willapa Bay. Further, we expect adults of these stocks may be exposed to carbaryl given that they occupy the estuary during the application periods and oyster plots occur throughout the Bay. It is possible that chum salmon from the Columbia River also occur in the Willipa Bay and Grays Harbor estuaries. However, we are unaware of documentation of their occurrence at those locations, and unlike juvenile Chinook salmon from the Columbia River, juvenile chum have not been found in recent sampling efforts of nearshore surf zones. We therefore conclude that Columbia River ESA-listed chum are unlikely to be present in Willapa Bay or Grays Harbor.

Measured Concentrations in Willapa Bay and Grays Harbor following Carbaryl Applications

Numerous monitoring studies have been conducted in coordination with applications of carbaryl to control burrowing shrimp in commercial oyster beds in Washington State. The results of the studies are quite variable, as are the study designs and objectives. Several investigations have documented water column concentrations exceeding several mg/L (which is substantially higher than the μg/L concentrations observed in freshwater aquatic habitats) following the first flood tide post-application (Hurlburt 1986; Creekman and Hurlburt 1987; Tufts 1989; Tufts 1990). Water column concentrations measured at Willapa Bay in 1984 detected a mean concentration of 10.6 mg/L upon initial flooding of the treated area (Hurlburt 1986). Concentrations were monitored at varying depths for several hours following flooding. The concentrations decreased rapidly but were detectable in the low μg/L range throughout sampling. In 1985 samples were collected along transects from treated areas following the direction of the tide to monitor the dispersal of carbaryl (Creekman and Hurlburt 1987). The data indicate that carbaryl is transported off the treated area by the tide. Average concentrations measured above the treated area ranged from 2.4 – 5.5 mg/L. The highest average concentration (7.9 mg/L) was collected 510 ft from the treated area. Average concentrations at the most distant sampling point (650 ft from initial treated area) were 2.5 mg/L. In 1996, further study evaluated carbaryl movement off the treated spray area (Tufts 1989). Samples were collected along a transect corresponding to the direction of the incoming tide. The peak concentration measured was 27.8 mg/L. Over one treated area, the concentration of carbaryl decreased from 13.2 mg/L to 9.3 mg/L as the water depth increased from 1.5 to 10 inches. The concentration further decreased to 600 μg/L when the water rose to 16 inches. The monitoring indicated concentrations of carbaryl decreased with increasing distances along the transect, but concentrations greater than 1 mg/L were detected in several instances at distances several hundred feet from treated plots. Carbaryl was found in the water column at the detection limit (0.1 mg/L) as far as 1,725 ft from the treated area.

In 1987, surface water monitoring was conducted in Willapa Bay to determine "the dilution pattern of carbaryl washed from sprayed oyster beds, and to determine carbaryl concentrations in shallow pools and streams where marine fish might be found" (Tufts 1990). Carbaryl concentrations in the mg/L range were frequently detected with the incoming tide. Peak concentration for one sample site was 18.8 mg/L upon initial flooding. The concentrations decreased to 0.2 mg/L when covered by 18 inches of water. Concentrations of carbaryl were variable among sites. For example, at one of the sampling sites carbaryl was not detected until the water was 7 inches deep. A peak concentration of 8 mg/L was detected at this site when the water depth was 11 inches. This sample station was located 300 ft from a treated area. Maximum carbaryl concentrations detected at other sites were 17.4, 7.8, and 4 mg/L with samples collected at depths of 6, 4, and 13 inches from the bottom. Carbaryl's primary toxic degradate, 1-naphthol, was detected at concentrations as high as 1.4 mg/L. Average concentrations of carbaryl detected in tide pools and small streams ranged from 3.6 to 11.2 mg/L. The 11.2 mg/L detection was associated with application of 5 lbs carbaryl/acre. An average concentration of 7.8 mg/L in tide pools and streams was associated with an application rate of 4 lbs carbaryl/acre, approximately half the maximum rate allowed. Similarly, a more recent study found a peak of 820 µg/L in the water column 50 ft from the application site following a typical treatment application at the maximum rate of 8 lbs/acre (n=3) (Weisskopf and Felsot 1998).

Several studies demonstrate that carbaryl dissipates fairly rapidly from over the treatment site due to degradation, metabolism, dilution, and off-site transport (Hurlburt 1986; Creekman and Hurlburt 1987; Tufts 1989; Tufts 1990; Weisskopf and Felsot 1998). In 2006 and 2007, samples collected at the mouth of channels adjacent to treated oyster beds in Willapa Bay had maximum concentrations of 29.1 µg/L after 6 h (high tide), 38 µg/L after 12 h (low tide), and 21.1 µg/L after 24 h (low tide) (Major, Grue et al. 2005).

The NPDES permit for Willapa Bay and Grays Harbor requires annual monitoring of water column concentrations in treated areas. It specifies an acute effluent limit of 3 µg/L and a chronic limit of 0.06 µg/L. However, those data are of highly questionable value because the monitoring plan specifies that monitoring is suspended for the first 48 h following application to assess the acute effluent, and further suspends monitoring for 30 days from the last application to assess the chronic limit. We expect that most of the carbaryl will be degraded and transported to other locations by the time carbaryl monitoring is initiated.

Carbaryl sprayed on mud flats can be transported substantial distances at concentrations that may have ecological impacts. Researchers found that close to 100% of Dungeness crabs were killed up to 100 m off the carbaryl application area (Doty, Armstrong et al. 1990). Levels decrease to below 1 mg/L when transported more than 200m or more. Washington DOE reports that carbaryl concentrations in the "potential effects threshold range" of 0.1-0.7 μg/L have been detected at locations several miles from oyster beds soon after large areas were treated (Johnson 2001).

Measured Concentrations in Invertebrates

Samples of crustaceans from carbaryl treated areas in 1984 showed high tissue levels (Table 68). A single ghost shrimp and Dungeness crab were analyzed and contained concentrations of 24.9 and 41.9 mg/kg (ppm), respectively (Hurlburt 1986). Analysis of dead shrimp in 1985 following treatment at 7.5 lbs carbaryl/acre revealed average concentrations of approximately 4.5 mg/kg. When left on the treated oyster beds the concentrations declined to approximately 10% of the initial concentration 24 h post-treatment, then remained relatively stable at the 48, 72, and 96 h sampling events. Concentrations in shrimp and annelid worms were investigated following the 1985 applications and associated with different application rates (Tufts 1989). Average concentrations in shrimp ranged from 5.3 to 13.8 mg/kg. The concentration in worms ranged from 57-58.6 mg/kg. The 10 lb application rate is less relevant as the current label restricts applications to 8 lbs a.i./acre. Concentrations in other, more relevant salmonid prey items were apparently not assessed although we assume they may be comparable for other organisms that occupy similar habitat.

Table 68. Carbaryl concentrations in aquatic organisms on treated sites (mg/kg).

Year	Dungeness Crab	Burrowing Shrimp	Annelid Worms
1984	41.9	24.9 (rate not reported)	-
1985	-	4.5 (7.5 lb a.i./acre)	-
1986	-	13.8 (10 lb a.i./acre) 8.7 (7.5 lb a.i./acre) 5.3 (5 lb a.i./acre)	75.5 (5 lb a.i./acre) 57 (7.5 lb a.i./acre) 58.6 (5 lb a.i./acre)

Ecological Effects

There have been concerns that carbaryl applications in Willapa Bay and Grays Harbor may have adverse effects to the commercial Dungeness crab fishery (Feldman, Armstrong et al. 2000). Consequently, Washington Department of Fisheries conducted several surveys to estimate the number of crabs killed (Hurlburt 1986; Creekman and Hurlburt 1987; Tufts 1989; Tufts 1990).

Transect surveys conducted on treatment beds the day following applications indicate large numbers of crabs are killed by the applications (Table 69). Other acute mortalities to non-target organisms were generally not reported. However, transect surveys were conducted during 1986 and 1987 due to concerns over potential impacts to fish given staff observations from the Washington Department of Fisheries reported that fish mortality was "routinely" noted, but not quantitatively assessed following applications of carbaryl (Tufts 1989). Mortalities were characterized as small fish apparently trapped in shallow pools during low tide and directly exposed during carbaryl treatments. The surveys indicate several thousand fish were killed each year from carbaryl applications to approximately 400 acres at rates of ≤ 7.5 lbs a.i./acre (Table 70). Currently, the NPDES permit allow for treatment of up to 600 acres. The current 24 (c) label does not specify acreage restrictions and allows for applications of carbaryl up to 8 lbs a.i./acre.

Table 69. Estimated dungeness crab mortalities resulting from carbaryl applications in Willapa Bay and Grays Harbor, Washington

Year	Maximum Application Rates*	Total Acres Treated**	Dungeness Crab Killed
1984	≤ 10 lbs a.i. / acre	490	38,410
1985	≤ 7.5 lbs a.i./acre	391	59,933
1986	≤ 7.5 lbs a.i./acre	398	16,286
1987	≤ 7.5 lbs a.i./acre	434	44,053

* Current 24(c) label allows for application of up to 8 lbs a.i./acre
** Current NPDES permit allows for a total of 600 hundred acres to be treated in Willapa Bay and Grays Harbor, WA. The Label places no restrictions on acres or geographic restrictions for use of carbaryl on oyster beds in Washington.

Given that the fish that reside in standing water on mud flats are likely to be the most vulnerable to carbaryl exposure, staff from the Washington Department of Fisheries also estimated the available marine fish habitat that existed on exposed mudflats of treated areas. They characterized water that was at least 2 inches in depth, a depth that is adequate for salmon fry, as marine habitat. This habitat comprised substantial portions of the treated areas. In 1986, surveys indicate 135 acres of the 398 of the treated area (approximately 34%) were marine fish habitat (Tufts 1989). In 1987, 67 of the 434 (15%) acres were characterized by Washington Department of Fisheries as marine fish habitat. The current NPDES permit specifies that there be a 200 ft buffer zone for sloughs and channels when carbaryl is applied by helicopter. A 50 ft buffer is required for those aquatic habitats when carbaryl is applied by hand sprayer. AgDrift estimates for aerial application at an application rate of 8 lbs a.i./acre with a 200 ft buffer indicate the average initial concentration from drift to a body of water that is 2 inches deep (~ 5 cm) and 10 m wide would be 771 μg/L. It seems highly unlikely that helicopter applications would be able to successfully avoid direct overspray of some of the marine fish habitats on the exposed mudflats. Additionally, the 24(C) labels do not contain buffer zone restrictions. A direct overspray of 2 inches of water would result in an average initial concentration of approximately 18 mg/L, a concentration comparable to measured concentrations associated with initial tidal inundation of treated mud flats (see surface water detections above). Acute exposure to these concentrations are expected to kill a portion of exposed salmonids in a matter of hours *i.e.,* 24 h LC50s for carbaryl range from 948-8,000 μg/L, n= 60 (Mayer and Ellersieck 1986), and significantly reduce AChE activity that will lead to myriad sublethal effects such as impaired swimming.

Table 70. Estimated fish mortalities resulting from carbaryl applications in Willapa Bay and Grays Harbor, Washington

Year	Maximum Application Rates*	Total Acres Treated**	Fish Killed
1986	≤ 7.5 lbs a.i./acre	398	14, 954
1987	≤ 7.5 lbs a.i./acre	434	8,041

* Current 24(c) label allows for application of up to 8 lbs a.i./acre
** Current NPDES permit allows for a total of 600 hundred acres to be treated in Willapa Bay and Grays Harbor, WA. The Label places no restrictions on acres or geographic restrictions for use of carbaryl on Oyster beds in Washington.

Salmonids were not reported in the sampling data, which was taken over two seasons. The authors report that the fish kills were extremely variable and unpredictable. The fish mortality reported for transect surveys included only four species in 1986: saddleback gunnel, threespine stickleback, staghorn sculpin, and arrow goby. In 1987, mortalities included English sole, sand sole, kelp greenlings, shiner perch, saddleback gunnels, staghorn sculpin, and arrow goby.

Response of benthic community

Samples were collected in 1985 to assess impacts of carbaryl on the intertidal invertebrate community (Tufts 1990). Treated area showed decreases in numbers of infaunal benthic crustaceans at 15 and 30 d post-treatment while a concurrent increase in crustacean abundance was observed on control plots (Table 71). Abundance of benthic crustaceans was low on both control and treated plots 5 months after application in December. The authors suggest the data may represent a seasonal decline in benthic crustaceans.

Table 71. Total number of benthic crustaceans (tanaids, cumaceans, amphipods, copepods, and ostracods; adapted from Tufts 1990).

Date	Sampling	Control Plot (change from pretreatment)	Treated Plot (change from pretreatment)
June 30	Pretreatment	75	33
July 16	15 days post-treatment	96 (+28%)	2 (-94%)
August 01	30 days post-treatment	136(+181%)	4 (-82%)
December 10	161 days post-treatment	2	6

Field incidents reported in EPA incident database

NMFS reviewed reported incidents of fish deaths from field observations throughout the U.S. because the information reflects real world scenarios of pesticide applications and corresponding death of freshwater fish. We recognize that much of the information is not described in sufficient detail to attribute an incident to a label-permitted use leading to the death of fish, or to make conclusions regarding the frequency of fish kills that may be associated with the use of pesticides. NMFS uses the information as a component to evaluate a line of evidence- whether or not fish kills have been observed from labeled uses of the three pesticide products. EPA categorizes incidents in the database into one of five levels of certainty: highly probable, probable, possible, unlikely, or unrelated. The certainty level indicates the likelihood that a particular pesticide caused the observed effects. EPA uses the following definitions to classify fish kill incidents:

- Highly probable (4): Pesticide was confirmed as the cause through residue analysis or other reliable evidence, or the circumstances of the incident along with knowledge of the pesticides toxicity or history of previous incidents give strong support that this pesticide was the cause.

- Probable (3): Circumstances of the incident and properties of the pesticide indicate that this pesticide was the cause, but confirming evidence is lacking.

- Possible (2): The pesticide possibly could have caused the incident, but there are possible explanations that are at least as plausible. Often used when organisms were exposed to more than one pesticide.
- Unlikely (1): Evidence exists that a stressor other than exposure to this pesticide caused the incident, but that evidence is not conclusive.

- Unrelated (0): Conclusive evidence exists that a stressor other than exposure to the given pesticide caused the incident.

NMFS reviewed several incident reports provided by EPA from OPP's incident database. This database is populated with reports received by EPA from registrants that are defined as reportable under FIFRA 6(a)(2) and includes other information received from registrants and other sources.

Relatively few incidents involving lethality to fish were documented in the database, considering the length of time these products have been registered and their toxicity profiles. EPA provided the following table summarizing known mortality incidents (Table 72). NMFS is uncertain as to how representative this database is of known fish kills and the level of coordination that has occurred with various state and federal agencies that investigate these incidents. The mortality events discussed above for carbaryl applications in Willapa Bay and Grays Harbor were not recorded in the database. Several of the incidents provided by EPA are discussed in more detail below.

Table 72. EPA summary of field incident data with carbaryl, carbofuran, and methomyl; highlighted incidents (*) are discussed.

Carbaryl		Carbofuran		Methomyl	
Incident ID	**Certainty**	**Incident ID**	**Certainty**	**Incident ID**	**Certainty**
B0000-246-01*	Probable	I000165-052	Possible	B0000-216-19	Unlikely
B0000-501-92*	Probable	I000599-008*	Probable	B0000-501-36	Unlikely
I000799-003	Possible	I001605-003	Probable	I000108-001*	Probable
I000910-001	Unlikely	I005416-001	Highly Probable[1]		
I007720-020	Probable	I012265-001	Possible		
I015419-644	Possible	I012265-003	Probable		
I018436-001	Possible	I012265-004	Possible		
I013436-001*		I013436-001*	Possible	I013436-001*	Possible

[1]Waterfowl mortality incident in flooded rice

Incident I013436-001: A report of a large fish kill on October 16, 2001, was investigated by the California Department of Fish and Game. The kill involved several thousand fish. Although salmonids were not identified, it occurred in the San Joaquin River, which is used by listed salmonids. EPA characterized this event as "possible" regarding its association with carbaryl, carbofuran, and methomyl. Methomyl was detected in fish gills and several other pesticides were detected in the water including azinphos-methyl, another cholinesterase inhibitor, and several chlorinated hydrocarbons (chlordane, DDT, dieldrin, methoxychlor, mirex, and others). It is likely that the kill was in part due to mixture toxicity, particularly with the combinations of carbamates and OP insecticides.

Incident I000599-008: A report by the California Department of Fish and Game indicates approximately 3,000 fish, 4,000 crayfish and frogs, 200 birds, and at least 5,000 invertebrates were killed on North Temple Creek in San Joaquin County, California. The adjacent vineyard had been treated with Furadan 4F (carbofuran). Chemical analysis confirmed carbofuran in the gizzards of starling up to 9.8 mg/kg, crayfish collected on site had 0.6 mg/kg, the stomach contents of great egrets, which were primarily crayfish, contained 0.3 mg/kg, and surface water had detections up to 7,800 μg/L. The California Department of Fish and Game concluded that the preponderance of evidence indicated carbofuran was the cause of death. EPA characterized the cause of death as probable.

Incident B0000-246-01: According to a 1975 report by EPA, 22,000 catfish were killed in 7 miles of stream at an agricultural site in Oklahoma. Minimal information on the incident was reported. Chemical analysis was not conducted but EPA classified this association with carbaryl as probable. Other a.i.s were identified as possibly contributing to the kill were toxaphene, methyl parathion, and endrin.

Incident B0000-501-92: EPA concluded that a fish kill in a New Jersey pond was caused by drift of carbaryl following the application of Sevin in 2000. This incident was characterized as probable. No chemical analysis was conducted.

FMC reported several incidents called in to their emergency telephone number in the second quarter of 1992. One incident reported a small number of dead bluegill observed in a drain pond adjacent to corn and tobacco fields. The caller indicated Furadan 15G (carbofuran), Counter (chlorpyrifos), and Temik (aldicarb) had been applied to the crops. Chlorpyrifos is an AChE inhibiting organophosphate insecticide. Aldicarb and carbaryl are both cholinesterase inhibiting *N*-methyl carbamates. Dead fish were observed following heavy rains. No analysis was done on the fish to determine the cause of death. EPA characterized carbofuran as a “possible” cause of the incident. It seems likely that the three cholinesterase inhibiting insecticides may have resulted in a cumulative exposure that was sufficient to cause the fish kill due to mixture toxicity.

Incident I000108-001: DuPont reported a fish kill in 1992 in southern Georgia that occurred following an application of Lannate LV (methomyl). One hundred and twenty-five fish were found dead in a pond located 50 to 75 yards from a large field of sweet corn. The 200 acre field was treated with Lannate LV five times between June 1 and June 16. Additionally, Lorsban insecticide (chlorpyrifos) was applied four times during the same interval including co-applications of four fertilizer treatments. On the day of the fish kill (June 16), applications of both Lannate and Lorsban were made. Lannate LV was applied at rates of 1.5 pints/acre and Lorsban was applied at a rate of 1 pint/acre. During the first 16 days of June a total of 10.52 inches of rain were received. The State of Georgia took water samples on June 17. Analysis indicated 136 µg/L of methomyl in the pond. Chlorpyrifos concentrations were not reported and it is not known if it was an analyte. Water temperatures were relatively high (reported to be about 90ºF) but dissolved oxygen was reported to be “normal (7.2-10)”. DuPont suggested the high temperatures, fertilizers, and suspended solids could be stressful to fish and low oxygen content at night might have played a part in the deaths. The species involved in the incident (carp and bluegill) were warm water fishes. EPA characterized methomyl as a “probable” cause of the fish kill. Given the reported rapid dissipation of methomyl it is likely that the concentrations measured do not represent peak concentrations that occurred in the pond. It seems likely that chlorpyrifos may have also been a contributing factor in this incident.

Mixture Analysis of Carbaryl, Carbofuran, and Methomyl

As noted earlier, pesticides most often occur in the aquatic environment as mixtures. In our review and synthesis of the available exposure and response information, we find the three *N*-methyl carbamates carbaryl, carbofuran, and methomyl share the same mechanism of toxic action (AChE inhibition) and are expected to co-occur in salmonid habitats. Therefore, we employ a simple mixture analysis derived from empirical data with Pacific salmonids to predict potential effects to individual salmonid’s AChE activity and their survival from short-term exposures. The analysis is predicated on the toxic potencies of the three insecticides added together to predict the resulting cumulative effect to AChE activity and mortality.

Mixture toxicity is typically described by three general responses: antagonistic, additive, or synergistic. Antagonism and synergism are where the toxic response is not predicted by the individual potencies of the pesticides found in the mixture. Antagonistic effects of a mixture lead to less than expected toxicity on the organismal endpoint. Mechanistically, the pesticides are likely interacting with one another to reduce the toxic potency of individual pesticides. Synergistic effects of a mixture lead to a greater than expected effect on the organismal endpoint and the pesticides within the mixture enhance the toxicity of one another. The third general type of mixture toxicity and the one most frequently reported is additivity (known also as dose-addition or concentration-addition). This type of response is defined by adding the individual potencies of pesticides together to predict the effect on the biological endpoint. Additivity has been demonstrated for many pesticide classes as well as other organic compounds such as PAHs, PCBs, and dioxins.

Additive toxicity of chemicals that share a mode or mechanism of toxic action is well established in the scientific literature, and as a result has been informing regulatory decisions for more than a decade. In 1996, the National Academy of Sciences recommended a dose-additive approach to assessing risks to human infants and children from pesticide exposure. EPA currently assesses human risk of pesticide mixtures for pesticides that share a common mechanism of toxic action *e.g., N*-methyl carbamates [such as carbaryl, carbofuran, methomyl], organophosphorus insecticides, chloroacetanilide and triazine herbicides, as mandated by FQPA. The analysis EPA conducts is predicated on additive toxicity and applies dose-addition to set tolerance limits of pesticide residues on food. For example, the toxic potencies of the *N*-methyl carbamates are added together to determine pesticide tolerance limits for edible crops. Although additive toxicity is evaluated when determining risk to humans, EPA OPP has yet to apply a similar approach to address cumulative toxicity of pesticides that share a common mode or mechanism of action in the evaluation of terrestrial and aquatic species. That said, the use of dose-addition for mixtures containing acetylcholinesterase-inhibiting pesticides is well established and has been extended to protection of aquatic life (Belden, Gilliom et al. 2007).

Dose-addition assumes the cumulative toxicity of the mixture can be predicted from the sum of the individual toxic potencies of each component of the mixture. Within the past decade, government regulatory bodies started to use dose-addition models to predict toxicity for those chemicals that share a common mode of action. In California, the CVRWQB used dose-addition models (based on the toxic-unit approach) to develop TMDLs for the OP insecticides diazinon and malathion. NMFS Biological Opinions have also recognized the environmental reality of co-occurring pesticides in species' aquatic habitats and applied additive toxicity models to predict potential responses of salmonids (NMFS 2004; NMFS 2005; NMFS 2005; NMFS 2005; NMFS 2008).

In salmon, dose-additive inhibition of brain AChE activity by mixtures of OPs and carbamates was demonstrated *in vitro* (Scholz, Truelove et al. 2006). More recently, it has been found that salmonid responses to OP and carbamate mixtures vary *in vivo*; some interactions were synergistic, rather than just additive (Laetz, Baldwin et al. 2009). We used the dose-addition method to predict responses by applying the modeling exposure estimates and measured concentrations of carbaryl, carbofuran, and methomyl presented in the *Exposure Analysis*. Effects of the three carbamates individually and in combination on AChE inhibition Figure 42 (A) and survival (B), are shown below. Based on additivity, the mixture is expected to be more toxic than the individual carbamates for both endpoints. Due to the steep slopes of the two dose-response curves, and especially the mortality slope, small changes in concentrations elicit large changes in observed toxicity. The exposure values represent concentrations from EPA PRZM-EXAMs 60 d average modeling estimates for surface waters (carbaryl: 4 aerial applications at 5 lbs a.i./acre, citrus in FL; carbofuran: 1 ground application at 2 lbs a.i./acre, artichokes in CA; methomyl: 3 aerial applications at 1.8 lbs a.i./acre, peaches). We recognize that this approach is likely to under-predict toxicity for mixtures that produce synergistic rather than additive responses (Laetz, Baldwin et al. 2009).

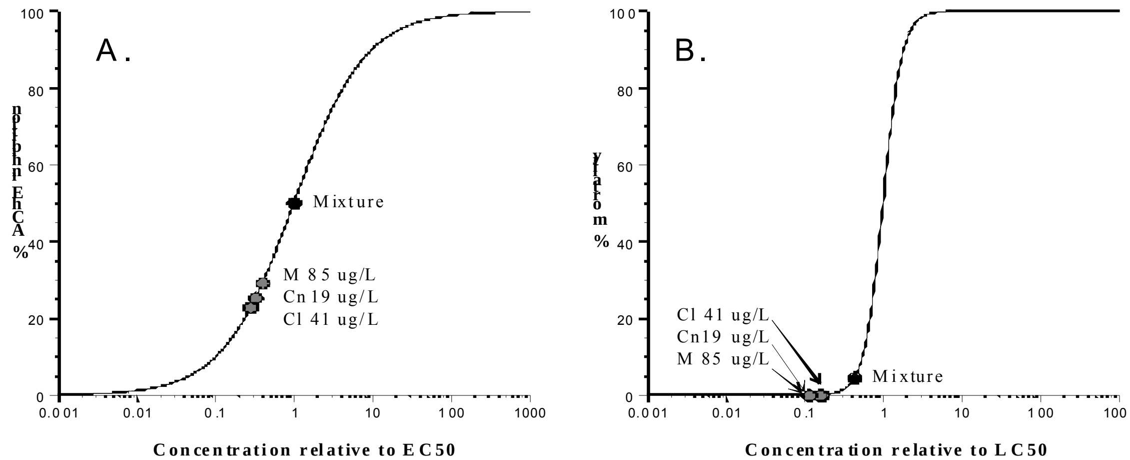

Figure 42. Percent AChE inhibition (A.) and percent mortality (B.) for salmonids expected from exposure to carbaryl (Cl), carbofuran (Cn), and methomyl (M) as separate constituents and as mixtures (Cl 41 μg/L, Cn 19 μg/L, and M 85 μg/L)[4].

We used a variety of exposure estimates and monitoring data to evaluate responses to different mixtures of carbaryl, carbofuran, and methomyl (Table 73). The predicted additive responses from these mixtures ranged from 16-74% inhibition of AChE and 0.01-74% mortality. The predicted additive response to AChE inhibition is likely to result in increased behavioral consequences to salmonids. What is not captured in these responses is the likelihood of exposure to the various mixture concentrations. The PRZM-EXAMS values were estimates selected from EPA simulations of western crops. The scenarios were representative of use rates and numbers of applications on current product labels. Additionally, we used 60 d, time-weighted averages of exposure rather than predicted peak concentrations as exposure to multiple pesticides would be expected to occur more frequently over chronic durations. This may underestimate effects as responses assumed 96 h exposure. Site specific considerations will also have an influence on the frequency of exposure.

[4] EPA's default pesticide slope was used for acute mortality (3.63 or probit slope of 4.5) [EPA 2004]. The slope used for AChE inhibition was based on pooling data from five cholinesterase-inhibiting insecticides; including carbofuran, carbaryl, chlorpyrifos, diazinon, and malathion Laetz, C. A., D. H. Baldwin, et al. (2009). "The synergistic toxicity of pesticide mixtures: Implications for risk assessment and the conservation of endangered Pacific salmon." Environmental Health Perspectives **117**: 348-353..

Table 73. Predicted AChE inhibition and mortality from estimated and measured exposure to carbaryl, carbofuran, and methomyl.

	Concentration (µg/L)	% AChE Inhibition	% Mortality
Modeling: PRZM-EXAMS 60-day averages[1] (from Table 49)			
Carbaryl	12	8.34	0.00
Carbofuran	19	25.39	0.04
Methomyl	81	28.33	0.09
Additive response		**44.30**	**1.38**
Modeling: GENEEC 90-day averages (from Table 50)			
Sweet Corn			
Carbaryl	229	60.67	42.10
Carbofuran	53	47.67	1.63
Methomyl	49	19.61	0.01
Additive response		**72.24**	**73.62**
Potatoes			
Carbaryl	101	41.28	3.59
Carbofuran	106	63.93	17.02
Methomyl	34	14.66	0.00
Additive response		**71.95**	**59.44**
Citrus			
Carbaryl	280	65.25	60.14
Methomyl	21	9.76	0.00
Additive response		**66.25**	**62.97**
Monitoring: NAWQA maxima in 4 states (from Table 54)			
Carbaryl	33.5	19.59	0.07
Carbofuran	32.2	36.02	0.27
Methomyl	0.82	0.48	0.00
Additive response		**44.22**	**1.79**
Monitoring: California CDPR database maxima (from Table 55)			
Carbaryl	8.4	6.07	0.00
Carbofuran	5.2	8.93	0.00
Methomyl	5.4	2.85	0.00
Additive response		**15.58**	**0.01**
Monitoring: Washington EIM database maxima (from Table 56)			
Carbaryl	10	7.09	0.00
Carbofuran	2.3	4.29	0.00
Methomyl	0.17	0.11	0.00
Additive response		**10.60**	**0.00**

[1]PRZM-EXAMS estimates for carbaryl in California peaches (2 applications at 7 lb a.i./acre), carbofuran in California artichokes (1 application at 2 lb a.i./acre), and methomyl in lettuce (10 applications at 0.9 lb a.i./acre). Although current labeling may not be consistent for these uses (peaches, artichokes, and lettuce), the use rate and number of applications are consistent with other labeled uses within the action area.

The GENEEC estimates are 90 d, time-weighted averages that were based on labeled uses of the three compounds in sweet corn, potatoes, and citrus. We found no restrictions that would prevent co-application or sequential applications of carbaryl, carbofuran, and methomyl. The application rates assumed were consistent with current labels and generally representative of use rates authorized for many crop and non-crop uses. An exception was the assumed carbaryl use rate in citrus (12 lb a.i./acre), which is substantially higher than the maximum use rate approved for most crops.

The NAWQA, CDPR, and EIM monitoring values represent the maximum concentrations found in the respective databases. The values cited were all measured within the four states and the vast majority is from waters that contain or drain to listed salmonid habitats. Most of the detections in these, and other monitoring studies that did not target specific applications of the three chemicals occurred at or below the µg/L level. We expect that exposure at these levels will be common in drainages where the three products are used extensively.

Evaluation of Risk Hypotheses: Individual Salmonids

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. We also highlight general uncertainties and data gaps associated with the data. In some instances there may be no information related to a given hypothesis. If the evidence supports the hypothesis we determine whether it warrants an assessment either at the population-level, or affects PCEs to such a degree, to warrant an analysis on the potential to reduce the conservation value of designated critical habitat.

1. Exposure to carbaryl, carbofuran, and methomyl is sufficient to:

A. Kill salmonids from direct, acute exposure.

A large body of laboratory toxicity data indicates that anadromous salmonids die following short-term (< 96 h) exposure to the three insecticides at levels above 200 µg/L. We expect concentrations of carbaryl, carbofuran, and methomyl in salmonid off-channel 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 off-channel habitats during aerial applications or severe runoff events death is possible, particularly from carbaryl and carbofuran. The available monitoring data, if representative of salmonid habitats, indicated that concentrations rarely achieve LC50 values for the three compounds in freshwaters. However, it is unlikely that peak concentrations are reflected in the monitoring data, and given the acutely toxic nature of carbamates, a brief exposure would be sufficient to cause effects. As described in the *Exposure Analysis*, monitoring data are limited when compared to the range of habitats used by salmonids. Few data were found that targeted applications and subsequent concentrations in edge of field habitats which typically show much higher concentrations than weekly, monthly, or seasonal monitoring efforts. Although we found no information on egg survival following acute exposures, we do not expect death of eggs from these insecticides as entry into the eggs via the water column is unlikely. Further support for acute lethality to fish is found in field incidences of death attributed to carbaryl, carbofuran, and methomyl that EPA ranked as "probable". We located several incidents showing death of freshwater fish following exposures to *N*-methyl carbamates. We expect juveniles of listed salmonids to be at the highest risk of death when in small freshwater off-channel and edge habitats, and secondarily in estuarine habitats where carbaryl is applied directly to mudflats. In conclusion, the available information on measured and expected concentrations of the three insecticides supports this hypothesis. We translate the fitness level consequences of reduced survival from mortality of juvenile salmonids to potential population-level consequences using population models (see population modeling section below).

B. Reduce salmonid survival through impacts to growth.

Fish growth is reduced following long-term exposures to carbofuran and methomyl in fathead minnows and rainbow trout, respectively. EPA reported on a single test that measured growth effects to fathead minnows following 30 and 60 d exposure to carbaryl at 0, 8, 17, 62, 210, and 680 µg/L (Carlson 1971). No statistically significant effects on growth were reported in this study. It is difficult to extrapolate from this one study with a warm water species to potential growth effects to ESA-listed salmonids especially given that salmonids appear substantially more sensitive to carbaryl's acute toxicity than fathead minnows (96 h LC50s for salmonids range from 250-4,500µg/L carbaryl and for fathead minnows 7,700-14,600 µg/L). Additionally, 20% and 50% inhibition of AChE in salmonids occurs at concentrations as low as 23 and 185 µg/L, respectively (Labenia, Baldwin et al. 2007) and this inhibition was found to affect swimming behavior (Labenia, Baldwin et al. 2007). Reduced growth occurred at 56 μg/L for carbofuran and 142 μg/L for methomyl. Only one test result was reported for each pesticide. We did not identify any studies that provided a quantitative relationship between growth and fish survival in the field or lab. However, there is abundant literature that shows salmonids that are smaller in size have reduced first year survival (Appendix 1). Additionally, exposure to sublethal concentrations of other AChE inhibitors (chlorpyrifos and diazinon) for acute durations does cause reduced feeding success, which likely results in impacts to growth (Scholz, Truelove et al. 2000; Sandahl, Baldwin et al. 2005). We expect that juvenile fish exposed to carbaryl, carbofuran, and methomyl during their freshwater residency will feed less successfully, resulting in lower growth rates and reduced sizes. Exposure concentrations will likely vary temporally and spatially for salmonids depending on life history, pesticide use, and environmental conditions. The available information support that growth is likely reduced where salmonids are exposed to more than 56 μg/L carbofuran and to concentrations below LC50 for carbaryl and methomyl, although the exact concentrations that may cause growth reduction are currently unknown. The weight of evidence supports the conclusion that fitness level consequences from reduced size are likely to occur in individual salmonids exposed to the three *N*-methyl carbamates. Therefore, we address the potential for population-level repercussions due to reduced growth using species-specific population models.

C. Reduce salmonid growth through impacts on the availability and quantity of salmonid prey

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 three carbamates. These primarily involved evaluating laboratory experimental results that reported on incidences of death or sublethal effects. We located a total of 10, 4, and 14 survival estimates (24, 48, 72, and 96 h EC/LC50s) for carbaryl, carbofuran, and methomyl, respectively. Based on an evaluation of the assessment endpoints, we found a robust body of exposure and toxicity data that indicated salmonid aquatic prey are highly sensitive and affected by expected exposures to each of the insecticides as well as from mixtures containing the three insecticides. We expect death and a variety of sublethal effects to salmonid prey items.

The second line of evidence is whether field level reductions in aquatic invertebrates correlate to *N*-methyl carbamate insecticide use and/or concentrations in salmonid habitats. We found several examples supporting this line of evidence. The available laboratory and field data show reductions in aquatic invertebrate taxa and reductions in invertebrate abundances following applications of carbofuran and carbaryl (Table 67). Furthermore, field studies explicitly investigated whether real world applications and subsequent pesticide drift and/or runoff into aquatic environments affected receiving water aquatic communities. One compelling study measured pre- and post-application aquatic community responses from field applications of carbaryl at 0.75 and 1 lb/acre (these rates are at the lower end of approved maximum application rates, up to 12 lbs/acre) (Courtemanch and Gibbs 1980). Applications correlated to substantial drift of salmonid prey type aquatic invertebrates, of which most were dead or dying. More striking was that population reductions of plecoptera (stoneflies) persisted for sixty days (Courtemanch and Gibbs 1980). Runoff from fields treated with carbofuran at 2.68 lbs a.i./ acre contained concentrations as high as 264 μg/L in the field ditches that drained to a stream where carbofuran was measured at 27 μg/L (Matthiessen, Shearan et al. 1995). Caged amphipods present in the stream all died during peak runoff following a rain event one month after the carbofuran application. This indicated that carbofuran can persist in soil and mobilize by rain into subsequent runoff and ultimately elicit toxic responses from stream invertebrates (Matthiessen, Shearan et al. 1995). Although we found no field studies with methomyl, we expect that it would similarly reduce aquatic invertebrate populations at higher concentrations than carbaryl and carbofuran based on its toxicity profile.

The third line of evidence we evaluated was whether salmonids showed reduced growth in areas of low prey availability, particularly those areas coinciding with use of carbaryl, carbofuran, and methomyl. An evaluation of this line is complicated by multiple factors affecting habitat quality *i.e.,* water quantity, quality, riparian zone condition, etc., which in turn affects prey items and salmonids. We were unable to locate information attributing reduced growth in salmonids to specific insecticide exposures that reduced prey, as most studies focused on measuring direct effects on salmonids or direct effects on invertebrates (see review by Schulz 2004). However, there are multiple field experiments and studies that demonstrate reduced fish growth resulting from reduced prey availability (Brazner and Kline 1990; Metcalfe, Fraser et al. 1999; Baxter, Fresh et al. 2007) or document fish growth rates below maximal potential growth rates when prey are limited (Dineen, Harrison et al. 2007).

One study, in particular, tested the hypothesis that single applications of the OP insecticide chlorpyrifos (0.5, 5, 20 μg/L) to outdoor ponds (littoral enclosures) would reduce the abundance of invertebrates and cause diet changes that would result in reduced growth rates of juvenile fish (Brazner and Kline 1990). The results are direct, empirical evidence that support this hypothesis. Growth rates of fathead minnow larvae were reduced significantly in all chlorpyrifos-containing treatments due to reduction in prey abundance. At 15 d post-treatment, the reductions in growth rate compared to control fish were the most pronounced and coincided with the greatest reductions in invertebrates. Stomach contents of minnows were identified throughout the experiment. By day 7 mean numbers of protozoans, chironomids, rotifers, cladocerans, mean total number of prey being eaten per fish, and mean species richness were greater in unexposed treatments compared to some of the other treatments. On day 15, most of the differences were more pronounced. The results strongly support the conclusion that foraging opportunities were better in untreated enclosures and unexposed larvae grew significantly more compared to chlorpyrifos-treated enclosures. Furthermore, the reductions in prey items in diets mirrored the reduction in prey items in the enclosures. We did not find any study results with *N*-methyl carbamates, but we make the inference that concentrations of the three *N*-methyl carbamates that are sufficient to reduce aquatic prey would also lead to reduced fish growth. This further supports the hypothesis that reduction in prey abundances translates to reductions in subsequent ration as well as individual growth. The authors concluded that "low levels of contaminants that induce slower growth in young-of-the-year fish through food chain effects or other means may eventually reduce the survival and recruitment of these fish."

Collectively, the lines of evidence strongly support the overall hypothesis. Thus, we carry reduced prey impacts to the next level of analysis (*i.e.,* the population-level). We conducted population modeling exercises based on reduced abundances of salmonid prey, presented in the next section, (*Effects to Salmonid Populations from the Proposed Action*).

D. Impair swimming which leads to reduced growth (via reductions in feeding), delayed and interrupted migration patterns, survival (via reduced predator avoidance), and reproduction (reduced spawning success).

Swimming is a critical function for anadromous salmonids. The primary line of evidence for this hypothesis is impaired swimming behaviors following exposure to carbaryl, carbofuran, and methomyl. A secondary line of evidence for this hypothesis is studies showing swimming behavior modification following exposure to other AChE-inhibiting chemicals such as the OPs, as we anticipate the results are similar in nature. Several studies regarding the effects of carbaryl on the swimming related behaviors were reviewed, with effects on predator avoidance, schooling, and feeding behaviors occurring at carbaryl concentrations of ~100-1,000 μg/L in laboratory studies (Weis and Weis 1974; Arunachalam, Jeyalakshmi et al. 1980; Little, Archeski et al. 1990; Carlson, Bradbury et al. 1998; Labenia, Baldwin et al. 2007). Only one study was available for carbofuran showing effects to social behaviors at 5 μg/L (Saglio, Trijasse et al. 1996), and no studies were located for methomyl. Concentrations at which effects were noted for carbaryl and carbofuran are within the ranges of concentrations estimated by the various modeling methods, especially for the off-channel habitat locations. We anticipate similar effects will occur with methomyl because it has the same mechanism of action as carbaryl and carbofuran, inhibition of AChE (which is correlated to swimming impacts). However, we expect it will take greater concentrations of methomyl compared to carbaryl and carbofuran to affect swimming. We expect that these levels are likely from concentrations resulting from aerial drift into off-channel habitats. This is because concentrations of methomyl necessary for AChE inhibition and acute lethality (LC50) are greater than carbaryl and carbofuran concentrations necessary to inhibit AChE and kill fish. We also discussed compelling evidence that OPs impair salmonid swimming behaviors and also show associated reductions in AChE activity.

