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

NOV 1 8 2008

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

Dear Ms. Edwards:

Enclosed is the National Oceanic Atmospheric Administration 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 chlorpyrifos, diazinon, and malathion on endangered species, threatened species, and critical habitat that has been designated for those species. This Opinion assesses the effects of all pesticides containing chlorpyrifos, diazinon, or malathion 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 the proposed action is likely to jeopardize the continued existence of 27 listed Pacific salmonids as described in the attached Opinion. NMFS also concluded that the effects of chloropyrifos, diazinon and malathion may adversely affect Ozette Lake Sockeye salmon. We further conclude that the proposed action is likely to destroy or adversely modify critical habitat for 25 of 26 listed Pacific salmonids with designated critical habitat. The proposed action will not destroy or adversely modify designated critical habitat for Ozette Lake Sockeye salmon. As NMFS did not designate critical habitat for the Lower Columbia River coho salmon or Puget Sound steelhead, the Opinion presents no analysis of critical habitat pertaining to these species.

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's 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,

James H. Lecky
Director, Office of Protected Resources

National Marine Fisheries Service
Endangered Species Act Section 7 Consultation

Biological Opinion

Environmental Protection Agency Registration of Pesticides Containing Chlorpyrifos, Diazinon, and Malathion

Photograph: Tom Maurer, USFWS

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Consultation Conducted by:	Endangered Species Division of the Office of Protected Resources, National Marine Fisheries Service
Approved by:	
Date:	NOV 18 2008

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 and NMFS or the USFWS concur with that conclusion (50 CFR §420.14(b)).

The United States (U.S.) Environmental Protection Agency (EPA) initiated consultation with NMFS on its proposal to authorize use, pursuant to the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA), 7 U.S.C. 136 *et seq.*, of pesticide products containing the active ingredients of chlorpyrifos, diazinon, and malathion on April 14, 2003, November 29, 2002, and December 1, 2004, respectively. 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 active ingredients on the listed ESUs, plus on two newly listed ESUs. 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 ESUs, 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 2002, 2003, and 2004 requests for formal consultation on the proposed authorization of the above active ingredients. 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, biological 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 active ingredients for chlorpyrifos, diazinon, and malathion.

NMFS also considered information and comments provided by EPA and by the registrants identified as applicants by EPA. We also considered comments on the draft Opinion provided to EPA by others after review of the draft Opinion.

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 Evolutionarily Significant Units (ESUs) of listed Pacific salmonids of its continuing approval of 54 pesticide active ingredients.

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 about effects to the salmonids on all 54 active ingredients by December 2004.

In December 2002, EPA and the U.S. Fish and Wildlife Service 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 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 Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA).

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, 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 active ingredients.

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 active ingredients. (EPA had concluded that 17 of the 54 active ingredients at issue in the first litigation would not affect any listed salmonid species or any of their designated critical habitat, and so did initiate consultation on those active ingredients.) This first consultation evaluates three organophosphates: chlorpyrifos, diazinon, and malathion.

Consultation History

On November 29, 2002, the EPA sent a letter to NMFS' Office of Protected Resources (OPR) requesting section 7 consultation for the registration of the active ingredient diazinon and its effects on 26 ESUs of Pacific salmonids listed at that time. In that same letter, EPA's Office of Pesticide Program (OPP) determined that the use of diazinon "may affect but is not likely to adversely affect" 4 ESUs and "may affect" 22 ESUs of listed salmonids.

On April 14, 2003, the EPA sent a letter to NMFS' OPR requesting section 7 consultation for the registration of the active ingredient chlorpyrifos and its effects on 26 ESUs of Pacific salmonids listed at that time, as well as on the Central Valley Fall/Late Fall-run Chinook salmon ESU that was proposed for listing as (NMFS later determined not to list this ESU). In that same letter, the EPA's OPP determined that the use of chlorpyrifos will have "no effect" for 2 ESUs; "may affect but is not likely to adversely affect" 6 ESUs; and "may affect" 19 ESUs of listed salmonids. EPA's "no effect" determinations for chlorpyrifos applied to the Columbia River Chum salmon and Ozette Lake Sockeye salmon ESUs.

On December 1, 2004, the EPA sent a letter to NMFS' OPR requesting section 7 consultation for the registration of the active ingredient malathion and its effects on 26 ESUs of Pacific salmonids listed at that time. In that same letter, EPA's OPP determined that the use of malathion will have "no effect" for 2 ESUs; "may affect but is not likely to adversely affect" 6 ESUs, and "may affect" 18 ESUs of listed salmonids. EPA's "no effect" determinations applied to the California Coastal Chinook salmon and Northern California steelhead ESUs.

On June 28, 2005, NMFS listed the Lower Columbia River coho salmon ESU as endangered. Given this recent listing, EPA's 2002, 2003, and 2004 effects determinations for chlorpyrifos, diazinon, and malathion on listed Pacific salmonids lack an effect determination for the Lower Columbia River coho salmon.

On May 22, 2007, NMFS listed the Puget Sound Steelhead Distinct Population Segment (DPS) as threatened. Given this recent listing, EPA's 2002, 2003, and 2004 effect determinations for chlorpyrifos, diazinon, and malathion on listed Pacific salmonids lack an effect determination for the Puget Sound Steelhead.

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

On February 11, 2008, NMFS listed the Oregon coast coho salmon ESU as threatened. EPA's 2002, 2003, and 2004 initiation packages for chlorpyrifos, diazinon, and malathion provided an effect determination for the Oregon coast coho salmon ESU. This ESU was previously listed in 1998 and its ESA status was in-flux until 2008.

From March 2008 through April 2008, NMFS requested dose-response information from EPA for chlorpyrifos, diazinon, and malathion.

On April 3, 2008, EPA provided some of the requested information to NMFS (diazinon acute study information).

On July 31, 2008, NMFS provided EPA its draft Opinion on the impacts to the Pacific salmon ESUs from the proposed reregistration of pesticide products containing active ingredients chlorpyrifos, diazinon, and malathion and their formulations in the U.S. and its affiliated territories. NMFS' draft Opinion concluded EPA's proposed action will jeopardize all 28 listed Pacific salmon ESUs and destroy or adversely modify designated critical habitat for 26 salmon ESUs. The draft Opinion did not provide reasonable and prudent alternatives (RPAs) as NMFS intended to develop RPAs in cooperation with EPA. NMFS' transmittal memo accompanying the draft Opinion also requested EPA feedback on the document by September 2, 2008.

On August 8, 2008, NMFS contacted EPA to discuss joint development of RPAs. The agencies scheduled a meeting for August 20, 2008.

On August 14, 2008, EPA posted NMFS' draft Opinion onto its website and opened a docket for the document on http\\:www.regulations.gov. This posting allowed for public comment on the draft Opinion as part of EPA's registration process. EPA also conveyed questions in advance of the August 20, 2008 meeting, to NMFS via e-mail. Questions pertained to potential risk reduction measures to avert jeopardy, NMFS' deadline for receipt of EPA comments, applicant involvement in the development of RPAs, and NMFS' briefing on the conclusion reached on the draft Opinion to the applicants.

On that same date, EPA also informed NMFS via e-mail of two applicants affected by the current consultation. Both applicants are represented by the same legal counsel and that counsel requested a meeting on behalf of his clients with both agencies on August 29, 2008. The applicants are Dow AgroSciences, LLC (DAS) and Makhteshim Agan of North American (MANA).

On August 20, 2008, EPA and NMFS met and discussed RPAs, EPA's authorities under FIFRA, NMFS' settlement agreement timeline, and preparation for the August 29, 2008, meeting with the applicants. EPA requested NMFS develop target concentrations for the three active ingredients prior to EPA engaging in RPA discussions. EPA also requested an extension to NMFS' September 2, 2008, deadline for comments on the draft Opinion. NMFS agreed to a revised deadline of September 15, 2008. During planning discussions for the August 29, 2008, meeting, EPA informed NMFS of a third applicant, Cheminova. NMFS began its work with the identified applicants in accordance with the section 7 regulations. The agencies developed an agenda that included NMFS presenting its evaluation and review of the conclusion reached in the draft Opinion at the onset of the meeting.

On August 29, 2008, EPA, NMFS, and the applicants met. NMFS presented its evaluation and review of EPA's proposed action and the conclusion reached in the draft Opinion. The applicants also provided feedback on NMFS' draft Opinion via four separate presentations. NMFS asked the applicants for their advice on the development of RPAs. The applicants stated their belief that RPA discussions were premature as they believed that NMFS' evaluation was incomplete and based on outdated information. The applicants offered to provide additional information, including confidential business information (CBI), for NMFS' consideration. The applicants and EPA also requested an extension to NMFS' September 2, 2008, deadline for comments on the draft Opinion. NMFS responded that receipt of comments by September 15, 2008 would increase its ability to consider the comments because of NMFS' October 31, 2008 stipulated settlement agreement deadline. At this same meeting, the applicants and NMFS asked EPA to identify applicants early during the consultation process and for future pesticide consultations. The applicants offered to answer questions from NMFS on the provided materials. The applicants further requested NMFS consider the supplemental information they provided in its analysis. The parties agreed to participate in a teleconference on

September 11, 2008 to address questions from NMFS on the supplemental material provided by the applicants. NMFS agreed to provide its list of questions to EPA and the applicants in advance of that meeting.

On September 4, 2008, NMFS requested two scientific studies for diazinon from MANA.

On September 5, 2008, NMFS and EPA agreed to continue RPA discussions. EPA requested the RPA discussions occur after the September 11, 2008, meeting. EPA also reiterated its request for target concentrations from NMFS as a starting point for the RPA discussions. NMFS agreed to provide draft target concentrations in advance of the next RPA meeting. NMFS also informed EPA of its request for additional studies from the applicants. On that same date, NMFS received the requested diazinon studies.

On September 9, 2008, NMFS requested scientific studies for malathion from Cheminova. NMFS received the malathion studies on that same date. NMFS also provided its list of questions based on its review of the applicant materials received to date.

On September 11, 2008, EPA, NMFS, and the applicants discussed questions raised by NMFS during its review of the supplemental materials provided by the applicants over the previous two weeks. NMFS also requested incident reports from EPA for all three active ingredients. EPA indicated it would send the reports to NMFS. The participants also discussed CBI clearance for NMFS' review of such data and procedures to secure this information. At that meeting, EPA informed NMFS that it must send a letter to EPA indicating that NMFS staff will comply with FIFRA requirements as part of the clearance process. This requirement was in addition to completion of the FIFRA on-line training and application process that NMFS staff had already completed. EPA agreed to send an example letter for NMFS to model its request.

On September 15, 2008, EPA and the applicants provided comments to NMFS on the draft Opinion. On that same date, NMFS provided draft RPAs to EPA.

On September 17, 2008, NMFS received two sets of comments (six boxes total) from the applicants on its draft Opinion. A transmittal letter accompanying the boxes explained the contents of the delivery and the applicants' position on the draft Opinion. On that same date, NMFS requested EPA approval of CBI clearance 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. 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. EPA and NMFS also discussed whether the draft RPAs will be released to parties beyond EPA and the applicants. NMFS confirmed that it did not intend to release the draft RPAs beyond the action agency and applicants.

On September 23, 2008, NMFS staff received notification of CBI clearance. NMFS immediately requested CBI data the applicants had previously provided to EPA. On that same date, EPA sent incident reports (dated post-2002) for the three active ingredients to NMFS.

On September 28, 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. NMFS' schedule for receipt of the requested information was also provided.

On October 2, 2008, EPA, NMFS, and the applicants met. NMFS presented its rationale for development of the target concentrations for the active ingredients. EPA and the applicants provided feedback on NMFS' approach. NMFS conveyed it was not comfortable providing a single point estimate as a threshold in the RPA given multiple uncertainties and existing baseline conditions. However, NMFS did so in response to EPA's request. As NMFS was still reviewing the comments received from EPA and the applicants, the parties agreed to a subsequent meeting involving all scientists from NMFS, EPA, and the applicant companies. The meeting was scheduled for October 16, 2008.

On October 3, 2008, NMFS received pre-2002 incident reports for the three active ingredients.

On October 7, 2008, NMFS received three boxes of comments provided by the applicants to EPA.

On October 10, 2008, NMFS and the applicants developed a draft agenda for the October 16, 2008 meeting. EPA was expected to participate via conference call but did not participate. The parties at this meeting agreed to a common meeting objective whereby NMFS scientists would ask additional questions of scientists from the applicants' companies.

On October 16, 2008, scientists from EPA, NMFS, and the applicant companies met and discussed reports and scientific studies provided by the applicants to NMFS. Lawyers representing the applicants and NOAA were also in attendance.

On October 20, 2008, NMFS requested (via e-mail) information from DAS on fish kills that occurred in association with chlorpyrifos applications for two terrestrial field studies conducted by Wildlife International. On that same date, NMFS also requested the full citation for a European Union (EU) mesocosm study from Cheminova or the actual report with MRID numbers. This study was discussed at the October 16, 2008, meeting. At the time of NMFS' request, it assumed requested reports would be sent directly to NMFS. Cheminova promptly responded to NMFS' query and provided the citation and MRID information to NMFS. Cheminova further clarified that NMFS must request copies of proprietary studies subject to FIFRA section 12(g) protections directly from EPA. On that same date, EPA provided the certainty data pertaining to incident information in NMFS' possession. The certainty data is a measure that indicates EPA's impression of the reliability and accuracy of a given incident report.

On October 21, 2008, EPA confirmed receipt of NMFS' request for the EU mesocosm study and MRID number via e-mail. EPA stated it would notify NMFS when the report is sent to NMFS.

On October 22, 2008, the legal counsel for DAS and MANA provided copies of the two slide presentations given at the October 16, 2008, meeting.

On October 23, 2008, the legal counsel for DAS and MANA provided an additional slide from the slide presentations

On October 24, 2008, NMFS sent an email to EPA seeking clarification on several approved uses of malathion referenced in the 2006 malathion RED document. NMFS has not received a response to this request for information.

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 chlorpyrifos, diazinon, or malathion[1]. 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 ESU/DPSs of all authorized uses by EPA, regardless of whether those uses have historically occurred. EPA's pesticide registration 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. Pesticides must be registered or exempted by EPA's OPP before they may be sold or distributed in the U.S. Once registered, a pesticide may not legally be used unless the use is consistent with the approved directions for use on the pesticide's label or labeling (http:www.epa.gov/pesticides/regulating/registering/index.htm).

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. The statutory authority for EPA's proposed action is FIFRA. FIFRA governs the sale and use of pesticides by directing EPA to regulate pesticides through a registration process. A pesticide generally may not be sold or used in the U.S. unless it is

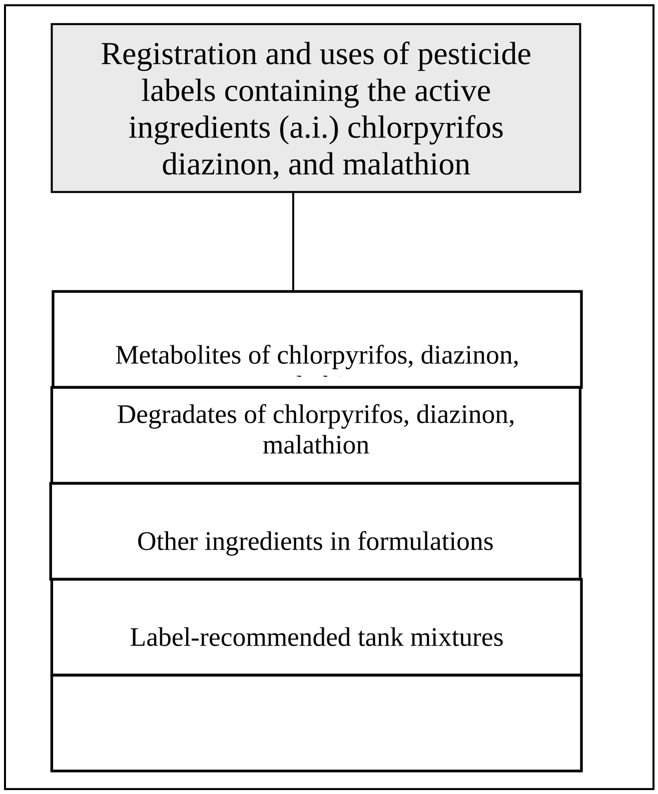
[1] December 12, 2007- EPA, NMFS, and FWS agreed that the Federal Action for EPA's FIFRA registration actions will be defined as the "authorization for use or uses described in labeling of a pesticide product containing a particular pesticide ingredient."

registered by EPA and has an approved label authorizing a given use (7 U.S.C. §136a (c)(5). Additionally, FIFRA requires product labels to specify where and how pesticide products may be used and applied. EPA authorization of pesticide uses are categorized as FIFRA sections 3 (new product registrations), 4 (re-registrations and special review), 18 (emergency use), or 24(c) (special local actions). This consultation addresses all EPA authorized uses of pesticide products containing chlorpyrifos, diazinon, and malathion.

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 is to 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." 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 (21 U.S.C. §346a)."

Pesticide Labels. For this consultation, EPA's proposed action encompasses all approved product labels containing chlorpyrifos, diazinon, and malathion; 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 (See Figure 1). The three biological evaluations (BEs) indicate that chlorpyrifos, diazinon, and malathion are labeled for a variety of uses and use sites such as pest control in agricultural crops, on structures, residential and industrial uses, animal applications, and vector control for public health programs (EPA 2002; EPA 2003; EPA 2004b). Significant modifications have been made or are planned for new product labels containing chlorpyrifos, diazinon, and malathion 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 risks from organophosphates (OP). EPA's review of individual OPs prior to July 2006 resulted in the issuance of Interim Reregistration Decisions (IREDs) for 22 OPs including chlorpyrifos and diazinon and a Reregistration Eligibility Decision (RED) for one OP, malathion. When EPA concluded the OP cumulative assessment in July 2006, all RED decisions for individual OP pesticides were considered complete. Therefore, OP IREDs are considered completed REDs. EPA has indicated that the chlorpyrifos and diazinon IREDs (EPA 2002b; EPA 2004a) and the malathion RED (EPA 2006b) provide a complete summary of all authorized uses of pesticides containing the three active ingredients (EPA 2008a; EPA 2008c; EPA 2008e). Notable changes resulting from these reregistration activities include reductions in non-crop uses of chlorpyrifos and diazinon, modifications to maximum labeled application rates, reductions in the number of applications, and specification of minimum application intervals. Despite label changes, there are very few areas within the action area where presumed use of all three compounds can be excluded.

Figure 1. Stressors of the Action



Mode of Action of Organophosphorus (OP) Insecticides. Chlorpyrifos, diazinon, and malathion share the same mechanism of action. They are neurotoxicants to the central and peripheral nervous systems of animals. In fish and aquatic invertebrates, the parent OPs are transformed into more toxic metabolites, sometimes called oxons. The active ingredient (a.i.) and their oxon metabolites inhibit the enzyme acetylcholinesterase found in brain and muscle tissue of invertebrates and vertebrates. Thus, OPs belong to a class of insecticides known as acetylcholinesterase inhibitors. Inhibition of acetylcholinesterase 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 acetylcholinesterase. 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 behavioral modification to death. NMFS batched the consultations on these three active ingredients 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. Additionally, cumulative exposure to the three active ingredients is expected given they have overlapping uses and occur together in surface water samples.

Active and Other Ingredients. Chlorpyrifos, diazinon, and malathion 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. 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 also refers to inert ingredients as "other ingredients". 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 the interfacial or surface tension of a system or a surface-active substance [e.g., a group of non-ionic surfactant is the alkylphenol polyethoxylates (APEs)]. Nonylphenol is 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.