The most sensitive swimming endpoints are those associated with swimming activity compared to those that measure swimming capacity (Little and Finger 1990; Little, Archeski et al. 1990). The ecological consequences to salmonids from aberrant swimming behaviors are implied primarily through the impairment of feeding, translating to reduced growth. Impaired swimming behavior correlated with both AChE inhibition and increased mortality from predation (Labenia, Baldwin et al. 2007). Although NMFS was unable to locate results from field or laboratory experiments for the other remaining endpoints of this hypothesis, we conclude that swimming behaviors are affected by the three insecticides. Adverse effects to swimming-associated behaviors are directly attributed to AChE inhibition, leading to potential reductions in an individual's fitness (*i.e.,* growth, migration, survival, and reproduction). We therefore translate impaired swimming to potential impacts on salmonid populations. Based on concentrations generated in modeling, NMFS believes concentrations of carbaryl and carbofuran, applied in accordance with current labels, are sufficient to impair swimming behavior of salmonids in some environments. As we located no studies that evaluated the effects of methomyl on swimming behaviors in fish, we do not know the exact levels sufficient to impair swimming except to say that levels well below reported LC50s can impair swimming. Swimming-mediated behaviors are frequently impaired at 0.3 – 5.0% of reported fish LC50s, and that 75% of reported adverse effects to swimming occurred at concentrations lower than reported LC50s (Little and Finger 1990). Taken together this information supports methomyl affecting swimming behaviors below reported LC50s. We expect methomyl at hundreds of μg/L (expected from currently approved applications) to impair swimming.

E. Reduce olfactory-mediated behaviors resulting in consequences to survival, migration, and reproduction.

In the Opinion regarding effects of chlorpyrifos, diazinon, and malathion on listed salmonids (NMFS 2008), sufficient data were available to conclude that olfactory-mediated behaviors were affected by those pesticides. Fewer data were available to assess the effects of carbaryl, carbofuran, and methomyl on these endpoints. Evidence is unclear as to mode of action for insecticide-mediated olfactory impairment. Thus, conclusions drawn based on data for OPs may not necessarily be applicable to *N*-methyl carbamates. Data currently available are conflicting, with the one study available for carbaryl showing no apparent effect on olfaction on juvenile cutthroat trout at concentrations of up to 500 μg/L. Three studies regarding olfactory effects of carbofuran were located and reviewed, and these indicated olfactory effects in several species of fish (including two salmonids) at concentrations ranging from 1-10 μg/L. These concentrations are within both the ranges estimated by modeling, and the ranges measured in all monitoring data sets. No data were available for methomyl. While data are not conclusive, based on what is known, and giving the benefit of the doubt to the species, NMFS believes it is reasonable to assume that these types of effects will occur from carbofuran exposures. Therefore, we discuss the potential implications at the population-level.

F. Exposure to mixtures of carbaryl, carbofuran, and methomyl can act in combination to increase adverse effects to salmonids and salmonid habitat.

The exposure and toxicity information we compiled, reviewed, and analyzed support the risk hypothesis, although less mixture data were available for the carbamates addressed in this Opinion than the OPs addressed in the previous Opinion (NMFS 2008). Evidence of additive and synergistic effects on survival and AChE inhibition in salmonids were identified. Multiple, independent study results supported additive toxicity from measured AChE inhibition. We therefore conducted an analysis of potential mixtures on the levels of AChE inhibition and the potential for a greater reduction in survival predicated on simple additivity. The analysis showed that both survival and AChE inhibition of individuals is likely affected to a greater degree than from exposure to a single chemical alone. We also expect that assessment endpoints influenced by AChE inhibition are likely affected to a greater degree when in the presence of more than one of the three insecticides. Considerable uncertainty arises as to the level of impairment caused by mixtures for some endpoints, as dose responses have not been characterized for some pesticide combinations. We conclude that this hypothesis is supported by the available information and we assess the potential for population-level consequences below.

G. Exposure to other stressors of the action including degradates, adjuvants, tank mixtures, and other active and other ingredients in pesticide products containing carbaryl, carbofuran, and methomyl cause adverse effects to salmonids and their habitat.

Fifteen of the carbamate formulations contain other a.i.s, including malathion (OP insecticide), bifenthrin (pyrethroid insecticide), rotenone (insecticide/piscicide), metaldehyde (mulloscicide), captan (phthalidimide fungicide), and cupric sulfate (toxic to algae, aquatic invertebrates, and fish). No data regarding the joint toxicity of these multi-a.i. products were presented in the BEs or located in open literature. However, given that these pesticides are also toxic to aquatic life it is reasonable to assume the salmonids and/or their prey items will exhibit a greater toxic response when exposed to multiple a.i.s than to a single one. We are unable to estimate the magnitude of such a response, and other than for malathion, are not currently able to evaluate whether the response would fall into an additive, synergistic, or antagonistic category. Based on mixture data discussed elsewhere in this Opinion, we assume additivity for carbaryl, carbofuran, and methomyl.

Some toxicity data were available in the BEs for formulations not containing additional a.i.s. Based on that data, no specific conclusions can be drawn as to whether formulations are, in general, more or less toxic than the technical grade a.i. However, it should be noted that adjuvants which increase either uptake (such as penetrants or surfactants that make physiological membranes more permeable) or length of exposure (attractants, or emulsions that increase time in the water column) are likely to increase toxicity of the a.i.

Only a limited amount of data were available to evaluate toxicity of the degradates of carbaryl, carbofuran, and methomyl. Monitoring data indicate that 1-napthol, a degradate of carbaryl, and 3-hydoxycarbofuran, a degradate of carbofuran, occur in measurable quantities in the environment. Based on the monitoring data, 1-napthol co-occurs with carbaryl in both water column and sediment. Toxicity information presented in the BE indicates that 1-napthol is approximately an order of magnitude less toxic than parent carbaryl for aquatic invertebrates. No data were given in the BE regarding the toxicity of 1-napthol to fish. We did locate one open literature study evaluating the effects of 1-napthol on fish, which indicated it was 2-5 times more toxic than the parent carbaryl (Shea and Berry 1983). No toxicity data were located for 3-hydroxycarbofuran or 3-ketocarbofuran, but based on structural similarities, the toxicity is likely similar to the parent carbofuran. Based on fate data presented in the BEs, 3-hydroxycarbofuran and 3-ketocarbofuran appear to occur in soil rather than in the water column. Both are shown as being non-detectable to representing 2-5% of applied parent material. Together, they could represent 6-7% of applied, assuming they were measured in the same test. Neither the BE nor the Science Chapter specify if that is the case.

Overall, while NMFS cannot quantify the increased toxicity of exposure to other stressors of the action such as additional a.i.s, degradates, and adjuvants, the existing body of information indicates these compounds are likely to cause and/or exacerbate adverse effects on salmonids and their habitat caused by carbaryl, carbofuran, and methomyl.

We did not receive a complete list of the currently registered formulations containing carbaryl, carbofuran, and methomyl from EPA. Thus, we cannot make any definitive conclusions for every stressor of the action. However, we did evaluate the exposure and response to a commonly used surfactant/adjuvant mixed with, or found in pesticide formulations. We reasoned if the data support adverse effects from this one of more than 4,000 substances, then other unidentified inert ingredients could also be toxic and may pose a significant risk to salmonids and their habitat. We selected NP polyethoxylates and NP because of their widespread use in pesticide formulations and abundance of information regarding environmental concentrations and adverse effects to salmonids and their prey. The data indicated that these surfactants can kill outright, disrupt endocrine systems, particularly reproductive physiology, and bioaccumulate in benthic invertebrates from expected concentrations in the environment (Arsenault, Fairchild et al. 2004; Madsen, Skovbolling et al. 2004; Jardine, MacLatchy et al. 2005; Luo, Ban et al. 2005; McCormick, O'Dea et al. 2005; Segner 2005; Hutchinson, Ankley et al. 2006; Lerner, Bjornsson et al. 2007; Lerner, Bjornsson et al. 2007). Importantly, we found studies that linked Atlantic salmon population crashes in Canada to use of NP in insecticide formulations. However, the BEs did not provide any information as to the prevalence of this material in formulations of the three OP insecticides that pertain to this consultation. We did receive confirmation from the technical registrant of methomyl that no methomyl containing formulation they make contains NP. Significant uncertainty surrounds the number and type of compounds, as well as the toxicity of these other materials used in pesticide formulations. As a result, we must caveat our conclusions regarding population-level responses with the uncertainty that the actual risk posed to listed salmonids and their habitat is likely greater when all ingredients are taken into account.

H. Exposure to other pesticides present in the action area can act in combination with carbaryl, carbofuran, and methomyl to increase effects to salmonids and their habitat.

The available toxicity and exposure data support the hypothesis. Other carbamates and OPs found in the action area likely result in additive or synergistic effects to exposed salmonids and aquatic invertebrates. The magnitude of effects will depend on the duration and concentrations of exposure.

I. Exposure to elevated temperatures can enhance the toxicity of the stressors of the action.

We found no consistent correlation with elevated temperature and toxicity of the three *N*-methyl carbamates. However, salmonids are coldwater species, and exposure to elevated temperatures increases physiological stress, thus making them more susceptible to other stressors. Additionally, other AChE inhibitors such as OPs are more toxic to salmonids and aquatic invertebrates exposed to elevated temperatures, so where elevated temperatures co-occur with OPs and carbamates the effect to aquatic life is likely greater than at lower temperatures. Many salmonid populations reside in watersheds which have been listed by the four western states as impaired due to temperature exceedances. We therefore discuss qualitatively temperature impacts on salmonids population responses to the stressors of the action.

J. Exposure to specific pH ranges can affect the toxicity of the stressors of the action.
Some data indicate acute toxicity of carbaryl and carbofuran increases as pH increased based on the available freshwater fish assays (Mayer and Ellersieck 1986). Other data indicated that toxicity was reduced in pH greater than 9 due to rapid hydrolysis *i.e.,* a half-life of 30 minutes in pond mesocosms. For methomyl, pH seems to have less of an influence on hydrolysis rates. Within the Pacific Northwest and California pH varies seasonally and typically may range from 6 – >9. We expect that salmonids exposed to both pH ranges at or near physiological tolerance limits and the three insecticides concurrently may die at relatively lower concentrations compared to salmonids exposed in laboratory assays. We therefore discuss qualitatively pH impacts on salmonids population responses to the stressors of the action.

Effects to Salmonid Populations from the Proposed Action

Here we translate individual fitness consequences to potential population-level effects using both quantitative and qualitative methods. We quantitatively translate reduced survival of juveniles based on four-day exposures to four populations of salmonids including ocean-type Chinook, stream-type Chinook, coho, and sockeye salmon. We employ a life history population model that incorporates changes in first-year juvenile survival rates and then translates them into predicted changes in the modeled population's intrinsic rate of growth, *i.e.,* lambda (Appendix 1). We discuss the percent change in lambda in the context of expected concentrations of the three insecticides in salmonid habitats. We focus on the concentrations at which a significant departure occurs from the unexposed population and compare them to expected environmental concentrations described in the *Exposure Analysis*. We also discuss in general terms the likelihood of exposure to the range of pesticide concentrations that occur in salmonid habitats.

In a second modeling exercise, we translate reductions in growth of juvenile salmon from AChE inhibition and from reduced prey abundances to potential population impacts using individual-based growth and life history population models (Appendix 1). These two endpoints (AChE inhibition and reduced prey abundance) are combined in the model to evaluate population-level effects due to reductions in first year survival of juveniles (Appendix 1). Similar to the survival models, percent change in lambda is the output. We discuss the significance of population changes in the context of departures from normal variability of the unexposed population and expected environmental concentrations. We conclude this section on population-level effects with a discussion of population-level responses to other affected salmonid endpoints that were not modeled. These include effects from other stressors of the proposed action, mixture effects, and effects to behaviors from impaired olfaction and AChE inhibition such as swimming behaviors.

Salmonid Population Models

We selected four generalized life history strategies to model (Appendix 1). We ran general life history matrix models for coho salmon (*Oncorhynchus kisutch*), sockeye salmon (*O. nerka*) and ocean-type and stream-type Chinook salmon (*O. tshawytscha*). We did not construct a steelhead (*O. mykiss*) life history model due to the lack of demographic information. Chum salmon (*O. keta*) were omitted from the growth model exercise because they migrate to marine systems soon after emerging from the gravel and the model assesses early life stage growth effects over a minimum of 140 d in freshwater systems. The basic salmonid life history we modeled consisted of hatching and rearing in freshwater, smoltification in estuaries, migration to the ocean, maturation at sea, and returning to the natal freshwater stream for spawning followed shortly by death. For specific information on how we constructed the models see Appendix 1.

Effects to salmonid populations from death of sub-yearling juveniles

An acute toxicity model was constructed that estimated the population-level impacts of subyearling juvenile (referred to as juveniles within this section) mortality resulting from exposure to concentrations of carbaryl, carbofuran, and methomyl. These models excluded sublethal and indirect effects of the pesticide exposures and focused on the population-level outcomes resulting from an annual 4 d exposure of all juveniles in the population to carbaryl, carbofuran, and methomyl from single exposures. Death of juveniles was implemented as a change in first-year survival rate for each of the salmon life-history strategies modeled. We also addressed mixture toxicity of the three insecticides in the model using exposure concentrations derived from EPA modeling estimates and from NMFS' modeling estimates of off-channel habitats. Model output is displayed in Tables 74-77.

The percent changes in lambdas increased as concentrations of the three carbamates increased. Increases in direct mortality during the first year of life produced large impacts on the population growth rates for all the life history strategies. Model results for stream-type Chinook salmon showed significant impacts at lower concentrations than the other modeled populations. This result is primarily due to the size of the standard deviation of the unexposed population. Percent changes in lambda were deemed significant if they were outside of one standard deviation from the unexposed population. The relative sensitivity of the life history models producing the greatest to the least changes in population growth rate for equivalent impact on survival rates was coho salmon, ocean-type Chinook salmon, stream-type Chinook salmon, and sockeye salmon. We note that the choice of LC50 is an important driver for these results. Therefore, an LC50 above or below the ones used here will result in a different dose-response. We selected the lowest reported salmonid LC50 from the available information to ensure that risk is not underestimated. However, if the actual environmental 96 h LC50 is lower, then the model will under-predict mortality. If the actual environmental acute LC50 is higher, then the model will over-predict mortality.

These results indicate that exposure of salmonid populations to carbaryl, carbofuran, and methomyl for four days at the reported LC50s would have severe consequences to the population's growth rate. If exposure occurred every year for each new cohort, population growth rate would be reduced and recovery efforts would be slowed. For those natural populations with current lambdas of less than one, risk of extinction would increase substantially, especially if several successive generations were exposed. For each of the combinations of species and insecticide, we denoted the relative concentration at which the percent change in lambda is deemed significantly different from the unexposed populations *e.g.,* a 9.1% change in lambda is estimated at 190 μg/L carbaryl for ocean-type Chinook salmon. These population effect thresholds assume exposure to all the juveniles in the population.

Carbaryl's thresholds for the four species ranged from 129-190 μg/L, carbofuran's from 89-126 μg/L, and methomyl's from 287-431 μg/L. These results can be compared to expected concentrations shown in Figure 39, Figure 40, and Figure 41. All of the thresholds overlap with expected concentrations for the three insecticides. The overlap primarily occurs with NMFS modeled estimates for off-channel habitats.

When we compare the population threshold concentrations to expected levels in salmonid habitats described in the exposure section, it is likely that some individuals within a population will be exposed during their freshwater juvenile life stage, particularly those juveniles exposed while using off-channel habitats. Additionally, four day averages from GENEEC runs, but not from PRZM-EXAMS, exceed these threshold concentrations. It is uncertain how appropriate EPA's model estimates are for salmonid habitats, but taken at face value juveniles exposed to GENEEC-derived concentrations would die. Population-level effects are expected from these exposures if the majority of the individuals that comprise the populations are exposed during their freshwater residency. The likelihood of population effects from death of juveniles increases for those populations that spend longer periods in freshwaters such as steelhead, stream-type Chinook, and coho salmon. We also expect additional acute mortalities from juveniles that are exposed to 24(c) carbaryl applications in Washington estuaries, although we do not know how many individuals are exposed each year. For those populations with lambdas greater than one, reductions in lambda from death of juveniles can also lead to consequences to abundance and productivity. Consequently, attainment of recovery goals would take longer to achieve for populations with reduced lambdas. Many of the populations that are categorized as core populations or are important to individual strata, have lambdas just above one and are essential to survival and recovery goals. Slight changes in lambda, even as small as 3-4%, would result in reduced abundances and/or increased time to meet population recovery goals.

Table 74. Modeled output for Ocean-type Chinook salmon exposed to 4 d exposures of carbaryl, carbofuran, and methomyl reporting the impacted factors of survival as percent dead, lambda and standard deviation, and percent change in lambda compared to an unexposed population.

<u>Carbaryl</u>	0 µg/L	50 µg/L	100 µg/L	200 µg/L	250 µg/L	350 µg/L	500 µg/L	750 µg/L
% dead	0	0	3	31	50	77	93	98
Lambda (STD)	1.09 (0.1)	1.08 (0.1)	1.08 (0.1)	0.98 (0.09)	0.89 (0.08)	0.71 (0.06)	.53 (0.05)	0.36 (0.03)
% change in lambda	NA	NS	NS (-1)	-10	-18	-34	-52	-67
Threshold for significant change in lambda	-9.1 % ~ 190 µg/L							
<u>Carbofuran</u>	0 µg/L	50 µg/L	100 µg/L	150 µg/L	164 µg/L	200 µg/L	250 µg/L	350 µg/L
% dead	0	1	14	42	50	67	82	94
Lambda (STD)	1.09 (0.10)	1.09 (0.10)	1.04 (0.10)	0.93 (0.08)	0.89 (0.08)	0.78 (0.07)	0.67 (0.06)	0.5 (0.04)
% change in lambda	NA	NS	NS (-4)	-15	-18	-27	-39	-54
Threshold for significant change in lambda	-9.1 % ~ 126 µg/L							
<u>Methomyl</u>	0 µg/L	250 µg/L	400 µg/L	500 µg/L	560 µg/L	700 µg/L	900 µg/L	1,100 µg/L
% dead	0	5	23	40	50	69	85	92
Lambda (STD)	1.09 (0.10)	1.07 (0.10)	1.08 (0.9)	0.94 (0.08)	0.89 (0.08)	0.78 (0.07)	0.64 (0.06)	0.54 (0.05)
% change in lambda	NA	NS(-1)	NS(-7)	-14	-18	-29	-41	-51
Threshold for significant change in lambda	-9.1 % ~ 431 µg/L							

NA denotes non applicable; NS denotes values less than one standard deviation of lambda expressed as the percent of lambda. (Calculated value, omitted when less than or equal to one)

Table 75. Modeled output for Stream-type Chinook salmon exposed to 4 d exposures of carbaryl, carbofuran, and methomyl reporting the impacted factors of survival as percent dead, lambda and standard deviation, and percent change in lambda compared to an unexposed population

Carbaryl	0 μg/L	50 μg/L	100 μg/L	200 μg/L	250 μg/L	350 μg/L	500 μg/L	750 μg/L
% dead	0	0	3	31	50	77	93	98
Lambda (STD)	1.00 (0.03)	1.00 (0.03)	0.99 (0.03)	0.91 (0.03)	0.84 (0.03)	0.69 (0.02)	0.53 (0.02)	0.37 (0.01)
% change in lambda	NA	NS	NS (-1)	-9	-16	-31	-47	-63
Threshold for significant change in lambda	-3.1 % ~ 129 μg/L							
Carbofuran	0 μg/L	50 μg/L	100 μg/L	150 μg/L	164 μg/L	200 μg/L	250 μg/L	350 μg/L
% dead	0	1	14	42	50	67	82	94
Lambda (STD)	1.0 (0.03)	1.0 (0.03)	0.96 (0.03)	0.87 (0.03)	0.84 (0.03)	0.76 (0.02)	0.65 (0.02)	0.5 (0.01)
% change in lambda	NA	NS	-4	-13	-16	-24	-35	-50
Threshold for significant change in lambda	-3.1 % ~ 89 μg/L							
Methomyl	0 μg/L	250 μg/L	400 μg/L	500 μg/L	560 μg/L	700 μg/L	900 μg/L	1,100 μg/L
% dead	0	5	23	40	50	69	85	92
Lambda (STD)	1.0 (0.03)	0.99 (0.03)	0.94 (0.03)	0.88 (0.03)	0.84 (0.03)	0.75 (0.02)	0.63 (0.02)	0.54 (0.02)
% change in lambda	NA	NS(-1)	-6	-12	-16	-25	-37	-46
Threshold for significant change in lambda	-3.1 % ~ 287 μg/L							

NA denotes non applicable; NS denotes values less than one standard deviation of lambda expressed as the percent of lambda. (Calculated value, omitted when less than or equal to one)

Table 76. Modeled output for Coho salmon exposed to 4 d exposures of carbaryl, carbofuran, and methomyl reporting the impacted factors of survival as percent dead, lambda and standard deviation, and percent change in lambda compared to an unexposed population.

Carbaryl	0 μg/L	50 μg/L	100 μg/L	200 μg/L	250 μg/L	350 μg/L	500 μg/L	750 μg/L
% dead	0	0	3	31	50	77	93	98
Lambda (STD)	1.03 (0.05)	1.03 (0.05)	1.02 (0.05)	0.91 (0.05)	0.82 (0.04)	0.69 (0.03)	0.42 (0.02)	0.27 (0.01)
% change in lambda	NA	NS	NS(-1)	-12	-21	-39	-58	-74
Threshold for significant change in lambda	-5.3 % ~ 144 μg/L							
Carbofuran	0 μg/L	50 μg/L	100 μg/L	150 μg/L	164 μg/L	200 μg/L	250 μg/L	350 μg/L
% dead	0	1	14	42	50	67	82	94
Lambda (STD)	1.03 (0.05)	1.02 (0.05)	0.98 (0.05)	0.86 (0.05)	0.82 (0.04)	0.71 (0.04)	0.58 (0.03)	0.40 (0.02)
% change in lambda	NA	NS(-1)	NS(-5)	-17	-21	-31	-44	-61
Threshold for significant change in lambda	-5.3 % ~ 98 μg/L							
Methomyl	0 μg/L	250 μg/L	400 μg/L	500 μg/L	560 μg/L	700 μg/L	900 μg/L	1,100 μg/L
% dead	0	5	23	40	50	69	85	92
Lambda (STD)	1.03 (0.05)	1.01 (0.05)	0.94 (0.05)	0.87 (0.05)	0.82 (0.04)	0.69 (0.04)	0.55 (0.03)	0.44 (0.02)
% change in lambda	NA	NS(-2)	-8	-16	-21	-32	-47	-57
Threshold for significant change in lambda	-5.6 % ~ 338 μg/L							

NA denotes non applicable; NS denotes values less than one standard deviation of lambda expressed as the percent of lambda. (Calculated value, omitted when less than or equal to one)

Table 77. Modeled output for Sockeye salmon exposed to 4 d exposures of carbaryl, carbofuran, and methomyl reporting the impacted factors of survival as percent dead, lambda and standard deviation, and percent change in lambda compared to an unexposed population

Carbaryl	0 μg/L	50 μg/L	100 μg/L	200 μg/L	250 μg/L	350 μg/L	500 μg/L	750 μg/L
% dead	0	0	3	31	50	77	93	98
Lambda (STD)	1.01 (0.06)	1.01 (0.06)	1.00 (0.06)	0.92 (0.05)	0.86 (0.05)	0.71 (0.04)	0.55 (0.03)	0.40 (0.02)
% change in lambda	NA	NS	NS(-1)	-8	-15	-30	-46	-61
Threshold for significant change in lambda	-5.6 % ~ 169 μg/L							
Carbofuran	0 μg/L	50 μg/L	100 μg/L	150 μg/L	164 μg/L	200 μg/L	250 μg/L	350 μg/L
% dead	0	1	14	42	50	67	82	94
Lambda (STD)	1.01 (0.06)	1.01 (0.06)	0.97 (0.05)	0.89 (0.05)	0.86 (0.05)	0.78 (0.04)	0.67 (0.04)	0.52 (0.03)
% change in lambda	NA	NS	NS(-4)	-12	-15	-23	-34	-48
Threshold for significant change in lambda	-5.6% ~ 114 μg/L							
Methomyl	0 μg/L	250 μg/L	400 μg/L	500 μg/L	560 μg/L	700 μg/L	900 μg/L	1,100 μg/L
% dead	0	5	23	40	50	69	85	92
Lambda (STD)	1.01 (0.06)	1.00 (0.06)	0.95 (0.05)	0.89 (0.05)	0.86 (0.05)	0.76 (0.04)	0.65 (0.04)	0.56 (0.03)
% change in lambda	NA	NS(-1)	-6	-11	-15	-24	-36	-45
Threshold for significant change in lambda	-5.3 % ~ 391 μg/L							

NA denotes non applicable; NS denotes values less than one standard deviation of lambda expressed as the percent of lambda. (Calculated value, omitted when less than or equal to one)

The population exercises discussed thus far focused on the effects from exposures to a single annual application of an insecticide; however, we know that these insecticides likely occur together in environmental mixtures and are frequently applied multiple times. To address the potential population-level effects to environmental mixtures of the three insecticides, we ran the same models with a calculated percent mortality predicted using dose-addition. We applied the same additivity model described in the previous *Mixtures* section to predict the cumulative percentage of death in an exposed population. Table 78 shows the model's prediction from three scenarios. We did not use the data from ambient monitoring programs to devise a scenario because the programs were not designed to capture peak concentrations from drift or runoff into juvenile salmonid rearing areas. We did run a few modeling runs with median values of the three insecticides taken from ambient monitoring data. The results showed no statistically significant reductions in lambda for the four populations. In scenario 1, 5% of exposed individuals are expected to die following 4 d exposures to the estimated concentrations from PRZM-EXAMS 24 h averages from EPA's BEs. For carbaryl, we selected a 2 lbs/acre applied aerially with four applications to apples in Oregon which resulted in a concentration of 19 μg/L. For carbofuran, we selected a 2 lb/acre ground application to artichokes in California which resulted in a concentration of 35 μg/L. For methomyl, we selected 0.9 lbs/acre applied ten times to lettuce in California which resulted in a concentration of 88 μg/L. If these events were to occur in a watershed, 5% of the individuals are expected to die, which would not lead to a population-level effect (*i.e.,* reduction in lambda), based on juvenile mortality.

In contrast, the other two scenarios showed substantial and severe percent reductions in lambda for each of the modeled populations; reductions in lambda ranged from 27-52% (Table 78) In scenario 2, we selected 90 d GENEEC-modeled scenarios for corn from Table 50. Concentrations of the three insecticides were as follows: carbaryl = 229 μg/L, carbofuran = 53 μg/L, and methomyl = 49 μg/L. The toxicity is largely a result of the carbaryl concentration, as it was close to the 96 h LC50 of 250 μg/L. In scenario 3, concentrations from drift into off-channel habitats (0.5 m deep) were calculated using the AgDrift model and included a 100 ft no-spray buffer for methomyl. For carbaryl, 5 lb/acre applied once aerially with a fine-medium spray droplet size resulted in a concentration of 335 μg/L. For carbofuran, 1 lb/acre applied once also with a fine-medium spray droplet size resulted in a concentration of 67 μg/L. For methomyl, 0.9 lbs/acre applied once resulted in a concentration of 17.1 μg/L. Cumulatively, these three concentrations would result in 89% mortality of exposed juveniles which reduce lambdas of the four salmonid populations by 41-52%, a substantial effect.

The likelihood of scenarios 2 and 3 occurring is difficult to predict due to the lack of detailed information on watershed characteristics, salmonid presence, the numbers of salmonids exposed, the duration of exposure, and the climatic variables leading to drift. Scenarios 2 and 3 may represent infrequent events, but if a substantial part of a population of listed salmonids is exposed to these mixtures, a severe reduction in a population's abundance is expected.

Table 78. Modeled output for Ocean-type Chinook, Stream-type Chinook, Sockeye, and Coho salmon exposed to 4 d exposures of carbaryl-, carbofuran-, and methomyl-containing mixtures. The table denotes the impacted factors of survival as percent dead, lambda and standard deviation, and percent change in lambda compared to an unexposed population

Scenario 1: PRZM-EXAMS 24-h averages	Ocean-type Chinook	Stream-type Chinook	Sockeye	Coho
% dead	5	5	5	5
Lambda (SD)	1.07 (0.10)	0.99 (0.03)	1.00 (0.06)	1.01 (0.05)
% change in lambda	NS(-1)	NS(-1)	NS(-1)	NS(-2)
Scenario 2: GENEEC 90-d averages	Ocean-type Chinook	Stream-type Chinook	Sockeye	Coho
% dead	74	74	74	74
Lambda (SD)	0.74 (0.07)	0.72 (0.02)	0.74 (0.04)	0.66 (0.04)
% change in lambda	-32	-28	-27	-36
Scenario 3: Offchannel habitats 0.5 m deep	Ocean-type Chinook	Stream-type Chinook	Sockeye	Coho
% dead	89	89	89	89
Lambda (SD)	0.59 (0.05)	0.58 (0.02)	0.60 (0.03)	0.49 (0.03)
% change in lambda	-46	-42	-41	-52

NA denotes non applicable; NS denotes values less than one standard deviation of lambda expressed as the percent of lambda. (Calculated value, omitted when less than or equal to one)

Effects to salmonid populations from reduced size of juveniles due to impaired feeding and reduced abundance of aquatic prey

To evaluate the potential for adverse effects to juvenile growth resulting from carbaryl, carbofuran, and methomyl on Pacific salmonid populations, we developed a model (Appendix 1). The model links AChE inhibition, feeding behavior, prey availability, and somatic growth of individual salmon to the productivity of salmon populations expressed as a percent change in lambda (a population's intrinsic rate of growth). The model scenarios (single insecticide or multiple pulses of single insecticides) assume annual exposure of all the subyearling juveniles in the population and their prey to the insecticide. Similar to the survival model, we developed the growth model for four populations of salmonids: ocean-type Chinook, stream-type Chinook, sockeye, and coho salmon.

We integrated two avenues of effect to juvenile salmonids' growth from exposure to the three *N*-methyl carbamates (Appendix 1). The first avenue is a result of AChE inhibition on the feeding success and subsequent effects to growth of juvenile salmonids. Study results with juvenile salmonids show that feeding success is reduced following exposures to AChE inhibitors (Sandahl, Baldwin et al. 2005). The second avenue the model addresses is the potential for reductions in juvenile growth due to reduction in available prey. Salmon are often found to be food limited in freshwater aquatic habitats, suggesting that a reduction in prey due to insecticide exposure may further stress salmon and lead to reduced growth rates. Field mesocosm data support this assertion, showing reduced growth of juvenile fish following exposure to the AChE inhibitor, chlorpyrifos (Brazner and Kline 1990). Furthermore, based on our review of the sensitivities of aquatic invertebrates to the three insecticides, we expect reductions in densities and altered composition of the salmonid prey communities.

Reductions in aquatic prey are included in the model because of the high relative toxicity of pesticides to salmonid prey and the extended duration of effects on prey communities. Juvenile salmonids are largely opportunistic, feeding on a diverse community of aquatic and terrestrial invertebrate taxa that are entrained in the water column or on the surface (Higgs, Macdonald et al. 1995). As a group, these invertebrates are among the more sensitive taxa for which there is toxicity data, but within this group, there is a wide range of sensitivities (Table 64, Table 65).

The three insecticides are highly toxic to aquatic macroinvertebrates; and concentrations that are not expected to kill salmonids are often lethal for their invertebrate prey (*e.g.,* for carbaryl, range of mean LC50s for salmonids = 250-3,000 μg/L, vs. range of geometric mean EC50s for water fleas = 3-120 μg/L). In particular, prey items that are preferred by small juvenile salmonids (including midge larvae, water fleas, mayflies, caddisflies, and stoneflies) are among the most sensitive aquatic macroinvertebrates. In addition, effects on the prey community can persist for extended periods of time (weeks, months, years), resulting in effects on fish feeding and growth long after an exposure has ended (Ward, Arthington et al. 1995; Van den Brink, van Wijngaarden et al. 1996; Liess and Schulz 1999; Colville, Jones et al. 2008).

Selection of aquatic invertebrate toxicity values to represent salmonid prey items

The model requires for each insecticide an EC50 (defined as a 50% reduction in the biomass of salmonid prey items) and a corresponding slope (Appendix 1). The term "EC50" will be used in this section to describe short-term survival data for aquatic invertebrates (death and immobility). To determine what levels of the three pesticides reduce aquatic invertebrate numbers, we reviewed the available field and laboratory studies. We found robust data for carbaryl, carbofuran, and methomyl with respect to laboratory acute toxicity tests that measured survival at 24-, 48-, 72-, and 96 h with an array of aquatic invertebrates. We did not locate a field study that measured aquatic community response to a range of concentrations of the three insecticides. Therefore, we did not select concentration data from field experiments as we did in NMFS' 2008 Opinion on the registration of chlorpyrifos, diazinon, and malathion (NMFS 2008).

To determine a single effect concentration to use in the model analyses, a search was completed using the EPA's ECOTOX database for each pesticide (http://cfpub.epa.gov/ecotox/). Several criteria were used to determine which reported effect concentrations were included in the final analysis. The data included were from studies on taxa that are known to be salmonid prey (or are functionally similar to salmonid prey); these include a diverse group of aquatic insects and worms and fresh and saltwater crustaceans. Studies with exposures of at least 24 h and not more than 96 h were included. Studies examining shorter and longer exposure times are known to affect invertebrates (Peterson, Jepson et al. 2001), but these were excluded so that estimated EC50s would be comparable. Studies reporting survival EC50s in which mortality or immobilization was the recorded endpoint were included. Data derived for sublethal endpoints (*e.g.*, growth or reproduction) were not included. If specific data were represented more than once in the ECOTOX output, duplicates were eliminated. Data from recent peer-reviewed studies that report survival EC50s were also included (Norberg-King, Durhan et al. 1991; Takahashi and Hanazato 2007). Next, we calculated the geometric mean when more than one survival EC50 was reported for a species.

Probability distributions of aquatic prey survival toxicity values (EC50s)

We plotted the survival EC50 data for each of the three insecticides using cumulative probability distributions. We also plotted the data based on all test results for the species without taking the geometric means. From the distributions of the data, a single effect concentration and slope were derived to best represent the diverse community of prey available in juvenile salmonid freshwater and estuarine habitats. The distributions of individual EC50s and the geometric means of EC50s by taxa were analyzed to estimate the 50th, 10th, and 5th percentiles. Figure 43 shows the distributions of geometric means of EC50s by taxa. Specifically, for each pesticide, a probability plot was used to graph the EC50 concentrations normalized to a normal probability distribution. For each plot, the X axis is scaled in probability (between zero and 100%) and shows the percentage of entire data whose value is less than the data point. The Y axis displays the range of the data on a log scale. The results of a linear regression of the log-transformed concentrations are shown and highlight the lognormal distribution of the data (Figure 43). In the regression equation, the normsinv function returns the inverse of the standard normal cumulative distribution. The standard normal distribution has a mean of zero and a standard deviation of one. For example, given a percentile value of 50 (*i.e.,* a probability of 0.5), normsinv(50) returns a value of zero. The plots and regressions were performed using KaleidaGraph 4.03 (Synergy Software).

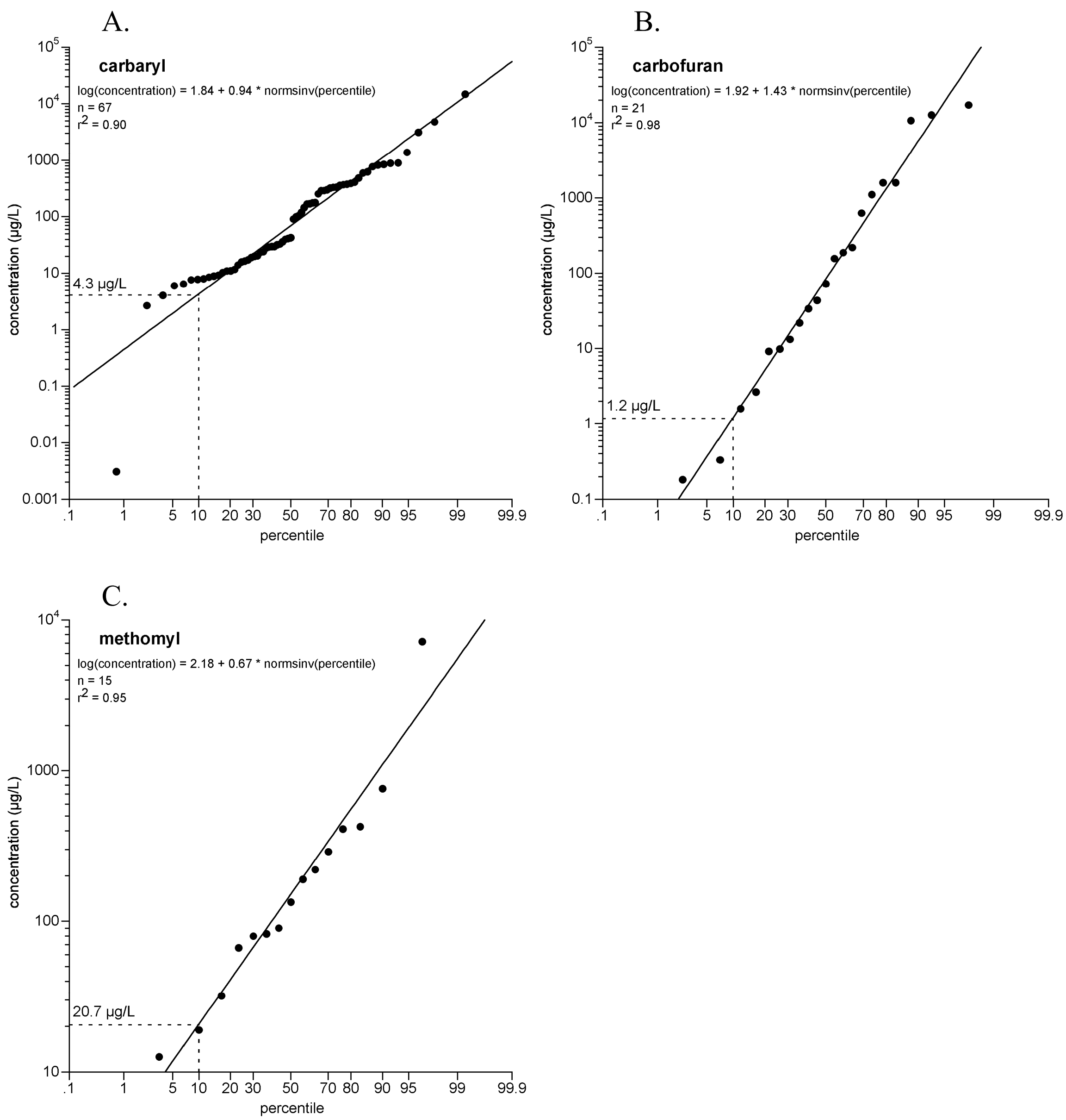

Figure 43. Probability plot for each pesticide showing the distribution of the geometric means of the EC50s for each aquatic invertebrate species. The straight line shows the result of a linear regression. In the regression equation, the normsinv() function returns the inverse of the standard normal cumulative distribution. See text for more details. A) Plot of carbaryl EC50s. B) Plot of carbofuran EC50s. C) Plot of methomyl EC50s.

In Table 79, concentrations are reported for each insecticide associated with either the 50th, 10th, or 5th percentiles derived from probability distribution plots with all study results (plots not shown) or with species geometric means. Percentiles from the species geometric means show higher concentrations compared to the "all studies" plots. This is likely a reflection that when individual study results are considered separately, the species with the greater number of EC50 study results is found at the lower end of the distribution.

Table 79. Carbaryl, carbofuran, and methomyl survival EC50 concentrations at 50th , 10th , and 5th percentiles from probability distribution plots.

	Concentration of EC50 at each percentile (µg/L)		
	50%	10%	5%
All studies probability distribution plot			
Carbaryl	45.23	2.29	0.98
Carbofuran	58.95	0.94	0.29
Methomyl	128.9	12.93	6.74
Species geometric means probability distribution plot			
Carbaryl	69.53	4.33	1.97
Carbofuran	89.95	1.22	0.37
Methomyl	150.76	20.74	11.82

We selected the 10th percentile from each of the pesticide plots to represent the survival EC50 for salmonid prey. The associated 10th percentile for carbaryl (4.33 µg/L), carbofuran (1.22 µg/L), and methomyl (20.74 µg/L) was used as input for the population growth model exercises. The 10th percentile is a reasonable selection because the data included in the meta-analysis were limited to concentrations that caused mortality or immobilization within a short period of time (1-4 days). A growing number of studies on a variety of insecticides have reported that concentrations well below LC50s can cause delayed mortality or sublethal effects that may scale up to affect aquatic invertebrate populations, especially in scenarios with multiple exposures and/or other stressors. Evidence for ecologically significant sublethal or delayed effects to aquatic invertebrates includes reduced growth rates (Schulz and Liess 2001; Forbes and Cold 2005), altered behavior (Johnson, Jepson et al. 2008), reduced emergence (Schulz and Liess 2001; Johnson, Jepson et al. 2008), reduced reproduction (Cold and Forbes 2004; Forbes and Cold 2005), and reduced predator defenses (Sakamoto, Chang et al. 2006; Johnson, Jepson et al. 2008).

Additionally, the available toxicity data – and therefore the data included for these analyses– are from studies using taxa hearty enough to survive laboratory conditions. Studies specifically examining salmonid prey that are more difficult to rear in the laboratory have documented relatively low survival EC50 values when exposed to current use insecticides (Johnson, Jepson et al. 2008), including carbaryl (Peterson, Jepson et al. 2001; Johnson, Jepson et al. 2008).

The selection of the 10th percentile is a reasonable choice in keeping with general risk management practices of protecting the aquatic community. The standard procedure the U.S. EPA Office of Water uses in establishing Aquatic Life Criteria is to protect to the 5th percentile of a species sensitivity distribution that includes all genera of aquatic species for which valid data are available (EPA 1995). Their procedure involves combining species data into a genus mean value so that multiple tests on one sensitive species do not unduly influence the outcome, and evaluating a specified group of taxa. As we had already selected for the sensitive end of the spectrum by using salmonid prey items, and did not collapse species data into a genus mean, we used the 10th percentile. In the probabilistic risk assessment for carbofuran (EPA 2005), OPP evaluated effects on the 5th, 50th, and 95th percentiles of separate species sensitivity distributions for fish and invertebrates, although it was not clear in the document which was the preferred percentile for risk management.