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 nontarget 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 on specific mixing instructions for use with compatible mixes. Labels may also recommend specific tank-mixes. Pursuant to FIFRA, EPA's 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. In June 2006, EPA issued REDs for the active ingredients of chlorpyrifos, diazinon, and malathion. The REDs include various mitigation measures such as phase out and/or cancellation of certain uses of malathion, diazinon, and chlorpyrifos. These mitigation components were considered part of the proposed action.

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

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

Registration Information of Pesticide Active Ingredients under Consultation. As discussed above, the proposed action encompasses EPA's registration of the uses (as described by product labels) of all pesticides containing chlorpyrifos, diazinon, or malathion. However, EPA did not provide copies of all product labels containing chlorpyrifos, diazinon, and malathion. The following descriptions represent information acquired from review of a sample of current product labels as well as information conveyed in the BEs and EPA RED/IRED documents.

Chlorpyrifos

Chlorpyrifos is a chlorinated organophosphorus insecticide, acaricide, and nematicide widely used in agriculture and non-agricultural settings. Chlorpyrifos was first registered in 1965 for control of foliage and soil-borne insect pests on a variety of crops.

Chlorpyrifos is a white crystalline solid with a melting point of 41.5-42.5ºC. This compound is relatively stable to hydrolysis in neutral pH and acidic aqueous solutions. However, stability decreases with increasing pH. Chlorpyrifos has a half-life of 16 days at pH 9. The hydrolytic stability coupled with the aqueous photolysis half-life of 30 days, low volatilization, and degradation under aerobic conditions indicate chlorpyrifos may be persistent in the water columns of some aqueous systems with relatively long hydrological residence times.

Chlorpyrifos is also soluble in most organic solvents (i.e., acetone, xylene, and methylene chloride). Chlorpyrifos is not particularly volatile based on its low vapor pressure of 1.87 x 10^{-5} mm HG at 20ºC (Merck Index, 11th edition). Its maximum attainable vapor concentration is 25 ppb at 25ºC.

Chlorpyrifos' most common trade names are Dursban®, Lorsban®, Empire®, Equity®, Whitmire PT 270®.

The National Pesticide Information Retrieval System (http://ppis.ceris.purdue.edu/htbin/epachem.com) suggests that there are currently 24 registrants with active registrations of pesticide products containing chlorpyrifos. Several labels mention other active ingredients in chlorpyrifos containing formulations. According to the BE, in 2003, there were 312 chlorpyrifos labels (i.e., registered products), including 83 "Special Local Needs" registrations. Section 24 (c) of FIFRA grants states the authority to identify a "Special Local Needs" to address an existing or imminent pest problem. The biological evaluation indicated there are forty chlorpyrifos Special Local Needs registrations for California, Idaho, Oregon, and Washington. However, the specifics information on what the 24(c) registrations authorize was not provided. Six registrants produced “manufacturing use products” to be formulated into “end use products”. However, EPA indicated that it lacks a database that is 100% accurate in terms of providing all current labels for existing product uses (EPA 2008e). Registrants producing chlorpyrifos have end use products for agricultural uses. Many of the end use product registrations by smaller registrants are for golf courses, residential containerized ant baits, industrial plants, and termiticide uses.

NMFS is unaware of any chlorpyrifos products approved for use on crops that contain more than one active ingredient. Chlorpyrifos is commonly formulated with pyrethroids for indoor uses in plants, warehouses, and ships, etc. One mosquito adulticide (a compound that kills the adult life phase of the target pest) product also contains permethrin. Several of the granular golf course and road median turf products are formulated with fertilizers and herbicides such as trifluralin and benfluralin. One product has dichlorvos and may be used on ornamentals in road medians, golf courses, and industrial plant surfaces. One product for indoor greenhouse use contains cyfluthrin. A wood preservative for “finished” wood has an anti-mildew agent. Cattle ear tags impregnated with chlorpyrifos are used to kill cattle pests. These ear tags may also be impregnated with diazinon, cypermethrin, or permethrin.

There are registered uses of chlorpyrifos products throughout the freshwater range of threatened and endangered salmonids in the West Coast.

Usage Information.
Chlorpyrifos is one of the most widely used organophosphate insecticides in the U.S. From 1997-1998, about 21 to 24 million pounds (lbs) of the a.i. chlorpyrifos were used annually for 8 million acres treated in the U.S. About 10 million lbs and 11 million lbs are applied annually in agricultural and non-agricultural settings, respectively. The leading agricultural uses are on corn and fruit trees. The largest agricultural market for chlorpyrifos in terms of total lbs a.i. is corn (5.5 million). The largest non-agricultural market in terms of total lbs of a.i. applied were pest control operators for termite control (5 million) and turf (2.5 million). Dow AgroSciences, one of the registrants of chlorpyrifos products, indicated that the average use of chlorpyrifos dropped from over 15 million lbs per year for the period of 1998-2002 to less than 9 million lbs per year for the period of 1993-2006 (EPA 2008a). The reduction in use during this period was primarily attributed to significant reductions in non-crop uses. In 2000, registrants agreed to cancel or phase out most residential uses (EPA 2004a). The most recent statistics available indicate approximately 1.8 to 2.0 million lbs of chlorpyrifos is applied to approximately 36,000 – 40,000 acres per year in California for agricultural uses (CDPR 2004; CDPR 2005; CDPR 2006). Similar statistics were not available for the other states where listed salmon and steelhead are distributed.

Examples of Registered Uses.
Agricultural Uses. Chlorpyrifos is used on a myriad of crops. Examples of crops currently proposed for continued chlorpyrifos use and which are grown in areas with Pacific salmon and steelhead include alfalfa, almonds, apples, asparagus, broccoli, cabbage, carrots (grown for seed only), cauliflower, cherries, citrus, corn, cotton, figs, filberts, grapes, grass seed, nectarines, onions, pears, peaches, pecans, peppermint, plums and prunes, radishes, snap beans (seed treatment), sorghum, spearmint, strawberries, sugar beets, sunflowers, sweet potatoes, turnips, other vegetables, walnuts, wheat, pulp wood, and Christmas trees (nurseries and plantations).

Non-agricultural Uses. Chlorpyrifos was formerly registered for various indoor and outdoor uses in and around residential areas. EPA indicated that some of these were cancelled (EPA 2003). Indoor uses that remain are residential use of containerized baits, and use in ship holds, railroad boxcars, industrial plants, manufacturing plants, and food processing plants. Outdoor residential uses include adult mosquito control, fire ant control, use on golf courses, pulpwood production, nursery and green house uses, animal premises, cattle ear tags, sod farms, industrial plants, road median strips, and non-structural wood treatments such as poles and fence posts. Earlier, EPA stated the use of chlorpyrifos products for structural termite control may be prohibited after December 31, 2005 (EPA 2003). The IRED indicates pre-construction termiticide use may be allowed beyond 2005 if data are submitted that show that residential post application risks from this use are not a concern (EPA 2004a). Examples of existing uses of chlorpyrifos that can be applied to non-crop areas include cattle ear tags, golf course turf, containerized ant baits, spot and/or crack and crevice treatments in industrial buildings, and mosquito adulticides (EPA 2008e).

Examples of Registered Formulation Types. Chlorpyrifos formulations include liquid emulsifiable concentrates, granular, wettable powder, dry flowable, pressurized liquids, dusts, ready-to-use solutions, microencapsulated material, pellets/tablets, soluble concentrates and impregnated materials (cattle ear tags).

Examples of Methods and Rates of Application.

Methods. Examples of approved application methods include: aerial applications, chemigation, groundboom, tractor-drawn granular spreader, airblast sprayer, low and high pressure hand wands, hydraulic hand-held sprayer, shaker can, belly grinder, large tank sprayer, compressed air sprayer, hose-end sprayer, aerosol sprayer, hand and ear tags.

Chlorpyrifos may be applied to plants foliar surfaces, bark, seed and can be soil-incorporated or applied in broadcast treatments.

Rates. Maximum application rates found range from 0.5 lb a.i./ acre to 6 lb a.i./ acre for existing product labels. New product labels required following completion of the RED in 2006 include the following changes to maximum single application rates:

- Golf course turf: 1 lb/Acre
- Citrus crops: 6 lbs/Acre in Fresno, Tulare, Kern, Kings, and Madera counties in California and 4 lbs/Acre in all other areas
- Tobacco: 2 lbs/Acre (liquid formulations), 3 lbs/acre (granular formulations)
- Corn: 1 lb/Acre

Application timing is dependent on use, but may occur throughout the year.

Metabolites and Degradates.
The major degradate of chlorpyrifos in the environment under most conditions is 3,5,6-trichloro-2-pyridinol (TCP). TCP appears to be more persistent than chlorpyrifos (substantial amounts remain 365 days post-application) and it exhibits much lower soil/water partitioning than chlorpyrifos. Consequently, substantial amounts of TCP are available for runoff for longer periods than chlorpyrifos. TCP is moderately to slightly toxic to freshwater fish and invertebrate species. The degradate is considerably less toxic to fish and invertebrates than chlorpyrifos. Chlorpyrifos may also oxidize to its active metabolite chlorpyrifos-oxon, a more toxic compound than chlorpyrifos.

Diazinon

The National Pesticide Information Retrieval System (http://ppis.ceris.purdue.edu/htbin/epachem.com) suggests that there are currently nine registrants that have active registrations of 16 pesticide products containing diazinon. Diazinon is an organophosphate insecticide, acaricide and nematicide used to control a variety of organisms. It was first registered in 1956 as an insecticide for use on fruit, vegetables, and forage and field crops. Diazinon has veterinary uses for fleas and ticks. Diazinon has also been used for control of household insects, grubs, nematodes in turf, seed treatments, and fly control. As of March 29, 1988, diazinon uses on golf courses and sod farms were canceled due to numerous bird kills.

Pure diazinon is a colorless oil which is formulated into “stabilized” technical diazinon. Technical diazinon (> 90% pure) is an amber to brown liquid with a boiling point of 83-84ºC. Technical diazinon is practically insoluble in water (40 parts per million at 20 ºC). Although technical diazinon is completely miscible in acetone, benzene, dichloromethane, ethanol, 1-octanol, toluene, and xylene, it is soluble in petroleum oils.

Usage Information. Based on available usage information from 1987 through 1997, total annual domestic usage of diazinon is over 13 million lbs a.i./year. Usage during that period was for outdoor residential uses by homeowners (39%), lawn care operators (19%), pest control operators (11%), and agriculture (31%). About four million lbs of the a.i. diazinon are used annually on agricultural sites (EPA 2002b). Use is highest on almonds and stone fruits. About 69% was used in and around residential and associated areas. There are multiple formulations containing diazinon currently registered, i.e., approximately 430 (EPA 2002b). Diazinon is used widely throughout the U.S. The states of California, Florida, and Texas have the highest usage of diazinon.

According to MANA, diazinon use nationwide has dropped by more than 90% following a December 2000 Memorandum of Agreement (MOA) to phase out and cancel all residential uses of diazinon by September 2003 (EPA 2008a). The MOA also canceled 20 existing uses on food crops and the total use of diazinon following implementation of label changes in the IRED is less than 750,000 lbs annually (EPA 2008a). The most recent statistics available indicate approximately 400,000 to 500,000 lbs of diazinon is applied to approximately 400,000 – 500,000 acres per year in California for agricultural uses (CDPR 2004; CDPR 2005; CDPR 2006). Similar statistics were not available for the other states where listed salmon and steelhead are distributed.

Examples of Registered Uses.
Agriculture. Registered uses of diazinon include food crop sites for almonds, apples, apricots, bananas*, beets (red, table), blackberries, blueberries, carrots, celery*, cherries, cranberries, cucumbers*, endive (escarole), figs, filberts, ginseng, grapes, kale, lettuce, loganberries, melons, nectarines, onions, parsley*, parsnips*, peaches, pears, peppers, pineapples, plums, Irish potatoes*, prunes, radishes, raspberries, rutabaga, squash (winter and summer)*, spinach, strawberry, sweet potatoes*, Swiss chard, tomato, turnip,(roots and tops)*, vegetables (Brassica leafy group), and watercress. An asterisk (*) denotes only 24(c) Special Local Need registrations.

Non-agriculture. Outdoor residential uses of diazinon by homeowners, lawn care operators, and pest control operators were phased out or canceled as of December 31, 2004 (EPA 2008a). It is legal for homeowners to continue to use existing stocks that were purchased prior to the phase out. All new diazinon product labels are for agricultural crops. The only exceptions include: 1) tree trunk wraps for commercial agriculture and horticulture, 2) outdoor applications to ornamental plants, and 3) cattle ear tags (EPA 2002b; EPA 2008e).

Examples of Registered Formulation Types. Formulation types include dusts, emulsifiable concentrates, granules, impregnated materials, liquid, microencapsulated, pressurized sprays, soluble concentrates, flowable concentrates, wettable powders, ready-to-use solutions, and seed dressings.

Examples of approved Methods and Rates of Application.
Equipment. Liquid diazinon (liquid formulations or formulated from wettable powder) can be applied by airblast sprayer, aircraft, airless sprayer, backpack sprayer, backpack/low pressure hand wand equipment, chemigation, handheld spray equipment, hydraulic sprayer with hand gun, groundboom sprayer high pressure hand wand, and paint brush. Aerial application to food crops is only authorized for lettuce in California. According to the diazinon IRED (EPA 2002b), all granular uses on food crops are canceled or phased out by 2008. The only exceptions are two current Section 24(c)

registrations held by Oregon and Washington for control of the cranberry girdler (EPA 2008a).

Method and Rate. Diazinon can be applied as a foliar or soil treatment via aerial application, air blast, ground boom, tractor and push-type granular spreaders and hand-held spray equipment. Rates vary according to method and type of application and pest. Typical vegetable crop rates range from foliar application of 0.5 lb a.i./acre to soil incorporate rates of up to 4 lb a.i./acre; granular application up to 4 lb a.i./acre; and fruit and nut trees with 1 to 3 lb a.i./acre. According to the current labels, diazinon of the 14-G, 50 WP, and 48 EC formulations is applied foliarly or as a soil treatment using ground or aerial equipment followed by incorporation in some uses.

Timing. The timing of application is dependent on use, but may occur throughout the year.

Metabolites and Degradates. Diazinon is moderately persistent and mobile in the environment. Diazinon appears to degrade by hydrolysis in water and by photolysis and microbial metabolism. It also dissipates by volatilization from impervious surfaces. Diazinon degrades by hydrolysis at all pHs tested.

Hydrolysis is rapid under acidic condition with a half-life of 12 days at pH 5. Under neutral and alkaline conditions, diazinon hydrolyzed more slowly with abiotic hydrolysis half-lives of 138 days at pH 7 and 77 days at pH 9. Diazinon is stable to photolysis in water. However, diazinon was shown to degrade with a half-life of less than two days on soil. This indicates that photodegradation may be important under certain circumstances.

Diazinon is activated internally to become diazoxon, a more potent cholinesterase inhibitor than diazinon (Tsuda 1997). Diazinon and its degradates may occur in both groundwater and surface waters. Diazinon is moderately mobile and persistent. Laboratory data indicate diazinon will not persist in acidic water. However, in neutral and alkaline waters residues may be quite persistent. Oxypyrimidine is the main soil and water degradate. Diazoxon, a toxic degradate, rapidly hydrolyzes to oxypyrimidine. Based on a 1997 killifish study, the toxicity of diazoxon is 20 times more toxic than diazinon (Tsuda 1997).

Malathion

The National Pesticide Information Retrieval System (http://ppis.ceris.purdue.edu/htbin/epachem.com) suggests that there are currently 38 registrants with active registrations for more than 100 products containing malathion. Malathion is a broad spectrum organophosphate insecticide first registered in 1956. It is widely used in agriculture for various food and feed crops, homeowner outdoor uses, ornamental nursery stock, building perimeters, pastures and rangeland, and regional pest eradication programs. Malathion is applied to foliage to kill sucking and chewing insects that damage crops.

Malathion in its liquid state is a clear amber color with a boiling point of 156-157ºC. Malathion is soluble in water and is readily soluble in most alcohols, esters, aromatic solvents, and ketones. Malathion is only slightly soluble in aliphatic hydrocarbons. This compound hydrolyzes rapidly and has a half-life of 6.21 days under neutral and alkaline conditions. Malathion remains hydrolytically stable with a half-life of 107 days in a buffered acidic environment. Malathion is persistent in the environment with a half-life of up to 11 days.

Usage Information

In 2000, about 11-13 million lbs of malathion were used annually in the U.S. As of July 2006, 15 million lbs were used annually (EPA 2006). Percentage of malathion use include: U.S. Department of Agriculture (USDA) - 59 to 61%; general agriculture -16 to 20%; public health 8 to 15%; and home and garden use – 10%. These use percentages likely vary with fluctuations in pest pressure or concerns for public health such as mosquito control following natural disasters, e.g., a hurricane or a major flood.

About 10.2 million lbs a.i. are applied through the USDA Boll Weevil Eradication Program (BWEP). Additionally, 1.5 million lbs are applied to agricultural crops, and 300,000 lbs are applied as post harvest grain treatment to corn, wheat, and oats. About 500,000 lbs a.i. is used on non-agricultural sites, such as around buildings, roads, and ditches. About 1.5 million lbs are applied in quarantine programs and public health programs that target the adult life phase of pest insects. One million lbs are used in the residential/home owner market.

The most recent statistics available indicate approximately 400,000 to 500,000 lbs of malathion is applied to approximately 200,000 – 250,000 acres per year in California for agricultural uses (CDPR 2004; CDPR 2005; CDPR 2006). Similar statistics were not available for the other states where listed salmon and steelhead are distributed.

Examples of Registered Uses

Agriculture

Malathion is registered for food and feed crops such as alfalfa; apricot; asparagus; avocado; barley; bean (succulent and dry); beets (garden); blackberry; blueberry; boysenberry; broccoli; broccoli raab; Brussels sprout; cabbage (including Chinese); carrot; cauliflower; celery; cherry; chestnut; clover; collards; corn (field, sweet, and pop); cotton; currant; dandelion; date; dewberry; eggplant; endive; escarole; potato; fig; garlic; grape; grapefruit; guava; hay grass; hops; horseradish; kale; kohlrabi; kumquat; leek; lemon; lespedeza; lettuce (head and leaf); lime; loganberry; lupine; macadamia nut; mango; melon; mint; mushroom; mustard greens; nectarines; oats; okra; onion; orange; papaya; parsley; parsley; parsnip; passion fruit; pasture and rangeland; pea; peach; pear; pecan; pepper; pineapple; pumpkin; radish; raspberry; rice; rutabaga; rye; salsify; shallot; sorghum; spinach; squash; strawberry; sweet potato; Swiss chard; tangelo; tangerine; tomato (including tomatillo); vetch; walnut; watercress; watermelon; what (spring and winter); wild rice; and yam; indoor stored commodity treatment and empty storage facilities for barley, corn, oats, rye, and wheat; and uncultivated areas at agricultural sites.

Non-agriculture
Malathion is approved for use on a variety of non-agricultural use sites including: Christmas tree plantations; cull piles; drainage systems; fence rows and hedge rows; greenhouse; the perimeter of households and domestic dwellings; intermittently flooded areas; outdoor building structures; rights of way/fencerows; uncultivated areas/soil; shade trees; ornamentals (trees, herbaceous plants, non-flowering plants, woody shrubs and vines); pine seed orchards; outdoor solid waste containers; outdoor solid waste sites; swamps/marshes/stagnant water; wide area public health uses (EPA 2006b).

Regional Pest Eradication Programs.
This category includes the BWEP, Medfly control, and Mosquito control programs.

Pharmaceutical Malathion.
There is a pharmaceutical use of malathion as a pediculicide for the treatment of head lice and their ova on humans, which is regulated by the Food and Drug Administration.

Examples of Registered Formulations and Types.