Modeling availability of unaffected prey

Reductions in benthic invertebrate densities can lead to long-term reductions in prey availability and reductions in fish growth (Davies and Cook 1993). That said, prey densities are not usually reduced to zero (Wallace, Lugthart et al. 1989). Therefore, it is assumed that regardless of the exposure scenario, prey abundance would not drop below a specific "floor" of prey availability. This floor is included in the model to reflect an assumption that a minimal yet constant terrestrial subsidy of prey and/or an aquatic community with tolerant individuals would be available as prey, regardless of pesticide exposure and in addition to the constant recovery rate (see below).

Therefore, even in extreme exposure scenarios, some prey will be available, as determined by the value assigned to the floor; in some highly degraded systems this may or may not be the case. No studies have quantified this floor for the purpose of estimating prey availability, but several studies have documented reductions in overall benthic insect densities of 75-98% (Wallace, Lugthart et al. 1989; Anderson, Hunt et al. 2003; Anderson, Phillips et al. 2006). Because benthic densities are typically correlated with drift densities (Hildebrand 1974; Waters and Hokenstrom 1980), these reductions likely result in similar reductions of prey. Therefore, assuming there is also some constant rate of terrestrial invertebrate subsidy in addition to a residual aquatic community, a floor of 0.20, or 20% of fish ration, is reasonable. The model does not include any additional impacts to fish via dietary exposure from contaminated prey, or any potential synergistic or additive effects to the aquatic invertebrates that may be result from multiple stressors (Schulz and Liess 2001).

Modeling spikes in invertebrate drift following insecticide exposure

"Catastrophic drift" of invertebrates, due to acute mortality and/or emigration of benthic prey into the water column is frequently observed following exposure to insecticides (Davies and Cook 1993; Schulz and Liess 2001; Schulz 2004). Drift rates within hours of exposure can be more than 10,000 times the natural background drift (Cuffney 1984), and fish have been found to exploit this by feeding beyond satiation (Haines 1981; Davies and Cook 1993). The duration and magnitude of the spike in drift of prey is dependent in part on the physical properties and dose of the pesticide; however, the spike is generally ephemeral and returns to natural, background levels within hours to days (Haines 1981; Kreutzweiser and Sibley 1991). Likewise, the magnitude of the spike is dependent in part on the benthic density of prey; the spike in drift from communities that have been reduced by previous exposures is smaller than the spike from previously undisturbed communities (Cuffney 1984; Wallace, Huryn et al. 1991). To reflect this temporary increase in prey availability, the model includes a one-day prey spike for the day following an exposure (Appendix 1). The model also accounts for this short-term increase in prey availability by allowing fish to feed at a maximum rate of 1.5 times their normal, optimal ration.

Modeling recovery of salmonid prey

We selected a 1% recovery in prey biomass per day. Reports of recovery of invertebrate prey populations, once pesticide exposure has ended, range from within days to more than a year (Cuffney 1984; Kreutzweiser and Sibley 1991; Pusey, Arthington et al. 1994; Ward, Arthington et al. 1995; Van den Brink, van Wijngaarden et al. 1996; Liess and Schulz 1999; Colville, Jones et al. 2008). The dynamics of recovery are complicated by several factors, including the details of the pesticide exposure(s) as well as habitat and landscape conditions (Liess and Schulz 1999) (Van den Brink, Baveco et al. 2007). In watersheds with undisturbed upstream habitats, recovery can be rapid due to a healthy source of invertebrates that can immigrate via drift and/or aerial colonization (for adult insects) (Heckmann and Friberg 2005). However, in watersheds dominated by agricultural or urban land uses, healthy upstream or nearby habitats may be limited and consequently, recolonization by salmonid prey is likely reduced (Liess and Von der Ohe 2005; Schriever, Ball et al. 2007). Additionally, many large, high-quality prey take a year or more to develop (Merritt and Cummins 1995) indicating that recovery of biomass (as compared to prey density) is likely a limiting factor (Cuffney 1984). Recovery to pre-disturbance levels is unlikely in aquatic habitats where invertebrate abundances are repeatedly reduced by stressors. We consider a 1% (control prey abundance per day) recovery rate as ecologically realistic to represent recolonization by invertebrates in salmonid habitats (Ward, Arthington et al. 1995; Van den Brink, van Wijngaarden et al. 1996; Colville, Jones et al. 2008).

Growth model results

Exposure to single insecticides for 4-, 21- , and 60 day exposure durations

Population model outputs for the four salmon populations are summarized as dose-response curves in Figures 44-47. As expected, greater reductions in population growth resulted from longer exposures to the insecticides. The primary factor driving the magnitude of change in lambda was the Prey Abundance parameter for each insecticide *i.e.*, the 10th percentile survival EC50 for salmonid prey. The AChE parameter for each of the insecticides was a secondary factor compared to Prey Abundance. This is largely because the salmonid EC50s for AChE were much higher, typically by an order of magnitude, than the prey survival EC50s.

Similar trends in effects were seen for each pesticide across all four life history strategies modeled. This is apparent by the similar shape of the dose-response curves across species. The curves plateau when there is no more reduction possible in the aquatic community *i.e.,* the 20% biomass of the aquatic invertebrate community is reached. Once that plateau is achieved, further reductions in lambda are minimal with increasing concentrations. Clearly, the most toxic of the pesticides affected salmon populations at lower concentrations, as is observed with carbaryl and carbofuran where concentrations in the low μg/L range are sufficient to reduce populations' growth rates compared to methomyl where 20 μg/L or greater are needed to reduce lambdas. One factor that contributed to the similar responses observed was the use of the same surrogate toxicity values for all four life history strategies. The stream-type Chinook salmon (Figure 45) and sockeye salmon **(**Figure 46) models produced very similar results as measured as the final output of percent change in population growth rate. The ocean-type Chinook salmon model output (Figure 44) produced the next most extreme response, and coho salmon output (Figure 46) showed the greatest changes in lambda resulting from the pesticide exposures. When looking for similarities in parameters to explain the ranking, no single life history parameter or characteristic, such as lifespan, reproductive ages, age distribution, lambda and standard deviation, or first-year survival show a pattern that matches this consistent output **(**Appendix 1**)**. Combining these factors into the transition matrix for each life history and conducting the sensitivity and elasticity analyses revealed that changes in first-year survival produced the greatest changes in lambda. While some life history characteristics may lead a population to be more vulnerable to an impact, the culmination of age structure, survival and reproductive rates as a whole strongly influences the population-level response.

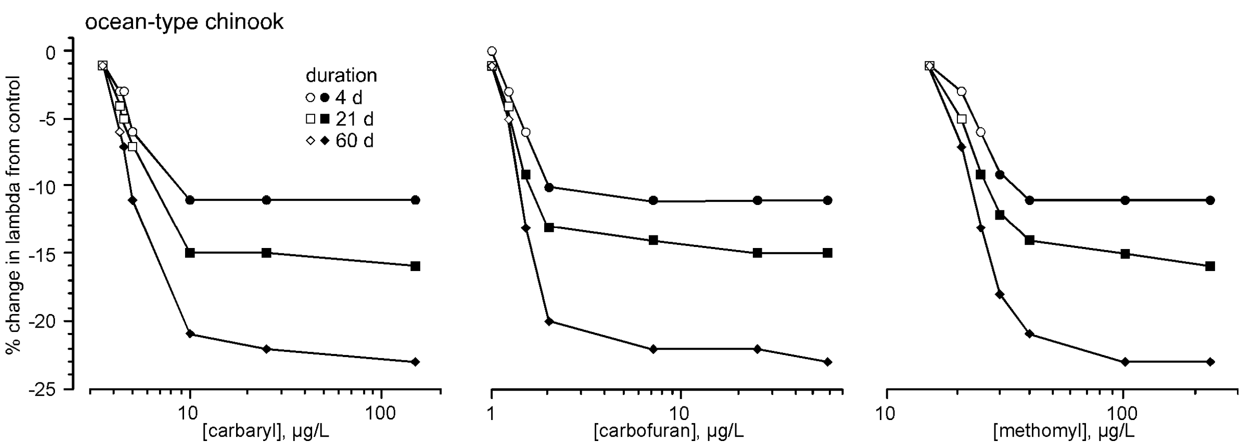

Figure 44. Percent change in lambda for Ocean-type Chinook salmon following 4 d, 21 d, and 60 d exposures to carbaryl, carbofuran, and methomyl. Open symbols denote a percent change in lambda of less than one standard deviation from control population. Closed symbols represent a percent change in lambda of more than one standard deviation from control population.

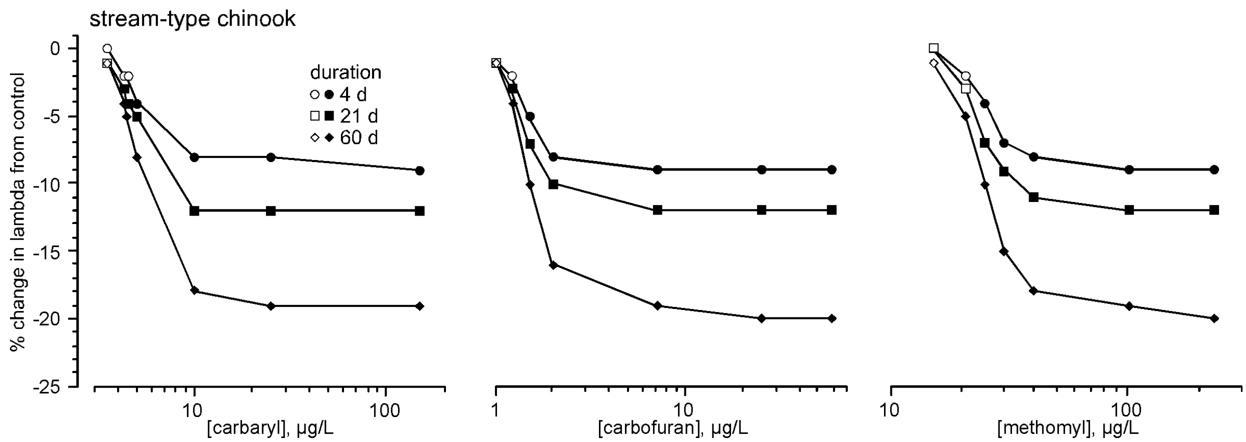

Figure 45. Percent change in lambda for Stream-type Chinook salmon following 4 d, 21 d, and 60 d exposures to carbaryl, carbofuran, and methomyl. Open symbols denote a percent change in lambda of less than one standard deviation from control population. Closed symbols represent a percent change in lambda of more than one standard deviation from control population.

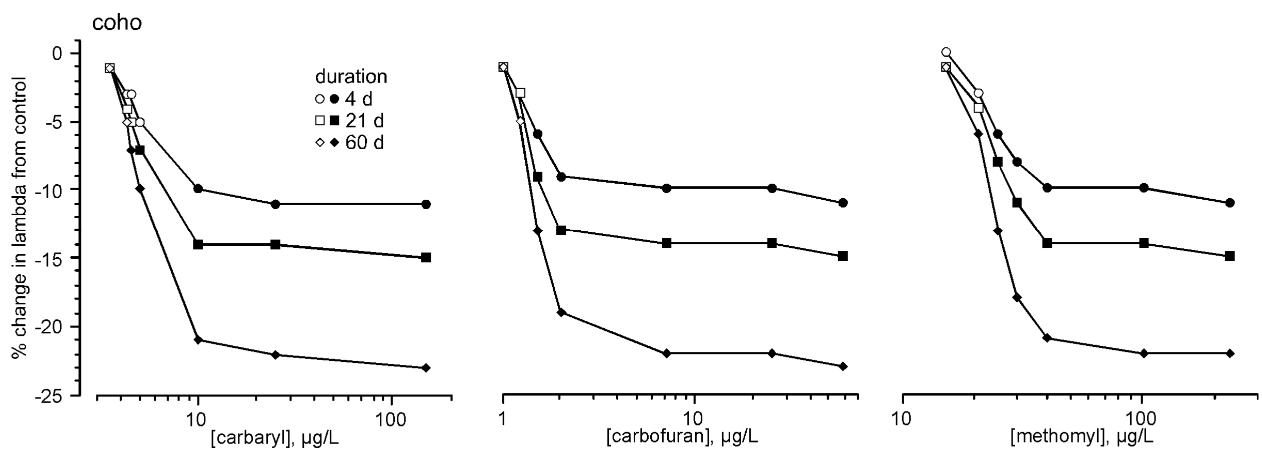

Figure 46. Percent change in lambda for Coho salmon following 4 d, 21 d, and 60 d exposures to carbaryl, carbofuran, and methomyl. Open symbols denote a percent change in lambda of less than one standard deviation from control population. Closed symbols represent a percent change in lambda of more than one standard deviation from control population.

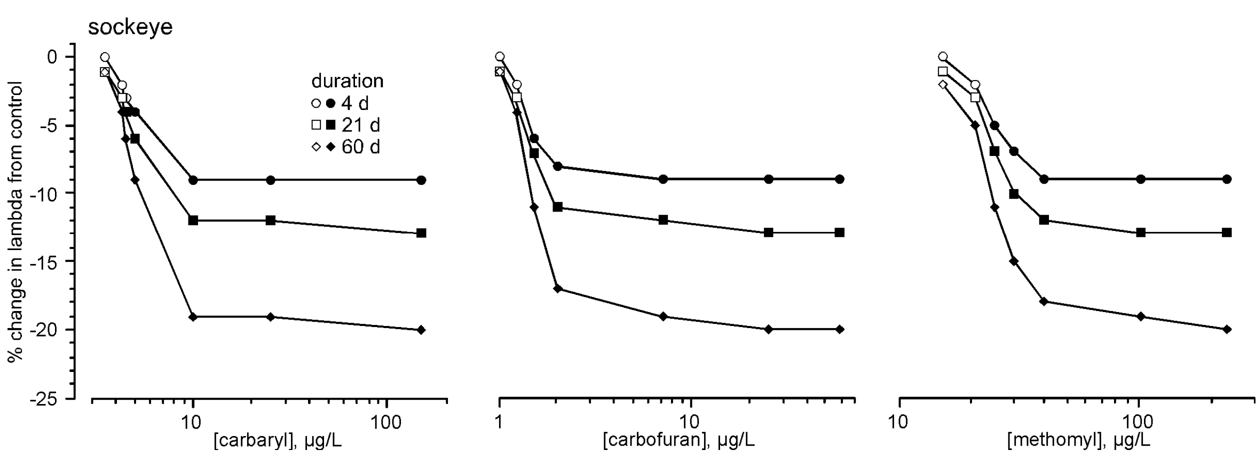

Figure 47. Percent change in lambda for sockeye salmon following 4 d, 21 d, and 60 d exposures to carbaryl, carbofuran, methomyl. Open symbols denote a percent change in lambda of less than one standard deviation from control population. Closed symbols represent a percent change in lambda of more than one standard deviation from control population.

By applying some of these changes in lambda to known threatened and endangered populations' lambdas from Appendix 2, significant reductions in population viabilities are anticipated. For example, if the Puget Sound Chinook salmon Green River population with a lambda of 0.67 is exposed to carbaryl at 10.0 μg/L for 4 d, a concentration attainable in off-channel habitats, we would expect a reduction in lambda by 11% (Table 74) or 8% (Table 75) depending whether the individuals exhibit ocean-type or stream-type life histories. These reductions would be severe to the population and are primarily a result of reductions in juvenile growth from impacts to salmonid prey. Even for those lambdas that are well above one such as Central Valley Chinook salmon Spring Runs' Butte Creek population (lambda = 1.3), reductions of 11% would have major consequences to a population's viability from reduced growth of juveniles. The repercussions to these populations' viabilities are increased with increasing concentrations, durations, multiple applications, and when mixtures are incorporated.

Exposure to multiple applications of carbaryl or methomyl

We constructed two scenarios to evaluate the potential population effects from multiple applications of carbaryl and methomyl (Table 80). We used concentrations from AgDrift model outputs to estimate drift contributions of carbaryl into an off-channel habitat (Table 52). Populations exposed to carbaryl drift within off-channel habitats following four applications would experience severe declines in population growth rate ranging from 15-19%. Interestingly, a single application would result in notable reductions in the four modeled population's lambdas as well, ranging from 8-11%. Similar to the carbaryl modeled scenario, methomyl applied to sweet corn ten times (the label allows for 14 applications at 1 d intervals) resulted in significant reductions in lambdas ranging from 6-8%. A single application did not result in a significant reduction in lambda for any of the four populations.

Table 80. Multiple application scenarios for carbaryl and methomyl and predicted percent change in lambdas for salmon populations

	Carbaryl	Methomyl
Crop examples	Almonds, chestnuts, pecans, filberts, walnuts, pistachios	Sweet corn
Application rate	5 lbs a.i./acre	0.45 lbs a.i./acre
Number of applications/yr	4	10
Application interval	14 days	3 days
Method of application	Aerial (fine-medium droplet distribution)	aerial (fine-medium droplet distribution)
No-application Buffer	none	100 ft
Off-channel habitat characteristics	water depth = 0.5 m Initial average concentration 335 μg/L; 24 h exposure	water depth = 0.5 m Initial average concentration 8.55 μg/L; 96 h exposure
% change in Lambda		
Ocean-type Chinook	-19%	-8%
Stream-type Chinook	-15%	-6%
Sockeye	-16%	-7%
Coho	-18%	-8%

Population-level consequences from other affected salmonid assessment endpoints and other stressors of the action

In this section we present the population-level consequences from individual effects that are not amenable to population modeling. In most cases we lack the empirical data to conduct population modeling for these endpoints. Thus, we use qualitative methods to infer population-level responses. We focus on the population metrics of abundance and productivity. Both are metrics used by NMFS to assess a population's viability and both can be compromised by the chemicals assessed in this Opinion. Individual fitness consequences that reduce survival, growth, reproduction, or migration can lead to reduced salmonid population viability if sufficient numbers of individuals comprising a population are affected, and are more pronounced when affected over multiple generations. If the reductions in fitness result in reducing a population's survival or recovery potential, then we consider whether the ESU or DPS is impacted (See *Integration and Synthesis* section).

With the proposed action it is difficult to place an exact number on the percentage of a population that is affected or how frequently a population is affected because of the lack of information on and wide variability in the spatial and temporal uses of the registered formulations containing carbaryl, carbofuran, and methomyl, compounded by the imperfect data on where salmonids are at any given time. However, NMFS has sufficient information to make reasonable inferences from the available use, exposure, and response data on the likelihood of population-level consequences. Below we address whether the remaining fitness level consequences identified from the risk hypotheses affect the viability of salmonid populations. As mentioned earlier, we focus on the potential for reduced population abundance and productivity.

Impaired swimming and olfactory–mediated behaviors

All life stages of salmonids rely on their inherent ability to smell, to swim, and to navigate through a variety of habitats over their life span and to ultimately spawn successfully in natal waters and complete their life cycle. We have shown that exposure to carbaryl and carbofuran, compared to effects concentrations necessary to impair swimming behaviors, appears sufficient to do so in some environments, especially in rearing locations for the juveniles. Although no data were available to evaluate the effect of methomyl on swimming behavior, we find it reasonable to apply the conclusions drawn for carbaryl and carbofuran due to the chemicals sharing a similar mechanism of toxic action. Specifically, we expect that salmonids with impaired swimming behaviors from AChE inhibition will show reduced feeding, delayed or interrupted migration, reduced survival, and reduced reproductive success. We conclude that exposed populations are likely to have reduced abundance and productivity as a result of impaired swimming.

Based on the information we reviewed for carbaryl, carbofuran, and methomyl on salmonid olfaction, we find differences in expected responses. For carbaryl, we conclude that it is unlikely to affect salmonid olfaction at estimated concentrations. For methomyl, we located no information on its effects to fish olfaction, and given the variation in olfactory responses measured from other AChE inhibitors (OPs and carbamates) it is uncertain whether methomyl will affect olfaction. For carbofuran, definitive evidence shows that olfaction in fish is affected at low μg/L concentrations.

Because olfaction plays an important role in a suite of ecologically relevant behaviors that are affected when an individual salmonid's olfaction is impaired, we include this endpoint in our analysis. Lack of predator avoidance behaviors by juvenile and adult salmonids reduces the probability of surviving predation events. Juvenile salmonids with impaired olfaction may fail to properly imprint on their natal waters, which later in life leads to adults straying *i.e.,* migrating into and spawning in streams other than their natal stream. Adults that do not return to natal waters are a functional loss to recruitment of a population. Adult male salmonids that do find their way back to natal stream or river reaches and are subsequently exposed to the three insecticides may lose some or all of their olfactory capacity, even from a short-term exposure. Female salmonids release odorants to trigger male priming hormones and to alert males of a female's spawning condition. However, male fish with reduced olfactory capacity may not detect these cues, as demonstrated in a study on carbofuran (Bretaud, Saglio et al. 2002). Thus spawning synchronization could be compromised and recently laid eggs may go unfertilized. Unfertilized eggs may result in reduced productivity and abundance for a population if sufficient numbers of spawning events are missed. Again, we find it difficult to accurately predict when these impairments and missed spawning opportunities occur, primarily as a result of incomplete pesticide use information, difficulty in conducting field experiments with adult salmonids, and uncertainties surrounding extent of effects and concentrations which may trigger them. Because imprinting, avoiding predators, homing, and spawning are likely affected when exposed to carbofuran, we conclude these additional effects cannot be dismissed. Therefore, we expect populations exposed to carbofuran will show reduced reproductive rates, reduced return rates, and reduced intrinsic rates of growth when sufficient numbers of individuals are affected.

Starvation during a critical life stage transition

Salmonids emerge from redds (nests) with a yolk-sac as their initial food source (yolk-sac fry). Once the yolk-sac has been completely absorbed, they must begin exogenous feeding. Fry have limited energy reserves, and if they are unable to properly swim or detect and capture prey the onset of starvation occurs rapidly. Because juvenile salmon are limited by gape width, prey at this stage are limited to very small aquatic invertebrates. The stressors of the action likely affect this critical life stage transition in several ways, leading to increased early life stage mortality. Impaired swimming and olfaction affects the fry's ability to detect and capture prey. Prey may be killed outright by the stressors of the action leading to reduced prey availability or the complete absence of prey. These same areas also have off-channel habitats where fry seek shelter and food and those areas are highly susceptible to the highest concentrations of the three insecticides. Therefore, we expect reductions in a population's abundance where transitioning yolk-sac fry are exposed to the stressors of the action. All salmonid ESUs share this common life stage transition and therefore are at risk.

Death of returning adults

We discussed and analyzed with models the importance of juveniles to population viability. However, we did not address possible implications of returning adults dying from exposure to the stressors of the action. An adult returning from the ocean to natal freshwaters is important to a population's survival and recovery for many reasons. Notably, less than one percent of adults generally complete their life cycle. For populations with lambdas well below 1, every adult is crucial to a population's viability. We expect that some sensitive adults will die from short-term exposures before they spawn within some of the populations, particularly those that spawn in intensive agricultural watersheds and urban/suburban environments where elevated temperatures, other AChE-inhibiting insecticides, and low pHs are present. The lower end concentrations that kill 50% of exposed salmonids fall in the hundreds of μg/L. For methomyl, we expect that fewer adults would die compared to carbofuran and carbaryl based on differences in toxic potencies. We still expect that sensitive individuals exposed to concentrations below the LC50 will die. These concentrations are expected in habitats receiving drift from aerial applications. The persistence of these concentrations will vary with habitat, but for those habitats with lower pHs and minimal flow sensitive adult salmonids are expected to be killed. We cannot quantify the number of adults lost to a given population in a given year. For those populations where each adult salmonid is important to viability, we expect reductions in both productivity and abundance.

Synergistic toxicity

With certain combinations and specific concentrations of carbaryl and carbofuran, synergism occurs, translating into increased inhibition of AChE and in some cases increased rates of mortality among exposed salmonids. We have no data either supporting or refuting synergism for methomyl. We have no predictive models for synergistic toxicity. However, where we expect co-occurrence of the three insecticides, we expect synergism may occur if high enough concentrations exist. Generally, these concentrations must exceed individual LC50s for the three compounds, which is most likely to occur in areas with extensive crop uses where applications overlap in space and time. In these areas, even more fish would die from synergistic effects than predicted based on the additive toxicity for carbaryl, carbofuran, and methomyl. Juveniles and returning adults may experience synergistic toxicity. Whether or not death occurs is dependent on exposure duration and concentrations of the three insecticides. Typically, adults are less sensitive than early lifestages, however it is very difficult to conduct toxicity assays with pre-spawn adult salmoninds. Pre-spawn adults have used up most of their accumulated fat, converting it into gamete production, and will die soon after spawning. We are unsure how sensitive these adults are to toxic pesticides, but expect in their physiological state that they will be as or more sensitive than juveniles. We expect that some adults that occupy or spawn in shallow, low flow systems will be impacted from synergistic toxicity. Therefore, population-level effects could be more pronounced, depending upon the number of individuals and the importance of those individuals to the survival and recovery of the population. We conclude that based on the expected environmental concentrations of the three insecticides, synergism is likely in many off-channel habitats resulting in increased rates of death to juveniles and to adults.

Toxicity from other stressors of the action

We identified inert ingredients, adjuvants (NP), tank mixtures (recommended on pesticide product labels), degradates (1-napthol, 3-hydroxycarbofuran), and other pesticide a.i.s (malathion, bifenthrin, rotenone, metaldehyde, captan, and cupric sulfate) as toxic to salmonids and their prey. There remain substantial data gaps on the expected concentrations of these chemicals in salmonid habitats. However, some chemicals are detected at concentrations that pose substantial risk to listed salmonids and their prey (*e.g.,* malathion, NP). The risk posed by these other stressors to salmonid populations is complicated by the same factors we discussed for carbaryl, carbofuran, and methomyl (*i.e.,* the numbers of individuals exposed, the uncertainty surrounding the temporal and spatial uses of these chemicals, etc.). Severe population reductions of Atlantic salmon in Canada were attributed specifically to the use of NP within a pesticide formulation (Fairchild, Swansburg et al. 1999; Brown and Fairchild 2003). We conclude that given the use and co-application of these chemicals with carbaryl, carbofuran, and methomyl, exposed individuals are at increased risk of the suite of toxic effects expected from these substances. Substantial uncertainty also exists regarding the identity of other ingredients in formulations further complicating our ability to predict overall toxicity to salmonids and their prey as a result of pesticide use. Exposed populations are at increased risk of reduced abundance and productivity from these chemicals. However, NMFS is unable to accurately describe the level of risk.

Conclusion on population-level effects

We conclude that many of the populations of threatened and endangered salmonids covered by this consultation will likely show reductions in viability, particularly those that are comprised of juvenile life histories that rear for weeks to years in freshwater habitats found in intensive agricultural and residential/urban areas. Juvenile coho salmon, steelhead, and ocean- and stream-type Chinook salmon use these types of rearing areas for extended periods which overlaps with pesticide applications. Of greatest concern are for those independent populations for each ESU or DPS distributed in high use areas of carbaryl, carbofuran, and methomyl.

Based on these facts, we expect that where the geographic range of listed populations overlaps with intensive cropping patterns and urban/residential areas severe effects to abundance and productivity are anticipated from exposure to the three pesticides. However, due to the impending phase out of carbofuran, we expect the likelihood of exposure and therefore toxicity to decline rapidly as existing products stocks are depleted. Population effects are largely a result of reduced salmonid prey abundances and subsequent reduced growth of juveniles. Although less of a factor, individual fitness consequences are also likely in some areas due to impaired swimming.

Effects to Designated Critical Habitat: Evaluation of Risk Hypotheses

Presently, critical habitat has been designated for 26 of the 28 listed salmonid ESUs, all of which is located within the action area. Designated critical habitat within the action area includes spawning and rearing areas, freshwater migratory corridors, and nearshore and estuarine areas, and includes essential physical and biological features. When NMFS designated critical habitat for the 26 salmonid ESU/DPSs, NMFS identified PCEs essential for the conservation of the species. The PCEs NMFS identified are those sites and habitat components that function to support one or more lifestages. For this opinion, the primary PCEs potentially affected are prey availability and water quality degradation in spawning and rearing areas, freshwater migratory corridors, and nearshore and estuarine areas.

The effects of the proposed action on prey and water quality PCEs are addressed below by weighing the available evidence either supporting or refuting the critical habitat-related risk hypotheses. NMFS reviewed and presented the toxicological information available for habitat assessment endpoints within the *Response Analysis* section. Included are the PCEs prey availability and water quality. If these PCEs are likely impacted by the stressors of the action, we address the potential for reductions to the associated conservation value of the designated critical habitat in the *Effects of the Proposed Action to Designated Critical Habitat* Section.

Critical Habitat Risk Hypotheses based on potential effects to PCEs:

> *Risk Hypothesis 1. Exposure to the stressors of the action is sufficient to reduce abundances of aquatic prey items of salmonids.*

We evaluated two lines of evidence to determine whether this hypothesis is supported by the available information. The first is whether data support the occurrence of adverse effects to salmonid prey items from the stressors of the action. The second is whether reductions in abundances of salmonid prey items occur in areas of documented exposure to the stressors of the action. We found overwhelming evidence in support of the first line of evidence. The stressors of the action are expected to kill large numbers and types of aquatic species that serve as prey to salmonids, especially when carbaryl, carbofuran, and methomyl are present together and/or co-occur with other insecticides. The concentrations we summarized indicate that alone each of the insecticides can also kill prey at expected environmental concentrations.

The IBI and other metrics of aquatic community health were reviewed to evaluate the second line of evidence. In areas of intensive agriculture, where we expect use of the stressors of the action, biological integrity is often significantly reduced (Cuffney et al 1997). Many of the preferred salmonid prey items are present only in low numbers or absent altogether in these areas. We see similar depauperate communities in urban areas. We recognize many other limiting factors contribute to poor condition of these aquatic communities. However, these insecticides and their formulations may be responsible for a substantial portion. In fact, several studies have shown toxicity to salmonid prey items from field collected waters and sediment due to pesticide residues (Cuffney, Meador et al. 1997; Hall, Killen et al. 2006).

We therefore expect that spawning, rearing, and migratory areas will be affected by the stressors of the action resulting in reduced abundances of aquatic prey items. The magnitude in reduction of prey is difficult to ascertain, however where designated critical habitats overlap with high potential use areas such as agricultural or urban dominated flood plains, we expect pronounced reductions. These areas support fry and juvenile growth and development which are predicated on an abundant and diverse source of prey. Reductions in prey availability may affect the conservation value and the ability of the habitat features to support salmonids. In summary, the available information shows prey items of ESA-listed salmonids are affected by the stressors of the action to an extent warranting an analysis of whether the conservation value of designated critical habitat is negatively affected.

Risk hypothesis 2. Exposure to the stressors of the action is sufficient to degrade water quality in designated critical habitat.

We addressed this hypothesis by comparing exposure concentrations to toxicity data. The results of the comparison indicate that expected concentrations from the proposed action trigger adverse effect levels for salmonids and their prey (see *Exposure Analysis* and *Response Analysis* sections). Based on pesticide uses described in the *Description of the Action* section coupled with expected concentrations presented in the *Exposure Analysis* section (*i.e.,* concentrations derived from surface water monitoring data, EPA and NMFS modeling estimates), we expect these concentrations to be present in designated critical habitat and therefore to degrade water quality. In many of the watersheds containing designated critical habitats, water quality is identified as a major limiting factor to salmonid production. The proposed action is likely to further degrade water quality in spawning, rearing, and migratory areas. Collectively, this information supports the conclusion that designated critical habitats are likely degraded throughout the four states and further analysis is warranted to determine the potential to reduce the conservation value of designated critical habitats.

We attempted to compare expected concentrations of these pesticides to U.S. Water Quality Criteria designated pursuant the Clean Water Act, however none are published. We then evaluated whether any of the state waters within designated critical habitat are listed as impaired by carbaryl, carbofuran, and/or methomyl by searching state 303(d) lists (discussed in the *Environmental Baseline* section). Currently, state water quality standards have been developed for carbaryl and carbofuran, but not for methomyl. Unfortunately, surface water sampling is not conducted on all freshwater habitats (including designated critical habitats) within the action area and as a result we do not discount areas not sampled. For these reasons, it is also not possible to determine from 303(d) lists the magnitude of water quality degradation to designated critical habitat. We also do not rely on 303(d) lists as the sole source of information for degradation of water quality. In California, Oregon, and Washington some streams have been placed on the 303(d) list for carbaryl and carbofuran (see *Environmental Baseline* section) which indicates that water quality can be degraded by the stressors of the action in salmonid habitats. We located no listings for methomyl, and we located no listings for the three pesticides in the state of Idaho.

Areas of Uncertainty

In this section we list the predominant uncertainties and data gaps uncovered by our analysis of the effects of the proposed action. We do not discuss the entire suite of uncertainties, but highlight those likely have the most influence on the present analysis.

- Description of the action. We lacked a complete description of EPA-authorized uses of pesticides containing carbaryl, carbofuran, and methomyl as described in labeling of all pesticide products containing these a.i.s.

- Exposure resulting from non-agricultural uses. We lacked exposure estimates of stressors of the action associated with non-agricultural uses of these pesticides.

- Exposure to and toxicity of pesticide formulations and adjuvants. Minimal information was found on formulations, adjuvants, and on other/inert ingredients within registered formulations.

- Exposure to mixtures. We lacked information on permitted tank mixtures. Additionally, given that relatively few tank mix combinations are prohibited, it was not feasible to evaluate all potential combinations of tank mixtures. Pesticide mixtures are found in freshwater throughout the listed-salmonid habitat areas. However, mixture constituents and concentrations are highly variable.

- Toxicity of mixtures. The toxicity of most environmental mixtures is unknown.

- Synergistic responses. Exposure to combinations of carbaryl, carbofuran, methomyl, and/or other combinations of OP and carbamate insecticides can result in synergistic responses. However, we are not aware of a method to predict synergistic responses.

Cumulative Effects

Cumulative effects include the effects of future state, tribal, local, or private actions that are reasonably certain to occur in the action area considered by this Opinion. Future federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to section 7 of the ESA.

During this consultation, NMFS searched for information on future state, tribal, local, or private actions that were reasonably certain to occur in the action area. NMFS conducted electronic searches of business journals, trade journals, and newspapers using Google and other electronic search engines. Those searches produced reports on projected population growth, commercial and industrial growth, and global warming. Trends described below highlight the effects of population growth on existing populations and habitats for all 28 ESUs/DPSs. Changes in the near-term (five-years; 2014) are more likely to occur than longer-term projects (10-years; 2019). Projections are based upon recognized organizations producing best available information and reasonable rough-trend estimates of change stemming from these data. NMFS analysis provides a snapshot of the effects from these future trends on listed ESUs.

States along the Pacific west coast, which also 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. This is particularly true for coastal states. 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 [which also includes four additional states (Arizona, Utah, Nevada, and Alaska) beyond the action area] had a projected population of 65.6 million people in 2005. This figure will eventually grow to 70.0 million in 2010 and 74.4 million in 2015. At this rate, such growth will make the Pacific coast states the most populous region in the nation.

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 42 metropolitan areas that experienced a 10% growth or greater between 2000 and 2007, 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 (1.83%/year), Riverside-San Bernadino-Ontario, CA (3.63%/year), and Sacramento-Arden-Arcade-Roseville, CA (2.34%/year).

Urban Growth

As these cities border coastal or riverine systems, diffuse and extensive growth will increase overall volume of contaminant loading from wastewater treatment plants and sediments from sprawling urban and suburban development into riverine, estuarine, and marine habitats. Urban runoff from impervious surfaces and roadways may also contain oil, heavy metals, PAHs, and other chemical pollutants and flow into state surface waters. Inputs of these point and non-point pollution sources into numerous rivers and their tributaries will affect water quality in available spawning and rearing habitat for salmon. Based on the increase in human population growth, we expect an associated increase in the number of NPDES permits issued and the potential listing of more 303(d) waters with high pollutant concentrations in state surface waters.

Mining

Mining has historically been a major component of western state economies. With national output for metals increasing at 4.3% annually (little oil, but some gas is drawn from western states), output of western mines should increase markedly (Woods and Figueroa 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.

Agriculture

As the western states have large tracts of irrigated agriculture, a rise in agricultural output is anticipated. Impacts from heightened agricultural production will likely result in two negative impacts on listed Pacific salmonids (Woods and Figueroa 2007). 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.

Recreation

The western states are widely known for scenic and natural beauty. Increasing resident and tourist use will place additional strain on the natural state of park and nature areas that are also utilized 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 undesirable negative effects on water quality. They include increases in sedimentation, loss of riparian shade (increasing temperatures), increased point and non-point pollution discharges, decreased infiltration of rainwater (leading to decreases in shallow groundwater recharge, decreases in hyporheic flow, and decreases in summer low flows).

Nevertheless, there are also non-federal actions likely to occur in or near surface waters in the action area that may have beneficial effects on the 28 ESUs. They include implementation of riparian improvement measures, fish habitat restoration projects, and BMPs (*e.g.,* associated with timber harvest, grazing, agricultural activities, urban development, road building, recreational activities, and other non-point source pollution controls).

NMFS expects many of the current anthropogenic effects described in the *Environmental Baseline* will continue. Listed Pacific salmonids are exposed to harvest, hatchery, hydropower, and habitat degradation activities. With regard to water quality, fish are continually exposed to pesticides, contaminants, and other pollutants during their early life history phase and during adult migratory returns to their natal streams for spawning.

NMFS also expects the natural phenomena in the action area (*e.g.,* oceanographic features, ongoing and future climate change, storms, natural mortality) will continue to influence listed Pacific salmonids as described in the *Environmental Baseline*. Climate change effects are expected to be evident as alterations to water yield, peak flows, and stream temperature. Other effects, such as increased vulnerability to catastrophic wildfires, may occur as climate change alters the structure and distribution of forest and aquatic systems.

Coupled with EPA's registration of carbaryl, carbofuran, and methomyl, climate change, and 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

The Integration and Synthesis section is the final step of NMFS' assessment of the risk posed to listed Pacific salmonids and critical habitat as a result of EPA's registration of carbaryl, carbofuran, and methomyl. In this section, we perform two evaluations: whether it is reasonable to expect the proposed action is likely to (1) reduce both survival and recovery of the species in the wild and (2) result in the destruction or adverse modification of designated critical habitat.

To address species survival and recovery, we discuss the likelihood of the proposed action to reduce the viability of the salmonid species to such an extent that increased extinction rates are likely. Specifically, we address whether we expect reductions in a population's abundance and productivity from the stressors of the action to affect a species (ESU or DPS). We also address whether the conservation value of the physical and biological features of critical habitat is retained. We address whether we expect reductions in the PCEs for prey availability and degradation of water quality from the stressors of the action to reduce the conservation value of designated critical habitat. Conclusions for each ESU/DPS and associated designated critical habitat are found in the *Conclusion* section.

Effects of the Proposed Action at the Species Level

In our *Effects of the Proposed Action* section we assessed the effects of the federal action to listed Pacific salmonids. We discussed the exposure, response, and risks to individuals when they co-occur with the allowable uses of carbaryl, carbofuran, and methomyl. In addition to the a.i.s, we also considered their metabolites and degradates, other active and inert ingredients included in their product formulations, tank mixtures, and adjuvants authorized on their product labels within the action area. Our analysis also considered the possibility of co-occurrence of carbaryl, carbofuran, or methomyl with other AChE-inhibiting pesticides resulting in additive or synergistic effects to salmonids within freshwater habitats. We assessed the stressors of the action in the context of differing life histories, declining populations, and degraded water quality and habitat conditions within the distribution of listed species.

The life history of listed salmonids plays a large role in determining exposure. Adult salmon and steelhead spend weeks to several months in freshwater habitats during their migration and spawning. Immediately after emerging from the gravel substrate, salmonid fry move to habitat where they can swim freely and forage. Chinook salmon, coho salmon, and steelhead fry typically select off-channel habitats associated with their natal rivers and streams. Off-channel habitats provide fry with protection from the primary water flow within rivers and streams. Diverse invertebrate communities also populate off-channel habitats, adding to juvenile salmonids' reliance on these areas. Coho salmon and steelhead rear in freshwater for more than a year. Sockeye salmon fry most frequently distribute to shallow beach areas in the littoral zones of lakes before moving into deeper water and taking on a more pelagic existence. In contrast, chum salmon migrate downstream to estuaries near the mouth of their parent stream almost immediately. Fry reside in estuaries for as little as one or two weeks before moving offshore or into deeper habitats within the nearshore environment.

Based on general knowledge of field runoff and drift from pesticide applications, we expect higher pesticide concentrations in edge-of-field, low-flow, and shallow aquatic systems. Therefore, Chinook salmon, coho salmon, and steelhead have an increased risk of exposure due to their preference for off-channel habitats. Coho salmon and steelhead are at greater risk of exposure because of their extended residency periods.

We expect carbaryl, carbofuran, and methomyl found in surface water runoff and pesticide drift will cause acute lethality and sublethal effects to juvenile salmonids. Based on exposure for four days at the reported LC50s, a severe consequence to a salmonid population's growth rate is expected. The most pronounced effects are within off-channel habitats based on NMFS exposure estimates. The likelihood of population-level effects as a result of mortalities of juvenile salmonids increases as populations spend longer periods in freshwater adjacent to agricultural or developed areas. Species at the greatest risk include steelhead, coho, stream-type Chinook, and ocean-type Chinook salmon.

Methomyl concentrations required to result in population-level impacts are not expected to occur in the majority of aquatic habitats used by juveniles. However, we expect that carbaryl and carbofuran applications could cause population-level effects through acute mortality of juveniles, particularly in areas of intensive agriculture and large urban or residential areas. We also expect acute mortalities of juveniles from applications of carbaryl to forests and from exposure to 24(c) carbaryl applications in estuaries in Washington State. However, we do not know how many individuals are exposed each year due to these uses.