Malathion is formulated as an emulsifiable concentrate, dust, wettable powder, ready-to-use liquid, and as a pressurized liquid. The emulsifiable concentrate and ready-to-use formulations may contain up to 82% and 96.8% a.i., respectively. Several of the 96.8 a.i. ready-to-use liquids are intended for ultra-low volume application with the use of aerial or ground equipment. Malathion is typically applied as multiple foliar treatments as needed to control various pest species.

Examples of Methods and Rates of Application.

Application Rate Ranges

• General Agriculture	0.175 – 7.5 lb a.i./acre
• Christmas tree plantations	3.2 lb a.i. / acre
• Cull piles	6.857 lb/1000 ft^2 (299 lb a.i./acre)
• Household perimeter	0.2439 lb/1000 ft^2 (10.6 lb a.i./acre)
• Intermittently flooded areas	0.5078 lb a.i./ acre
• Building surfaces	0.2057 lb a.i. / 1000 ft^2 (9.0 lb a.i./acre)
• Fence rows/uncultivated areas	0.9281 lb a.i./ acre
• Ornamentals	2.5 lb a.i./ acre
• Outdoor solid waste sites/containers	0.2439 lb/1000 ft^2 (10.6 lb a.i./acre)
• Wide area – public health	0.23 lb a.i./ acre
• Swamps/marshes/stagnant water	0.5075 lb a.i./acre

Application Equipment
Equipment includes aircraft (fixed wing and rotary), duster, fogger, ground boom, irrigation, shaker can, sprayer, and spreader.

Target Organisms
Organisms include ants, aphids, apple mealybug, armyworm, bagworm, beetle, borer, casebearer, blackheadded fireworm, blueberry maggot, cadelle, caterpillars, cattle lice, cherry fruitworm, cockroaches, corn earworm, corn rootworms, cotton fleahopper, cotton leaf perforator, cotton cankerworm, fleahoppers, fleas, flies, fruit flies, fungus gnats, garden webworm, brain borer, grape phylloxera, grasshoppers, green cloverworm, greenbug, groundpearls, hornets, imported cabbage worm, imported currantworm, ked, leafhoppers, leafrollers, leafminer, looper, millipedes, mites, mosquitos (adult, larvae), moths, kermes, mushroom flies, omnivorous leaftier, onion maggot, orange tortrix, orange worms, pear psylla, pecan phylloxera, pepper maggot, pickleworm, pillbugs, pine needle sheathminer, plant bugs, plum curculio, poultry lice, rose chafer, sawflies, scales, scorpions, silver fish sorghum midge, sowbugs, spiders, spittlebugs, springtails, strawberry leafroller, sugar beet root maggot, tadpole shrimp, thrips, ticks, tingids, tomato fruitworm, vetch, bruchid, wasps, weevil, whiteflies, and wild rice worm.

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.

Metabolites and Degradates
Malaoxon and isomalathion are two of multiple degradates resulting from oxidation and isomerization of malathion, respectively. Their presence increases the level of toxicity created by the a.i. malathion.

Malaoxon is the primary metabolite of malathion following biotransformation in invertebrates and vertebrates. Under certain conditions, malaoxon is formed as an environmental degradation product of malathion. Malaoxon is a neuroactive agent with a higher acute toxicity than malathion. When malathion degrades, malaoxon is created in small quantities. Malaoxon can occur via oxidation during water treatment process or through reaction with the ambient air. When administered to animals directly, malaoxon is a more potent cholinesterase inhibitor than malathion. EPA has limited data on malaoxon, the oxon analogue, and the other impurities/degradates of malathion.

Isomalathion is a known impurity present as a component of malathion during the manufacturing process. The current upper certified limit of isomalathion in the technical product is 0.2 % by weight. Data submitted by the technical registrant indicate that the presence of isomalathion, as a percent of the product, increases when malathion is stored under high temperatures, for long periods of time, or a combination of these two variables. Current guideline data indicate that malathion is stable for one year at 25ºC (77ºF). Under these conditions, the percent of isomalathion remains below the certified limit. EPA has limited toxicity data on isomalathion alone or in products containing elevated levels of isomalathion. The limited data available suggest that isomalathion increases the toxicity of malathion (Anderson et al. 2007).

Species

EPA's BEs considered effects of chlorpyrifos, diazinon, and malathion to 26 species of listed Pacific salmonids and their designated critical habitat. EPA determined that chlorpyrifos, diazinon, and malathion may affect and are likely to adversely affect most of these species. Exceptions follow:

EPA concluded that the registration of chlorpyrifos products would have no effect on Columbia River Chum salmon and Ozette Lake Sockeye salmon. Additionally, EPA concluded the registratration of chlorpyrifos products may affect, but is not likely to adversely affect California Coastal Chinook salmon, Central California Coho salmon, Hood Canal Summer-run Chinook salmon, Snake River Sockeye salmon, Northern California steelhead, and Central California Coast steelhead.

EPA concluded the registration of diazinon products may affect, but is not likely to adversely affect Hood Canal Summer-run Chinook salmon and Ozette Lake Sockeye salmon.

EPA concluded the registration of malathion products would have no effect on California Coastal Chinook salmon and Northern California Steelhead.

Even though EPA has determined that its action in registering pesticides containing the three active ingredients are not likely to adversely affect certain ESUs 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. When an action agency concludes that its action will not affect any listed species or critical habitat, then no section 7 consultation is necessary. 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. However, once formal consultation is triggered, NMFS evaluates the Federal action and its impacts to all listed species and critical habitat. Therefore, in this Opinion, NMFS will analyze the impacts to all ESUs of Pacific salmonids present in the action area, including those ESUs identified by EPA as being unaffected or not likely to be adversely affected and the two ESUs listed after EPA provided its effect determination.

Approach to this Assessment

NMFS approaches its section 7 analyses through a series of steps. The first step identifies those aspects of proposed actions that are likely to have direct and indirect effect on the physical, chemical, and biotic environment of an action area. As part of this step, we identify the spatial extent of these direct and indirect effects, including changes in that spatial extent over time. The result of this step represents the action area for the consultation. The second step of our analyses identifies the listed resources that are likely to co-occur with these effects in space and time and the nature of that co-occurrence (these represent our *exposure analyses*). In this step of our analyses, we try to identify the number, age (or life stage), gender, and life-histories of the individuals that are likely to be exposed to an action's effects and the populations or subpopulations those individuals represent. Once we identify which listed resources are likely to be exposed to an action's effects and the nature of that exposure, 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 in the *Risk Characterization* section. 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 (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 populations live, die, grow, mature, migrate, and reproduce (or fail to do so). Our adverse modification or destruction of designated critical habitat determinations will be based on an action's effects on reductions in the conservation value of critical habitat. These reductions in the conservation value of critical habitat can be in the quantity, quality, or availability of physical, chemical, or biotic resources in the habitat [i.e., primary constituent elements (PCEs)].

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. The analyses then translate individual-level effects to population level consequences. The analyses conclude by determining the consequences of those population-level risks to the species those populations comprise.

We measure risks to listed individuals using the individual's "fitness" which is measured using 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 responses to an action's effects on the environment (which we identify during our response analyses) are likely to have consequences to an individual's fitness.

Reductions in abundance, reproduction rates, or growth rates (or increase variance in one or more of these rates) of individuals 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 reduction in fitness, we would not expect the action to have adverse consequences on the viability of the populations those individuals represent or the species those populations comprise (Anderson et al. 2006; Mills and Beatty 1979; Stearns 1982). If we conclude that listed plants or animals are *not* likely to experience reduction in their fitness, we would conclude our assessment.

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. In this step of our analyses, we use the species' status (established in the *Status of Listed Resources* section of this Opinion) as our point of reference.

Evidence Available for the Consultation

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

- EPA's BEs, REDs, 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
- 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 several hundred pesticide formulations (pesticide products) containing the active ingredients chlorpyrifos, diazinon, and malathion through its authority to register pesticides under the 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 has initiated consultation on registration of chlorpyrifos, diazinon, and malathion and diazinon for 26 listed ESUs of Pacific salmonids. Since EPA has initiated consultation, NMFS has listed one additional coho ESU and one additional steelhead distinct population segment (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.

NMFS' evaluation proceeds by asking if endangered species, threatened species, and designated critical habitat are likely to be exposed to the direct and indirect effects of the proposed actions (Figure 1). If those listed resources are not likely to be exposed to these activities, we would conclude that EPA's actions are 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. The independent population is the fundamental unit of evaluation in determining the risk of extinction of salmon in the 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, 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 II 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 et al. 2007, McElhaney et al. 2000).

In determining the effect of an action to populations, we translate individual fitness level consequences to effects on VSP parameters. If populations are likely to be adversely affected, i.e., VSP parameters, by the stressors of the action, 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.

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 a series of phases- problem formulation, analysis, and risk characterization (EPA 1998). 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 that is applied in the *Effects of the Proposed Action* section.

Problem Formulation

The first phase of the framework is the problem formulation phase. In this phase, we generate conceptual models from our initial evaluation of the relationship between stressors of the action and potential receptors (listed species, habitat). Conceptual models representing these relationships are 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 exposure 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 chemical stressors.

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

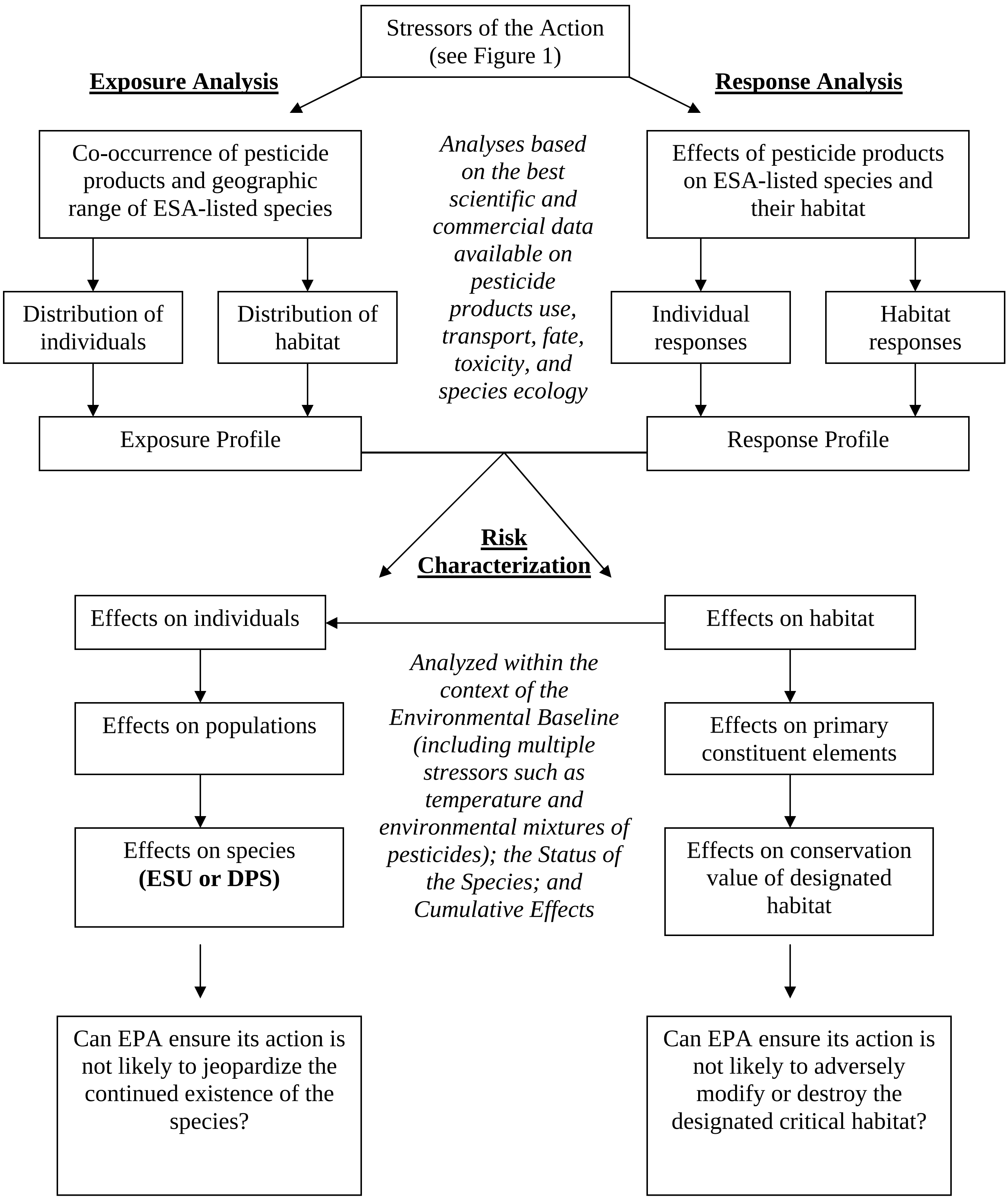
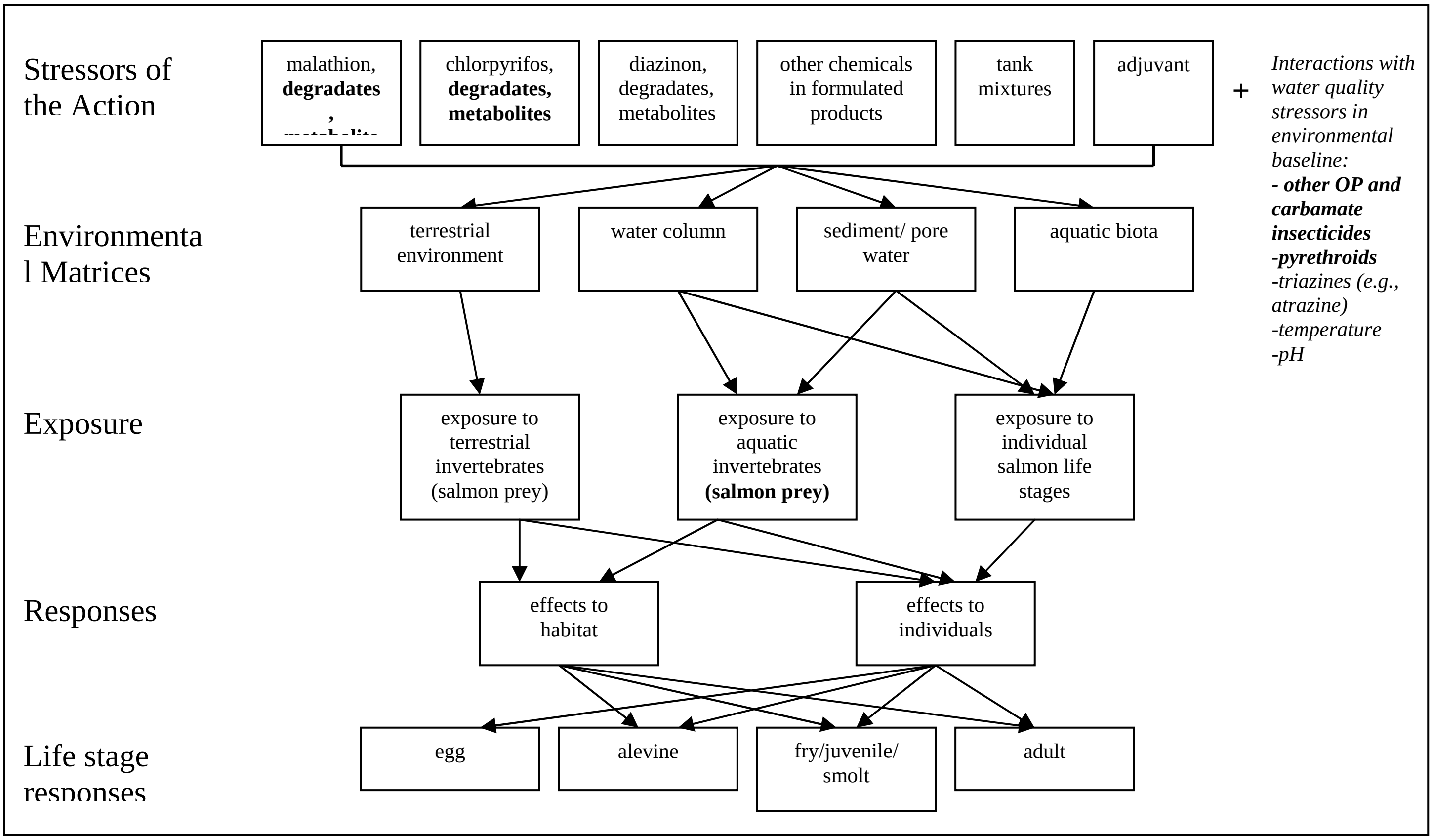


Figure 3. Exposure pathways to malathion, diazinon, and chlorpyrifos and general responses of listed Pacific salmonids and habitat.



Species Risk Hypotheses

We construct risk hypotheses by identifying biological requirements or assessment endpoints (Table 1) for listed resources in the action area. 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 we evaluate in the *Effects of the Proposed Action* section:

1. Exposure to chlorpyrifos, diazinon, and malathion 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).

   e. Reduce olfactory-mediated behaviors resulting in consequences to survival, migration, and reproduction.
2. Exposure to mixtures of chlorpyrifos, diazinon, and malathion can act in combination to increase adverse effects to salmonids and salmonid habitat.
3. Exposure to other stressors of the action including oxon degradates, adjuvants, tank mixtures, and other active and other ingredients in pesticide products containing chlorpyrifos, diazinon, and malathion cause adverse effects to salmonids and their habitat.
4. Exposure to other pesticides present in the action area can act in combination with chlorpyrifos, diazinon, and malathion to increase effects to salmonids and their habitat.
5. Exposure to elevated temperatures can enhance the toxicity of the stressors of the action.

Critical Habitat Risk Hypotheses

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

In risk hypothesis 1, aquatic exposure to chlorpyrifos, diazinon, and malathion can impair a species' nervous system and consequently affect swimming ability of fish. Swimming performance therefore is an assessment endpoint. Measurable changes in swimming speed would be a measure of performance or so called assessment measure. 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 active ingredients. 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 mixture toxicity. With vertebrates (fish and mammals) and invertebrates, the three active ingredients 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. In Table 1, assessment endpoints are identified for particular life stages. We assess the likelihood of these fitness level consequences occurring from exposure to the actions. Exposure estimates for our listed resources are derived from reviewing exposure data. This evaluation is conducted in the exposure analysis (Figure 2). We focused on the following physiological systems:

chemoreception, locomotion, feeding, reproduction, and growth. We did not locate any information on the remaining systems so they were not specifically addressed in our analysis.

The problem formulation phase concludes with the development of an analysis plan. The plan identifies how exposure will be assessed and which assessment endpoints will be evaluated. Therefore, the analysis plan is a road map for conducting the next phase of the assessment, called the analysis phase.