Sublethal effects caused by AChE inhibition, such as impaired swimming and olfactory-mediated behaviors, are also expected to result in adverse population-level effects. Swimming is a critical function for salmonids. Individuals with impaired swimming ability will show reduced feeding, delayed or interrupted migration, reduced survival, and reduced reproductive success. We expect that carbaryl, carbofuran, and methomyl will occur at concentrations high enough to impair swimming ability in occupied shallow water habitats. These effects may lead to population-level consequences that hinder the survival and recovery of listed species.

Olfactory-mediated behaviors are also vital for salmonid survival. These behaviors include detecting and avoiding predators, recognizing kin, locating natal waters for spawning, and reproduction. Juvenile salmonids with impaired olfactory-mediated behaviors can lose their ability to imprint on natal streams and avoid predators. Spawning is triggered by olfactory-mediated cues given and received by both sexes. If olfaction is inhibited, salmon may not recognize these cues, resulting in reduced reproduction. The available information suggests that carbaryl does not inhibit olfaction. Therefore, we do not expect to see effects to olfactory mediated behaviors from carbaryl exposure. Currently there is not enough information to determine the effect of methomyl on olfaction. However, exposure to carbofuran at low μg/L concentrations will likely result in the impairment of olfactory-mediated behaviors. That in turn will lead to increased predation and ultimately reduced juvenile and adult survivorship. As pesticide exposure increases we expect to see more pronounced effects due to additive and synergistic effects.

We also expect indirect effects from the a.i.s through reduced salmonid prey abundance and subsequent reduced juvenile growth and probability of survival. Feeding during the period immediately following emergence is vital for the survival of all ESA-listed Pacific salmonid fry.

Once fry have completely absorbed the yolk-sac, they are dependent on the local invertebrate population and may starve if sufficient food is not available. As salmonid prey are sensitive to the expected exposures to each of the insecticides, as well as from mixtures containing the three a.i.s, we expect death and a variety of sublethal effects to salmonid prey. If the salmonid prey base is limited while fry are in this life stage, juvenile growth will be significantly hindered. Many of the invertebrate communities are already degraded due to poor habitat conditions and poor water quality. These conditions make it more difficult for the prey base to recover following an exposure event. Thus, juvenile survivorship will be low and may consequently lead to a decrease in lambda for the affected population.

Based on the NMFS growth models, a four-day exposure to expected concentrations of carbaryl, carbofuran, and methomyl will substantially reduce a population's growth rate due to prey base reduction. All four life history types modeled demonstrated this effect. As exposure duration increases, we expect more pronounced effects on salmonid prey abundance and a longer prey recovery period leading to further reduced salmonid growth and decreased survival.

We expect that all effects will be more pronounced in drainage basins that include high use areas, *i.e.,* major agricultural, urban, or residential centers. To determine which ESUs/DPSs will be affected by the action, we compare the range of each species to the locations of high use land classes. Species that occupy major agricultural basins, such as the Willamette basin or California Central Valley, have a much higher risk of exposure than species found in uninhabited, non-agricultural areas.

California Coastal Chinook Salmon

The CC Chinook salmon ESU consists of 10 historically functionally independent populations in California from Humboldt County in the north to Sonoma County in the south. Historic annual escapement was around 73,000 fish with most of these being produced in the Eel River. Today, annual returns to the Eel River system is estimated at 150 to 2,800 fish. Runs in the Russian River may be viable, though the short-term trend is negative. Lack of data precludes development of population growth rates or trends for all other populations.

The *Status of Listed Resources* and *Environmental Baseline* sections indicate that fisheries, timber harvest, vineyards and other agriculture, non-native fish species, and migration barriers negatively affect this ESU. Adverse effects on Chinook salmon habitat include a high percentage of fines in the streams' bottom substrate, lack of large instream woody debris, reduced riparian vegetation, elevated water temperatures, degraded water quality, increased predation, and barriers that limit access to tributaries. The combined impacts from these multiple threats continue to affect the CC Chinook salmon. Accumulation of fine sediment in the bottom substrate, contamination of waters, lack of riparian vegetation, and non-natives fish species are expected to impact invertebrate abundance and composition in streams within the ESU.

Most of the agriculture and urban developments are concentrated in the Russian River valley and on alluvial plains of coastal rivers. Vineyards and smaller centers are dispersed along coastal watersheds. Based on the crop types, we expect carbaryl and methomyl are commonly applied throughout the growing season in the Russian River basin. We also expect use of the a.i.s on the alluvial plain of the lower Eel River and its estuary, and to a lesser extent along the lower portions and estuaries of other coastal streams. Carbaryl is expected to be used the entire year within urban/residential areas. Existing stocks of carbofuran are also expected to be applied throughout the growing season. In California, there are 61 pesticides that inhibit AChE approved for use and we expect that application of other AChE inhibiting pesticides will co-occur with the a.i.s, exacerbating adverse effects from AChE inhibition. However, monitoring data for pesticides in streams within the CC Chinook salmon ESU is lacking. Adults within Willapa Bay during 24(c) carbaryl application to oyster beds are likely exposed to this compound and may experience additional mortalities.

CC Chinook spawning occurs on gravel beds in the mainstem of rivers and larger tributaries. Fry use the floodplain, stream margins, and side channels to rear after emergence from the redds in December through mid-April. Downstream migration of juveniles starts as early as February and most enter marine waters before mid-summer.

We expect the proposed uses of carbaryl and carbofuran pesticide products that contaminate aquatic habitats will lead to individual fitness level consequences and subsequent population-level consequences. Land use and crop type data indicate that, while not a major agricultural center, the CC Chinook will be exposed to all three a.i.s. We expect that exposure to methomyl may lead to individual fitness consequences, but not to an extent that would affect population growth rates. The risk to this species' survival and recovery from the stressors of the action is high for carbaryl and carbofuran, but low for methomyl.

Central Valley Spring-run Chinook Salmon

The CV Spring-run Chinook salmon ESU includes four populations of Spring-run populations in the upper Sacramento River and its tributaries. The distribution within the Sacramento River basin is now mostly restricted to accessible areas below dams in the mainstem river and in three of its tributaries: Deer, Mill, and Butte Creeks. Abundance remains far below the estimated 700,000 once entering the Sacramento-San Joaquin Rivers system. The number of Spring-run Chinook salmon spawning in the Sacramento River has averaged about 9,800 annually since 2000. While most populations within the ESU are at or above replacement, the Sacramento River population has been steadily decreasing.

The major threats to this ESU identified in the *Status of Listed Resources* and *Environmental Baseline* sections include impaired or loss of habitat, predation, contamination, and water management. Reservoir dams in the Sacramento River have prevented the ESU from using its historic spawning locations. Physical channel habitat has been altered through sediment input from mining, levee construction, and removal of riparian vegetation for levee maintenance. Detected pesticides in the Sacramento River include thiobencarb, carbofuran, molinate, simazine, metolachlor, and dacthal, chlorpyrifos, carbaryl, and diazinon. State and federal water diversions in the south Sacramento-San Joaquin Delta (Delta) have resulted in increased mortality through prolonged migration and entrainment at the water diversion facilities. We also expect that application of other AChE inhibiting pesticides will co-occur with carbaryl, carbofuran, and methomyl in the waters within the ESU and exacerbate adverse effects from AChE inhibition.

With the high density of agriculture in the Sacramento River valley and the Sacramento-San Joaquin Delta (Delta), application of all three a.i.s is expected. Large urban centers occur along the Sacramento River and San Francisco Bay. Young and adult migrating Chinook salmon are also exposed to poor water quality from agricultural runoff that enters the Delta from the San Joaquin River. Carbaryl and methomyl can be applied to several crops throughout the growing season in the Central Valley. Existing stocks of carbofuran are also expected to be applied throughout the growing season. Carbaryl is approved for use throughout the year within urban/residential areas

The CV Spring-run Chinook salmon is categorized as an ocean-type fish. The young salmon emerge from the gravel from November through May and outmigration starts within four months. While the majority emigrates soon after emergence, some juveniles emigrate as yearlings with the onset of fall storms. Juveniles' seaward migration follows the mainstem Sacramento River, through the Delta, and across San Francisco Bay. Off-channel habitats within the lower Sacramento River floodplains, especially the Yollo Bypass, are particularity important for CV Chinook salmon juveniles during rearing and migration (Sommer, Nobriga et al. 2001; Sommer, Harrell et al. 2005).

The widespread uses of these pesticides have substantial overlap with rearing and migratory habitat of the CV Spring-run Chinook salmon ESU. The CV Spring-run Chinook salmon exhibit high variation in abundance, have restricted distribution, and are exposed to pesticides and other pollutants during rearing and downstream migration. We expect the proposed uses of the three carbamates may contaminate aquatic habitats and lead to both individual fitness and subsequentr population-level consequesnces. Therefore, the risk to this species' survival and recovery from the stressors of the action is high from carbaryl, carbofuran, and methomyl.

Lower Columbia River Chinook salmon

The LCR Chinook salmon ESU includes 32 historical populations in tributaries from the ocean to the Big White Salmon River, Washington and Hood River, Oregon. The ESU also includes 17 artificial propagation programs. LCR Chinook salmon numbers began to decline by the early 1900s from habitat degradation and excessive harvest rates. Many of these populations have low abundance. The annual population growth rates for 14 independent populations range from 0.93 to 1.037.

The major threats to this ESU identified in the *Status of Listed Resources* and *Environmental Baseline* sections include hydromorphological changes from hydropower development, loss of tidal marsh and swamp habitat, and reduced or eliminated access to subbasin headwaters from by the construction of non-federal dams. LCR Chinook salmon spawning and rearing habitats in tributary mainstems have been adversely affected by sedimentation, elevated water temperature, and reduced habitat diversity. 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.

NAWQA sampling in surface waters within the ESU range detected more than 50 pesticides in streams. Concentrations of ten pesticides, including carbaryl and carbofuran, also exceeded EPA's chronic toxicity aquatic life criteria (Wentz, Bonn et al. 1998). The combined impacts from these multiple threats continue to affect this ESU.

Most of the highly developed land and agricultural areas in this ESU's range are adjacent to salmonid habitat. Registered uses of carbaryl, carbofuran, and methomyl include applications to crop agricultural sites, residential sites, and urban sites. Registered 24(c) uses in Oregon include carbofuran application to potatoes, nursery stock, sugar beets, and watermelons. Based on land use patterns, we expect the highest exposure to these chemicals to occur in the Lewis River basin, Clackamas River basin, and Hood River basin.

Mature LCR Fall-run Chinook salmon enter freshwater in August through October to spawn in large river mainstems. After emergence, fry typically select off-channel habitats associated with their natal rivers and streams. Juveniles eventually emigrate from freshwater, usually within six months of hatching. LCR Spring-run Chinook salmon enter freshwater in March through June to spawn in upstream tributaries. These fish generally emigrate from freshwater as yearlings. As juveniles overwinter in shallow, freshwater habitats, they are likely to experience higher exposure to pesticides, other contaminants, and elevated temperature. In northern rivers, juveniles may rear in freshwater for two years or more. Given their long residence time in shallow freshwater habitats, LCR Chinook salmon are vulnerable to high pesticide exposures. Also, if adults and juveniles are within Willapa Bay during 24(c) carbaryl application to oyster beds they are likely exposed to this compound and may likely experience additional mortalities.

Given the life history of LCR Chinook salmon, we expect the proposed uses of carbaryl, carbofuran, and methomyl pesticide products may contaminate aquatic habitats and lead to individual fitness and subsequent population-level consequences. Therefore, the risk to this species' survival and recovery from the stressors of the action is high for carbaryl, carbofuran, and methomyl.

Upper Columbia River Spring-run Chinook salmon

The UCR Spring-run Chinook salmon ESU includes 11 populations and 7 artificial propagation programs in the state of Washington. The four known annual population growth rates range from 0.99 to 1.1. Based on 1980-2004 returns, the average annual population growth rate for this ESU is estimated at 0.93. One historical population is considered extinct.

The major threats to UCR Chinook salmon identified in the *Status of Listed Resources* and *Environmental Baseline* sections include reduced tributary stream flow and impaired fish passage from hydroelectric dams. Additionally, degradation of the tributary habitat and impaired water quality from development negatively affect this ESU. Pesticide detections in UCR Chinook salmon freshwater habitats are well documented. NAWQA sampling from 1992-1995 in the Central Columbia Plateau detected numerous pesticides in surface water (Williamson, Munn et al. 1998). Carbaryl was detected in 5% of samples, and carbofuran was detected in 6%. Methomyl was not detected. While detections of these three chemicals did not exceed water quality standards, concentrations of six other pesticides exceeded EPA criteria for the protection of aquatic life. Listed salmonids are commonly exposed to combinations of carbaryl, carbofuran, and methomyl, other compounds, and other mixtures of cholinesterase-inhibiting insecticides.

Registered uses of carbaryl, carbofuran, and methomyl include applications to crop agricultural sites, residential sites, and urban sites. Registered 24(c) uses in Washington State include carbofuran application to potatoes and spinach grown for seed. About 3% of the land has been developed and 3.5% has been cultivated for crops. Areas with the potential for high exposure in the Upper Columbia River basin are the Methow, Entiat, and Wenatchee River basins.

UCR Spring-run Chinook salmon begin returning from the ocean in the early spring. After migration, they hold in freshwater tributaries until spawning in mid- to late August. Fish spawn in the major tributaries leading to the Columbia River between Rock Island and Chief Joseph dams. UCR Spring-run Chinook salmon fry typically select off-channel habitats associated with their natal rivers and streams to rear. Juveniles spend a year in freshwater before migrating to the ocean in the spring of their second year of life. The duration of juvenile rearing in shallow freshwater habitats increases their susceptibility to higher exposures of pesticides, contaminants, and elevated temperature.

Given the life history of UCR Spring-run Chinook salmon, we expect the proposed uses of carbaryl, carbofuran, and methomyl pesticide products that contaminate aquatic habitats will lead to individual fitness and subsequent population-level consequences, *i.e.*, reductions in population viability. Therefore, the risk to this species' survival and recovery from the stressors of the action is high for carbaryl, carbofuran, and methomyl.

Puget Sound Chinook Salmon

The Puget Sound ESU includes all runs of Chinook salmon in the Puget Sound region from the North Fork Nooksack River to the Elwha River on the Olympic Peninsula. This includes 31 historic quasi-independent populations and 26 artificial propagations programs. Of the historic populations, only 22 are considered extant. The estimated total run size for this ESU in the early 1990s was 240,000 fish. During a recent five-year period, the geometric mean of natural spawners in populations of this ESU ranged from 222 to just over 9,489 fish (Good, Waples et al. 2005). Recent five-year and long-term productivity trends remain below replacement for the majority of the 22 extant populations of Puget Sound Chinook salmon. The annual population growth rate known for these populations ranged from 0.75 to 1.17.

The major threats to the Puget Sound Chinook salmon identified in the *Status of Listed Resources* and *Environmental Baseline* sections include degraded freshwater and marine habitat from agricultural activities and urbanization. Poor forestry practices have also reduced water quality in the upper river tributaries for this ESU. Elevated temperature, water diversions, and poor water quality across land use categories pose significant threats to the status of Puget Sound

Chinook salmon. Furthermore, there has been extensive urbanization in this region. Well over two million people inhabit the area, with most development occurring along rivers and coastline.

Pesticide use and detections in the ESU's watershed are well documented. NAWQA sampling conducted in 2006 in the Puget Sound basin detected numerous pesticides and other synthetic organic chemicals in streams and rivers. However, mixtures of chemicals found in agricultural and urban settings differ. Urban streams sampled in Puget Sound showed the highest detections for carbaryl, diazinon, and malathion. Carbaryl was detected at 60% of urban sampling sites (Ebbert, Embrey et al. 2000). Diazinon was detected at all urban sites, frequently at concentrations that exceeded EPA guidelines for protecting aquatic life (Bortleson and Ebbert 2000).

Registered uses of carbaryl, carbofuran, and methomyl include applications to crop agricultural sites, residential sites, and urban sites. In addition, Washington has 24(c) registrations for carbofuran use on spinach grown for seed and potatoes. Land classes indicate high-use of the three a.i.s in the following drainage basins: Nooksack, Skagit River, Stillaguamish, Skykomish, Snoqualimie, Snohomish, Lake Washington, Duwamish, and Puyallup. Roughly 7% of the lowland areas (below 1,000 ft elevation) in the Puget Sound region are covered by impervious surfaces, which increase urban runoff containing pollutants and contaminants into streams. Pollutants carried into streams from urban runoff include pesticides, heavy metals, PCBs, PBDEs, PAHs, pharmaceuticals, nutrients, and sediments. As such, we expect salmonid populations within the ESU to be exposed to carbaryl, carbofuran, and methomyl. Further, monitoring data shows that we can expect concurrent exposure with other AChE inhibiting chemicals, which will exacerbate the severity of effects. Adults within Willapa Bay during 24(c) carbaryl application to oyster beds are likely exposed to this compound and may experience additional mortalities.

Puget Sound stream-type Chinook salmon fry typically rear in shallow off-channel habitats associated with their natal rivers and streams. Juveniles generally have long freshwater residences of one or more years before migrating to the ocean.

Given the life history of the Puget Sound Chinook salmon, we expect that the proposed uses of carbaryl, carbofuran, and methomyl pesticide products may lead to individual fitness consequences and subsequent population-level consequences, *i.e.*, reductions in population viability. Therefore, the risk to this species' survival and recovery from the stressors of the action is high for carbaryl, carbofuran, and methomyl.

Sacramento River Winter-run Chinook salmon

The Sacramento River Winter-run Chinook salmon ESU includes only one population in Sacramento River, California. The current spawning distribution of Sacramento River Winter-run salmon is restricted to a short portion of the mainstem Sacramento River below Keswick Dam. Historic run estimates for the Sacramento River are as large as 200,000 fish (Brown et al. 1994). Estimated natural production has fluctuated greatly over the past two decades. In 2007, estimated natural production was 4,461 fish. The population's annual growth rate ranged from 0.870 to 1.090.

The major threats to this ESU identified in the *Status of Listed Resources* and *Environmental Baseline* sections indicate impaired or loss of habitat, predation, contamination, and water management negatively affect this ESU. Reservoir dams in the Sacramento River have eliminated the ESU from its historic spawning locations. Today, the ESU depends on the ability of the BOR to manage cold water through reservoir storage and releases to support adult holding, spawning, incubation, and rearing. The physical channel habitat has been altered through sediment input from mining, levee construction, and removal of riparian vegetation for levee maintenance. Pesticides are frequently detected in the Sacramento River including thiobencarb, carbofuran, molinate, simazine, metolachlor, dacthal, chlorpyrifos, carbaryl, and diazinon. Carbofuran is commonly detected in agricultural areas but less so in urban sites while carbaryl is commonly detected in all urban sites. Modification of hydrology has resulted in increased mortality through stranding, increased predation, prolonged migration, and entrainment at water diversion facilities.

About 10% of the land within this ESU is developed; large areas of urban centers occur along the Sacramento River and San Francisco Bay. As about 21% of land within the ESU is cultivated, all three a.i.s are expected to be applied to several crops within the ESU. Agriculture activity is prominent in the lower Sacramento River and within the Delta. Chinook salmon are also exposed to poor water quality from agricultural runoff that enters the Delta from the San Joaquin River. The Mediterranean climate in California, with dry summers and fall storms, may result in high concentration of these contaminants in run-off during the onset of the rainy season. These high concentrations can overlap with juvenile presence in the river system and movement into floodplains for rearing.

Winter-run adults enter the Sacramento River in early spring with spawning peaking in May and June. Spawning occurs in the Sacramento River downstream of the Keswick Dam. Fry rear in the Sacramento River for a few weeks to months before starting outmigration in late July, peaking in November and December. During outmigration, the young salmon migrate down the Sacramento River, through the Delta and San Francisco Bay. Juvenile winter-run Chinook salmon appear in the Delta from October to early May where they may rear in the fresher upstream portions for up to two months.

We expect that the proposed uses of carbaryl, carbofuran, and methomyl pesticide products will lead to both individual fitness level and subsequent population-level consequences. Registered uses of these a.i.s indicate overlap with the spawning, rearing, and migratory habitat of the one extant population in this ESU. Thus, the risk to this species' survival and recovery from the stressors of the proposed action is high for carbaryl, carbofuran, and methomyl.

Snake River Fall-run Chinook salmon

The SR Fall-run Chinook salmon ESU is comprised of a single population that spawns and rears in the mainstem Snake River and its tributaries below Hells Canyon Dam. Only 10 to 15% of the historical range of this ESU remains. Estimated historical returns from 1938 to 1949 were 72,000 fish annually (Bjornn and Horner 1980). The average abundance (1,273) of SR Fall-run Chinook salmon over the most recent 10-year period is below the 3,000 natural spawner average abundance thresholds identified as a minimum for recovery. The annual population growth rate for this single population is 1.02. Two historical populations are considered extirpated.

The major threats to this ESU identified in the *Status of Listed Resources* and *Environmental Baseline* sections include impaired stream flows and barriers to fish passage in tributaries from hydroelectric dams. During the 1960s and 1970s, approximately 80% of the ESU's historic habitat was eliminated or severely degraded by the construction of the Hells Canyon complex and the lower Snake River dams. Additionally, degraded freshwater habitats in the estuary, mainstem, and tributaries from development and land use activities negatively affect this ESU. Agricultural activities, urban communities, and industries are concentrated along the Snake River and near the mouths of major tributary valleys. Thus, stream water quality and biological communities in the downstream portion of the upper Snake River basin are degraded. The combined impacts from these multiple threats continue to affect SR Fall-run Chinook salmon.

Registered uses of carbaryl, carbofuran, and methomyl include applications to crop agricultural sites, residential sites, and urban sites. Registered 24(c) uses within this ESU include carbofuran application to potatoes and spinach grown for seed in Washington. The remaining spawning and rearing area of the SR Fall-run Chinook salmon is potentially a high-use area for these chemicals, as it is adjacent to agricultural and residential areas. About 14% of the land has been developed. Dryland agriculture occurs in the lower Clearwater Basin, with crops including wheat, peas, and lentils. Alfalfa, hay, and grasses are also grown in the lower Clearwater as well as the lower Salmon River basin. Thus, ESU exposure to these pesticides is likely. Adults within Willapa Bay during 24(c) carbaryl application to oyster beds are likely exposed to this compound and may experience additional mortalities.

Adult SR Fall-run Chinook salmon enter the Columbia River in July and August. Spawning occurs from October through November above Lower Granite Dam in the mainstem Snake River and in the lower reaches of the larger tributaries, such as the Salmon and Clearwater Rivers. Fry emerge from redds beginning in March or April, then rear for two months or more in the sandy littoral zone along the river margin. Juveniles begin migration in June along the edges of rivers, where they are at risk of exposure to higher concentrations of pesticides from drift and runoff. Their duration in shallow freshwater habitats increases their chances of higher exposure to pesticides, contaminants, and elevated temperature. Most fish exhibit an ocean-type life cycle, completing migration to the Pacific during the first year of life, though some overwinter inlow-velocity pools created by the hydroelectric dams.

Given the life history of SR Fall-run Chinook salmon, we expect the proposed uses of carbaryl and carbofuran pesticide products may lead to individual fitness level consequences and subsequent population-level consequences. We expect that exposure to methomyl may lead to individual fitness consequences, but not to an extent that would affect population growth rates. Land class data indicate that exposure of the carbamates to SR Fall-run Chinook salmon is possible. The risk to this species' survival and recovery from the stressors of the action is high for carbaryl and carbofuran, but low for methomyl.

Snake River Spring/Summer-run Chinook salmon

This ESU includes 32 historical populations in the Snake River basin which drains portions of southeastern Washington, northeastern Oregon, and north/central Idaho. Historically, the Salmon River system may have supported more than 40% of the total return of Spring/Summer-run Chinook salmon to the Columbia system (Fulton 1968). The long-term trends in productivity indicate a shrinking population. However, recent trends in productivity, buoyed by the last five years, are approaching replacement levels. The annual population growth known for 18 populations ranged from 0.97 to 1.1. Historical populations above Hells Canyon are considered extinct.

The major threats to this ESU identified in the *Status of Listed Resources* and *Environmental Baseline* sections include degraded water quality in the freshwater estuary, tributaries, and coastal habitats from land use activities and hydroelectric dams. Significant threats to SR Spring/Summer-run Chinook salmon include elevated temperature, water diversions, and poor water quality.

Some spawning and rearing areas of the SR Spring/Summer Run Chinook are in forested areas, where carbaryl may be applied. The likelihood of exposure in forested areas is unknown. Registered 24(c) uses within this ESU include carbofuran application to potatoes and spinach grown for seed in Washington. Dryland agriculture occurs in lower Clearwater Basin, with crops including wheat, peas, and lentils. Alfalfa, hay, and grasses are also grown in the lower Clearwater as well as the lower Salmon River basin. All three a.i.s are registered for use on alfalfa. Agricultural activities and urban communities, are concentrated along the Snake River and near the mouths of major tributary valleys, thus stream water quality and biological communities in the downstream portion of the upper Snake River basin are degraded.

SR Spring/Summer-run Chinook salmon spawn at high elevations in the headwater tributaries of the Clearwater, Grande, Ronde, Salmon, and Imnaha Rivers. Spawning is complete by the second week of September. Eggs incubate and hatch in late winter and early spring of the following year. The fry typically overwinter in shallow off-channel habitats associated with their natal rivers and streams. Juveniles become active seaward migrants during the following spring as yearlings (Connor, Sneva et al. 2005).

Given the life history of SR Spring/Summer-run Chinook salmon, we expect the proposed uses of carbaryl and carbofuran pesticide products that contaminate aquatic habitats will lead to individual fitness level consequences and subsequent population-level consequences. The uses of these materials along the migratory route of SR Spring/Summer-run Chinook salmon, may cause acute lethality or temporary AChE inhibition. We expect that exposure to methomyl may lead to individual fitness consequences, but not to an extent that would affect population growth rates. The risk to this species' survival and recovery from the stressors of the action is high for carbaryl and carbofuran, but low for methomyl.

Upper Willamette River Chinook Salmon

The UWR Chinook salmon ESU includes all eight naturally spawned populations residing in the Clackamas River and the Upper Willamette River above Willamette Falls. It also includes seven artificial propagation programs. The population in the McKenzie River is the only population that is naturally producing, and current estimates indicate a negative growth rate. Historically, the Upper Willamette River supported large numbers (exceeding 275,000 fish) of UWR Chinook salmon. Current abundance of natural-origin fish is estimated at less than 10,000.

The major threats to this ESU identified in the *Status of Listed Resources* and *Environmental Baseline* sections include habitat loss due to blockages from hydroelectric dams and irrigation diversions, and degraded water quality within the Willamette mainstem and the lower reaches of its tributaries. Elevated water temperature also poses a significant threat to the status of UWR Chinook salmon. 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 were detected in streams draining urban areas. Ten of these pesticides, including carbaryl and carbofuran, exceeded EPA criteria for the protection of freshwater aquatic life from chronic toxicity (Wentz, Bonn et al. 1998). The combined impacts from these threats continue to affect UWR Chinook salmon.

Based on the crop types, we expect that carbaryl and methomyl are commonly applied in the Willamette Valley throughout the growing season. Existing stocks of carbofuran are also expected to be applied throughout the growing season. Registered 24(c) uses in Oregon include carbofuran application to potatoes, nursery stock, sugar beets, and watermelons. The Willamette Basin is the largest agricultural area in Oregon. In 1992 the Willamette Basin accounted for 51% of Oregon's total gross farm sales and 58% of Oregon's crop sales. About one-third of the agricultural land is irrigated and most of it is adjacent to the mainstem Willamette River. Urban developments are also located primarily in the valley along the mainstem Willamette River. We expect carbaryl to be used throughout the year in urban and residential areas. Given that major urban and agricultural areas are located adjacent to the mainstem Willamette, ESU exposure to these pesticides is likely. We also expect that other AChE inhibiting pesticides will co-occur with carbaryl, carbofuran, and methomyl in the waters of the Willamette Valley and exacerbate adverse effects from AChE inhibition.

Chinook salmon fry typically select shallow off-channel habitats associated with their natal rivers and streams. Juveniles generally rear in freshwater for several months to more than one year before migrating to the ocean. Their duration in shallow freshwater habitats increases their susceptibility to higher exposures of pesticides, contaminants, and elevated temperature. UWR Chinook salmon exhibit an earlier time of entry into the Columbia River and estuary than other spring Chinook salmon ESUs (Meyers, Kope et al. 1998). Although most juveniles from interior spring Chinook salmon populations reach the mainstem migration corridor as yearlings, some juvenile Chinook salmon in the lower Willamette River are sub-yearlings (Friesen, Vile et al. 2004). Off-channel habitats within the Willamette Valley floodplain are particularity important for rearing fry and are actively being identified, reconnected and restored. We expect fry to be exposed to the three insecticides when applications overlap with fry occurrence and will further depress abundances of the available prey.

Based on land use and the life history of UWR Chinook salmon, we expect that this ESU will be compromised by reduced lambdas of affected populations. The risk to this species' survival and recovery from the stressors of the action is high for carbaryl, carbofuran, and methomyl.

Columbia River Chum Salmon

This ESU includes two remaining populations of 16 historical populations in the lower reaches (the Lower Gorge tributaries and Grays River) of the Columbia River. Thus, about 88% of the historic populations are extirpated or nearly so. In the early 1900s, the run numbered in the hundreds of thousands to a million returning adults. The size of the Lower Gorge population is estimated at 400-500 individuals, down from a historical level of greater than 8,900 (Good, Waples et al. 2005). Previous estimates of the Grays River population range from 331 to 812 individuals. However, the population increased in 2002 to as many as 10,000 individuals (Good, Waples et al. 2005). Overall, the lambda values indicate a long-term downward trend at 0.954 and 0.984, respectively.

The major threats to this ESU identified in the *Status of Listed Resources* and *Environmental Baseline* sections are hydromodification and habitat loss. Of the salmonids, chum salmon are most averse to negotiating obstacles in their migratory pathway. Thus, they are more highly impacted by the Columbia River hydropower system – specifically the Bonneville Dam (Johnson, Grant et al. 1997). The water quality in the lower Columbia River is poor. Recent USGS studies have demonstrated the presence of 25 pesticide compounds in surface waters, including carbaryl (Ebbert and Embry 2002). Although the habitat restoration project for the Grays River will likely provide some benefit to the population, we are unable to quantify the overall net effect for salmonids at this time.

Land use data indicate that the Columbia River chum salmon may be at risk of pesticide exposure. In addition to general uses, registered 24(c) uses in Oregon include carbofuran application to potatoes, nursery stock, sugar beets, and watermelons. The locations of high-pesticide use areas and the preferential use of river-edge habitat by chum salmon indicate that the species is at risk of pesticide exposure. The developed area surrounding the cities of Portland and Vancouver occurs along the migratory route of the Lower Gorge chum.

Columbia River chum salmon fry emerge between March and May and emigrate shortly thereafter to nearshore estuarine environments (Salo 1991). This is in sharp contrast to other salmonid behavior and indicates that chum salmon are less dependent on freshwater conditions for survival. After emergence, juvenile Columbia River chum salmon spend around 24 days feeding in the estuary. Adults return to spawn in the lower reaches of the Columbia River between the ages of two and five from mid-October through December. An average of ten days is spent in the freshwater by the spawning adults.

Given the life history of the Columbia River chum salmon, we expect that the proposed uses of carbaryl, carbofuran, and methomyl pesticide products may lead to individual fitness level consequences. We expect that exposure will occur, resulting in both acute lethality and sublethal olfactory-mediated effects. However, we do not expect these effects to occur at a scale that would have population-level effects. As chum fry are more precocious and quickly leave natal streams, they are less reliant on the local invertebrate population. Given the life history of Columbia River chum salmon, the risk to this species' survival and recovery from the proposed stressors of the action is low for carbaryl, carbofuran, and methomyl.

Hood Canal Summer-run Chum Salmon

This ESU includes 16 historical, naturally spawned populations of summer-run chum salmon in Olympic Peninsula Rivers between Hood Canal and Dungeness Bay, Washington, as well as eight artificial propagation programs. Of the historically existing populations, seven are believed to be extirpated. Most of the extirpated populations occurred on the eastern side of the canal. Only two of the remaining populations have long-term trends above replacement; long-term lambda values of the nine existing populations range from 0.85 to 1.39 (Good, Waples et al. 2005). The Hood Canal Summer-run chum salmon populations are the subject of an intense hatchery program intended to bolster numbers. As much as 60% of the spawning populations are hatchery-raised fish.

The major threat to this ESU identified in the *Status of Listed Resources* and *Environmental Baseline* sections is habitat degradation. The combined effects of degraded floodplains, estuarine, and riparian habitats, along with reduced stream flow and sedimentation, have had a profound negative impact on this ESU.

The land use and environmental data indicate that the Hood Canal Summer-run chum may be exposed to carbaryl, carbofuran, and methomyl. There is no cultivated crop land and less than 6% of the ESU is developed. Washington's 24(c) registrations for carbofuran use are not expected to be employed in this region. We do expect some exposure due to residential uses.

The Hood Canal Summer-run chum spawn from mid-September through mid-October (Tynan 1997). Emergence generally occurs from early February through mid April. Upon emerging, fry immediately commence downstream migration to estuaries (Tynan 1997). Upon arrival in the estuary, salmon fry inhabit nearshore areas in shallow water. In Puget Sound, they have been observed to reside in the top 6 inches of surface water and extremely close to the shoreline (Tynan 1997). This behavior increases the likelihood of acute exposure to drift and runoff events.

Given the life history of the Hood Canal Summer-run chum, we expect the proposed uses of carbofuran, carbaryl, and methomyl pesticide products may contaminate aquatic habitats and lead to individual fitness level consequences. We expect that exposure will occur, resulting in lethality and sublethal olfactory-mediated effects. We do not, however, expect that these effects will have population-level consequences. As chum fry are more precocious and quickly leave natal streams, they are less reliant on the local invertebrate population. Therefore, risk to this species' survival and recovery from the stressors of the action is low for carbaryl, carbofuran, and methomyl.

Central California Coast Coho Salmon

The CCC coho salmon ESU includes 11 historical independent populations within counties from Mendocino to Santa Cruz in California. Coho populations in three larger watersheds, as well as some in smaller watersheds, have been extirpated or are nearly so. Historical escapement has been estimated between 200,000 and 500,000 fish. Current escapements are not known, though a minimum of 6,570 adult coho salmon are estimated to return to coastal streams within the ESU. Long-term population trends do not exist for any of the populations in this ESU. More fish enter northern streams but variation in abundance between cohorts can be large with one cohort often dominating. Southern streams produce few naturally spawned fish of all cohorts.

CCC coho salmon populations have been adversely affected by loss of riparian cover, elevated water temperatures, alteration of channel morphology, loss of winter habitat, and siltation. High water temperatures prevent coho salmon from inhabiting several streams within the ESU. Pesticides are expected to enter rivers as drift during application in agricultural areas. 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. We expect that application of other AChE inhibiting pesticides will co-occur with carbaryl, carbofuran, and methomyl in the waters within the ESU and exacerbate adverse effects from AChE inhibition.

The majority of agricultural land use is concentrated in the Russian River watershed and watersheds south of the Golden Gate Bridge. High density urban development and urban centers occur in the San Francisco Bay and in the Russian River basin. All three a.i.s are expected to be applied to several crops within the Russian River basin and the Santa Cruz stratum during the growing season. Carbaryl is expected to be used within urban/residential areas throughout the entire year.

The movements of mature adults are influenced by stream flow; entry often occurs during the first large winter storms. Similarly, CCC coho salmon adults must wait in the lower river for sufficient flow to be able to reach spawning grounds. Fry emerge in spring and remain in the stream for up to 18 months. Newly emerged fry use backwater, side channels, and shallow channel edges. During winter, juveniles inhabit side channels, sloughs, backwater, and other protected channel features. The off-channel floodplain habitats provide feeding and growth opportunities that are important before smoltification and seaward migration in the spring. The presence of discrete brood years (BYs) makes coho salmon more vulnerable to environmental perturbations than other salmonids because a failed BY is unlikely to be replaced by other BYs.

Land use data indicate substantial overlap between high-use areas and fish runs in the Russian River, San Francisco Bay, and the Santa Cruz area. In several streams, one or more BYs are at the verge of extinction, and heavy pesticide exposure may result in the loss of a BY. We expect the proposed use of the three carbamates will lead to both individual fitness level consequences and subsequent population-level consequences. Therefore, the risk to this species' survival and recovery from the proposed action is high for carbaryl, carbofuran, and methomyl.

Lower Columbia River Coho Salmon

The LCR coho salmon ESU includes all naturally spawned coho salmon populations in streams and tributaries to the Columbia River in Washington and Oregon from the mouth of the Columbia up to and including the White Salmon and Hood rivers, and along the Willamette to Willamette Falls, Oregon. The ESU includes 26 historical populations and 25 artificial propagation programs. Over 90% of the historic populations of LCR coho salmon are considered extirpated. Most populations have very low numbers and have been replaced by hatchery production. Only two populations have a degree of natural spawning – the Sandy River and the Clackamas River. The annual population growth rates known for the Sandy River and Clackamas River are 1.102 and 1.028, respectively.

The major threats to LCR coho salmon identified in the *Status of Listed Resources* and *Environmental Baseline* sections include reduced water flow in the mainstem and estuary from irrigation diversions and hydroelectric dams. Additionally, degraded water quality in freshwater and tributary habitats negatively affects this ESU. Among the various types of habitat threats, elevated temperature, water diversions, and poor water quality have significant influence on the status of LCR coho salmon.

Pesticide use and detections in LCR coho salmon freshwater habitats are well documented. NAWQA sampling in surface waters within the ESU range detected more than 50 pesticides in streams within this ESU. Ten pesticides exceeded EPA's criteria for the protection of aquatic life from chronic toxicity, including carbaryl. The combined impacts from these multiple threats continue to affect this ESU.

Registered uses of carbaryl, carbofuran, and methomyl include applications to crop agricultural sites, residential sites, and urban sites. Registered 24(c) uses in Oregon include carbofuran application to potatoes, nursery stock, sugar beets, and watermelons. Agricultural and urban development occurs along the Columbia River and the basins of its major tributaries. Both the Sandy and Clackamas Rivers drain parts of the Willamette Basin – a major agricultural area. Thus, ESU exposure to these pesticides is likely.

LCR coho salmon enter freshwater from August through December. Coho salmon spawn in November and December: emergence from the redds occurs between March and July. Fry typically select off-channel habitats associated with their natal rivers and streams to rear. The juvenile coho salmon reside in shallow freshwater habitats for more than one year. The long residence in these habitats increases their likelihood of experiencing significant exposure to pesticides and other contaminants.

Given the life history of LCR coho salmon, we expect the proposed uses of carbaryl, carbofuran, and methomyl pesticide products may lead to both individual fitness level and subsequent population-level consequences. The risk to this species' survival and recovery from the stressors of the action is high for carbaryl, carbofuran, and methomyl.

SONCC *coho salmon*

The SONCC coho salmon ESU includes coho salmon in streams between Cape Blanco, Oregon, and Punta Gorda, California, and three artificial propagation programs in the Klamath, Trinity, and Rogue Rivers. Little information on abundance trends exists for the streams within the ESU. However, available information indicates that Eel River and Southern populations are at high risk from critically low abundances. Northern populations may have larger runs. Recent estimated escapement in the Klamath River is about 2,000 fish. The Rogue River spawner run ranged from 7,800 to 12,213 coho salmon from 1999 to 2001, though many are of hatchery origin.

The threats to this ESU include road crossings and other migration barriers, timber harvest and agricultural activities, and water management. Adverse effects on the SONCC coho salmon consist of barriers that limit access to tributaries, lack of large instream woody debris, reduced riparian vegetation, and elevated water temperature. The Klamath River, one of the largest rivers, is 303(d) listed because aquatic habitat has been degraded due to excessively high water temperatures and algae blooms associated with high nutrient loads, water impoundments, and agricultural water diversions (USEPA 1993). Pesticides used in land management activities are expected to enter stream mainstems and tributaries during application as drift and as runoff.

A large portion of the agricultural and municipal land uses within this ESU is concentrated in the upper Klamath and Rogue Rivers. A diverse array of crops is produced in these areas each year. Some agriculture and smaller urban centers are dispersed on the lower alluvial coastal plains and in rivers valleys of coastal rivers. Active forest management occurs throughout the watersheds within this ESU, and application of pesticide products is anticipated.

Spawning occurs from November through January, depending on the occurrence of fall and winter storms. Fry incubate for four to eight weeks before emergence in the spring. Newly emerged fry rear in backwater, side channels, and shallow channel edges for up to 18 months. During winter, juveniles inhabit side channels, sloughs, backwater, and other protected channel features. The off-channel and floodplain habitats provide feeding and growth opportunities that are important before smoltification and seaward migration in spring. The three-year life cycle of coho salmon, with limited exchange between cohorts, makes them more vulnerable to environmental perturbations than other salmonids.

Agricultural and urban land uses overlap substantially with this ESU, particularly in the Interior Rogue and Klamath rivers. Because of this overlap, we expect that the proposed uses of carbaryl, carbofuran, and methomyl will result in both individual fitness and population-level effects. In several streams, one or more BYs are at the verge of extinction. Exposure to the three a.i.s can result in loss of weak BYs. Low population abundance levels and the discrete cohort life history of the coho salmon make the SONCC ESU more vulnerable to environmental stressors. Therefore the risk to the survival and recovery of the SONCC ESU from the stressors of the action is high.

Oregon Coast Coho Salmon

The OC coho salmon ESU includes 11 naturally spawned populations in Oregon coastal streams south of the Columbia River and north of Cape Blanco and one hatchery stock. While none of the populations have become extinct, the ESU's current abundance levels are less than 10% of historic populations. OC coho salmon abundance estimates from 2000 to 2007 ranged from a low of 51,875 to a high of 260,000 naturally produced spawners. Long-term trends in ESU productivity remain strongly negative.

The major threats to this ESU identified in the *Status of Listed Resources* and *Environmental Baseline* sections are habitat degradation from logging, road construction, urban development, mining, agriculture, and recreation. Within the various types of habitat, elevated temperature, water diversions, and poor water quality also affect the status of OC coho salmon. The combined impacts from these multiple threats continue to affect OC coho salmon.

Crop rotation patterns and crop types influence the distribution and frequency of pesticides within an area. Based on limited agricultural activities, we expect that carbaryl and existing stocks of carbofuran may be applied on a limited scale throughout the growing season. Registered 24(c) uses in Oregon include carbofuran application to potatoes, nursery stock, sugar beets, and watermelons. We also expect carbaryl use within urban and residential areas throughout the entire year. However, we expect no to very low applications of methomyl throughout the growing season. Although carbaryl and carbofuran may be applied to pine seedlings, we expect no to low applications in these forests. As runoff from urban and agricultural areas may drain into adjacent streams, ESU exposure to these pesticides is likely. However, given the above land uses and expected low application of the three carbamates, we expect lower incidences of fish exposure to these compounds in surface waters following application.

OC coho salmon enter rivers in September or October; spawning occurs in December. Fry emerge between March and July, then move to shallow areas near the stream banks. Juvenile coho salmon are often found in small streams less than five ft wide, and may migrate considerable distances to rear in lakes and off-channel ponds. Generally, coho salmon spend 18 months rearing in freshwater before moving out into the ocean. Given this duration spent in shallow freshwater habitats, they are more likely to experience higher pesticide exposure and contaminants.

Given the limited applications of the three carbamates and low incidences within land uses that overlap with the range of OC coho, exposure is possible, although not likely to be extensive. We do not expect that exposure will have effects at the population level. Therefore, the risk to this species' survival and recovery from the stressors of the action is low for carbaryl, carbofuran, and methomyl.

Ozette Lake Sockeye Salmon

This ESU is made up of only one historic population. Natural spawning aggregations remain on two beaches of Ozette Lake. Two tributary spawning groups were initiated in1992 through hatchery programs. Peak run size in the 1940s has been estimated to be between 3,000 and 18,000 fish, and actual production (*i.e.*, including harvest) may have been as high as 50,000. Recent estimates put the population at 3,600 – 4,600 individuals (Haggerty, Ritchie et al. 2007). 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 likely low. Uncertainty in past population counts coupled with poorly documented historical abundance prevents calculation of population growth rates and trends.

Major threats to this population identified in the *Status of Listed Resources* and *Environmental Baseline* sections are siltation of spawning habitat from logging activities within the watershed and genetic effects from past interbreeding with kokanee. Almost 80% of the land cover for this

ESU is evergreen forest. Between 1940 and 1984, 85% of the basin was clear-cut logged (Blum 1988). Roughly 77% of the land in Ozette Basin is managed for timber production (Jacobs, Larson et al. 1996). The extent to which pesticide products are currently used by these companies is unknown.

Ozette Lake is in a sparsely populated area, with less than 1% of land developed. No crop land was identified in NLCD data (Table 32). The land use and environmental data indicate that the Ozette Lake sockeye salmon may be exposed to carbaryl or carbofuran if applied in the watershed. Carbaryl is registered for a number of non-agricultural applications which may take place in forested and residential areas. Additionally, the area has a small population, making residential use of carbaryl within the ESU a possibility. We do not expect 24(c) registrations of carbofuran will be used within this ESU, though it may be applied to pine seedlings. However, there are few data available on use and no monitoring data are currently available. We do not expect methomyl to be used within the boundaries of this ESU.

Ozette Lake sockeye salmon enter the lake between April and August, and spawning occurs late October through February. Fry emerge from gravel redds in the spring and emigrate to the open waters of the lake where they remain for a full year. They then smolt as 1-year olds and migrate to the open ocean. The majority of Ozette Lake sockeye salmon return to spawn as four-year old fish after spending two full years at sea.

Given the life history of the Ozette Lake sockeye, we expect that that the proposed uses of carbaryl and carbofuran pesticide products may contaminate aquatic habitats used by sockeye in a way that might lead to individual fitness level consequences. While the uses of these materials may lead to some overlap with the Ozette Lake sockeye, the existing and likely future land uses should limit the applications of pesticides containing the three carbamates. Consequently, the risk posed by the proposed action to Ozette Lake sockeye salmon's survival and recovery is low for carbaryl, carbofuran, and methomyl.

Snake River Sockeye Salmon

The SR sockeye salmon ESU historically includes populations in five Idaho lakes as well as artificially propagated sockeye salmon from the Redfish Lake Captive Broodstock Program. Only one hatchery-sustained population remains and is found in Redfish Lake. This population is listed as endangered and has an extremely high risk of extinction. Current smolt-to-adult survival of sockeye originating from the Stanley Basin lakes is rarely greater than 0.3% (Hebdon, Kline et al. 2004). No natural origin adults have returned to Redfish Lake to spawn since 1998; the population is maintained entirely by propagation efforts. Around 30 fish of hatchery origin return to spawn each year (FCRPS 2008).

The major threats to this ESU identified in the *Status of Listed Resources* and *Environmental Baseline* sections include impaired tributary flow and passage, migration barriers, degraded water quality, and hydromodification of the Columbia and Snake Rivers. Like the Ozette Lake ESU, the SR sockeye occupy a relatively undeveloped area with very little cropland (Table 37) However, the SR sockeye have the longest migration of any sockeye salmon, traveling 900 miles inland. These waters are contaminated by drift and runoff from both agricultural and urban areas. Exposure during migration likely adds to the low survivorship of smolts. The land use and environmental data indicate that the SR sockeye may be exposed to carbaryl, carbofuran, and methomyl during migration. Registered 24(c) uses within this ESU include carbofuran application to potatoes and spinach grown for seed in Washington.

Historically, sockeye salmon entered the Columbia River system in June and July, and arrived at Redfish Lake between August and September (FCRPS 2008). Spawning occurred in lakeshore gravel and generally peaked in October. Fry emerged in the spring (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. Adult sockeye spent two or three years in the open ocean before returning to Redfish Lake to spawn.

During adult and juvenile migrations the sockeye are at their greatest risk of exposure to the stressors of the action. Sockeye salmon making the 900 mile journey each way pass along many miles where agricultural crops are at the river's edge. Drift and runoff occurring in conjunction with sockeye salmon migration is expected to cause adverse effect.

Given the life history of the SR sockeye, we expect the proposed uses of carbaryl, carbofuran, and methomyl pesticide products may lead to individual fitness level consequences. However, we do not expect fry to be exposed to the a.i.s, although some exposure may occur during adult migration and lead to acute lethality or temporary AChE inhibition. However, this exposure is not expected to occur at a frequency that would cause effects at the population level. Therefore, the risk to this species' survival and recovery from the stressors of the action is low for carbaryl, carbofuran, and methomyl.

Central California Coast steelhead

The CCC steelhead DPS includes all naturally spawned steelhead in streams from the Russian River south to Aptos Creek. This area includes streams entering the San Francisco Bay. The DPS consists of nine historic independent populations. There is limited information on the abundance of these populations, but all are in severe decline.

The major threats to this DPS identified in the *Status of Listed Resources* and *Environmental Baseline* sections are dams and other migration barriers, urbanization and channel modification, agricultural activities, predators, hatcheries, and water diversions. Throughout the species' range, habitat conditions and quality have been degraded by a lack of channel complexity, eroded banks, turbid and contaminated water, low summer flow and high water temperatures, an array of contaminants found at toxic levels, and restricted access to cooler head waters from migration barriers. There is limited monitoring data from streams within this DPS. The combined impacts of these threats continue to affect the CCC steelhead.

Crop farming is concentrated in low laying areas and floodplains along the estuaries and stream valleys of the Russian River and Santa Cruz drainage basins. Christmas trees production occurs in these areas in addition to a variety of food crops. Carbaryl and methomyl are registered for a variety crops grown in the Russian River valley and in the San Mateo and Santa Cruz counties. Registered 24(c) uses for carbaryl in California include application to fruits and nuts, prickly pear cactus, ornamental plants, and non-food crops. Methomyl has 24(c) registrations for insect control on ornamentals, beans, soybeans, radishes, sweet potatoes, Chinese broccoli, broccoli raab, and pumpkins. The a.i.s are therefore expected to be used throughout the growing season. Existing stocks of carbofuran are also expected to be applied throughout the growing season. Further, 16% of the DPS' range consists of developed urban and residential land, mostly within the San Francisco Bay and Russian River basin. Carbaryl is expected to be used the entire year within the urban/residential areas.

The CCC steelhead populations are all of the winter-type. Fry emerge in spring and rear in smaller tributaries and off-channel habitats. During periods with high flows, they move into floodplain habitat. Juvenile steelhead remain in freshwater for one or more years before migrating downstream to smolt.

The registered uses of the three carbamates indicate substantial overlap with the CCC steelhead populations, especially in the San Francisco Bay, Russian River, and Santa Cruz area. We expect that the proposed uses of the three carbamates may contaminate spawning and rearing habitats and lead to both individual fitness level and population -level consequences. Thus, the risk to this species' survival and recovery from the stressors of the proposed action is high for carbaryl, carbofuran, and methomyl.

California Central Valley Steelhead

The CCV steelhead DPS includes all naturally spawned steelhead in the Sacramento River, San Joaquin River, and their tributaries. This area includes streams entering the Sacramento-San Joaquin Delta (Delta) east of Chipps Island. All populations rear and migrate through the Sacramento River, San Joaquin River, Delta, and San Francisco Bay. The current distribution is severely reduced and fragmented compared to historical distributions. About 6,000 river miles of river habitat have been reduced to 300 miles. Historical returns within the DPS may have approached two million adults annually. Current annual run size for the entire Sacramento-San Joaquin system today is estimated at less than 10,000 returning adults.

The *Status of Listed Resources* and *Environmental Baseline* sections indicate that dams and other migration barriers, urbanization and channel modification, agricultural activities, introduced non-native predators, hatcheries, and large scale water management and diversions negatively affect this DPS. Steelhead habitat has been highly degraded by reduced channel complexity, eroded banks, increased water temperature, migration barriers restricting access to cooler head waters, and decreased water quality from contaminants. Numerous NAWQA, CDPR, and other assessments found high concentration of contaminants in both the San Joaquin and Sacramento Rivers and their tributaries. In the San Joaquin Basin, seven pesticides exceeded EPA criteria for aquatic life. These pesticides include diuron, trifluralin, azinphos-methyl, carbaryl, chlorpyrifos, diazinon, and malathion. The combined impacts from these threats continue to affect the CCV steelhead.

Approximately 27% of the land within the DPS is developed for cultivation of crops (Table 38). High densities of crop farming occur throughout the San Joaquin Basin, in the Sacramento-San Joaquin Delta, and along lower Sacramento River. Further, 9.2% of the DPS consists of urban development. Based on the crop types, we expect carbaryl, carbofuran, and methomyl are commonly applied in the Central Valley throughout the growing season. In urban and residential areas, we expect carbaryl to be applied throughout the year. Monitoring in the San Joaquin basin found 48 of 83 pesticides tested for. Seven pesticides (diuron, trifluralin, azinphos-methyl, carbaryl, chlorpyrifos, diazinon, and malathion) exceeded criteria for the protection of aquatic life. Pesticides detected in the Sacramento River included thiobencarb, carbofuran, molinate, simazine, metolachlor, dacthal, chlorpyrifos, carbaryl, and diazinon. Carbofuran was most often detected in agricultural sites while carbaryl was also found in all urban sites tested.

All of the steelhead populations within this DPS exhibit the winter-type life history, though detailed information about the CCV steelhead life history is not available. Juvenile steelhead remain in freshwater for one or more years before migrating downstream to enter the ocean. The CCV steelhead use the lower Sacramento River and the Delta for rearing and as migration corridor. Some may utilize tidal marshes, non-tidal freshwater marshes, and other shallow areas in the Delta for rearing areas for short periods during outmigration to the ocean.

The registered uses of the three carbamates indicate substantial overlap with the CCV steelhead populations, especially in the lower Sacramento River and Delta. We expect that the proposed uses may contaminate spawning and rearing habitats and lead to both individual fitness and population-level consequences. Thus, the risk to this species' survival and recovery from the stressors of the proposed action is high for carbaryl, carbofuran, and methomyl.

Lower Columbia River Steelhead

The LCR Steelhead DPS includes 23 historical, naturally-spawned steelhead populations in Columbia River tributaries on the Washington side between the Cowlitz and Wind Rivers in Washington and on the Oregon side between the Willamette and Hood Rivers. Historical counts from the Cowlitz, Kalama, and Sandy Rivers suggest the ESU probably exceeded 20,000 fish. During the 1990s, fish abundance dropped to 1,000 to 2,000 fish. 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 annual population growth rate known for nine independent populations ranged from 0.945 to 1.06.

The major threats to this DPS identified in the *Status of Listed Resources* and *Environmental Baseline* sections include dams; water diversion; destruction or degradation of riparian habitat; and land use practices (logging, agriculture, and urbanization). Tributary hydropower development has created barriers and reduced fish access, ultimately affecting the spatial structure within the independent populations. Within the various types of habitat, water diversions, elevated temperature, and poor water quality also affect the status of LCR steelhead. Pesticides have also been detected in LCR steelhead habitats. NAWQA sampling from 1991-1995 in surface waters within the DPS range detected more than 50 pesticides in streams. Ten pesticides exceeded EPA's criteria for the protection of aquatic life from chronic toxicity. These pesticides include carbaryl, carbofuran, atrazine, chlorpyrifos, and malathion. The combined impacts from these multiple threats continue to affect this DPS.

Most of the highly developed land and agricultural areas in this DPS's range are adjacent to salmonid habitat. Registered uses of carbaryl, carbofuran, and methomyl include applications to crop agricultural sites, residential sites, and urban sites. Registered 24(c) uses in Oregon include carbofuran application to potatoes, nursery stock, sugar beets, and watermelons. Based on the crop types, we expect that carbaryl and methomyl are commonly applied in the Lewis River subbasin throughout the growing season and within urban and residential areas throughout the entire year for carbaryl. Existing stocks of carbofuran are also expected to be applied throughout the growing season. Thus, DPS exposure to these pesticides is likely.

This DPS includes winter- and summer-run types. Summer-run steelhead return to freshwater between May and November. They enter the Columbia River in a sexually immature condition and require several months in freshwater before spawning. Winter-run steelhead enter freshwater from November to April. These fish are close to sexual maturation and spawn shortly after arrival in their natal streams. Steelhead fry typically rear in off-channel habitats associated with their natal rivers and streams for more than a year. Given this duration, juveniles are likely to experience pesticide exposure.

Given the life history of LCR steelhead, we expect the proposed uses of carbaryl, carbofuran, and methomyl pesticide products may contaminate rearing habitats and lead to individual fitness and subsequent population-level consequences. The risk to this species' survival and recovery from the stressors of the action is high for carbaryl, carbofuran, and methomyl.

Middle Columbia River Steelhead

The MCR steelhead DPS includes 19 populations in Oregon and Washington subbasins upstream of the Hood and Wind River systems to and including the Yakima River. Historical run estimates for the Yakima River imply that annual species abundance may have exceeded 300,000 returning adults (Busby, Wainwright et al. 1996), whereas only 1,000 – 4,000 currently spawn. The most recent 10-year period indicated trends in abundance were positive for approximately half of the independent populations and negative for the remainder. Growth rates ranged from 0.97 to 1.02. Two historical populations are considered extinct.

Agricultural development is high within the range of MCR steelhead. Based on crop types and allowable uses, we expect that carbaryl and methomyl are commonly applied in the Yakima River basin, Walla Walla River basin, Deschutes River basin, and the John Day River basin throughout the growing season. Existing stocks of carbofuran are also expected to be applied throughout the growing season. In urban and residential areas, carbaryl will likely be applied throughout the entire year. Registered 24(c) uses within this ESU include carbofuran application to potatoes, nursery stock, sugar beets, and watermelons in Oregon and to potatoes and spinach grown for seed in Washington. Given the high amount of agriculture in these areas, we also expect other AChE inhibiting pesticides will co-occur with carbaryl, carbofuran, and methomyl and exacerbate adverse effects from AChE inhibition. Thus, DPS exposure to these pesticides is expected to be high

Mature adults (three to five years old) may enter rivers any month of the year and spawn in late winter or spring. Swim–up fry usually inhabit shallow water along banks of streams or aquatic habitats on stream margins. Steelhead rear in a variety of freshwater habitats and most remain in freshwater for two to three years. Some individuals, however, have stayed for as many as six to seven years. Most MCR steelhead smolt at two years and spend one to two years in the ocean prior to re-entering the freshwater to spawn.

Intense agricultural development has a high degree of overlap with the spawning and rearing areas of the MCR steelhead DPS. Given the long residency periods in freshwater, we expect that the proposed uses of of the three carbamates may lead to both individual fitness and population-level consequences. Therefore, the risk to this species' survival and recovery from the stressors of the action is high for carbaryl, carbofuran, and methomyl.

Northern California Steelhead

The NC steelhead DPS includes all naturally spawned steelhead in California coastal river basins from Redwood Creek southward to, but not including, the Russian River, as well as two artificial propagation programs. Historical estimates of annual production for this DPS are upwards of 200,000 fish. Exact information on abundance is lacking for most of the streams within the DPS, though most populations are in decline. The DPS includes 15 historically independent populations of winter steelhead and 10 populations of summer steelhead. None of the winter steelhead populations are viable due to low abundance and production.

The major threats to this DPS identified in the *Status of Listed Resources* and *Environmental Baseline* sections are timber harvest, forest roads, vineyard developments, and road crossings and dams. Stressors to the NC steelhead include lack of large instream woody debris, reduced riparian vegetation, elevated water temperature, increased predation, and barriers that limit access to tributaries. The combined impacts from these multiple threats continue to affect the NC steelhead.

About 85% of the land use for this DPS consists of forests and chaparral (Table 26). Atlhough carbaryl may be applied in forested areas, the extent of exposure is unknown. There are areas of Mendocino County with a high density of vineyards, and there is some crop farming in low laying areas and floodplains along the estuaries and lower reaches of coastal streams. Coastal communities are located at the mouths of several streams within the NC steelhead DPS. Based on the crop types, we expect carbaryl, carbofuran, and methomyl to be commonly applied throughout the growing season in these areas. Carbaryl is expected to be used the entire year within urban and residential areas. However, because agricultural and urban development is not concentrated in any one basin, the risk of exposure to any given population is decreased. Additionally, the majority of steelhead juveniles rear at higher elevations in the watersheds with less agricultural influence. The fry, then, are less likely to be exposed during critical life stages.

The winter-type steelhead enters rivers as mature adults between November and April to spawn. The summer-type enters the stream in immature condition between May and October. Early arriving summer steelehead move high into the upper watersheds where they hold in deep pools throughout the summer before spawning in fall. In northern California, juvenile steelhead remain in freshwater for two or more years before migrating downstream to smolt. Juvenile steelheads spend a variable amount of time in the estuary before entering the open ocean.

We expect the proposed uses of carbaryl, carbofuran, and methomyl pesticides products within this DPS may lead to individual fitness effects, ranging from acute lethality to temporary AChE inhibition. However, exposure of NC steelhead populations to the three carbamates will be limited given the amount of overlap between agriculture, residential and urban development and spawning and rearing habitat. As such, we do not expect exposure to result in population-level consequences. Therefore, the risk to the survival and recovery of NC steelhead from the proposed action is low for carbaryl, carbofuran, and methomyl.

Puget Sound Steelhead

The Puget Sound steelhead is comprised of 21 populations, some of which have both summer and winter runs. Steelhead occur in all major watersheds within the Sound and out to the Elwha River. Of the winter runs, 17 had declining and four had increasing population trends. Summer run abundance trends are not available. Estimated run size for Puget Sound steelhead in the early 1980s is approximately 100,000 winter-run steelhead and 20,000 summer-run steelhead.

The major threats to this DPS identified in the *Status of Listed Resources* and *Environmental Baseline* sections are habitat degradation from logging, road construction, urban development, mining, agriculture, and recreation; water diversions; and poor water quality. In particular, elevated temperature, water diversions, and poor water quality have the most significant influences on the status of Puget Sound steelhead. Furthermore, there has been extensive urbanization in this region. Well over two million people inhabit the area, with most development occurring along rivers and coastline.

Pesticide use and detections in the DPS's watershed are also well documented. 2006 NAWQA sampling in the Puget Sound basin detected 26 pesticides and 74 other synthetic organic chemicals in streams and rivers, with different mixtures of chemicals linked to agricultural and urban settings. Urban streams sampled in Puget Sound showed the highest detections for carbaryl, diazinon, and malathion. Diazinon was also frequently detected in urban streams at concentrations that exceeded EPA guidelines for protecting aquatic life (Bortleson and Ebbert 2000).

Registered uses of carbaryl, carbofuran, and methomyl include applications to crop agricultural sites, residential sites, and urban sites. In addition, Washington has 24(c) registrations for carbofuran use on spinach grown for seed and potatoes. Land classes indicate high use of the three a.i.s in the following drainage basins: Nooksack, Skagit River, Stillaguamish, Skykomish, Snoqualimie, Snohomish, Lake Washington, Duwamish, and Puyallup. Roughly 7% of the lowland areas (below 1,000 ft elevation) in the Puget Sound region are covered by impervious surfaces, which increase urban runoff containing pollutants and contaminants into streams. Pollutants carried into streams from urban runoff include pesticides, heavy metals, PCBs, PBDEs, PAHs, pharmaceuticals, nutrients, and sediments. Thus, we expect salmonid populations within the ESU to be exposed to carbaryl, carbofuran, and methomyl. Further, monitoring data shows that we can expect concurrent exposure with other AChE inhibiting chemicals, leading to a pronounced severity of effects.

Summer steelhead enter freshwater between May and October, while winter steelhead enter between November and April. Spawning generally occurs in late winter or spring. Immediately after leaving the gravel, swim-up fry usually inhabit shallow water along banks of stream or aquatic habitats on stream margins. Steelhead rear in a wide variety of freshwater habitats, generally for two to three years. However, they may possibly reside up to six to seven years in freshwater environments. Following the rearing period, they smolt and migrate to sea in the spring.

We expect the proposed uses of carbaryl, carbofuran, and methomyl pesticide products that contaminate aquatic habitats may lead to both individual fitness level and subsequent population-level consequences. The risk to this species' survival and recovery from the stressors of the action is high for carbaryl, carbofuran, and methomyl.

Snake River Basin Steelhead

The SR steelhead DPS includes 23 naturally spawned populations below impassable natural and man-made barriers in streams in the Columbia River Basin upstream from the Yakima River, Washington, to the U.S.-Canada border. The Snake River supports about 63% of the natural-origin production of steelhead in the Columbia River Basin. The 10-year average for natural-origin steelhead passing Lower Granite Dam between 1996 and 2005 is 28,303 adults. Annual population growth rates show mixed long- and short-term trends in abundance and productivity. The annual population growth rate is known for eight independent populations and range 0.89 to 1.08. One historical population is likely extirpated.

The major threats to this DPS identified in the *Status of Listed Resources* and *Environmental Baseline* sections include hydrosystem mortality, water diversions, excessive sediment, and degraded water quality. Elevated temperature also impacts the status of SR steelhead. Pesticides have been detected in SR steelhead freshwater habitats. NAWQA sampling in 1992-1995 in the DPS's watersheds detected Eptam, atrazine, desethylatrazine, metolachlor, and alachlor. Carbaryl and carbofuran were detected in only 1% of samples. The combined impacts from these multiple threats continue to affect SR steelhead.

Some spawning and rearing habitat of the Snake River steelhead is in forested areas, where carbaryl may be applied. However, the likelihood of exposure in forested areas is unknown. Registered 24(c) uses within this ESU include carbofuran application to potatoes and spinach grown for seed in Washington. Dryland agriculture occurs in lower Clearwater Basin, with crops including wheat, peas, and lentils. Alfalfa, hay, and grasses are also grown in the lower Clearwater as well as the lower Salmon River basin. All three a.i.s are registered for use on alfalfa. Agricultural activities and urban communities are concentrated along the Snake River and near the mouths of major tributary valleys, thus stream water quality and biological communities in the downstream portion of the upper Snake River basin are degraded. Given that urban and agricultural areas are located adjacent to streams in the Lower Clearwater River drainage, DPS exposure to these pesticides is likely.

Sexually immature adult Snake River summer steelheads enter the Columbia River from late June to October. Adults migrate upriver until they reach Snake River tributaries where they spawn between March and May of the following year. Emergence occurs by early June in low elevation streams and as late mid-July at higher elevations. After hatching, juvenile SR steelhead typically select off-channel habitats associated with their natal rivers and streams for rearing. They spend two to three years in freshwater before they smolt and migrate to the ocean. Juvenile steelhead are more likely to be exposed to pesticides and other contaminants because of their long freshwater residency period.

Given the life history of SR steelhead, we expect the proposed uses of carbaryl and carbofuran pesticide products that contaminate aquatic habitats will lead to individual fitness level consequences and subsequent population-level consequences. The uses of these materials in some natal areas and along migratory routes may cause acute lethality, temporary AChE inhibition, and reduced growth. We expect that exposure to methomyl may lead to individual fitness consequences, but not to an extent that would affect population growth rates. The risk to SR steelhead survival and recovery from the stressors of the action is high for carbaryl and carbofuran, but low for methomyl.

South-Central California Coast Steelhead

The S-CCC steelhead DPS includes all naturally spawned steelhead in streams from the Pajaro River to the Santa Maria River. The major basins in the S-CCC steelhead range are the Pajaro River system and the Salinas River system. Carmel River, a smaller basin, also supports a stable run of steelhead. Historic abundance estimates for the DPS imply an annual return may have been upwards of 20,000 fish. The estimated production in five of the major rivers indicates a return of less than 500 adults.

The major threats to this DPS identified in the *Status of Listed Resources* and *Environmental Baseline* sections are dams and other migration barriers, urbanization and channel modification, agricultural activities, and wildfires. These activities has resulted in reduced channel complexity, eroded banks, increased water temperature, migration barriers restricting access to cooler head waters, and decreased water quality from contaminants.

About 7% of the DPS is developed for cultivation of crops; a large portion of this land is in the Salinas River valley. Crops are concentrated in low laying areas and floodplains along the estuaries and lower reaches of streams. Based on the crop types, we expect carbaryl, carbofuran, and methomyl are commonly applied within the DPS throughout the growing season. Approximately 8% of the DPS has been developed for urban and residential purposes. Developed areas are located at the mouth of several streams within the S-CCC steelhead DPS. We expect carbaryl use in urban/residential areas throughout the entire year.

All S-CCC steelhead are a winter-run life history. Juvenile steelhead remain in freshwater for one or more years before migrating downstream to smolt. Steelhead in this area consists of two life history groups: one where the juveniles rear for one or two years mainly in freshwater and another where the juveniles rear at the upper end of coastal lagoons for the first or second summer.

In conclusion, based on the long freshwater residence time of steelhead and the considerable overlap between S-CCC steelhead distribution and expected pesticide use, we expect that the S-CCC may experience high levels of exposure. We expect that proposed uses of the three carbamates that contaminate spawning and rearing habitats may lead to both individual fitness and population-level consequences. Therefore, the risk to the survival and recovery of S-CCC from the proposed action is high for carbaryl, carbofuran, and methomyl.

Southern California Steelhead

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. It is estimated that the species' current distribution constitutes about 1% of the historical distribution. Current abundance is considerably reduced with an estimated escapement of 500 fish for four of the larger rivers. Long-term estimates and population trends are lacking for the streams within the DPS.

The major threats to this DPS identified in the *Status of Listed Resources* and *Environmental Baseline* sections are dams and other migration barriers, urbanization and channel modification, agricultural activities, and wildfires. As a result of these activities, the stream substrate contains a high proportion of fines, stream channels lack complexity, banks are eroding, migration barriers restrict fish access to cooler head waters and tributaries, and the water is turbid and contaminated.

Agriculture is concentrated in low laying areas and floodplains along the estuaries and lower reaches of streams. Carbaryl, carbofuran, and methomyl are expected to be commonly used for several crops during the growing season. Large areas of dense development exist in the counties of Santa Barbra to San Diego. Carbaryl is expected to be used within urban/residential areas throughout the entire year. The NAWQA analysis detected more than 58 pesticides in ground and surface waters within the heavily populated Santa Ana basin, including multiple AChE inhibitors. The invertebrate community in the basin is heavily altered by pesticide influence. Carbaryl was detected in 42% or urban samples but few detections exceeded standards for protection of aquatic life. Carbofuran was also detected but not in levels exceeding standards. The application of other AChE inhibiting pesticides is expect to co-occur with the a.i.s, and exacerbate adverse effects from AChE inhibition.

All steelhead populations in this DPS are winter-type. Juvenile steelheads remain in freshwater for one or more years before migrating downstream to smolt. SC steelhead consists of two life history groups: one where the juveniles rear for one or two years mainly in freshwater and another where the juveniles rear at the upper end of coastal lagoons for the first or second summer.

We expect the proposed uses of carbaryl, carbofuran, and methomyl pesticides products that contaminate aquatic habitat will lead to both individual fitness consequences and subsequent population-level consequences. There is substantial overlap between urban and agricultural areas and the spawning and rearing habitats of the SC steelhead populations. Therefore, the risk to this species' survival and recovery from the stressors of the action is high for carbaryl, carbofuran, and methomyl.

Upper Columbia River Steelhead

The UCR steelhead DPS includes all naturally spawned populations below natural and man-made impassable barriers in the Columbia River basin upstream from the Yakima River, Washington, to the U.S.-Canada border. The DPS is comprised of four naturally-spawned populations and six artificial propagation programs in Washington State. The historical independent populations are located within the Wenatchee, Entiat, Methow, and Okanogan river systems. Abundance data indicate that all populations are below the minimum threshold for recovery. The annual population growth rates range between 1.067 and 1.086. Overall adult returns are dominated by hatchery fish.

The major threats to this DPS identified in the *Status of Listed Resources* and *Environmental Baseline* sections include dams that block fish migration and alter river hydrology; water diversions; destruction or degradation of riparian habitat; and land use practices (logging, agriculture, urbanization). Elevated water temperature and poor water quality also affect the status of UCR steelhead. Pesticides have also been detected in UCR steelhead freshwater habitats. NAWQA sampling from 1992-1995 in the Central Columbia Plateau detected numerous pesticides in surface water (Williamson et al. 1998). Carbaryl was detected in 5% of samples, and carbofuran was detected in 6%. Methomyl was not detected. While detections of these three chemicals did not exceed water quality standards, concentrations of six other pesticides exceeded EPA criteria for the protection of aquatic life. The co-occurrence of atrazine with carbaryl and other OPs in aquatic habitats increases the likelihood of adverse responses in salmonids and their aquatic prey. The combined impacts from these multiple threats continue to affect UCR steelhead.

Land uses within this DPS range include agricultural, urban and residential development. Based on the crop types, we expect that carbaryl and methomyl are commonly applied in the Wenatchee, Methow, and Entiat subbasins throughout the growing season. Existing stocks of carbofuran are also expected to be applied throughout the growing season. As runoff containing pesticides and other contaminants from the above land uses drain into the Wenatchee, Entiat, and Methow Rivers, DPS exposure to these pesticides is likely. We also expect other AChE inhibiting pesticides will co-occur with carbaryl, carbofuran, and methomyl in the waters of the Wenatchee, Entiat, Methow, and Lower Crab Creek subbasins and exacerbate adverse effects from AChE inhibition.

UCR adults return to the Columbia River in the late summer and early fall. UCR steelhead spawn and rear in the major tributaries to the Columbia River between Rock Island and Chief Joseph dams. Adults reach spawning areas in late spring of the calendar year following entry into the river. Newly emerged fry move about considerably and seek suitable rearing habitat, such as stream margins or cascades. Steelhead fry typically select off-channel habitats associated with their natal rivers and stream for extended periods of rearing. Fry move downstream in the fall in search of suitable overwintering habitat (Chapman, Hillman et al.

1994). Most juvenile steelhead spend two or three years in freshwater before migrating to the ocean. Some juvenile steelhead may spend up to seven years rearing in freshwater before migrating to sea. Given the long duration in shallow freshwater habitats, juveniles are more likely to experience higher pesticide exposure, contaminants, and elevated temperature.

We expect that the proposed uses of carbaryl, carbofuran, and methomyl may contaminate the rearing habitat of UCR steelhead and lead to individual fitness and population-level effects. Therefore, the risk to this species' survival and recovery from the stressors of the action is high for carbaryl, carbofuran, and methomyl.

Upper Willamette River Steelhead

The UWR steelhead DPS includes all naturally spawned late-fall populations below natural and man-made impassable barriers in the Willamette River, Oregon, and its tributaries upstream from Willamette Falls to the Calapooia River. The DPS is comprised of four historical populations. Steelhead populations within this DPS have been declining, on average, since 1971 and have exhibited large fluctuations in abundance. Long-term trends in the annual population growth rate are less than one. The annual population growth rate of the four independent populations ranged from 0.97 to 1.023.

The major threats to this ESU identified in the *Status of Listed Resources* and *Environmental Baseline* sections include habitat loss due to blockages from hydroelectric dams and irrigation diversions, and degraded water quality within the Willamette mainstem and the lower reaches of its tributaries. Additionally, pesticide use and detections in UWR steelhead freshwater habitats are well documented. 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 carbaryl and carbofuran, exceeded EPA criteria for the protection of freshwater aquatic life for chronic toxicity (Wentz, Bonn et al. 1998). The combined impacts from these multiple threats continue to affect UWR steelhead.

Based on the crop types, we expect that carbaryl and methomyl are commonly applied in the Willamette Valley throughout the growing season. Existing stocks of carbofuran are also expected to be applied throughout the growing season. Registered 24(c) uses in Oregon include carbofuran application to potatoes, nursery stock, sugar beets, and watermelons. The Willamette Basin is the largest agricultural area in Oregon. In 1992 the Willamette Basin accounted for 51% of Oregon's total gross farm sales and 58% of Oregon's crop sales. About one-third of the agricultural land is irrigated and most of it is adjacent to the mainstem Willamette River. Urban developments are also located primarily in the valley along the mainstem Willamette River. We expect carbaryl to be used throughout the year in urban and residential areas. Given that major urban and agricultural areas are located adjacent to the mainstem Willamette, DPS exposure to these pesticides is likely. We also expect that other AChE inhibiting pesticides will co-occur with carbaryl, carbofuran, and methomyl in the waters of the Willamette Valley and exacerbate adverse effects from AChE inhibition.

UWR steelhead enter the Willamette River in January and February and ascend to their spawning areas in late March or April. After emergence, steelhead fry typically rear in off-channel habitats associated with their natal rivers and streams for two to three years. Smolt migration past Willamette Falls begins in early April and extends through early June, with peak migration in early to mid-May. Most spend two years in the ocean before re-entering freshwater to spawn.

Much of the Willamette Valley has been converted to agricultural purposes. Based on the co-occurrence of high use land classes with spawning and rearing habitats, we expect that the proposed uses of the three carbamates may contaminate spawning and rearing habitats and lead to both individual fitness and population-level effects. Therefore, the risk to this species' survival and recovery from the stressors of the action is high for carbaryl, carbofuran, and methomyl.

Summary of Species-Level Effects

In the preceding section NMFS described expected population-level effects in terms of reductions in annual growth rate, productivity (reproduction), and abundance (numbers of salmonids). We concluded that all but Ozette Lake sockeye salmon, Snake River sockeye salmon, Northern California steelhead, Columbia River chum salmon, Hood Canal summer-run chum salmon, and Oregon Coast coho salmon populations will likely show reductions in viability due to the reregistration of carbaryl and carbofuran. We also concluded that all but Ozette Lake sockeye salmon, Snake River sockeye salmon, Northern California steelhead, Columbia River chum salmon, Hood Canal summer-run chum salmon, Oregon Coast coho salmon, Snake River fall-run Chinook salmon, Snake River spring-summer-run Chinook salmon, California Coastal Chinook salmon, and Snake River steelhead populations will likely show reductions in viability due to the reregistration of methomyl. The effects of EPA's proposed action are first manifested at the individual-level where reductions in individual fitness are expected. We showed that an individual's survival, migration, and growth are all significantly reduced by the proposed action. We also showed that these reductions are likely intensified by co-occurring stressors in the action area including the presence of other carbamates and OP insecticides in the action area.

Therefore, given the severity of expected changes in the annual population growth rate for affected populations, it is likely that California Coastal Chinook salmon, Central Valley spring-run Chinook salmon, LCR Chinook salmon, Puget Sound Chinook salmon, Sacramento River winter-run Chinook salmon, Snake River fall-run Chinook salmon, Snake River spring/summer-run Chinook salmon, UCR spring-run Chinook salmon, Upper Willamette River Chinook salmon, Central California Coast coho salmon, Southern Oregon and Northern Coastal California coho salmon, Central California Coast steelhead, California Central Valley steelhead, LCR steelhead, MCR steelhead, Snake River Basin steelhead, South-Central California Coast steelhead, Southern California steelhead, UCR steelhead, and Upper Willamette River steelhead will experience reductions in viability, which ultimately reduces the likelihood of survival and recovery of these species. The Ozette Lake sockeye salmon, Snake River sockeye salmon, Northern California steelhead, Columbia River chum salmon, Hood Canal summer-run chum salmon, and Oregon Coast coho salmon will not likely experience reductions in viability.

Effects of the Proposed Action to Designated Critical Habitat

NMFS' critical habitat analysis determines whether the proposed action will destroy or adversely modify critical habitat for ESA-listed species by examining any change in the conservation value of the essential features of critical habitat. Our analysis does not rely on the regulatory definition of 'adverse modification or destruction' of critical habitat. Instead, this analysis focuses on statutory provisions of the ESA, including those in Section 3 that define "critical habitat" and "conservation," those in Section 4 that describe the designation process, and those in Section 7 setting forth the substantive protections and procedural aspects of consultation.

NMFS has designated critical habitat for all listed Pacific salmonids except for LCR coho salmon and Puget Sound steelhead. The action area encompasses all designated critical habitat areas considered in this Opinion. The PCEs for each listed species, where they have been designated, are described in the *Status of Listed Resources* section of this Opinion. The PCEs identify those physical or biological features that are essential to the conservation of the species that may require special management considerations or protections. As the species addressed in this Opinion have similar life history characteristics, they share many of the same PCEs. These PCEs include sites essential to support one or more life stages (sites for spawning, rearing, migration, and foraging) and contain physical or biological features essential to the conservation of the ESU/DPS, such as:

1. freshwater spawning sites with water quantity and quality conditions and substrate supporting spawning, incubation and larval development;
2. freshwater rearing sites with water quantity and floodplain connectivity to form and maintain physical habitat conditions and support juvenile growth and mobility; water quality and forage supporting juvenile development; and 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, along 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, along with water quality, water quantity, and salinity conditions supporting juvenile and adult physiological transitions between fresh and saltwater; natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, and side channels; and juvenile and adult forage, including aquatic invertebrates and fishes, supporting growth and maturation;
5. nearshore marine areas free of obstruction with water quality and quantity conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation; and

natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, and side channels; and
6. offshore marine areas with water quality conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation.

At the time that each habitat area was designated as critical habitat, that area contained one or more PCEs within the acceptable range of values required to support the biological processes for which the species use that habitat. We identified PCEs of prey availability and water quality in freshwater spawning sites, freshwater rearing sites, migratory corridors, and estuarine and nearshore marine areas as susceptible to the effects of the proposed action. We evaluated whether prey availability and water quality are likely affected by the proposed action and formulated risk hypotheses for these PCEs. We relied upon pesticide use information (see the *Description of the Action* section), surface water monitoring data, EPA modeling estimates, and NMFS modeling estimates (see *Exposure Analysis* section) to determine likely aquatic concentrations. Based on our analysis within the *Effects of the Proposed Action* section, the proposed action will reduce prey availability and degrade water quality within freshwater spawning habitat, freshwater rearing habitat, and migration corridors.

Direct exposure to carbaryl, carbofuran, and methomyl and the other chemical stressors of the action within freshwater will have an effect on Pacific salmonid critical habitat that overlaps with intense agricultural and residential/urban land uses. The *Environmental Baseline* discusses the extent of anthropogenic alteration of salmonid habitat within the action area. As established in *Effects of the Action to Listed Species* Section, agricultural, urban, and residential areas overlap with salmonids' geographic range to differing degrees between ESUs/DPSs. As noted in the *Effects of the Proposed Action* section, pesticides most often occur in the aquatic environment as mixtures. Carbaryl, carbofuran, and methomyl are found in environmental mixtures. Based on evidence of additive and synergistic effects of these compounds, we expect mortality of large numbers and types of aquatic insects, which are prey items for salmon. A reduction in salmonid prey abundance poses subsequent impacts on the overall growth of salmonid juveniles, especially during their first year of survival.

Reduction in Prey

Freshwater spawning, rearing, and migratory habitats must provide forage areas that support juvenile development. A reduction in the abundance of prey items will decrease the conservation value of these habitats, as they will support fewer individuals, especially during a salmonid's first year of survival. Given the environmental baseline conditions of the aquatic systems, the existing invertebrate community is already depauperate in many of these areas and requires longer periods for recovery following each pesticide application event. The a.i.s in surface runoff and pesticide drift entering freshwater rearing habitats will exacerbate reductions in insect communities that are salmonid prey items in these degraded systems. We expect that designated critical habitats that support juvenile feeding and growth will be contaminated by the proposed uses of carbaryl, carbofuran, and methomyl to the extent that the habitat is precluded from serving its intended role in the survival and recovery of listed salmonids. The overall conservation value of designated critical habitat, whether low, medium, or high, will not meet its intended role to support the survival and recovery of the species.