Figure 4. Physiological systems potentially affected by acetylchloinesterase inhibition

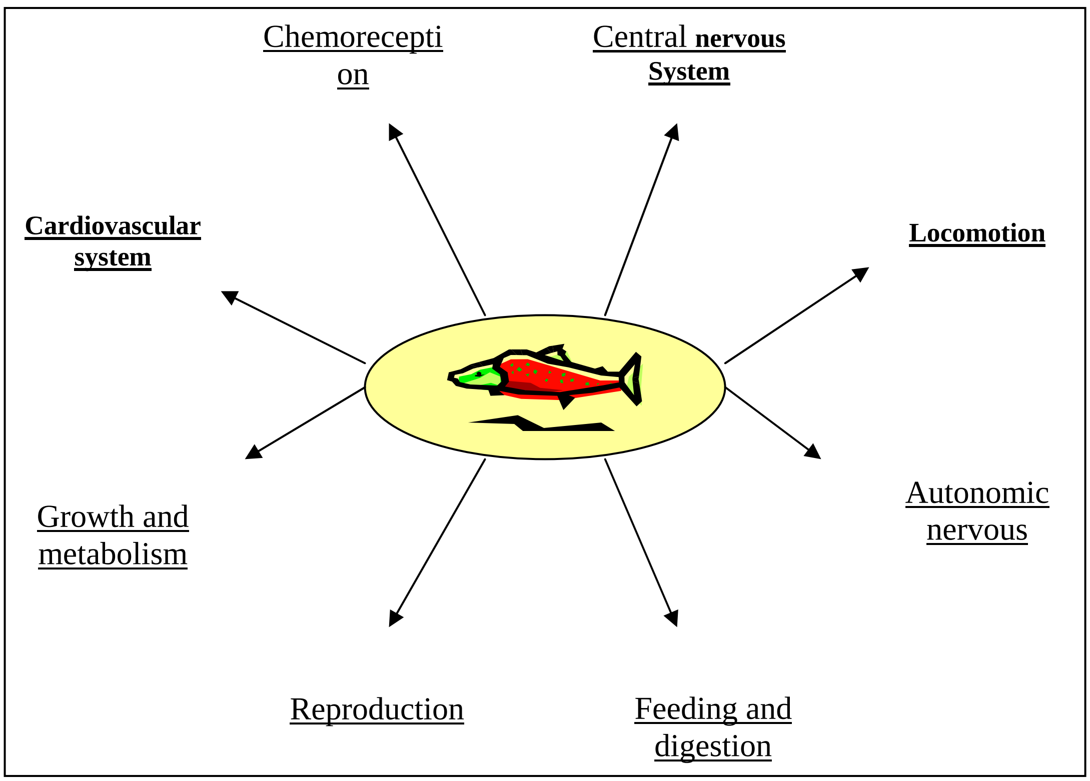


Table 1. Examples of salmonid lifestage responses to acetylcholinesterase inhibiting insecticides

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
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)
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) weight, length stomach contents, weight, length, starvation 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 stomach contents behavioral assays numbers of adult returns, behavioral assays numbers of eggs fertilized stomach contents histological assessment of ovaries/testis electro-olfactogram measurements 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 determine the type, likelihood, magnitude, and frequency of adverse responses resulting from predicted exposure. We evaluate species information and pesticide information to determine when, where, and at what concentrations listed salmonids and their habitat may be exposed. 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. We conduct a separate analysis to determine the potential for adverse modification or destruction of designated critical habitat.

To conclude consultation, cumulative effects are described and the extent to which species and habitat are affected is documented. Given the effects of the action, the condition of the *Status of Listed Resources*, and the *Environmental Baseline*, NMFS determines whether EPA's pesticide registration actions jeopardize the continued existence of the species. NMFS must also determine whether the action results in adverse modification or destruction of designated critical habitat.

Other Considerations

In this Opinion, we adapted EPA's 1998 framework for conducting ecological risk assessment to focus on ESA-specific considerations (EPA 1998). We evaluated lines of evidence constructed as species-specific risk hypotheses to ensure relevant endpoints were addressed. Ultimately, the analysis weighs each line of evidence by evaluating the best commercial and scientific data available that pertain to a given risk hypothesis. Overall, the analysis is a qualitative approach that uses some quantitative tools to provide examples of potential risks to listed salmonids and their habitat. Several other methods and tools currently exist for addressing contaminant-induced risk to the environment. Hazard-based assessments, probabilistic risk assessment techniques, combinations of the two, and deterministic approaches such as screening level assessments have been applied to questions of risk related to human health and the environment.

We considered the use of probabilistic risk assessment techniques for addressing risk at population and species (ESU and DPS) scales to the stressors of the action. However, we encountered significant limitations in available data regarding toxicity information, species information, and pesticide monitoring data. Examples of these limitations include issues with data collection, lack of data, non-normal distributions of data, and quality assurance and quality control. When these types of data limitations are coupled with the inherent complexity of EPA's proposed actions (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. At this time, the best available data do not support such an analysis and conclusions from such an analysis would be highly speculative.

We note that several ecological risk assessments have been conducted for a variety of reasons using probabilistic approaches to address aspects of risk to aquatic communities from chlorpyrifos, diazinon, and malathion (Geisy et al. 1999; Giddings et al. 2000; Hall 2002a; Hall 2002b; Hall and Anderson 2000). There is utility in some of the information within these assessments. A more detailed discussion of these assessments is presented in the *Effects of the Proposed Action* section. Risk assessments are conducted for an array of purposes to address specific management goals. The problem formulation phase of an assessment generally outlines the questions being posed and how the assessment will address them. Many of the previous assessments with the three active ingredients do not align well with the goals of an ecological risk assessment conducted under the ESA. For example, we located no probabilistic assessments that addressed risk to VSP parameters at the population scale, or addressed the probability of a species surviving and recovering from the stressors of the proposed action.

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 habitats 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 active ingredients and the effects on listed Pacific salmonids under NMFS' jurisdiction and associated designated critical habitats 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 habitats 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).

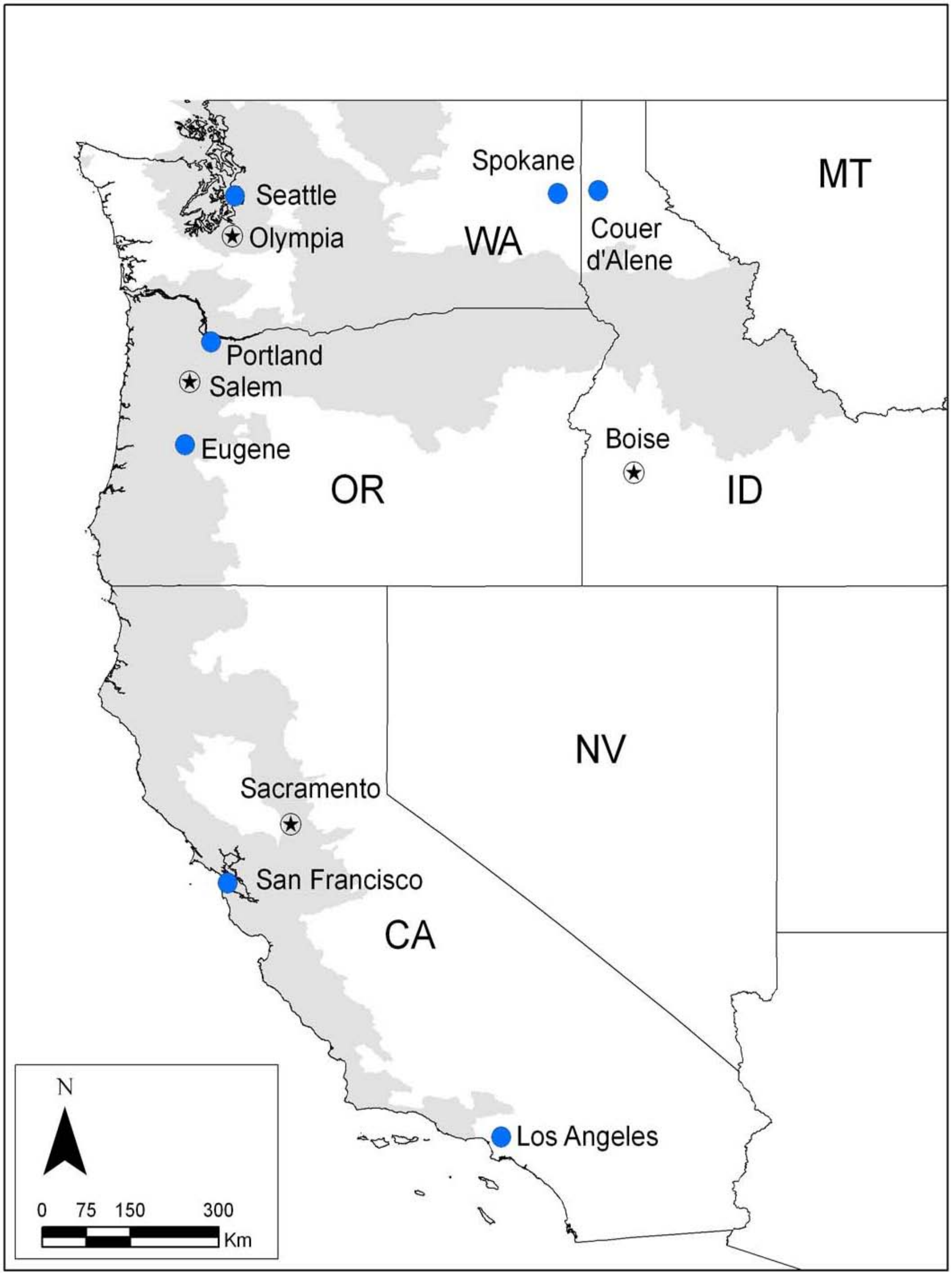


Figure 5. Map showing extent of inland action area with the range of all ESU and DPS boundaries for Endangered Species Act listed salmonids highlighted in gray.

Chlorpyrifos, diazinon, and malathion are the first 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 remains incomplete until it analyzes the effects of its authorization of pesticide product labels with chlorpyrifos, diazinon, and malathion 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 habitats under NMFS' jurisdiction throughout the U.S. and its territories.

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 chlorpyrifos, diazinon, and malathion - containing products (Table 2). More detailed information on the status of these species and critical habitat can be 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 2. 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 (Good et al. 2005; McClure et al. 2003). 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 2005b) 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 Chinook salmon includes all naturally-spawned coastal Chinook salmon spawning from Redwood Creek south through the Russian River as shown in (Figure 6).