Reduction in Water Quality

Drift and runoff from areas of intensive urban and agricultural development will likely contain carbaryl, carbofuran, and methomyl in addition to other pesticides - particularly other AChE-inhibiting pesticides, chemical pollutants, and sediments that also degrade water quality. Depending on the available water flow, amount of shade from LWD and intact riparian zones, and water temperature in aquatic habitats, the toxicity of carbaryl, carbofuran, and methomyl in tributary and stream waters may become more pronounced. Reductions in water quality may reduce the conservation value of designated habitats used for spawning, rearing, and migration. Furthermore restoration actions promoted in many of the salmonid recovery plans focus on increasing flood plain connectivity and creating new off-channel habitats. These actions are proposed in agricultural and urban flood plains that overlap with uses of the three insecticides. Water quality (as well as prey availability) may be degraded in these newly constructed habitats from the stressors of the action- effectively precluding intended benefits to rearing juvenile salmonids. We expect that proposed uses may contaminate these areas, thereby precluding habitat from its intended purpose in supporting the survival and recovery of listed Pacific salmonids.

The precise change in the conservation value of critical habitat within the ESU/DPS from the proposed action cannot be quantified and will likely vary according to the specific designated critical habitat. However, based on the effects described above, it is reasonably likely that the proposed action will have a large, local, negative reduction in that conservation value of designated critical habitat within highly developed agricultural, residential, and urban areas. The duration, frequency, and severity of these reductions will vary according to overall numbers and volume of applications of carbaryl, carbofuran, and methomyl in areas of designated critical habitat, among other variables.

Therefore, we expect that the registration of carbaryl and carbofuran will adversely affect designated critical habitat of all listed Pacific salmonids except for that of Ozette Lake sockeye salmon, Snake River sockeye salmon, Northern California steelhead, Columbia River chum salmon, Hood Canal summer-run chum salmon, and Oregon Coast coho salmon. We also expect that the reregistration of methomyl will adversely affect designated critical habitat of all listed Pacific salmonids, except for that of Ozette Lake sockeye salmon, Snake River sockeye salmon, Northern California steelhead, Columbia River chum salmon, Hood Canal summer-run chum salmon, Oregon Coast coho salmon, Snake River fall-run Chinook salmon, Snake River spring-summer-run Chinook salmon, California Coastal Chinook salmon, and Snake River steelhead.

Conclusion

Carbaryl and Carbofuran

After reviewing the current status of California Coastal Chinook salmon, Central Valley spring-run Chinook salmon, LCR Chinook salmon, Puget Sound Chinook salmon, Sacramento River winter-run Chinook salmon, Snake River fall-run Chinook salmon, Snake River spring/summer-run Chinook salmon, UCR spring-run Chinook salmon, Upper Willamette River Chinook salmon, Central California Coast coho salmon, LCR coho salmon, Southern Oregon and Northern Coastal California coho salmon, California Central Valley steelhead, Central California Coast steelhead, LCR steelhead, MCR steelhead, Puget Sound steelhead, Snake River Basin steelhead, South Central California coast steelhead, Southern California steelhead, UCR steelhead, and Upper Willamette River steelhead, the environmental baseline for the action area, the effects of the proposed action, and the cumulative effects, it is NMFS' Opinion that the registration of carbaryl and carbofuran is likely to jeopardize the continued existence of these endangered or threatened species (Table 81).

It is NMFS' Opinion that the registration of carbaryl and carbofuran is not likely to jeopardize the continued existence of Ozette Lake sockeye salmon, Snake River sockeye salmon, Northern California steelhead, Columbia River chum salmon, Hood Canal summer-run chum salmon, and Oregon Coast coho salmon (Table 81).

After reviewing the current status of designated critical habitat for California Coastal Chinook salmon, Central Valley spring-run Chinook salmon, LCR Chinook salmon, Puget Sound Chinook salmon, Sacramento River winter-run Chinook salmon, Snake River fall-run Chinook salmon, Snake River spring/summer-run Chinook salmon, UCR spring-run Chinook salmon, Upper Willamette River Chinook salmon, Columbia River chum salmon, Central California Coast coho salmon, Southern Oregon/Northern Coastal California coho salmon, California Central Valley steelhead, Central California Coast steelhead, LCR steelhead, MCR steelhead, Snake River Basin steelhead, South-Central California coast steelhead, Southern California steelhead, UCR steelhead, and Upper Willamette River steelhead, the environmental baseline for the action area, the effects of the proposed action, and the cumulative effects, it is NMFS' Opinion that the registration of carbaryl and carbofuran, is likely to result in the destruction or adverse modification of critical habitat of these endangered and threatened species (Table 82).

It is NMFS' Opinion that the registration of carbaryl and carbofuran is not likely to result in the destruction or adverse modification of critical habitat of Ozette Lake sockeye salmon, Snake River sockeye salmon, Northern California steelhead, Columbia River chum salmon, Hood Canal summer-run chum salmon, and Oregon Coast coho salmon (Table 82).

Methomyl

After reviewing the current status of Central Valley spring-run Chinook salmon, LCR Chinook salmon, Puget Sound Chinook salmon, Sacramento River winter-run Chinook salmon, UCR spring-run Chinook salmon, Upper Willamette River Chinook salmon, Central California Coast coho salmon, LCR coho salmon, Southern Oregon and Northern Coastal California coho salmon, California Central Valley steelhead, Central California Coast steelhead, LCR steelhead, MCR steelhead, Puget Sound steelhead, South Central California coast steelhead, Southern California steelhead, UCR steelhead, and Upper Willamette River steelhead, the environmental baseline for the action area, the effects of the proposed action, and the cumulative effects, it is NMFS' Opinion that the registration of methomyl is likely to jeopardize the continued existence of these endangered or threatened species (Table 81).

It is NMFS' Opinion that the registration of methomyl is not likely to jeopardize the continued existence of California Coastal Chinook salmon, Snake River fall-run Chinook salmon, Snake River spring/summer-run Chinook salmon, Ozette Lake sockeye salmon, Snake River sockeye salmon, Northern California steelhead, Columbia River chum salmon, Hood Canal summer-run chum salmon, Oregon Coast coho salmon, and Snake River steelhead (Table 81).

After reviewing the current status of designated critical habitat for Central Valley spring-run Chinook salmon, LCR Chinook salmon, Puget Sound Chinook salmon, Sacramento River winter-run Chinook salmon, UCR spring-run Chinook salmon, Upper Willamette River Chinook salmon, Central California Coast coho salmon, LCR coho salmon, Southern Oregon and Northern Coastal California coho salmon, California Central Valley steelhead, Central California Coast steelhead, LCR steelhead, MCR steelhead, Puget Sound steelhead, South Central California coast steelhead, Southern California steelhead, UCR steelhead, and Upper Willamette River steelhead, the environmental baseline for the action area, the effects of the proposed action, and the cumulative effects, it is NMFS' Opinion that the registration of methomyl is likely to result in the destruction or adverse modification of critical habitat of these endangered and threatened species (Table 82).

It is NMFS' Opinion that the registration of methomyl is not likely to result in the destruction or adverse modification of critical habitat of California Coastal Chinook salmon, Snake River fall-run Chinook salmon, Snake River spring/summer-run Chinook salmon, Ozette Lake sockeye salmon, Snake River sockeye salmon, Northern California steelhead, Columbia River chum salmon, Hood Canal summer-run chum salmon, Oregon Coast coho salmon, and Snake River steelhead (Table 82).

Table 81. Jeopardy and Non-Jeopardy Determinations for Listed Species

<table>
<tr><th>Common Name
(Scientific Name)</th><th>Distinct Population Segment or Evolutionarily Significant Unit</th><th>Carbaryl</th><th>Carbofuran</th><th>Methomyl</th></tr>
<tr><td rowspan="9">Chinook salmon
Oncorhynchus tshawytscha</td><td>California Coastal</td><td colspan="2">Jeopardy</td><td>Non-Jeopardy</td></tr>
<tr><td>Central Valley Spring-run</td><td colspan="3" rowspan="5">Jeopardy</td></tr>
<tr><td>Lower Columbia River</td></tr>
<tr><td>Upper Columbia River Spring-run</td></tr>
<tr><td>Puget Sound</td></tr>
<tr><td>Sacramento River Winter-run</td></tr>
<tr><td>Snake River Fall-run</td><td colspan="2" rowspan="2">Jeopardy</td><td rowspan="2">Non-Jeopardy</td></tr>
<tr><td>Snake River Spring/Summer-run</td></tr>
<tr><td>Upper Willamette River</td><td colspan="3">Jeopardy</td></tr>
<tr><td rowspan="2">Chum salmon
Oncorhynchus keta</td><td>Columbia River</td><td colspan="3" rowspan="2">Non-Jeopardy</td></tr>
<tr><td>Hood Canal Summer-run</td></tr>
<tr><td rowspan="4">Coho salmon
Oncorhynchus kisutch</td><td>Lower Columbia River</td><td colspan="3">Jeopardy</td></tr>
<tr><td>Oregon Coast</td><td colspan="3">Non-Jeopardy</td></tr>
<tr><td>Southern Oregon & Northern California Coast</td><td colspan="3" rowspan="2">Jeopardy</td></tr>
<tr><td>Central California Coast</td></tr>
<tr><td rowspan="2">Sockeye salmon
Oncorhynchus nerka</td><td>Ozette Lake</td><td colspan="3" rowspan="2">Non-Jeopardy</td></tr>
<tr><td>Snake River</td></tr>
<tr><td rowspan="11">Steelhead
Oncorhynchus mykiss</td><td>Central California Coast</td><td colspan="3" rowspan="4">Jeopardy</td></tr>
<tr><td>California Central Valley</td></tr>
<tr><td>Lower Columbia River</td></tr>
<tr><td>Middle Columbia River</td></tr>
<tr><td>Northern California</td><td colspan="3">Non-Jeopardy</td></tr>
<tr><td>Puget Sound</td><td colspan="3">Jeopardy</td></tr>
<tr><td>Snake River</td><td colspan="2">Jeopardy</td><td>Non-Jeopardy</td></tr>
<tr><td>South-Central California Coast</td><td colspan="3" rowspan="4">Jeopardy</td></tr>
<tr><td>Southern California</td></tr>
<tr><td>Upper Columbia River</td></tr>
<tr><td>Upper Willamette River</td></tr>
</table>

Table 82. Adverse Modification Determinations for Designated Critical Habitat of Listed Species

<table>
<tr><th>Common Name
(Scientific Name)</th><th>Distinct Population Segment or Evolutionarily Significant Unit</th><th>Carbaryl</th><th>Carbofuran</th><th>Methomyl</th></tr>
<tr><td rowspan="9">Chinook salmon
Oncorhynchus tshawytscha</td><td>California Coastal</td><td colspan="2">Adverse Modification</td><td>No Adverse Modification</td></tr>
<tr><td>Central Valley Spring-run</td><td colspan="3" rowspan="5">Adverse Modification</td></tr>
<tr><td>Lower Columbia River</td></tr>
<tr><td>Upper Columbia River Spring-run</td></tr>
<tr><td>Puget Sound</td></tr>
<tr><td>Sacramento River Winter-run</td></tr>
<tr><td>Snake River Fall-run</td><td colspan="2" rowspan="2">Adverse Modification</td><td rowspan="2">No Adverse Modification</td></tr>
<tr><td>Snake River Spring/Summer-run</td></tr>
<tr><td>Upper Willamette River</td><td colspan="3">Adverse Modification</td></tr>
<tr><td rowspan="2">Chum salmon
Oncorhynchus keta</td><td>Columbia River</td><td colspan="3" rowspan="2">No Adverse Modification</td></tr>
<tr><td>Hood Canal Summer-run</td></tr>
<tr><td rowspan="4">Coho salmon
Oncorhynchus kisutch</td><td>Lower Columbia River</td><td colspan="3">No Designated Critical Habitat</td></tr>
<tr><td>Oregon Coast</td><td colspan="3">No Adverse Modification</td></tr>
<tr><td>Southern Oregon & Northern California Coast</td><td colspan="3" rowspan="2">Adverse Modification</td></tr>
<tr><td>Central California Coast</td></tr>
<tr><td rowspan="2">Sockeye salmon
Oncorhynchus nerka</td><td>Ozette Lake</td><td colspan="3" rowspan="2">No Adverse Modification</td></tr>
<tr><td>Snake River</td></tr>
<tr><td rowspan="11">Steelhead
Oncorhynchus mykiss</td><td>Central California Coast</td><td colspan="3" rowspan="4">Adverse Modification</td></tr>
<tr><td>California Central Valley</td></tr>
<tr><td>Lower Columbia River</td></tr>
<tr><td>Middle Columbia River</td></tr>
<tr><td>Northern California</td><td colspan="3">No Adverse Modification</td></tr>
<tr><td>Puget Sound</td><td colspan="3">No Designated Critical Habitat</td></tr>
<tr><td>Snake River</td><td colspan="2">Adverse Modification</td><td>No Adverse Modification</td></tr>
<tr><td>South-Central California Coast</td><td colspan="3" rowspan="4">Adverse Modification</td></tr>
<tr><td>Southern California</td></tr>
<tr><td>Upper Columbia River</td></tr>
<tr><td>Upper Willamette River</td></tr>
</table>

Reasonable and Prudent Alternatives

This Opinion has concluded that EPA's proposed registration of pesticides containing carbaryl and carbofuran is likely to jeopardize the continued existence of 22 endangered and threatened Pacific salmonids and is likely to destroy or adversely modify designated critical habitat for 20 threatened and endangered salmonids. The Opinion further concluded that EPA's proposed registration of pesticides containing methomyl is likely to jeopardize the continued existence of 18 endangered and threatened Pacific salmonids and is likely to destroy or adversely modify designated critical habitat for 16 threatened and endangered salmonids. "Jeopardize the continued existence of" means "to engage in an action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species" (50 CFR §402.02).

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.

NMFS reached this conclusion because measured and predicted concentrations of the three a.i.s in salmonid habitats, particularly in off-channel habitats[5], are likely to cause adverse effects to listed Pacific salmonids including significant reductions in growth and survival. For carbaryl and carbofuran, 22 ESUs/DPSs of listed Pacific salmonids are likely to suffer reductions in

[5] 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.

Main channel –the stream channel that includes the thalweg (longitudinal continuous deepest portion of the channel.

viability given the severity of expected changes in abundance and productivity associated with the proposed action. Similarly for methomyl, 18 ESUs/DPSs of listed Pacific salmonids are likely to suffer reductions in viability. These adverse effects are expected to appreciably reduce the likelihood of both the survival and recovery of these listed Pacific salmonids. EPA's proposed registration of carbaryl and carbofuran is also likely to result in the destruction or adverse modification of critical habitat for 20 affected ESUs/DPSs because of adverse effects on salmonid prey and water quality in freshwater rearing, spawning, and foraging areas. EPA's proposed registration of methomyl is also likely to result in the destruction or adverse modification of critical habitat for 16 affected ESUs/DPSs because of adverse effects on salmonid prey and water quality in freshwater rearing, spawning, and foraging areas.

The Reasonable and Prudent Alternative (RPA) accounts for the following issues: (1) the action will result in exposure to other chemical stressors in addition to the a.i. that may increase the risk of the action to listed species, including unspecified inert ingredients, adjuvants, and tank mixes; (2) exposure to chemical mixtures containing carbaryl, carbofuran, and methomyl and other cholinesterase-inhibiting compounds result in additive and synergistic responses; and (3) exposure to other chemicals and physical stressors (*e.g.,* pH and temperature) in the baseline habitat will likely intensify response to carbaryl, carbofuran, and methomyl.

The action as implemented under the RPA will remove the likelihood of jeopardy and of destruction or adverse modification of 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. NMFS recognizes that carbofuran's registration is currently in the process of being cancelled. However, NMFS is uncertain when and if cancellation will occur. Furthermore, existing stocks under older labels would remain and are currently allowed to be applied. The RPA and RPMs therefore apply to carbofuran as well. The goal of the RPA is to reduce exposure to ensure that the action is not likely to jeopardize listed species or destroy or adversely modify critical habitat.

The RPA is comprised of six required elements that must be implemented in its entirety within one year of the receipt of EPA's Opinion to ensure the proposed 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 that has been designated for these species. All elements of the RPA apply only to those ESUs/DPSs where there is jeopardy or the destruction or adverse modification of critical habitat. These elements rely upon recognized practices for reducing drift and runoff of pesticide products into aquatic habitats.

Specific Elements of the Reasonable and Prudent Alternative

Elements 1-4 shall be specified on FIFRA labels of all pesticide products containing carbaryl, carbofuran, and methomyl used in California, Idaho, Oregon, and Washington. Alternatively, the label could direct pesticide users to the EPA's Endangered Species Protection Program (ESPP) bulletins that specify elements 1-4. For purposes of this RPA salmonid habitats are defined as freshwaters, estuarine habitats, and nearshore marine habitats including bays within the ESU/DPS ranges including migratory corridors. The freshwater habitats include intermittent streams and other temporally connected habitats to salmonid-bearing waters. Freshwater habitats also include all known types of off-channel habitats as well as drainages, ditches, and other man-made conveyances to salmonid habitats that lack salmonid exclusion devices (*e.g.*, screens).

Element 1.

Do not apply pesticide products[6] within specified buffers of salmonid habitats (Table 83). Buffers only apply to those salmonid habitats where NMFS concluded jeopardy or the destruction or adverse modification of designated critical habitat for listed Pacific salmonids. Buffers also only apply when water exists in the stream or habitat and shall be measured from the water's edge of salmonid habitat, including off-channel habitat, to the point of deposition (below spray nozzle).

Pesticide buffers are recognized tools to reduce pesticide loading into aquatic habitats from drift. EPA, USFWS, NMFS, courts, and state agencies routinely enlist buffers as pesticide load

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[6] Use of the term "pesticide products" in the Reasonable and Prudent Alternative section of the Opinion refers to pesticide products containing carbaryl, carbofuran, and methomyl.

reduction measures. EPA requires the use of buffers on end-use product labels for ground and/or aerial applications for some products that pose risk to aquatic systems. For example, many methomyl containing end-use products have mandated buffers of 25, 100, and 450 ft for ground, aerial, and Ultra Low Volume applications, respectively. CDPR has pesticide use limitations of 120 and 600 ft buffers for carbaryl, carbofuran, and methomyl-containing pesticides when the wind is blowing toward sensitive areas. On June 14, 1989, USFWS issued a Biological Opinion for 165 listed species and 112 pesticide a.i.s. Prescribed buffers under species-specific RPAs ranged from 60 ft (ground applications) to one half mile (aerial applications). Many of EPA's historical county bulletins for endangered species referenced a 60 ft buffer for ground applications and a 300 ft buffer for aerial spraying. One court decision prescribed mandatory 60 ft (ground) and 300 ft (aerial) buffers for applications within the ranges of ESA-listed Pacific salmonids. NMFS has prescribed a range of buffers in ESA consultations for herbicide and insecticide application actions by agencies such as the U.S. Forest Service and Bureau of Land Management overlapping with ESA-listed salmonid habitats. Herbicide buffers ranged from 0 ft to 500 ft depending on application type, rate, and frequency. Insecticide buffers ranged from 0 ft to 200 ft depending on application type, rate, and frequency.

NMFS generated estimated environmental concentrations for the three *N*-methyl carbamates for off-channel habitats using the AgDrift model (set to EPA Tier 1 simulation defaults). NMFS generated values for a range of buffer sizes in 100 ft increments for ground applications (0 – 1,000 ft), and aerial applications (0 – 1,000 ft). The dimensions of the off-channel habitat modeled were 32.8 ft (10 m) wide and 0.328 ft (0.1 m) deep. Key model assumptions include:

- Drift as the only pathway of exposure;
- A single application of each a.i.; and
- The estimates represent instantaneous average concentrations across the entire habitat modeled.

Table 83. Mandatory pesticide no application buffers for ground and aerial applications

Rate	Carbaryl	Carbofuran	Methomyl
lbs/ acre	No Application Buffer (ft)		
Ground Applications			
0 -1	200	200	50
≥1-3	300	300	NA
≥3-5	400	400	NA
≥5-10	500	500	NA
≥10	600	600	NA
Aerial Applications			
All rates	1000	1000	600

NA – Not Applicable. Methomyl has no current registrations permitting maximum applications ≥ 1 lb ai/A

The estimated concentrations decline as buffer size increases **(**Table 84**).** We note the disparity between the concentrations predicted by the two models. For example, at 100 ft. the predicted concentrations are 4.4 µg/L from a ground application and 92.9 µg/L from an aerial application. The two results are not directly comparable because the models use different methods to predict amount of drift.

Table 84. Estimated environmental concentrations of carbaryl, carbofuran, and methomyl applied at the rate of 1 lb per acre for ground and aerial applications.

Ground application, low boom, ASAE very fine-fine droplet distribution, 50th percentile estimates EPA Tier 1 simulations	
Buffer (ft)	Off-Channel (10 m * 0.1 m) (µg/L)
0	76.427
10	20.168
100	4.406
200	2.568
300	1.813
400	1.392
500	1.122
600	0.933
700	0.794
800	0.688
900	0.604
997	0.583
Aerial application, fine-medium droplet distribution. EPA Tier 1 simulation	
Buffer (ft)	Off-Channel (10 m * 0.1 m) (µg/L)
0	333.566
10	260.482
100	92.888
200	48.985
300	33.096
400	25.289
500	20.902
600	18.010
700	16.035
800	14.692
900	13.719
997	12.983

Based on the population growth modeling exercises, a four-day exposure to expected concentrations of carbaryl, carbofuran, and methomyl will substantially reduce a population's growth rate due to prey base reduction. All four life history types modeled demonstrated this effect. As exposure duration and application rate increases, we expect more pronounced effects on salmonid prey abundance and recovery timing leading to further reduced salmonid growth. We expect that some juvenile salmonids are likely to experience reductions in growth. However, the prescribed buffers are intended to avoid population-level effects. We also expect that prey items will die from these exposures. The likelihood of impacting salmonid prey availability, an identified PCE, is substantially reduced by these buffers.

The majority of buffers described earlier are smaller than the buffers prescribed in this element. Concentrations expected with smaller buffers would lead to a greater probability of affecting populations and PCEs especially in habitats that are compromised from a variety of stressors (described in the *Environmental Baseline s*ection).

The scenario we modeled with AgDrift in this RPA element is expected to occur when all of the modeled variables are present *e.g.*, specific wind speed, wind direction, release height, size of off-channel habitat, droplet size distribution, etc. The input variables are relevant to field conditions and the frequency of this exact scenario occurring remains unknown. We selected this scenario to represent off-channel habitats used by a sensitive salmonid life stage *i.e.*, juveniles. NMFS believes that these buffers will remove a substantial portion of risk attributed to pesticide drift.

*Element 2***.** Do not apply when wind speeds are greater than or equal to 10 mph as measured using an anemometer immediately prior to application. Because wind conditions may change during application of pesticide products, commence applications on the side nearest the aquatic habitat and proceed away from the aquatic habitat.

Element 3. For all uses do not apply pesticide products when soil moisture is at field capacity, or when a storm event likely to produce runoff from the treated area is forecasted by NOAA/NWS (National Weather Service), to occur within 48 h following application.

Element 4. Report all incidents of fish mortality that occur within four days of application and within the vicinity of the treatment area to EPA OPP (703-305-7695).

*Element 5***.** In addition to the labeling requirements above, EPA shall develop and implement a NMFS-approved effectiveness monitoring plan for off-channel habitats with annual reports. The plan shall identify representative off-channel habitats within agricultural areas prone to drift and runoff of pesticides. The number and locations of off-channel habitat sampling sites shall include currently used off-channel habitats by threatened and endangered Pacific salmonids identified by NMFS biologists and will include at least two sites for each general species (ESU or DPS; *i.e.,* coho salmon, chum salmon, steelhead, sockeye salmon, and ocean-type Chinook and stream-type Chinook salmon). The plan shall collect daily surface water samples targeting at least three periods during the application season for seven days. Collected water samples will be analyzed for current-use OPs and carbamates following USGS schedule for analytical chemistry. The report shall be submitted to NMFS OPR and will summarize annual monitoring data and provide all raw data.

*Element 6***.** This element is specific to Washington State's 24(c) for carbaryl applications on estuarine mudflats. As this use is specific for application on mudflats within estuaries, elements 1, 3, and 4 are not applicable for this use. Elements 2 and 5 are required. A monitoring program approved by NMFS OPR shall be established in Willapa Bay and Grays Harbor to determine the presence, AChE activity, and genetic source populations of captured juvenile salmonids. For example, a monitoring program shall be implemented to determine salmonid health at five locations using beach seines with the first incoming tide after carbaryl application. Additionally, fyke nets placed in five channels proximate to sprayed mudflats shall be used to determine and health at first outgoing tide after carbaryl application. All dead or dying salmonids, if any, shall be collected and genetically analyzed to determine source population origin. Additionally, muscle and brain AChE activity shall be measured in collected dead or dying salmonids. A subset of the live salmonids captured in seines and nets shall be genetically analyzed to determine source population origin. Annual reports shall be submitted to NMFS and include the raw data and summaries of the raw data including number of fish caught, species of salmonids, genetic analysis results, AChE activity results of brain and muscle samples, and number of dead or dying salmonids collected. The sampling effort shall be conducted annually as long as carbaryl is allowed for use in Willapa Bay and Grays Harbor.

Although NMFS has concluded that EPA's action for carbaryl and carbofuran is likely to jeopardize 22 listed ESUs/DPSs and destroy or adversely modify 20 designated critical habitats during the 15-year duration of the proposed action, NMFS does not believe that the effects of the action will attain this level in the year between issuance of this Opinion and EPA's implementation of the RPA. NMFS has further concluded that EPA's action for methomyl is likely to jeopardize 18 listed ESUs/DPSs and destroy or adversely modify designated critical habitats for 16 listed species during the 15-year duration of the proposed action. As with carbaryl and carbofuran, NMFS does not believe that the effects of the action for methomyl will attain this level in the year between issuance of this Opinion and EPA's implementation of the RPA. Products containing these three a.i.s have been in use for some time. NMFS believes that these products have contributed to ESU/DPS declines, but not to the extent that one year of additional use as now authorized would lead to likely jeopardy or adverse modification.

Based on the above conclusions on the effects of EPA's proposed registration of pesticide products containing carbaryl, carbofuran, and methomyl, the EPA is required to notify NMFS OPR of its final decision on implementation of the RPA.

Incidental Take Statement

Section 9 of the ESA and federal regulation pursuant to section 4(d) of the ESA prohibits the take of endangered and threatened species, respectively, without special exemption. 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. 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. 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 is not considered to be prohibited taking under the ESA provided that such taking is in compliance with the terms and conditions of this Incidental Take Statement.

Amount or Extent of Take Anticipated

As described earlier in this Opinion, this is a consultation on the EPA's registration of pesticide products containing carbaryl, carbofuran, and methomyl, and their formulations as they are used in the Pacific Northwest and California and the impacts of these applications on listed ESUs/DPSs of Pacific salmonids. The EPA authorizes use of these pesticide products for pest control purposes across multiple landscapes. The goal of this Opinion is to evaluate the impacts to NMFS' listed resources from the EPA's broad authorization of applied pesticide products. This Opinion is a partial consultation because pursuant to the court's order, EPA sought consultation on only 26 listed Pacific salmonids under NMFS' jurisdiction. However, even though the court's order did not address the two more recently listed ESUs and DPSs, NMFS analyzed the impacts of EPA's actions to them because they belong to the same taxon and the analysis requires consideration of the same information. Consultation with NMFS will be completed when EPA makes effect determinations on all remaining species under NMFS' jurisdiction and consults with NMFS as necessary.

For this Opinion, NMFS anticipates the general direct and indirect effects that would occur from EPA's registration of pesticide products across the states of California, Idaho, Oregon, and Washington to 28 listed Pacific salmonids under NMFS' jurisdiction during the 15-year duration of the proposed action. Recent and historical surveys indicate that listed salmonids occur in the action area, in places where they will be exposed to the stressors of the action. The RPAs are designed to reduce this exposure but not eliminate it. Pesticide runoff and drift of carbaryl, carbofuran, and methomyl are most likely to reach streams and other aquatic sites when they are applied to crops and other land use settings located adjacent to wetlands, riparian areas, ditches, off-channel habitats, and intermittent streams. These inputs into aquatic habitats are especially high when rainfall immediately follows applications. The effects of pesticides and other contaminants found in urban runoff especially from areas with a high degree of impervious surfaces may also exacerbate degraded water quality conditions of receiving waters used by salmon. Urban runoff is also generally warmer in temperature and elevated water temperature pose negative effects on certain life history phases for salmon, increasing susceptibility to chemical stressors. The range of effects of the three a.i.s on salmonids include reductions in growth, prey capture, and swimming ability, impaired olfaction affecting homing and reproductive behaviors, and increased susceptibility to predation and disease. Thus, we expect some exposed fish will respond to these effects by changing normal behaviors. In some cases, fish may die, be injured, or suffer sublethal effects. These results are not the purpose of the proposed action. Therefore, incidental take of listed salmonids is reasonably certain to occur over the 15-year duration of the proposed action.

Given the variability of real-life conditions, the broad nature and scope of the proposed action, and the migratory nature of salmon, the best scientific and commercial data available are not sufficient to enable NMFS to estimate a specific amount of incidental take associated with the proposed action. As explained in the *Description of the Proposed Action* and the *Effects of the Proposed Action* sections, NMFS identified multiple uncertainties associated with the proposed action. Areas of uncertainty include:

1. Incomplete information on the proposed action (*i.e.,* no master label summarizing all authorized uses of pesticide products containing carbaryl, carbofuran, and methomyl);
2. Limited use and exposure data on stressors of the action for non-agricultural uses of these pesticides;
3. Minimal information on exposure and toxicity for pesticide formulations, adjuvants, and other/inert ingredients within registered formulations;
4. No information on permitted tank mixtures and associated exposure estimates;
5. Limited data on toxicity of environmental mixtures;
6. No known method to predict synergistic responses from exposure to combinations of the three a.i.s;
7. Annual variable conditions regarding land use, crop cover, and pest pressure;
8. Variable temporal and spatial conditions within each ESU, especially at the population-level; and
9. Variable conditions of water bodies in which salmonids live.

NMFS therefore identifies as a surrogate for the allowable extent of take the ability of this action to proceed without any fish kills attributed to the use of carbaryl, carbofuran, or methomyl 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 are not to be listed salmonids. Salmonids appear to be more sensitive to these compounds, so that if there are kills of other freshwater fishes attributed to use of these pesticides, it is likely that salmonids have also died, even if no dead salmonids can be located. In addition, if stream conditions due to pesticide use kill less sensitive fishes in certain areas, the potential for lethal and non-lethal takes in downstream areas increases. A fish kill is considered attributable to one of these three ingredients, its metabolites, or degradates, if measured concentrations in surface waters are at levels expected to kill fish, if AChE measurements were taken of the fish carcass and correlate to fish death, if pesticides were applied in the general area, and if pesticide drift or runoff was witnessed or apparent.

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, and 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 carbaryl, carbofuran, and methomyl; (b) evaluate the direct, indirect, or cumulative impacts of pesticide misapplications in the aquatic habitats in which they occur; and (c) the consequences of those effects on listed Pacific salmonids under NMFS' jurisdiction. The purpose of the monitoring program is for the EPA to use the results of the monitoring data and modify the registration process in order to reduce exposure and minimize the effect of exposure where pesticides will occur in salmonid habitat.

The EPA shall:

1. Minimize the amount and extent of incidental take from use of pesticide products containing carbaryl, carbofuran, and methomyl by reducing the potential of chemicals reaching the water;
2. Monitor any incidental take or surrogate measure of take that occurs from the action; and
3. Report annually to NMFS OPR on the monitoring results from the previous season.

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 measure described above. These terms and conditions are non-discretionary.

1. a. EPA shall include the following instructions requiring reporting of fish kills either on the labels for all products containing carbaryl, carbofuran, and methomyl or in ESPP Bulletins:

   NOTICE: Incidents where salmon appear injured or killed as a result of pesticide applications shall be reported to NMFS OPR at 301-713-1401 and EPA at 703-305-7695. The finder should leave the fish alone, make note of any circumstances likely causing the death or injury, location and number of fish involved, and take photographs, if possible. Adult fish should generally not be disturbed unless circumstances arise where an adult fish is obviously injured or killed by pesticide exposure, or some unnatural cause. The finder may be asked to carry out instructions provided by NMFS OPR to collect specimens or take other measures to ensure that evidence intrinsic to the specimen is preserved.

   b. EPA shall report to NMFS OPR any incidences regarding carbaryl, carborfuran, or methomyl effects on aquatic ecosystems added to its incident database that it has classified as probable or highly probable.

   c. Do not apply pesticide products when wind speeds are greater than or equal to 10 mph as measured using an anemometer immediately prior to application. When applying pesticide products, commence applications on the side nearest the aquatic habitat and proceed away from the aquatic habitat.

2. For all uses do not apply pesticide products when soil moisture is at field capacity, or when a storm event likely to produce runoff from the treated area is forecasted by NOAA/NWS (National Weather Service), to occur within 48 h following application.

Conservation Recommendations

Section 7(a) (1) of the ESA directs federal agencies to use their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of endangered and threatened species. Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of a proposed action on listed species or critical habitat, to help implement recovery plans, or to develop information.

The following conservation recommendations would provide information for future consultations involving future authorizations of pesticide a.i.s that may affect listed species:

1. Conduct mixture toxicity analysis in screening-level and endangered species biological evaluations;
2. Develop models to estimate pesticide concentrations in off-channel habitats; and
3. Develop models to estimate pesticide concentrations in aquatic habitats associated with non-agricultural applications, particularly in residential and industrial environments.

In order for NMFS to be kept informed of actions minimizing or avoiding adverse effects or benefiting listed species or their habitats, the EPA should notify NMFS OPR of any conservation recommendations it implements in the final action.

Reinitiation Notice

This concludes formal consultation on the EPA's proposed registration of pesticide products containing carbaryl, carbofuran, and methomyl 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. If none of these reinitiation triggers are met within the next 15 years, then reinitiation will be required because the Opinion only covers the action for 15 years.

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Appendix 1: Population Modeling

Introduction

To assess the potential for adverse impacts of the anticholinesterase insecticides on Pacific salmon populations, a model was developed that explicitly links impairments in the biochemistry, behavior, prey availability and somatic growth of individual salmon to the productivity of salmon populations. More specifically, the model connects known effects of the pesticides on salmon physiology and behavior with community-level effects on salmon prey to estimate population-level effects on salmon. The model used here is an extension of one developed for investigating the direct effects of pesticides on the biochemistry, behavior, and growth of ocean-type Chinook salmon (Baldwin et al., in press).

In the freshwater portion of their life, Pacific salmon may be exposed to insecticides that act by inhibiting acetylcholinesterase (AChE). AChEe is a crucial enzyme in the proper functioning of cholinergic synapses in the central and peripheral nervous systems of vertebrates and invertebrates. Of consequence to salmon, anticholinesterase insecticides have been shown to interfere with salmon swimming behavior (Beauvais et al. 2000, Brewer et al. 2001, Sandahl et al. 2005), feeding behavior (Sandahl et al. 2005), foraging behavior (Morgan and Kiceniuk 1990), homing behavior (Scholz et al. 2000), anti-predator behaviors (Scholz et al. 2000), and reproductive physiology (Moore and Waring 1996, Waring and Moore 1997, Scholz et al. 2000).

Anticholinesterase insecticides have also been found to reduce benthic densities of aquatic invertebrates and alter the composition of aquatic communities (Liess and Schulz 1999, Schulz and Liess 1999, Schulz et al. 2002, Fleeger et al. 2003, Schulz 2004, Chang et al. 2005, Relyea 2005). Spray drift and runoff from agricultural and urban areas can expose aquatic invertebrates to relatively low concentrations of insecticides for as little as minutes or hours, but populations of many taxa can take months or even years to recover to pre-exposure or reference densities (Wallace et al. 1991, Liess and Schulz 1999, Anderson et al. 2003, Stark et al. 2004). For example, when an aquatic macroinvertebrate community in a German stream was exposed to runoff containing parathion (an AChE inhibitor) and fenvalerate (another commonly used insecticide), eight of eleven abundant species disappeared and the remaining three were reduced in abundance (Liess and Schulz 1999). Long-term changes in invertebrate densities and community composition likely result in reductions in salmon prey availability. Therefore, in addition to the direct impacts that AChE inhibitors have on salmon, there may also be, independently, significant indirect effects to salmon via their prey (Peterson et al. 2001a). Wild juvenile salmon feed primarily on invertebrates in the water column and those trapped on the water's surface, actively selecting the largest items available (Healey 1991, Quinn 2005). Salmon are often found to be food limited (Quinn 2005), suggesting that a reduction in prey number or size due to insecticide exposure may further stress salmon. For example, Davies and Cook (1993) found that several months following a spray drift event, benthic and drift densities were still reduced in exposed stream reaches. Consequently, brown trout in the exposed reaches fed less and grew at a slower rate compared to those in unexposed stream reaches (Davies and Cook 1993). Although the insecticide in their study was cypermethrin (a pyrethroid), similar reductions in macroinvertebrate density and recovery times have been found in studies with AChE inhibitors (Liess and Schulz 1999, Schulz et al. 2002), suggesting indirect effects to salmon via prey availability may be similar.

One likely biological consequence of reduced swimming, feeding, foraging, and prey availability is a reduction in food uptake and, subsequently, a reduction in somatic growth of exposed fish. Juvenile growth is a critical determinant of freshwater and marine survival for Chinook salmon (Higgs et al. 1995). Reductions in the somatic growth rate of salmon fry and smolts are believed to result in increased size-dependent mortality (Healey 1982, West and Larkin 1987, Zabel and Achord 2004). Zabel and Achord (2004) observed size-dependent survival for juvenile salmon during the freshwater phase of their outmigration. Mortality is also higher among smaller and slower growing salmon because they are more susceptible to predation during their first winter (Healey 1982, Holtby et al. 1990, Beamish and Mahnken 2001). These studies suggest that factors affecting the organism and reducing somatic growth, such as anticholinesterase insecticide exposure, could result in decreased first-year survival and, thus, reduce population productivity.

Changes to the size of juvenile salmon from exposure to carbaryl, carbofuran, and methomyl were linked to salmon population demographics. We used size-dependent survival of juveniles during a period of their first year of life. We did this by constructing and analyzing general life history matrix models for coho salmon (*Oncorhynchus kisutch*), sockeye salmon (*O. nerka*), and ocean-type and stream-type Chinook salmon (*O. tshawytscha*). A steelhead (*O. mykiss*) life history model was not constructed due to the lack of demographic information relating to the proportions of resident and anadromous individuals, the freshwater residence time of steelhead, and rates of repeated spawning. The basic salmonid life history modeled consisted of hatching and rearing in freshwater, smoltification in estuaries, migration to the ocean, maturation at sea, and returning to the natal freshwater stream for spawning followed shortly by death. Differences between the modeled strategies are lifespan of the female, time to reproductive maturity, and the number and relative contribution of the reproductive age classes (Figure 1). The coho females we modeled reach reproductive maturity at age 3 and provide all of the reproductive contribution. Sockeye females reach maturity at age 4 or 5, but the majority of reproductive contributions are provided by age 4 females. Chinook salmon females can mature at age 3, 4, or 5, with the majority of the reproductive contribution from ages 4 and 5. The primary difference between the ocean-type and stream-type Chinook salmon is the juvenile freshwater residence with ocean-type juveniles migrating to the ocean as sub-yearlings and stream-type overwintering in freshwater and migrating to the ocean as yearlings. The models depicted general populations representing each life history strategy and were constructed based upon literature data described below. Specific populations were not modeled due to the difficulty in finding sufficient demographic and reproductive data for a single population.

A separate acute toxicity model was constructed that estimated the population-level impacts of juvenile mortality resulting from exposure to lethal concentrations of carbaryl, carbofuran, and methomyl. These models excluded sublethal and indirect effects of the pesticide exposures and focused on the population-level outcomes resulting from an annual exposure of juveniles to a pesticide. The lethal impact was implemented as a change in first year survival for each of the salmon life history strategies.

The overall model endpoint used to assess population-level impacts for both the growth and acute lethality models was the percent change in the intrinsic population growth rate (lambda, λ) resulting from the pesticide exposure. Change in λ is an accepted population parameter often used in evaluating population productivity, status, and viability. The National Marine Fisheries Service uses changes in λ when estimating the status of species, conducting risk and viability assessments, developing endangered species recovery plans, composing biological opinions, and communicating with other federal, state and local agencies (McClure et al. 2003). While values of $\lambda<1.0$ indicate a declining population, negative changes in lambda greater than the natural variability for the population indicate a loss of productivity. This can be a cause for concern since the decline could make a population more susceptible to dropping below 1.0 due to impacts from multiple stressors.

The following models were developed to serve as a means to assess the potential effects on ESA-listed salmon populations from exposure to AChE inhibiting pesticides, including *N*-methyl carbamates and organophosphorus insecticides. The growth model focuses on the impacts to prey abundance and a salmon's ability to feed which are integrated into reductions in juvenile growth. Assessing the results from different pesticide exposure scenarios relative to a control (*i.e.*, unexposed) scenario can indicate the potential for sublethal pesticide exposures to lead to changes in the somatic growth and survival of individual sub-yearling salmon. Consequently, subsequent changes in salmon population dynamics as indicated by percent change in a population's intrinsic rate of increase assists us in forecasting the potential population-level impacts to listed populations. Also, the model helps us understand the potential influence of life history strategies that might explain differential results within the species modeled.

Methods

The model consists of two parts, an organismal portion and a population portion. The organismal portion of the model links AChE inhibition and reduced prey abundance from insecticide exposure to potential reductions in the growth of individual fish. The population portion of the model links the sizes of individual sub-yearling salmon to their survival and the subsequent growth of the population. Models were constructed using MATLAB 7.7.0 (R2008b) (The MathWorks, Inc. Natick, MA).

Organismal Model

For the organismal model a relationship between AChE activity and somatic growth of salmonid fingerlings was developed using a series of relationships between pesticide exposure, AChE activity, feeding behavior, food uptake, and somatic growth rate (Figures 2-4). The model incorporates empirical data when available. Since growth and toxicity data are limited, extrapolation from one salmon species to the others was done with the assumption that the salmon stocks would exhibit similar physiological and toxicological responses. Sigmoidal dose-response relationships based upon the AChE inhibition EC50 values and their slopes are used to determine the level of AChE activity (Figure 2A, 2B, 2C) from the exposure concentration of each pesticide exposure or pulse.

A linear relationship based on empirical data related AChE activity to feeding behavior (Sandahl et al. 2005, Figure 2D). Feeding behavior was then assumed to be directly proportional to food uptake, defined as potential ration (Figure 2E). The potential ration expresses the amount of food the organism can consume when prey abundance is not limiting. Potential ration over time (Figure 2F) depicts how the food intake of individual fish changes in response to the behavioral effects of the pesticide exposure over the modeled growth period. Potential ration is equal to final ration if no effects on prey abundance are incorporated (Figure 4). If effects of pesticide exposure on prey abundance are incorporated, final ration is the product of potential ration (relating to the fish's ability to capture prey, Figure 2) and the relative abundance of prey available following exposure (Figure 3). Next, additional empirical data (*e.g.*, Weatherley and Gill 1995) defined the relationship between final ration and somatic growth rate (Figure 4C). While the empirical relationship is more complex (*e.g.*, somatic growth rate plateaus at rations above maximum feeding), a linear model was considered sufficient for the overall purpose of this model. Finally, the model combines these linear models relating AChE activity to feeding behavior, feeding behavior to potential ration, and final ration to somatic growth rate to produce a linear relationship between AChE activity and somatic growth rate (Figure 4D). One important assumption of the model is that the relationships are stable, *i.e.*, do not change with time. The relationships would need to be modified to incorporate time as a variable if, for example, fish are shown to compensate over time for reduced AChE activity to improve their feeding behavior and increase food uptake.

Juvenile salmonids are largely opportunistic, feeding on a diverse community of aquatic and terrestrial invertebrate taxa that are entrained in the water column or on the surface (Higgs et al. 1995). As a group, these invertebrates are among the more sensitive taxa for which there is toxicity data, but within this group, there is a wide range of sensitivities. To determine a single effect concentration to use in the model analyses, a search was completed using the EPA's Ecotox database for each pesticide (http://cfpub.epa.gov/ecotox/). Several criteria were used to determine which reported effect concentrations were included in the final analysis. The data included were from studies on taxa that are known to be salmonid prey (or are functionally similar to salmonid prey); these include a diverse group of fresh and saltwater crustaceans, aquatic insects and worms. Studies with exposures of at least 24 hours (h) and not more than 96 h were included. Studies examining shorter and longer exposure times are known to affect invertebrates (*e.g.*, Peterson et al. 2001b), but these were excluded so that estimated EC50s would be comparable to previous analyses (NMFS 2008). Studies reporting invertebrate LC50s and EC50s in which mortality or immobilization of invertebrates was the recorded endpoint were included; the term "EC50" will be used in this report to describe all of these included data. Data derived for sublethal endpoints (*e.g.*, growth or reproduction) were not included. If specific data were represented more than once in the Ecotox output, duplicates were eliminated. Data from several recent peer-reviewed studies that are not yet included in the Ecotox data base, but report effect concentrations that caused mortality, were also included.

From the distributions of those data, a single effect concentration and slope were derived to best represent the diverse community of prey available in juvenile salmonid freshwater and estuarine habitats. The distributions of individual invertebrate EC50s and the geometric means of EC50s by taxa were analyzed to estimate the 50^{th}, 10^{th}, and 5^{th} percentiles. Figure 43shows the distribution of geometric means of EC50s by taxa for the 10^{th} percentile and Table 79shows the concentrations for the percentiles. Specifically, for each pesticide, a probability plot was used to graph the EC50 concentrations normalized to a normal probability distribution. For each plot, the X axis is scaled in probability (between zero and 100%) and shows the percentage of entire data whose value is less than the data point. The Y axis displays the range of the data on a log scale. The results of a linear regression of the log-transformed concentrations are shown and highlight the lognormal distribution of the data (Figure 43). In the regression equation, the normsinv() function returns the inverse of the standard normal cumulative distribution. The standard normal distribution has a mean of zero and a standard deviation of one. For example, given a percentile value of 50 (*i.e.*, a probability of 0.5), normsinv(50) returns a value of zero. The plots and regressions were performed using KaleidaGraph 4.03 (Synergy Software).

The decision to use the 10th percentile rather than the 50th percentile is consistent with previous designations by EPA, and is reasonable because of the relative sensitivity of invertebrates that are most likely consumed by juveniles. In addition, the 10th percentile is a reasonable threshold because the data included in the meta-analysis were limited to concentrations that caused mortality or immobilization within a short period of time (1-4 days). A growing number of studies on a variety of insecticides have reported that concentrations well below LC50s can cause delayed mortality or sublethal effects that may scale up to affect populations, especially in scenarios with multiple exposures and/or other stressors. Evidence for ecologically significant sublethal or delayed effects includes reduced growth rates (Schulz and Liess 2001b, Forbes and Cold 2005), altered behavior (Johnson et al. 2008), reduced emergence (Schulz and Liess 2001a), reduced reproduction (Cold and Forbes 2004, Sakamoto et al. 2006), and reduced predator defenses (Sakamoto et al. 2006, Johnson et al. 2008). Finally, the available toxicity data – and therefore the data included for these analyzes – are from studies using taxa hearty enough to survive laboratory conditions. Studies specifically examining salmonid prey that are more difficult to rear in the laboratory have documented relatively low LC50 or EC50 values when exposed to current-use pesticides (*e.g.*, Peterson et al. 2001a, Johnson et al. 2008). It is noteworthy that the most relevant invertebrate study for carbaryl – a study using a diverse group of seven salmonid prey taxa collected from streams in the Pacific Northwest – reports LC50s for carbaryl with a geometric mean of 26.28 μg/L (range 11.1 – 61.0) (Peterson et al. 2001a).

The models allow exposures that can include multiple AChE-inhibiting pesticides over various time pulses. Sigmoidal dose-response relationships, at steady-state, between each single pesticide exposure and 1) AChE activity and 2) relative prey abundance are modeled using specific EC50s and EC50s and slopes (Figure 2B and 3B). The timecourse for each exposure was built into the model as a pulse with a defined start and end during which the exposure remained constant (Figure 2A and 3A). The timecourse for AChE activity, on the other hand, was modeled using two single-order exponential functions, one for the time required for the exposure to reach full effect and the other for time required for complete recovery following the end of the exposure (time-to-effect$_{\text{AChE activity}}$ and time-to-recovery$_{\text{AChE activity}}$, respectively; Figure 2C). The apparent activity level was back-calculated to result in a relative concentration (concentration/ AChE inhibition EC50) for each day of the growth period for each pulse. The relative concentration for each day was summed across all the pulses to result in a total apparent concentration for each day. The sigmoid slope used in the calculation of AChE activity using the apparent concentration was the arithmetic mean of the sigmoid slopes for each pesticide present on each day. The timecourse for relative prey abundance was modeled incorporating a one day spike in prey drift relative to the toxicity and available prey base followed by a drop in abundance due to the toxic impacts (Figure 3C). Recovery is assumed to be due to a constant influx of invertebrates from connected habitats (aquatic and terrestrial) that are not exposed to the pesticide. Incoming organisms are subject to toxicity if pesticides are still present and this alters the rate of recovery during exposures. Incorporating dynamic effects and recovery variables allows the model to simulate differences in the pharmacokinetics (*e.g.*, the rates of uptake from the environment and of detoxification) of various pesticides and simulate differences in invertebrate community response and recovery rates (see below).

The relationship between final ration and somatic growth rate (Figure 4C) produces a relationship representing somatic growth rate over time (Figure 4D), which is then used to model individual growth rate and size over time. The growth models were run for 1,000 individual fish, with initial weight selected from a normal distribution with a mean of 1.0 g and standard deviation of 0.1 g. The size of 1.0 g was chosen to represent sub-yearling size in the spring prior to the onset of pesticide application. For each iteration of the model (one day for the organismal model), the somatic growth rate is calculated for each fish by selecting the parameter values from normal distributions with specified means and standard deviations (Table 1). The weight for each fish is then adjusted based on the calculated growth rate to generate a new weight for the next iteration. The length (days) to run the growth portion of the model was selected to represent the time from when the fish enter the linear portion of their growth trajectory in the mid to late spring until they change their growth pattern in the fall due to reductions in temperature and resources or until they migrate out of the system. The outputs of the organismal model that are handed to the population models consist of mean weights (with standard deviations) after the species-appropriate growth period (Table 2). A sensitivity analysis was run to determine the influence of the parameter values on the output of the growth model.

The parameter values defining control conditions that are constant for all the modeled species are listed in Table 1. Model parameters such as the length of the growth period and control daily growth rate that are species specific are listed in Table 2. Each exposure scenario was defined by a concentration and exposure time for each pesticide. The duration of time until full effect for the pesticides was assumed to be within a few days (Ferrari et al. 2004), with a half-life of 0.5 days.

For prey, it is assumed there is a constant, independent influx of prey from upstream habitats that will eventually (depending on the rate selected) return prey abundance to 1. As mentioned above, however, these invertebrates are subject to exposure once added to the system, and therefore prey recovery rate is a product of the influx rate as well as the exposure scenario. While recovery rates reported in the literature vary, it is assumed a 1% recovery rate is ecologically realistic (Ward et al. 1995, Van den Brink et al. 1996, Colville et al. 2008). It was also assumed that regardless of the exposure scenario, relative prey abundance would not drop below a specific floor (Figure 3B). This assumption depends on a minimal yet constant terrestrial subsidy of prey and/or an aquatic community with tolerant individuals that would be available as prey, regardless of pesticide exposure and in addition to the constant recovery rate. No studies specify floors per se, but studies quantifying invertebrate densities following highly toxic exposures indicate a floor of 0.2 is ecologically realistic (*i.e.*, regardless of the exposure, 20% of a fish's ration will be available daily; *e.g.*, Cuffney et al. 1984). Finally, because prey availability has been found to increase dramatically albeit briefly following pesticide exposures (due to immediate mortality and/or emigration of benthic prey into the water column; Davies and Cook 1993, Schulz 2004), a one-day prey spike is included for the day following an exposure. The relative magnitude of the spike is calculated as the product of the standing prey availability the day prior to exposure (minus the floor), the toxicity of the exposure, and a constant of 20. This calculation therefore accounts for the potential prey that are available and the severity of the exposure. The spike will be greater when more prey are available and/or the toxicity of the exposure is greater; alternatively, the spike will be small when few prey are available and/or the exposure toxicity is low.

Below are the mathematical equations used to derive Figures 2, 3, and 4.

Figures 2A and 3A use a step function:

time < start; exposure = 0

start ≤ time ≤ end; exposure = exposure concentration(s)

time > end; exposure = 0.

Figures 2B and 3B use a sigmoid function:

y = bottom + (top – bottom)/(1 + (exposure concentration/EC50)^slope).

For 2B, y = AChE activity, top = Ac, bottom = 0.

For Figure 3B, y = prey abundance, top = Pc (in this case 1), bottom = Pf.

Figures 2D, 2E, and 4C use a linear function (the point-slope form of a line):

y = m*(x – x1) + y1.

For 2D, m = Mfa, x1 = Ac, and y1 = Fc.

For 2E, m = Mrf (computed as Rc/Fc), x1 = Fc, and y1 = Rc.

For 4C, m = Mgr, x1 = Rc, and y1 = Gc.

Figure 2C uses a series of exponential functions:

time < start; y = c

start ≤ time ≤ end; y = c – (c – i)*(1 – exp(-ke*(time – start)))

time > end; ye = c – (c – i)*(1 – exp(-ke*(end – start)))

y = ye + (c – ye)*(1 – exp(-kr*(time – end))).

For Figure 2C, c = Ac, i = Ai, ke = ln(2)/AChE effect half-life, kr = ln(2)/AChE recovery half-life. For Figure 2C the value of ye is calculated to determine the amount of inhibition that is reached during the exposure time, which may not be long enough to reach the maximum level of inhibition.

For Figure 3C, an exposure pulse would result in a 1-day spike followed by a decline to the impacted level based upon the prey toxicity. During exposures resulting in low prey toxicity, toxicity-limited recovery can occur. After exposure ends a constant rate of recovery proceeds until control drift is reached or another exposure occurs

preyavail=preydrift(day-1)-floor;

preytox=1/(1+(concentration)^preyslope);

preyrecrate=0.01;

preydriftrec = preyrecrate*preytox.

time=start; spike=(-1+10^(1.654*preyavail))*(1-preytox)

preydrift =preydrift+spike

start ≤ time ≤ end; preydrift=(preyavail*preytox)+preyrdriftrec+floor;

time>end; preydrift = preydrift(day-1)+preydriftrec

Figure 2F is generated by using the output of Figure 2C for a given time as the input for 2D and using the resulting output of 2D as the input for 2E. The resulting output of 2E produces a single time point in the relationship in 2F. Performing this series of computations across multiple days produces the entire relationship in 2F. 4D is generated by taking the outputs of 4A and 4B for the same day. Note the relationship of 4A is equivalent to 2F. The resulting outputs of 4A and 4B are multiplied to produce a final ration for a given day. The prey abundance (4B) available for consumption during a prey spike is capped at a maximum of 1.5*control drift to provide a limited benefit to the individual fish. The final ration is used as input for 4C to generate 4D.

Population Model

The weight distributions from the organismal growth portion of the model are used to calculate size-dependent first-year survival for a life history matrix population model for each species and life history type. This incorporates the impact that reductions in size could have on population growth rate and abundance. The first-year survival element of the transition matrix incorporates a size-dependent survival rate for a three- or four-month interval (depending upon the species) which takes the juveniles up to 12 months of age. This time represents the 4-month early winter survival in freshwater for stream-type Chinook salmon, coho, and sockeye models. For ocean-type Chinook salmon, it is the 3-month period the sub-yearling smolt spend in the estuary and nearshore habitats (*i.e.*, estuary survival). The weight distributions from the organismal model are converted to length distributions by applying condition factors from data for each modeled species (cf; 0.0095 for sockeye and 0.0115 for all others) as shown in Equation L.

Equation L: length(mm) = ((fish weight(g)/cf)^(1/3))*10

The relationship between length and early winter or estuary survival rate was adapted from Zabel and Achord (2004) to match the survival rate for each control model population (Howell et al. 1985, Kostow 1995, Myers et al. 2006). The relationship is based on the length of a sub-yearling salmon relative to the mean length of other competing sub-yearling salmon of the same species in the system, Equation D, and relates that relative difference to size-dependent survival based upon Equation S. The values for α and resulting size-dependent survival (survival ϕ) for control runs for each species are listed in Table 2. The constant α is a species-specific parameter defined such that it produces the correct control survival ϕ value when Δlength equals zero.

Equation D: Δlength = fish length(mm) – mean length(mm)

Equation S: Survival $\phi = (e^{(\alpha+(0.0329*\Delta \text{length}))}) / (1 + e^{(\alpha+(0.0329*\Delta \text{length}))})$

Randomly selecting length values from the normal distribution calculated from the organismal model output size and applying equations 1 and 2 generates a size-dependent survival probability for each fish. This process was replicated 1,000 times for each exposure scenario and simultaneously 1,000 times for the paired control scenario and results in a mean size-dependent survival rate for each population. The resulting size-dependent survival rates are inserted in the calculation of first-year survival in the respective control and pesticide-exposed transition matrices.

The investigation of population-level responses to pesticide exposures uses life history projection matrix models. Individuals within a population exhibit various growth, reproduction, and survivorship rates depending on their developmental or life history stage or age. These age specific characteristics are depicted in the life history graph (Figure 1A-D) in which transitions are depicted as arrows. The nonzero matrix elements represent transitions corresponding to reproductive contribution or survival, located in the top row and the subdiagonal of the matrix, respectively (Figure 1E). The survival transitions in the life history graph are incorporated into the n x n square matrix (A) by assigning each age a number (1 through n) and each transition from age i to age j becomes the element a_{ij} of matrix A (i = row, j = column) and represent the proportion of the individuals in each age passing to the next age as a result of survival. The reproductive element (a_{1j}) gives the number of offspring that hatch per individual in the contributing age, j. The reproductive element value incorporates the proportion of females in each age, the proportion of females in the age that are sexually mature, fecundity, fertilization success, and hatch success.

In order to understand the relative impacts of a short-term exposure of a single pesticide on exposed vs. unexposed fish, we used parameters for an idealized control population that exhibits an increasing population growth rate. All characteristics exhibit density independent dynamics. The models assume closed systems, allowing no migration impact on population size. No stochastic impacts are included beyond natural variability as represented by selecting parameter values from a normal distribution about a mean each model iteration (year). Ocean conditions, freshwater habitat, fishing pressure, and marine resource availability were assumed constant and density independent.

In the model an individual fish experiences an exposure once as a sub-yearling (during its first spring) and never again. The pesticide exposure is assumed to occur annually. All sub-yearlings within a given population are assumed to be exposed to the pesticide. No other age classes experience the exposure. The model integrates this as every brood class being exposed as sub-yearlings and thus the vital demographic rates of the transition matrix are continually impacted in the same manner. Regardless of the level of AChE inhibition due to the direct exposure, only the sublethal effects are incorporated in the models at this time.

The model recalculates first-year survival for each run using a size-dependent survival value selected from a normal distribution with the mean and standard deviation produced by Equation S. Population model output consists of the percent change in lambda from the unexposed control populations derived from the mean of two thousand calculations of both the unexposed control population and the pesticide exposed population. Change in lambda, representing alterations to the population productivity, was selected as the primary model output for reasons outlined previously.

A prospective analysis of the transition matrix, A, (Caswell 2001) explored the intrinsic population growth rate as a function of the vital rates. The intrinsic population growth rate, λ, equals the dominant eigenvalue of A and was calculated using matrix analysis software (MATLAB version 7.7.0 by The Math Works Inc., Natick, MA). Therefore, λ is calculated directly from the matrix and running projections of abundances over time is redundant and unnecessary. The stable age distribution, the proportional distribution of individuals among the ages when the population is at equilibrium, is calculated as the right normalized eigenvector corresponding to the dominant eigenvalue λ. Variability was integrated by repeating the calculation of λ 2,000 times selecting the values in the transition matrix from their normal distribution defined by the mean standard deviation. The influence of each matrix element, a_{ij}, on λ was assessed by calculating the sensitivity values for A. The sensitivity of matrix element a_{ij} equals the rate of change in λ with respect to a_{ij}, defined by $\delta\lambda / \delta a_{ij}$. Higher sensitivity values indicate greater influence on λ. The elasticity of matrix element a_{ij} is defined as the proportional change in λ relative to the proportional change in a_{ij}, and equals (a_{ij}/λ) times the sensitivity of a_{ij}. One characteristic of elasticity analysis is that the elasticity values for a transition matrix sum to unity (one). The unity characteristic also allows comparison of the influence of transition elements and comparison across matrices.

Due to differences in the life history strategies, specifically lifespan, age at reproduction and first year residence and migration habits, four life history models were constructed. This was done to encompass the different responses to freshwater pesticide exposures and assess potentially different population-level responses. Separate models were constructed for coho, sockeye, ocean-type and stream-type Chinook salmon. In all cases transition values were determined from literature data on survival and reproductive characteristics of each species.

A life history model was constructed for coho salmon (*O. kisutch*) with a maximum age of 3. Spawning occurs in late fall and early winter with emergence from March to May. Fry spend 14-18 months in freshwater, smolt and spend 16-20 months in the saltwater before returning to spawn (Pess et al. 2002). Survival numbers were summarized in Knudsen et al. (2002) as follows. The average fecundity of each female is 4,500 with a standard deviation of 500. The observed number of males:females was 1:1. Survival from spawning to emergence is 0.3 (0.07). Survival from emergence to smolt is 0.0296 (0.00029) and marine survival is 0.05 (0.01). All parameters followed a normal distribution (Knudson et al. 2002). The calculated values used in the matrix are listed in Table 4. The growth period for first year coho was set at 180 days to represent the time from mid-spring to mid-fall when the temperatures and resources drop and somatic growth slows (Knudson et al. 2002).

Life history models for sockeye salmon (*O. nerka*) were based upon the lake wintering populations of Lake Washington, Washington, USA. These female sockeye salmon spend one winter in freshwater, then migrate to the ocean to spend three to four winters before returning to spawn at ages 4 or 5. Jacks return at age 2 after only one winter in the ocean. The age proportion of returning adults is 0.03, 0.82, and 0.15 for ages 3, 4 and 5, respectively (Gustafson et al.1997). All age 3 returning adults are males. Hatch rate and first-year survival were calculated from brood year data on escapement, resulting presmolts and returning adults (Pauley et al. 1989), and fecundity (McGurk 2000). Fecundity values for age 4 females were 3,374 (473) and for age 5 females were 4,058 (557) (McGurk 2000). First-year survival rates were 0.737/month (Gustafson et al. 1997). Ocean survival rates were calculated based upon brood data and the findings that 90% of ocean mortality occurs during the first 4 months of ocean residence (Pauley et al. 1989). Matrix values used in the sockeye baseline model are listed in Table 4. The 168 day growth period represents the time from lake entry to early fall when the temperature drops and somatic growth slows (Gustafson et al. 1997).

A life history model was constructed for ocean-type Chinook salmon (*O. tshawytscha*) with a maximum female age of 5 and reproductive maturity at ages 3, 4, or 5. Ocean-type Chinook salmon migrate from their natal stream within a couple months of hatching and spend several months rearing in estuary and nearshore habitats before continuing on to the open ocean. Transition values were determined from literature data on survival and reproductive characteristics from several ocean-type Chinook salmon populations in the Columbia River system (Healey and Heard 1984, Howell et al. 1985, Roni and Quinn 1995, Ratner et al. 1997, PSCCTC 2002, Green and Beechie 2004). The sex ratio of spawners was approximately 1:1. Estimated size-based fecundity of 4,511(65), 5,184(89), and 5,812(102) was calculated based on data from Howell et al., 1985, using length-fecundity relationships from Healy and Heard (1984). Control matrix values for the Chinook model are listed in Table 4. The growth period of 140 days encompasses the time the fish rear in freshwater prior to entering the estuary and open ocean. The first three months of estuary/ocean survival are the size-dependent stage. Size data for determining sub-yearling Chinook salmon condition indices came from data collected in the lower Columbia River and estuary (Johnson et al. 2007).

An age-structured life history matrix model for stream-type Chinook salmon with a maximum age of 5 was defined based upon literature data on Yakima River spring Chinook from Knudsen et al. (2006) and Fast et al. (1988), with sex ratios of 0.035, 0.62 and 0.62 for females spawning at ages 3, 4, and 5, respectively. Length data from Fast et al. (1988) was used to calculate fecundity from the length-fecundity relationships in Healy and Heard (1984). The 184-day growth period produces control fish with a mean size of 96 mm, within the observed range documented in the fall prior to the first winter (Beckman et al. 2000). The size-dependent survival encompasses the 4 early winter months, up until the fish are 12 months old.

Acute Toxicity Model

In order to estimate the population-level responses of exposure to lethal pesticide concentrations, acute mortality models were constructed based upon the control life history matrices described above. The acute responses are modeled as direct reduction in the first-year survival rate (S1). All sub-yearling salmon are assumed to be exposed in each scenario. Exposures are assumed to result in a cumulative reduction in survival as defined by the concentration and the dose-response curve as defined by the LC50 and slope for each pesticide. A sigmoid dose-response relationship is used to accurately handle responses well away from LC50 and to be consistent with other does-response relationships. The model inputs for each scenario are the exposure concentration and acute fish LC50, as well as the sigmoid slope for the LC50. For a given concentration a pesticide survival rate (1-mortality) is calculated and is multiplied by the control first-year survival rate, producing an exposed scenario first-year survival for the life history matrix. Variability is incorporated as described above using mean and standard deviation of normally distributed survival and reproductive rates and model output consists of the percent change in lambda from unexposed control populations derived from the mean of 1,000 calculations of both the unexposed control population and the pesticide exposed population. The percent change in lambda is considered different from control when the difference is greater than the percent of one standard deviation from the control lambda.

Results

Sensitivity Analysis

A sensitivity analysis conducted on the organismal model revealed that changes in the control somatic growth rate had the greatest influence on the final weights (Table 1). While this parameter value was experimentally derived for another species (sockeye salmon; Brett et al. 1969), this value was adapted for each model species and is within the variability reported in the literature for other salmonids (reviewed in Weatherley and Gill 1995). Other parameters related to the daily growth rate calculation, including the growth to ration slope (Mgr) and the control ration produced strong sensitivity values. Initial weight, the prey recovery rate and the prey floor also strongly influenced the final weight values (Table 1). Large changes (0.5 to 2X) in the other key parameters produced proportionate changes in final weight.

The sensitivity analysis of all four of the control population matrices predicted the greatest changes in population growth rate (λ) result from changes in first-year survival. Parameter values and their corresponding sensitivity values are listed in Table 4. The elasticity values for the transition matrices also corresponded to the driving influence of first-year survival, with contributions to lambda of 0.33 for coho, 0.29 for ocean-type Chinook salmon, 0.25 for stream-type Chinook salmon, and 0.24 for sockeye.

Model Output

Organismal and population model outputs for all scenarios are shown in Tables 74-78 and were summarized in Figures 44-47 in the main text of the Opinion. As expected, greater changes in population growth resulted from longer exposures to the pesticides. The factors driving the level of change in lambda were the Prey Drift and relative AChE Activity parameters determined by the toxicity values for each pesticide (Table 3). The low Prey Abundance EC_{50} values drive the effects for all three compounds at most concentrations investigated since they have higher AChE EC50 values (Table 3). Output from the acute toxicity models was presented in the Risk Characterization section of the main text. Increases in direct mortality during the first year of life produced large impacts on the population growth rates for all the life history strategies.

While strong trends in effects were seen for each pesticide across all four life history strategies modeled, some slight differences were apparent. The similarity in patterns likely stems from using the same toxicity values for all four models, while the differences are consequences of distinctions between the life history matrices. The stream-type Chinook salmon and sockeye models produced very similar results as measured as the percent change in population growth rate. The ocean-type Chinook salmon and coho models output produced the greatest changes in lambda resulting from the pesticide exposures. When looking for similarities in parameters to explain the ranking, no single life history parameter or characteristic, such as lifespan, reproductive ages, age distribution, lambda and standard deviation, or first-year survival show a pattern that matches this consistent output. Combining these factors into the transition matrix for each life history and conducting the sensitivity and elasticity analyses revealed that changes in first-year survival produced the greatest changes in lambda. In addition, the elasticity analysis can be used to predict relative contribution to lambda from changes in first-year survival on a per unit basis. As detailed by the elasticity values reported above, the same change in first-year survival will produce a slightly greater change in the population growth rate for coho and ocean-type Chinook salmon than for stream-type Chinook salmon and sockeye. While some life history characteristics may lead a population to be more vulnerable to an impact, the culmination of age structure, survival and reproductive rates as a whole strongly influences the population-level response.

Figure 1: Life-History Graphs and Transition Matrix for coho (A), sockeye (B) and Chinook salmon (C). The life history graph for a population labeled by age, with each transition element labeled according to the matrix position, a_{ij}, i row and j column. Dashed lines represent reproductive contribution and solid lines represent survival transitions. D) The transition matrix for the life history graph depicted in C.

Figure 2: Relationships used to link anticholinesterase exposure to the organism's ability to acquire food (potential ration). See text for details. Relationships in B, C, and D use empirical data. Closed circles represent control conditions. Open circles represent the exposed (inhibited) condition. A) Representation of a constant level of anticholinesterase pesticide exposure (either a single compound or mixtures). B) Sigmoidal relationship between exposure concentration and steady-state AChE activity showing a dose-dependent reduction defined by control activity (horizontal line, Ac), sigmoidal (*i.e.*, hille) slope (AChE slope), and the concentration producing 50% inhibition (vertical line, EC50). C) Timecourse of AChE inhibition based on modeling the time-to-effect and time-to-recovery as single exponential curves with different time-constants. At the start of the exposure AChE activity will be at control and then decline toward the inhibited activity (Ai) based on Panel B. D) Linear model relating AChE activity to feeding behavior using a line that passes through the feeding (Fc) and activity (Ac) control conditions with a slope of Mfa. E) The relationship between feeding behavior and the potential ratio an organism could acquire (if not food limited) used a line passing through the control conditions (Fc as in Panel D and the control ration, Rc) and through the origin producing a slope (Mrf) equal to Rc/Fc. F) Timecourse for effect of exposure to anticholinesterase on potential ration produced by combining C and E.

Figure 3: Relationships used to link anticholinesterase exposure to the availability of prey. See text for details. Relationships in B and C utilize empirical data. Closed circles represent control conditions. Open circles represent the exposed (inhibited) condition. A) Representation of a constant level of anticholinesterase pesticide exposure (either single compound or mixtures). B) Sigmoidal relationship between exposure concentration and relative prey abundance showing a dose-dependent reduction defined by control abundance (horizontal line at 1, Pc), sigmoid (*i.e.*, hille) slope (prey slope), the concentration producing a 50% reduction in prey (vertical line,

EC_{50}), and a minimum abundance always present (horizontal line denoted as floor, Pf). C) Timecourse of prey abundance including a 1-day spike in prey drift relative to the available prey and the level of toxicity followed by a drop to the level of impact or the floor whichever is greater. During extended exposures at low toxicity recovery can begin at the constant prey influx rate multiplied by the current level of toxicity. After exposure recovery to control prey drift is at the constant rate of influx from upstream habitats.

Figure 4: Relationships used to link anticholinesterase exposure to growth rate relating to long-term weight gain of each fish. See text for details. Relationships in A, B, and C utilize empirical data. Closed circles represent control conditions. Open circles (*e.g.*, Ai) represent the exposed (inhibited) condition. A&B) Relationships describing the timecourse of the effects of anticholinesterase exposure on the organisms ability to capture food (Panel A, potential ration) and the availability of food to capture (Panel B, relative prey abundance). The figures are the same as those in Figures 2F and 3C, respectively. For a given exposure concentration and time, multiplying potential ration by relative prey abundance yields the final ration acquired by the organism. C) A linear model was used to relate final ration to growth rate using a line passing through the control conditions and through the maintenance condition with a slope denoted by Mgr. D) Timecourse for effect of exposure to anticholinesterase on growth rate produced by combining A, B, & C.

Table 1. List of values used for control parameters to model organismal growth and the model sensitivity to changes in the parameter.

Parameter	Value[1]	Error[2]	Sensitivity[3]
acetylcholinesterase activity (Ac)	1.0[4,5]	0.06[5]	-0.167
feeding (Fc)	1.0[4,5]	0.05[5]	0.088
ration (Rc)	5% weight/day[6]	0.05[7]	-0.547
feeding vs. activity slope (Mfa)	1.0[5]	0.1[5]	-0.047
ration vs. feeding slope (Mrf)	5 (Rc/Fc)	-	-
growth vs. ration slope (Mgr)	0.35[6]	0.02[6]	-0.547
growth vs. activity slope (Mga)	1.75 (Mfa*Mrf*Mgr)	-	-
initial weight	1 gram[8]	0.1[8]	1.00
control prey drift	1.0[4]	0.05[11]	0.116
AChE impact time-to-effect ($t_{1/2}$)	0.5 day[9]	n/a	0.005
AChE time-to-recovery ($t_{1/2}$)	0.25 days[10]	n/a	-0.0001
prey floor	0.20[11]	n/a	0.178
prey recovery rate	0.01[12]	n/a	0.323
somatic growth rate (Gc)	1.3[13]	0.06[6]	2.531

[1] mean value of a normal distribution used in the model or constant value when no corresponding error is listed

[2] standard deviation of the normal distribution used in the model

[3] mean sensitivity when baseline parameter is changed over range of 0.5 to 2-fold

[4] other values relative to control

[5] derived from Sandahl et al., (2005)

[6] derived from Brett et al., (1969)

[7] data from Brett et al., (1969) has no variability (ration was the independent variable) so a variability of 1% was selected to introduce some variability

[8] consistent with field-collected data for juvenile Chinook salmon (Nelson et al., 2004)

[9] estimated from Ferrari et al., 2004

[10] consistent with Labenia et al., 2007

[11] estimated from Van den Brink et al., 1996

[12] derived from Ward et al., 1995, Van den Brink et al., 1996, Colville et al., 2008

[13] derived from Brett and adapted for ocean-type Chinook salmon, used for sensitivity analysis

Table 2. Species specific control parameters to model organismal growth and survival rates. Growth period and survival rate are determined from the literature data listed for each species. Gc and α were calculated to make the basic model produce the appropriate size and survival values from the literature.

	Chinook Stream-type[1]	Chinook Ocean-type[2]	Coho[3]	Sockeye[4]
days to run organismal growth model	184	140	184	168
growth rate % body wt/day (Gc)	1.28	1.30	0.90	1.183
α from equation S	-0.33	-1.99	-0.802	-0.871
Control Survival ϕ	0.418	0.169	0.310	0.295

[1] Values from data in Healy and Heard 1984, Fast et al., 1988, Beckman et al., 2000, Knudsen et al., 2006
[2] Values from data in Healey and Heard 1984, Howell et al., 1985, Roni and Quinn 1995, Ratner et al.,1997, PSCCTC 2002, Green and Beechie, 2004, Johnson et al., 2007
[3] Values from data in Pess et al., 2002, Knudsen et al., 2002
[4] Values from data in Pauley et al., 1989, Gustafson et al., 1997, McGurk 2000

Table 3. Effects values (µg/L) and slopes for AChE activity, acute fish lethality, and prey abundance dose-response curves.

compound	AChE Activity EC_{50}[1] □g/L	AChE Activity slope	Fish lethality LC_{50}[2] □g/L	Fish lethality slope[3]	Prey Abundance EC_{50}[4] □g/L	Prey Abundance Slope[5]
carbaryl	145.8	0.95	250	3.63	4.33	5.5
carbofuran	58.4	0.95	164	3.63	1.22	5.5
methomyl	213	0.95	560	3.63	20.74	5.5

[1] Values from Laetz et. al 2009
[2] Values from EPA BEs
[3] sigmoidal slope that produces responses with a probit slope of Peterson, see text.
[4] Values from analysis of global search of reported LC50 and EC50s reported in EPA's Ecotox database. See text.
[5] Values from Peterson et al., 2001

Table 4. Matrix transition element and sensitivity (S) and elasticity (E) values for each model species. These control values are listed by the transition element taken from the life history graphs as depicted in Figure 1 and the literature data described in the method text. Blank cells indicate elements that are not in the transition matrix for a particular species. The influence of each matrix element on λ was assessed by calculating the sensitivity (S) and elasticity (E) values for A. The sensitivity of matrix element a_{ij} equals the rate of change in λ with respect to the transition element, defined by $\delta\lambda / \delta a$. The elasticity of transition element a_{ij} is defined as the proportional change in λ relative to the proportional change in a_{ij}, and equals (a_{ij}/λ) times the sensitivity of a_{ij}. Elasticity values allow comparison of the influence of individual transition elements and comparison across matrices.

Transition Element	Chinook Stream-type			Chinook Ocean-type			Coho			Sockeye		
	Value[1]	S	E	Value[2]	S	E	Value[3]	S	E	Value[4]	S	E
S1	0.0643	3.844	0.247	0.0056	57.13	0.292	0.0296	11.59	0.333	0.0257	9.441	0.239
S2	0.1160	2.132	0.247	0.48	0.670	0.292	0.0505	6.809	0.333	0.183	1.326	0.239
S3	0.17005	1.448	0.246	0.246	0.476	0.106				0.499	0.486	0.239
S4	0.04	0.319	0.0127	0.136	0.136	0.0168				0.1377	0.322	0.0437
R3	0.5807	0.00184	0.0011	313.8	0.0006	0.186	732.8	0.000469	0.333			
R4	746.73	0.000313	0.233	677.1	0.000146	0.0896				379.57	0.000537	0.195
R5	1020.36	1.25E-05	0.0127	1028	1.80E-05	0.0168				608.7	7.28E-05	0.0437

[1] Value calculated from data in Healy and Heard 1984, Fast et al., 1988, Beckman et al., 2000, Knudsen et al., 2006

[2] Value calculated from data in Healey and Heard 1984, Howell et al., 1985, Roni and Quinn 1995, Ratner et al., 1997, PSCCTC 2002, Green and Beechie, 2004, Johnson et al., 2007

[3] Value calculated from data in Pess et al., 2002, Knudsen et al., 2002

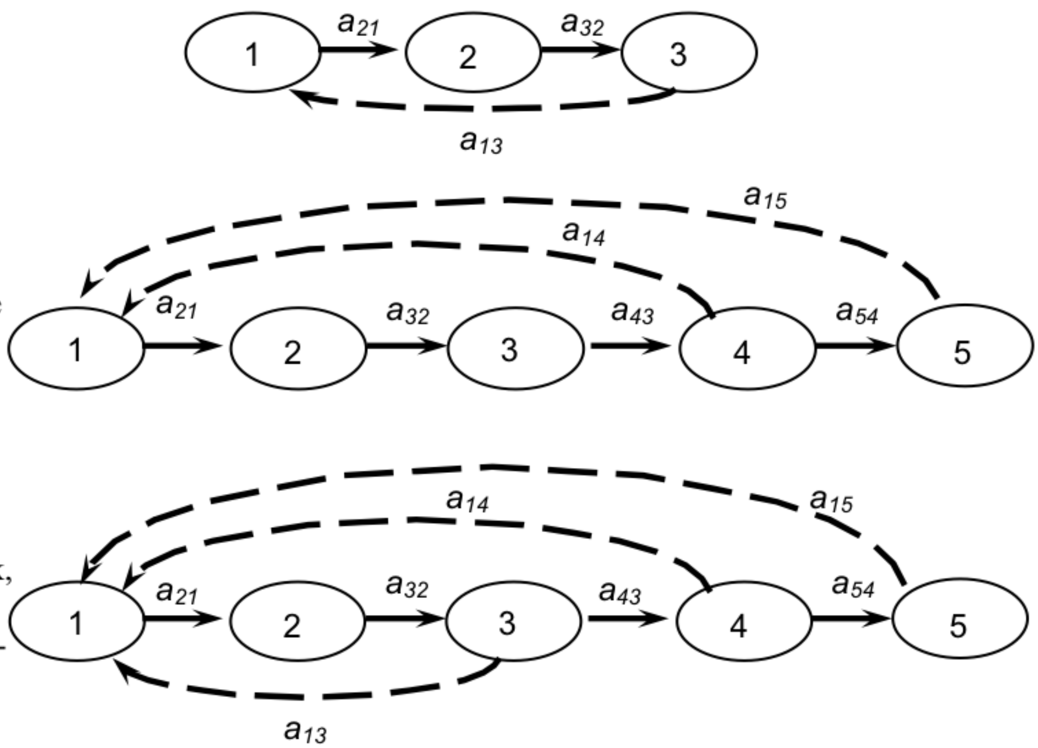
[4] Value calculated from data in Pauley et al., 1989, Gustafson et al., 1997, McGurk 2000

Figure 1.

A. Coho

B. Sockeye

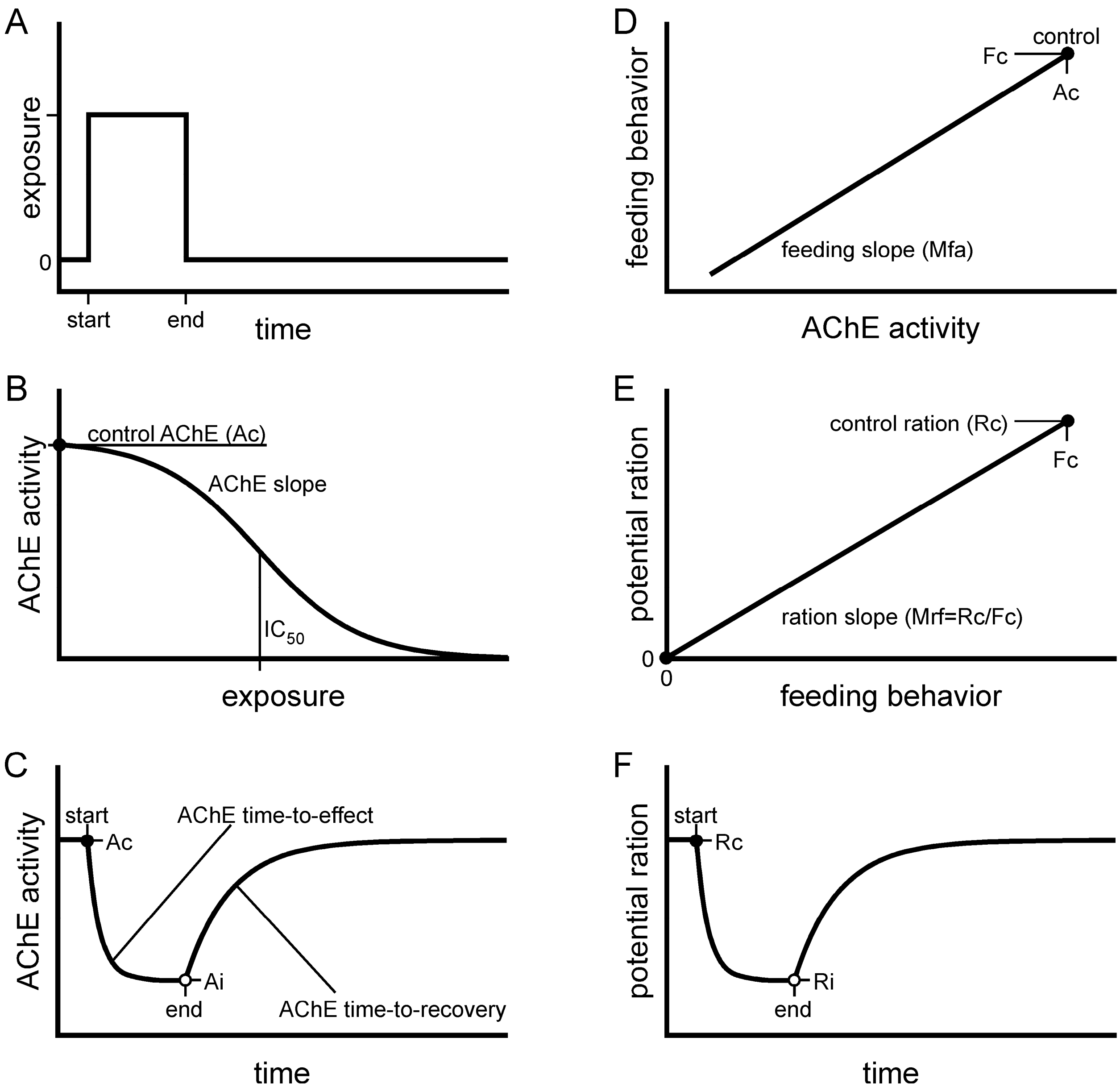
C. Chinook, ocean-type and stream-type



D. Transition Matrix

$$A = \begin{bmatrix} 0 & 0 & R3=a_{13} & R4=a_{14} & R5=a_{15} \\ S1=a_{21} & 0 & 0 & 0 & 0 \\ 0 & S2=a_{32} & 0 & 0 & 0 \\ 0 & 0 & S3=a_{43} & 0 & 0 \\ 0 & 0 & 0 & S4=a_{54} & 0 \end{bmatrix}$$

Figure 2.


A
exposure
0
start
end
time
B
AChE activity
control AChE (Ac)
AChE slope
IC50
exposure
C
AChE activity
start
Ac
AChE time-to-effect
Ai
end
AChE time-to-recovery
time
D
feeding behavior
control
Fc
Ac
feeding slope (Mfa)
AChE activity
E
potential ration
control ration (Rc)
Fc
ration slope (Mrf=Rc/Fc)
0
0
feeding behavior
F
potential ration
start
Rc
Ri
end
time

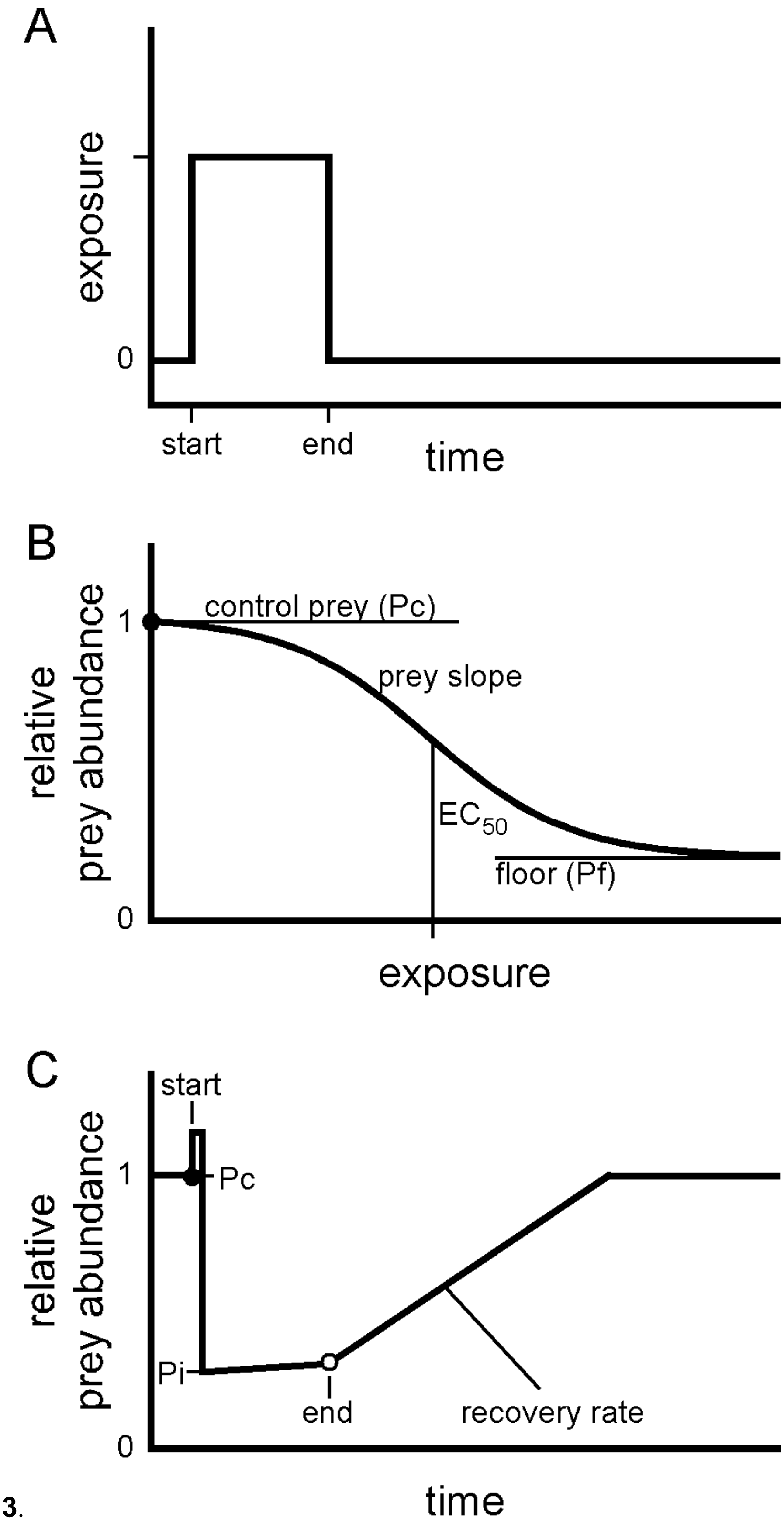

A
exposure
0
start
end
time
B
relative
prey abundance
1
0
control prey (Pc)
prey slope
EC50
floor (Pf)
exposure
C
relative
prey abundance
start
1
Pc
Pi
end
recovery rate
0
time

Figure 3.

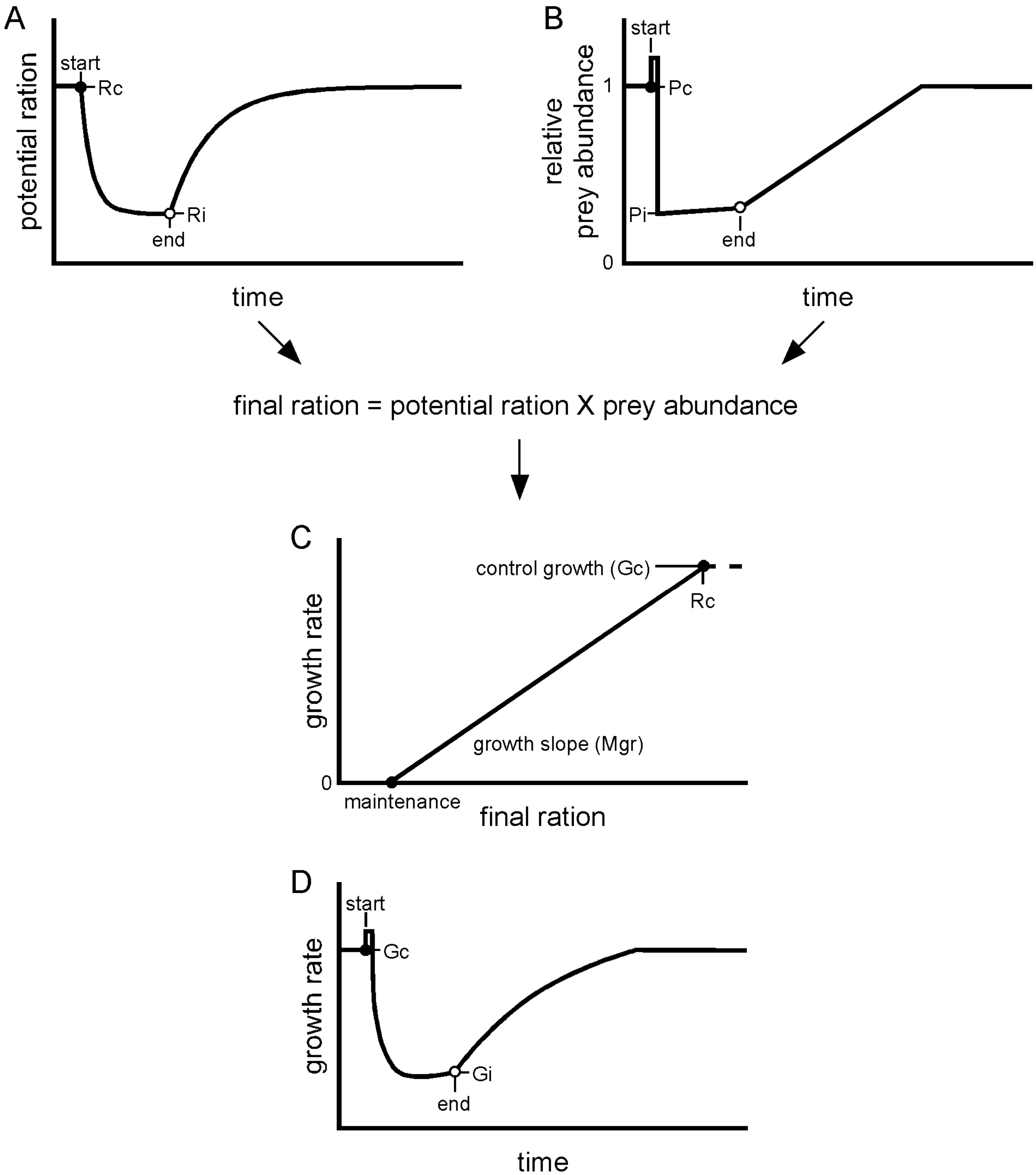

A
start
Rc
Ri
end
potential ration
time
B
start
1
Pc
relative prey abundance
Pi
end
0
time
final ration = potential ration X prey abundance
C
control growth (Gc)
Rc
growth rate
growth slope (Mgr)
0
maintenance
final ration
D
start
Gc
growth rate
Gi
end
time

Figure 4.

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Appendix 2. Species and Population Annual Rates of Growth

Chinook Salmon

ESU	Population	λ - H=0	95% CI -lower	95% CI - upper
California Coastal	Eel River	N/A	N/A	N/A
	Redwood Creek	N/A	N/A	N/A
	Mad River	N/A	N/A	N/A
	Humboldt Bay tributaries	N/A	N/A	N/A
	Bear River	N/A	N/A	N/A
	Mattole River	N/A	N/A	N/A
	Tenmile to Gualala	N/A	N/A	N/A
	Russain River	N/A	N/A	N/A
Central Valley Spring - Run (Good et al., 2005 - 90% CI)	Butte Creek - spring run	1.300	1.060	1.600
	Deer Creek - spring run	1.170	1.040	1.350
	Mill Creek - spring run	1.190	1.000	1.470
Lower Columbia River (Good et al., 2005) (# = McElhany et al., 2007)	Youngs Bay	N/A	N/A	N/A
	Grays River - fall run	0.944	0.739	1.204
	Big Creek	N/A	N/A	N/A
	Elochoman River - fall run	1.037	0.813	1.323
	Clatskanie River #	0.990	0.824	1.189
	Mill, Abernathy, Germany Creeks - fall run	0.981	0.769	1.252
	Scappose Creek	N/A	N/A	N/A
	Coweeman River - fall run	1.092	0.855	1.393
	Lower Cowlitz River - fall run	0.998	0.776	1.282
	Upper Cowlitz River - fall run	N/A	N/A	N/A
	Toutle River - fall run	N/A	N/A	N/A
	Kalamaha River - fall run	0.937	0.763	1.242
	Salmon Creek / Lewis River - fall run	0.984	0.771	1.256
	Clackamas River - fall run	N/A	N/A	N/A
	Washougal River - fall run	1.025	0.803	1.308
	Sandy River - fall run	N/A	N/A	N/A
	Lower Gorge tributaries	N/A	N/A	N/A
	Upper Gorge tributaries - fall run	0.959	0.751	1.224
	Hood River - fall run	N/A	N/A	N/A
	Big White Salmon River - fall run	0.963	0.755	1.229
	Sandy River - late fall run	0.943	0.715	1.243
	North Fork Lewis River - late fall run	0.968	0.756	1.204
	Upper Cowlitz River - spring run	N/A	N/A	N/A
	Cispus River	N/A	N/A	N/A
	Tilton River	N/A	N/A	N/A
	Toutle River - spring run	N/A	N/A	N/A
	Kalamaha River - spring run	N/A	N/A	N/A
	Lewis River - spring run	N/A	N/A	N/A
	Sandy River - spring run #	0.961	0.853	1.083
	Big White Salmon River - spring run	N/A	N/A	N/A
	Hood River - spring run	N/A	N/A	N/A

Chinook Salmon (continued)

ESU	Population	λ - H=0	95% CI -lower	95% CI - upper
Upper Columbia River Spring - Run (FCRPS)	Methow River	1.100	N/A	N/A
	Twisp River	N/A	N/A	N/A
	Chewuch River	N/A	N/A	N/A
	Lost / Early River	N/A	N/A	N/A
	Entiat River	0.990	N/A	N/A
	Wenatchee River	1.010	N/A	N/A
	Chiawawa River	N/A	N/A	N/A
	Nason River	N/A	N/A	N/A
	Upper Wenatchee River	N/A	N/A	N/A
	White River	N/A	N/A	N/A
	Little Wenatchee River	N/A	N/A	N/A
Puget Sound (only have λ where hatchery fish = native fish), (Good et al., 2005)	Nooksack - North Fork	0.750	0.680	0.820
	Nooksack - South Fork	0.940	0.880	0.990
	Lower Skagit	1.050	0.960	1.140
	Upper Skagit	1.050	0.990	1.110
	Upper Cascade	1.060	1.010	1.110
	Lower Sauk	1.010	0.890	1.130
	Upper Sauk	0.960	0.900	1.020
	Suiattle	0.990	0.930	1.050
	Stillaguamish - North Fork	0.920	0.880	0.960
	Stillaguamish - South Fork	0.990	0.970	1.010
	Skykomish	0.870	0.840	0.900
	Snoqualmie	1.000	0.960	1.040
	North Lake Washington	1.070	1.000	1.140
	Cedar	0.990	0.920	1.060
	Green	0.670	0.610	0.730
	White	1.160	1.100	1.220
	Puyallup	0.950	0.890	1.010
	Nisqually	1.040	0.970	1.110
	Skokomish	1.040	1.000	1.080
	Dosewallips	1.170	1.070	1.270
	Duckabush	N/A	N/A	N/A
	Hamma Hamma	N/A	N/A	N/A
	Mid Hood Canal	N/A	N/A	N/A
	Dungeness	1.090	0.980	1.200
	Elwha	0.950	0.840	1.060
Sacramento River Winter - Run (Good, 2005 - 90% CI))	Sacramento River - winter run	0.970	0.870	1.090

Chinook Salmon (continued)

ESU	Population	λ - H=0	95% CI -lower	95% CI - upper
Snake River Fall - Run (Good, 2005)	Lower Snake River	1.024	N/A	N/A
Snake River Spring/Summer - Run (FCRPS)	Tucannon River	1.000	N/A	N/A
	Wenaha River	1.100	N/A	N/A
	Wallowa River	N/A	N/A	N/A
	Lostine River	1.050	N/A	N/A
	Minam River	1.050	N/A	N/A
	Catherine Creek	0.970	N/A	N/A
	Upper Grande Ronde River	N/A	N/A	N/A
	South Fork Salmon River	1.110	N/A	N/A
	Secesh River	1.070	N/A	N/A
	Johnson Creek	N/A	N/A	N/A
	Big Creek Spring Run	1.090	N/A	N/A
	Big Creek Summer Run	1.090	N/A	N/A
	Loon Creek	N/A	N/A	N/A
	Marsh Creek	1.080	N/A	N/A
	Bear Valley / Elk Creek	1.100	N/A	N/A
	North Fork Salmon River	N/A	N/A	N/A
	Lemhi River	1.020	N/A	N/A
	Pahsimeroi River	1.080	N/A	N/A
	East Fork Salmon Spring Run	1.040	N/A	N/A
	East Fork Salmon Summer Run	1.040	N/A	N/A
	Yankee Fork Spring Run	N/A	N/A	N/A
	Yankee Fork Summer Run	N/A	N/A	N/A
	Valley Creek Spring Run	N/A	N/A	N/A
	Valley Creek Summer Run	N/A	N/A	N/A
	Upper Salmon Spring Run	1.060	N/A	N/A
	Upper Salmon Summer Run	1.060	N/A	N/A
	Alturas Lake Creek	N/A	N/A	N/A
	Imnaha River	1.050	N/A	N/A
	Big Sheep Creek	N/A	N/A	N/A
	Lick Creek	N/A	N/A	N/A
Upper Williamette River (McElhany et al., 2007)	Clackamas River	0.967	0.849	1.102
	Molalla River	N/A	N/A	N/A
	North Santiam River	N/A	N/A	N/A
	South Santiam River	N/A	N/A	N/A
	Calapooia River	N/A	N/A	N/A
	McKenzie River	0.927	0.761	1.129
	Middle Fork Williamette River	N/A	N/A	N/A
	Upper Fork Williamette River	N/A	N/A	N/A

Chum Salmon

ESU	Population	λ - H=0	95% CI -lower	95% CI - upper
Columbia River	Youngs Bay	N/A	N/A	N/A
	Grays River	0.954	0.855	1.064
	Big Creek	N/A	N/A	N/A
	Elochoman River	N/A	N/A	N/A
	Clatskanie River	N/A	N/A	N/A
	Mill, Abernathy and German Creeks	N/A	N/A	N/A
	Scappose Creek	N/A	N/A	N/A
	Cowlitz River	N/A	N/A	N/A
	Kalama River	N/A	N/A	N/A
	Lewis River	N/A	N/A	N/A
	Salmon Creek	N/A	N/A	N/A
	Clackamus River	N/A	N/A	N/A
	Sandy River	N/A	N/A	N/A
	Washougal River	N/A	N/A	N/A
	Lower Gorge tributaries	0.984	0.883	1.096
	Upper Gorge tributaries	N/A	N/A	N/A
Hood Canal Summer - Run (only have λ where hatchery fish reproductive potential = native fish; Good et. al., 2005)	Jimmycomelately Creek	0.850	0.690	1.010
	Salmon / Snow Creeks	1.230	1.130	1.330
	Big / Little Quilcene rivers	1.390	1.170	1.610
	Lilliwaup Creek	1.190	0.750	1.630
	Hamma Hamma River	1.300	1.110	1.490
	Duckabush River	1.100	0.930	1.270
	Dosewallips River	1.170	0.930	1.410
	Union River	1.150	1.050	1.250
	Chimacum Creek	N/A	N/A	N/A
	Big Beef Creek	N/A	N/A	N/A
	Dewetto Creek	N/A	N/A	N/A

Coho Salmon

ESU	Population	λ - H=0	95% CI -lower	95% CI - upper
Central California Coast	Ten Mile River	N/A	N/A	N/A
	Noyo River	N/A	N/A	N/A
	Big River	N/A	N/A	N/A
	Navarro River	N/A	N/A	N/A
	Garcia River	N/A	N/A	N/A
	Other Mendacino County Rivers	N/A	N/A	N/A
	Gualala River	N/A	N/A	N/A
	Russain River	N/A	N/A	N/A
	Other Sonoma County Rivers	N/A	N/A	N/A
	Martin County	N/A	N/A	N/A
	San Mateo County	N/A	N/A	N/A
	Santa Cruz County	N/A	N/A	N/A
	San Lorenzo River	N/A	N/A	N/A
Lower Columbia River (Good et al., 2005)	Youngs Bay	N/A	N/A	N/A
	Grays River	N/A	N/A	N/A
	Elochoman River	N/A	N/A	N/A
	Clatskanie River	N/A	N/A	N/A
	Mill, Abernathy, Germany Creeks	N/A	N/A	N/A
	Scappose Creek	N/A	N/A	N/A
	Cispus River	N/A	N/A	N/A
	Tilton River	N/A	N/A	N/A
	Upper Cowlitz River	N/A	N/A	N/A
	Lower Cowlitz River	N/A	N/A	N/A
	North Fork Toutle River	N/A	N/A	N/A
	South Fork Toutle River	N/A	N/A	N/A
	Coweeman River	N/A	N/A	N/A
	Kalama River	N/A	N/A	N/A
	North Fork Lewis River	N/A	N/A	N/A
	East Fork Lewis River	N/A	N/A	N/A
	Upper Clackamas River	1.028	0.898	1.177
	Lower Clackamas River	N/A	N/A	N/A
	Salmon Creek	N/A	N/A	N/A
	Upper Sandy River	1.102	0.874	1.172
	Lower Sandy River	N/A	N/A	N/A
	Washougal River	N/A	N/A	N/A
	Lower Columbia River gorge tributaries	N/A	N/A	N/A
	White Salmon	N/A	N/A	N/A
	Upper Columbia River gorge tributaries	N/A	N/A	N/A
	Hood River	N/A	N/A	N/A

Coho Salmon (continued)

ESU	Population	λ - H=0	95% CI -lower	95% CI - upper
Southern Oregon and Northern California Coast	Southern Oregon and Northern California Coast	N/A	N/A	N/A
Oregon Coast	Necanicum	N/A	N/A	N/A
	Nehalem	N/A	N/A	N/A
	Tillamook	N/A	N/A	N/A
	Nestucca	N/A	N/A	N/A
	Siletz	N/A	N/A	N/A
	Yaquima	N/A	N/A	N/A
	Alsea	N/A	N/A	N/A
	Siuslaw	N/A	N/A	N/A
	Umpqua	N/A	N/A	N/A
	Coos	N/A	N/A	N/A
	Coquille	N/A	N/A	N/A

Sockeye Salmon

ESU	Population	λ - H=0	95% CI -lower	95% CI - upper
Ozette Lake	Ozette Lake	N/A	N/A	N/A
Snake River	Snake River	N/A	N/A	N/A

Steelhead

DPS	Population	λ - H=0	95% CI -lower	95% CI - upper
Central California Coast (Good et al., 2005)	Russain River	N/A	N/A	N/A
	Lagunitas	N/A	N/A	N/A
	San Gregorio	N/A	N/A	N/A
	Waddell Creek	N/A	N/A	N/A
	Scott Creek	N/A	N/A	N/A
	San Vincente Creek	N/A	N/A	N/A
	San Lorenzo River	N/A	N/A	N/A
	Soquel Creek	N/A	N/A	N/A
	Aptos Creek	N/A	N/A	N/A
California Central Valley (Good et al., 2005)	Sacramento River	0.950	0.900	1.020
Lower Columbia River (Good et al., 2005)	Cispus River	N/A	N/A	N/A
	Tilton River	N/A	N/A	N/A
	Upper Cowlitz River	N/A	N/A	N/A
	Lower Cowlitz River	N/A	N/A	N/A
	Coweeman River	0.908	0.792	1.041
	South Fork Toutle River	0.938	0.830	1.059
	North Fork Toutle River	1.062	0.915	1.233
	Kalama River - winter run	1.010	9.130	1.117
	Kalama River - summer run	0.981	0.889	1.083
	North Fork Lewis River - winter run	N/A	N/A	N/A
	North Fork Lewis River - summer run	N/A	N/A	N/A
	East Fork Lewis River - winter run	N/A	N/A	N/A
	East Fork Lewis River - summer run	N/A	N/A	N/A
	Salmon Creek	N/A	N/A	N/A
	Washougal River - winter run	N/A	N/A	N/A
	Washougal River - summer run	1.003	0.884	1.138
	Clackamas River	0.971	0.901	1.047
	Sandy River	0.945	0.850	1.051
	Lower Columbia gorge tributaries	N/A	N/A	N/A
	Upper Columbia gorge tributaries	N/A	N/A	N/A

Steelhead (continued)

DPS	Population	λ - H=0	95% CI -lower	95% CI - upper
Middle Columbia River (Good et al., 2005)	Klickitat River	N/A	N/A	N/A
	Yakima River	1.009	N/A	N/A
	Fifteenmile Creek	0.981	N/A	N/A
	Deschutes River	1.022	N/A	N/A
	John Day - upper main stream	0.975	N/A	N/A
	John Day - lower main stream	0.981	N/A	N/A
	John Day - upper north fork	1.011	N/A	N/A
	John Day - lower north fork	1.013	N/A	N/A
	John Day - middle fork	0.966	N/A	N/A
	John Day - south fork	0.967	N/A	N/A
	Umatilla River	1.007	N/A	N/A
	Touchet River	0.961	N/A	N/A
Northern California (Good et al., 2005)	Redwood Creek	N/A	N/A	N/A
	Mad River - winter run	1.000	0.930	1.050
	Eel River - summer run	0.980	0.930	1.040
	Mattole River	N/A	N/A	N/A
	Ten Mile river	N/A	N/A	N/A
	Noyo River	N/A	N/A	N/A
	Big River	N/A	N/A	N/A
	Navarro River	N/A	N/A	N/A
	Garcia River	N/A	N/A	N/A
	Gualala River	N/A	N/A	N/A
	Other Humboldt County streams	N/A	N/A	N/A
	Other Mendocino County streams	N/A	N/A	N/A
Puget Sound*	Puget Sound	N/A	N/A	N/A
Snake River (Good et al., 2005)	Tucannon River	0.886	N/A	N/A
	Lower Granite run	0.994	N/A	N/A
	Snake A run	0.998	N/A	N/A
	Snake B run	0.927	N/A	N/A
	Asotin Creek	N/A	N/A	N/A
	Upper Grande Ronde River	0.967	N/A	N/A
	Joseph Creek	1.069	N/A	N/A
	Imnaha River	1.045	N/A	N/A
	Camp Creek	1.077	N/A	N/A
South-Central California Coast	South-Central California Coast	N/A	N/A	N/A
Southern California	Santa Ynez River	N/A	N/A	N/A
	Ventura River	N/A	N/A	N/A
	Matilija River	N/A	N/A	N/A
	Creek River	N/A	N/A	N/A
	Santa Clara River	N/A	N/A	N/A

Steelhead (continued)

DPS	Population	λ - H=0	95% CI -lower	95% CI - upper
Upper Columbia River (Good et al., 2005)	Wenatchee / Entiat Rivers	1.067	N/A	N/A
	Methow / Okanogan Rivers	1.086	N/A	N/A
Upper Williamette River (McElhany et al., 2007)	Molalla River	0.988	0.790	1.235
	North Santiam River	0.983	0.789	1.231
	South Santiam River	0.976	0.855	1.114
	Calapooia River	1.023	0.743	1.409

Appendix 3: Abbreviations

7-DADMax	7-day average of the daily maximum
ACA	Alternative Conservation Agreement
AChE	acetylcholinesterase
a.i.	active ingredient
APEs	alkylphenol ethoxylates
APHIS	U.S. Department of Agriculture Animal Plant and Health Inspection Service
BE	Biological Evaluation
BEAD	Biological and Economic Analysis Divsion
BLM	Bureau of Land Management
BMP	Best Management Practices
BOR	Bureau of Reclaimation
BOR	Bureau of Reclamation
BPA	Bonneville Power Administration
BRT	Biological Review Team (NOAA Fisheries)
BY	Brood Years
CAISMP	California Aquatic Invasive Species Management Plan
CALFED	CALFED Bay-Delta Program (California Resource Agency)
CBFWA	Columbia Basin Fish and Wildlife Authority
CBI	Confidential Business Information
CC	California Coastal
CCC	Central California Coast
CCV	Central California Valley
CDPR	California Department of Pesticide Regulation
CHART	Critical Habitat Assessment Review Team
CIDMP	Comprehensive Irrigation District Management Plan
CFR	Code of Federal Regulations
cfs	cubic feet per second
CDFG	California Department of Fish and Game
Corps	U.S. Department of the Army Corps of Engineers

CSOs	combined sewer/stormwater overflows
CSWP	California State Water Project
CURES	Coalition for Urban/Rural Environmental Stewardship
CVP	Central Valley Projects
CVRWQCB	Central Valley Regional Water Quality Control Board
CWA	Clean Water Act
d	day
DCI	Date Call-Ins
DDD	Dichloro Diphenyl Dichloroethane
DDE	Diphenyl Dichlorethylene
DDT	Dichloro Diphenyl Trichloroethane
DER	Data Evaluation Review
DEQ	Oregon Department of Environmental Quality
DIP	Demographically Independent Population
DOE	Washington State Department of Ecology
DPS	Distinct Population Segment
EC	Emulsifiable Concentrate Pesticide Formulation
EC_{50}	Median Effect Concentration
EEC	Estimated Environmental Concentration
EFED	Environmental Fate and Effects Division
EIM	Environmental Information Management
EPA	U.S. Environmental Protection Agency
ESPP	Endangered Species Protection Program
ESA	Endangered Species Act
ESU	Evolutionarily Significant Unit
EU	European Union
EXAMS	Tier II Surface Water Computer Model
FERC	Federal Energy Regulatory Commission
FCRPS	Federal Columbia River Power System
FFDCA	Federal Food and Drug Cosmetic Act
FIFRA	Federal Insecticide, Fungicide, and Rodenticide Act

FQPA	Food Quality Protection Act
ft	feet
GENEEC	Generic Estimated Exposure Concentration
h	hour
HCP	Habitat Conservation Plan
HSRG	Hatchery Scientific Review Group
HUC	Hydrological Unit Code
IBI	Indices of Biological Integrity
ICTRT	Interior Columbia Technical Recovery Team
ILWP	Irrigated Lands Waiver Program
IPCC	Intergovernmental Panel on Climate Change
IRED	Interim Re-registration Decision
LCFRB	Lower Columbia Fish Recovery Board
ISG	Independent Science Group
ITS	Incidental Take Statement
km	kilometer
Lbs	Pounds
LC_{50}	Median Lethal Concentration.
LCR	Lower Columbia River
LOAEC	Lowest Observed Adverse Effect Concentration.
LOEL	Lowest Observed Adverse Effect level
LOC	Level of Concern
LOEC	Lowest Observed Effect Concentration
LOQ	Limit of Quantification
LWD	Large Woody Debris
m	meter
MCR	Middle Columbia River
mg/L	milligrams per liter
MOA	Memorandum of Agreement
MPG	Major Population Group
MRID	Master Record Identification Number

MTBE	Methyl tert-butyl ether
NASA	National Aeronautics and Space Administration
NAWQA	U.S. Geological Survey National Water-Quality Assessment
NC	Northern California
NEPA	National Environmental Protection Agency
NLCD	Natural Land Cover Data
NP	Nonylphenol
NPDES	National Pollutant Discharge Elimination System
NPS	National Parks Services
NRCS	Natural Resources Conservation Service
NWS	National Weather Service
NEPA	National Environmental Policy Act
NMA	National Mining Association
NMC	*N*-methyl carbamates
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
NOAEC	No Observed Adverse Effect Concentration
NPDES	National Pollution Discharge Eliminating System
NPIRS	National Pesticide Information Retrieval System
NRC	National Research Council
OC	Oregon Coast
ODFW	Oregon Division of Fish and Wildlife
OP	Organophosphates
Opinion	Biological Opinion
OPP	EPA Office of Pesticide Program
PAH	polyaromatic hydrocarbons
PBDEs	polybrominated diphenyl ethers
PCBs	polychlorinated biphenyls
PCEs	primary constituent elements
POP	Persistent Organic Pollutants
ppb	Parts Per Billion

PPE	Personal Protection Equipment
PSP	Pesticide Stewardship Partnerships
PSAMP	Puget Sound Assessment and Monitoring Program
PSAT	Puget Sound Action Team
PRIA	Pesticide Registration Improvement Act
PRZM	Pesticide Root Zone Model
PUR	Pesticide Use Reporting
QA/QC	Quality Assurance/Quality Control
RED	Re-registration Eligibility Decision
REI	Restricted Entry Interval
RPA	Reasonable and Prudent Alternatives
RPM	reasonable and prudent measures
RQ	Risk Quotient
SAP	Scientific Advisory Panel
SAR	smolt-to-adult return rate
SASSI	Salmon and Steelhead Stock Inventory
SC	Southern California
S-CCC	South-Central California Coast
SONCC	Southern Oregon Northern California Coast
SLN	Special Local Need (Registrations under Section 24(c) of FIFRA)
SR	Snake River
TCE	Trichloroethylene
TCP	3,5,6-trichloro-2-pyridinal
TGAI	Technical Grade Active Ingredient
TIE	Toxicity Identification Evaluation
TMDL	Total Maximum Daily Load
TRT	Technical Recovery Team
UCR	Upper Columbia River
USFS	United States Forest Service
USC	United States Code
USFWS	United States Fish and Wildlife Service

USGS	United States Geological Survey
UWR	Upper Willamette River
VOC	Volatile Organic Compounds
VSP	Viable Salmonid Population
WDFW	Washington Department of Fish and Wildlife
WLCRTRT	Willamette/Lower Columbia River Technical Review Team
WQS	Water Quality Standards
WWTIT	Western Washington Treaty Indian Tribes
WWTP	Wastewater Treatment Plant

Appendix 4: Glossary

303(d) waters Section 303 of the federal Clean Water Act requires states to prepare a list of all surface waters in the state for which beneficial uses – such as drinking, recreation, aquatic habitat, and industrial use - are impaired by pollutants. These are water quality limited estuaries, lakes, and streams that do not meet the state's surface water quality standards and are not expected to improve within the next two years. After water bodies are put on the 303(d) list they enter into a Total Maximum Daily Load Clean Up Plan.

Active ingredient	The component(s) that kills or otherwise affects the pest. A.i.s are always listed on the label (FIFRA 2(a)).
Adulticide	A compound that kills the adult life stage of the pest insect.
Anadromous Fish	Species that are hatched in freshwater migrate to and mature in salt water and return to freshwater to spawn.
Adjuvant	A compound that aides the operation or improves the effectiveness of a pesticide.
Alevin	Life history stage of a salmonid immediately after hatching and before the yolk-sac is absorbed. Alevins usually remain buried in the gravel in or near the egg nest (redd) until their yolk sac is absorbed when they swim up and enter the water column.
Anadromy	The life history pattern that features egg incubation and early juvenile development in freshwater migration to sea water for adult development, and a return to freshwater for spawning.
Assessment Endpoint	Explicit expression of the actual ecological value that is to be protected (*e.g.*, growth of juvenile salmonids).

Bioaccumulation	Accumulation through the food chain (*i.e.*, consumption of food, water/sediment) or direct water and/or sediment exposure.
Bioconcentration	Uptake of a chemical across membranes, generally used in reference to waterborne exposures.
Biomagnification	Transfer of chemicals via the food chain through two or more trophic levels as a result of bioconcentration and bioaccumulation.
Degradates	New compounds formed by the transformation of a pesticide by chemical or biological reactions.
Distinct Population Segment	A listable entity under the ESA that meets tests of discreteness and significance according to USFWS and NMFS policy. A population is considered distinct (and hence a "species" for purposes of conservation under the ESA) if it is discrete fro an significant to the remainder of its species based n factors such as physical, behavioral, or genetic characteristics, it occupies an unusual or unique ecological setting, or its loss would represent a significant gap in the species' range.
Escapement	The number of fish that survive to reach the spawning grounds or hatcheries. The escapement plus the number of fish removed by harvest form the total run size.
Evolutionarily Significant Unit	A group of Pacific salmon or steelhead trout that is (1) substantially reproductively isolated from other conspecific units and (2) represent an important component of the evolutionary legacy of the species.
Fall Chinook	This salmon stock returns from the ocean in late summer and early

Salmon	fall to head upriver to its spawning grounds, distinguishing it from other stocks which migrate in different seasons.
Fate	Dispersal of a material in various environmental compartments (sediment, water air, biota) as a result of transport, transformation, and degradation.
Flowable	A pesticide formulation that can be mixed with water to form a suspension in a spray tank.
Fry	Stage in salmonid life history when the juvenile has absorbed its yolk sac and leaves the gravel of the redd to swim up into the water column. The fry stage follows the alevin stage and in most salmonid species is followed by the parr, fingerling, and smolt stages. However, chum salmon juveniles share characteristics of both the fry and smolt stages and can enter sea water almost immediately after becoming fry.
Half-pounder	A life history trait of steelhead exhibited in the Rogue, Klamath, Mad, and Eel Rivers of southern Oregon and northern California. Following smoltification, half-pounders spend only 2-4 months in the ocean, then return to fresh water. They overwinter in fresh water and emigrate to salt water again the following spring. This is often termed a false spawning migration, as few half-pounders are sexually mature.
Hatchery	Salmon hatcheries use artificial procedures to spawn adults and raise the resulting progeny in fresh water for release into the natural environment, either directly from the hatchery or by transfer into another area. In some cases, fertilized eggs are outplanted (usually in "hatch-boxes"), but it is more common to release fry or smolts.
Inert ingredients	"an ingredient which is not active" (FIFRA 2(m)). It may be toxic or enhance the toxicity of the active ingredient.

Iteroparous	Capable of spawning more than once before death
Jacks	Male salmon that return from the ocean to spawn one or more years before full-sized adults return. For coho salmon in California, Oregon, Washington, and southern British Columbia, jacks are 2 years old, having spent only 6 months in the ocean, in contrast to adults, which are 3 years old after spending 1 ½ years in the ocean.
Jills	Female salmon that return from the ocean to spawn one or more years before full-sized adult returns. For sockeye salmon in Oregon, Washington, and southern British Columbia, jills are 3 years old (age 1.1), having spent only one winter in the ocean in contrast to more typical sockeye salmon that are age 1.2, 1.32.2, or 2.3 on return.
Kokanee	The self-perpetuating, non-anadromous form of *O. nerka* that occurs in balanced sex ration populations and whose parents, for several generations back, have spent their whole lives in freshwater.
Lambda	Also known as Population growth rate, or the rate at which the abundance of fish in a population increases or decreases.
LRL	Laboratory Reporting Level (USGS NAWQA data)- Generally equal to twice the yearly determined LT-MDL. The LRL controls false negative error. The probability of falsely reporting a non-detection for a sample that contained an analyte at a concentration equal to or greater that the LRL is predicted to be less than or equal to 1 percent.
Major Population Group (MPG)	A group of salmonid populations that are geographically and genetically cohesive. The MPG is a level of organization between demographically independent populations and the ESU.

Main channel — The stream channel that includes the thalweg (longitudinal continuous deepest portion of the channel.

Metabolite — A transformation product resulting from metabolism.

Mode of Action — A series of key processes that begins with the interaction of a pesticide with a receptor site and proceeds through operational and anatomical changes in an organisms that result in sublethal or lethal effects.

Natural fish — A fish that is produced by parents spawning in a stream or lake bed, as opposed to a controlled environment such as a hatchery.

Nonylphenols — A type of APE and is an example of an adjuvant that may be present as an ingredient of a formulated product or added to a tank mix prior to application.

Off-channel habitat — Water bodies and/or inundated areas that are connected (accessible to salmonid juveniles) seasonally or annually to the main channel of a stream including but not limited to features such as side channels, alcoves, ox bows, ditches, and floodplains.

Parr — The stage in anadromous salmonid development between absorption of the yolk sac and transformation to smolt before migration seaward.

Persistence — The tendency of a compound to remain in its original chemical form in the environment.

Pesticide — Any substance or mixture of substances intended for preventing, destroying, repelling or mitigating any pest.

Reasonable and Prudent Alternative (RPA)	Recommended alternative actins identified during formal consultation that can be implemented in a manner consistent with the scope of the Federal agency's legal authority an jurisdiction, that are economically an technologically feasible, an that the Services believes would avoid the likelihood of jeopardizing the continued existence of the listed species or the destruction or adverse modification of designated critical habitat.
Redd	A nest constructed by female salmonids in streambed gravels where eggs are deposited and fertilization occurs.
Riparian area	Area with distinctive soils an vegetation between a stream or other body of water and the adjacent upland. It includes wetlands and those portions of flood plains an valley bottoms that support riparian vegetation.
Risk	The probability of harm from actual or predicted concentrations of a chemical in the aquatic environment – a scientific judgement.
Salmonid	Fish of the family *Salmonidae*, including salmon, trout, chars, grayling, and whitefish. In general usage, the term usually refers to salmon, trout, and chars.
SASSI	A cooperative program by WDFW and WWTIT to inventory and evaluate the status of Pacific salmonids in Washington State. The SASSI report is a series of publications from this program.
Semelparous	The condition in an individual organism of reproducing only once in a lifetime.

Smolt	A juvenile salmon or steelhead migrating to the ocean and undergoing physiological changes to adapt from freshwater to a saltwater environment.
Sublethal	Below the concentration that directly causes death. Exposure to sublethal concentrations of a material may produce less obvious effect on behavior, biochemical, and/or physiological function of the organism often leading to indirect death.
Surfactant	A substance that reduces the interfacial or surface tension of a system or a surface-active substance.
Synergism	A phenomenon in which the toxicity of a mixture of chemicals is greater than that which would be expected from a simple summation of the toxicities of the individual chemicals present in the mixture.
Technical Grade Active Ingredient (TGAI)	Pure or almost pure active ingredient. Available to formulators. Most toxicology data are developed with the TGAI. The percent AI is listed on all labels.
Technical Recovery Teams (TRT)	Teams convened by NOAA Fisheries to develop technical products related to recovery planning. TRTs are complemented by planning forums unique to specific states, tribes, or reigns, which use TRT and other technical products to identify recovery actions.
Teratogenic	Effects produced during gestation that evidence themselves as altered structural or functional processes in offspring.
Total Maximum	defines how much of a pollutant a water body can tolerate (absorb)

Daily Load (TMDL)	daily and remain compliant with applicable water quality standards. All pollutant sources in the watershed combined, including non-point sources, are limited to discharging no more than the TMDL.
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 Act." Each state is responsible for maintaining water quality standards.