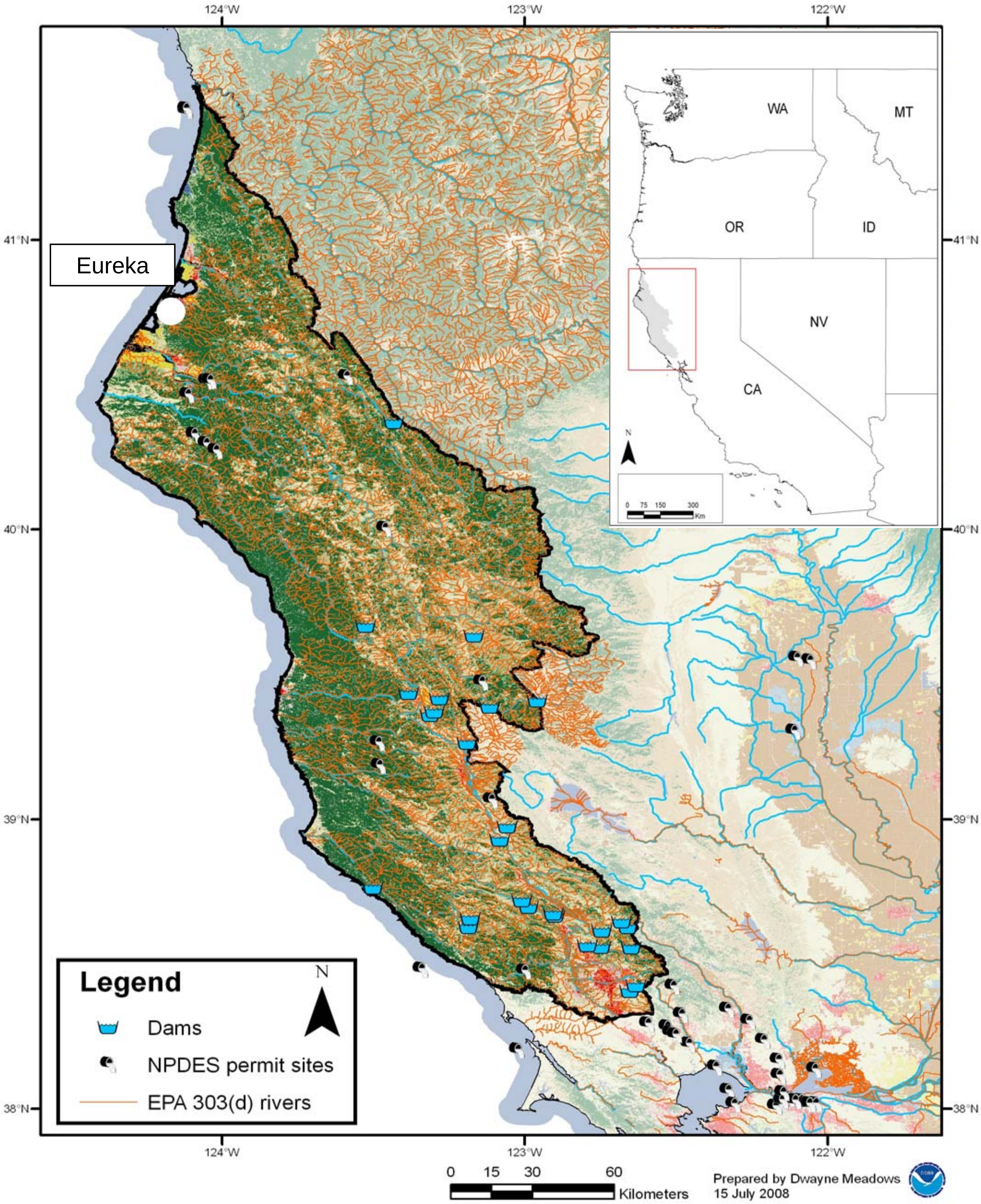


Figure 6. California Coastal Chinook salmon distribution. The Legend for the Land Cover Class categories is found in Figure 7 .

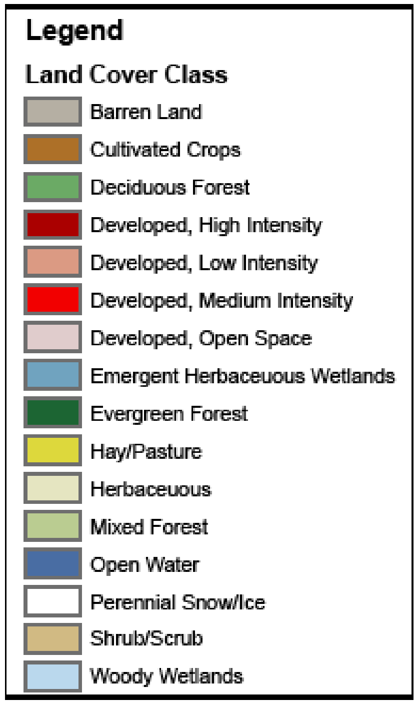


Figure 7. Legend for the Land Cover Class categories found in species distribution maps. Land cover is based on the 2001 National Land Cover Data and classifications. http://www.mrlc.gov/index.php.

California Coastal Chinook salmon are a fall-run, ocean-type fish. Although a spring-run (river-type) component existed historically, it is now considered extinct (Bjorkstedt et al. 2005). Table 3 identifies populations within the California Coastal Chinook salmon ESU, their abundances, and hatchery input.

Table 3. California Coastal Chinook salmon--preliminary population structure, abundances, and hatchery contributions (Good 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

California Coastal 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 California Coastal 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 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 ESUs (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 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 Chinook salmon includes all naturally spawned populations of spring-run Chinook salmon in the Sacramento River and its tributaries in California (Figure 8).
Table **4** identifies populations within the Central Valley spring-run Chinook salmon ESU, their abundances, and hatchery input.

Table 4. Central Valley Spring-Run Chinook salmon--preliminary population structure, abundances, and hatchery contributions (Good 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

Central Valley spring-run 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).

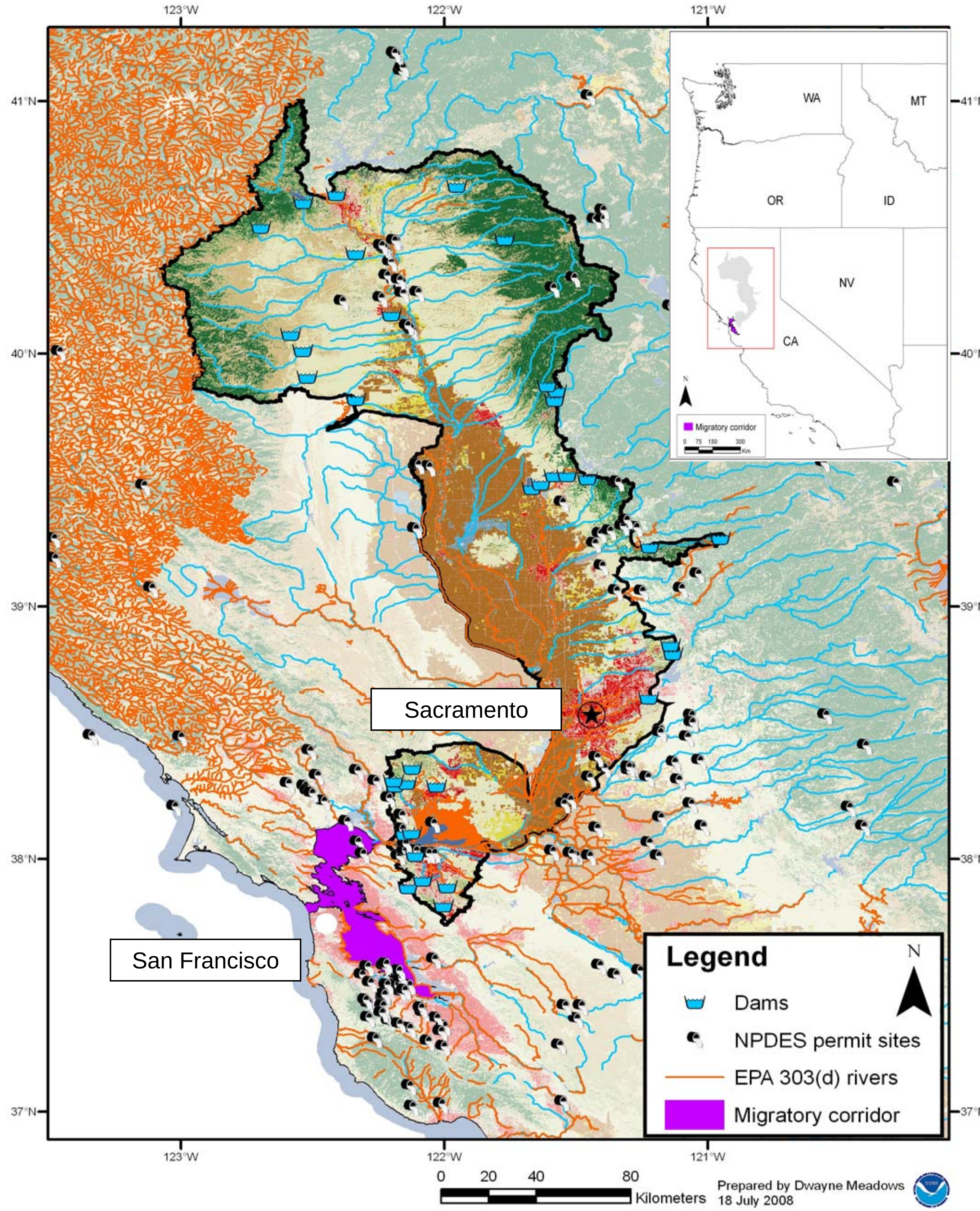


Figure 8. Central Valley Spring-Run Chinook salmon distribution. The Legend for the Land Cover Class categories is found in Figure 7 .

Status and Trends

Central Valley spring-run 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 Central Valley. 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 (Clarke 1929; Rutter 1904; Stone 1874).

The Central Valley 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 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 Central Valley spring-run 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 Central Valley spring-run 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 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. (2006), abundance time series data for Mill, Deer, Butte, and

Big Chico creeks spring-run Chinook salmon (updated through 2001) confirm that population increases 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 Central Valley have long-and short-term lambdas >1, indicating population growth. Central Valley spring-run Chinook salmon have some of the highest population growth rates in the Central Valley. However, population sizes are relatively small compared to fall-run Chinook salmon populations. Finally, 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.

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 Central Valley water (e.g., through use of CALFED EWA and Central Valley 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. 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.

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 5 identifies populations within the LCR Chinook salmon ESU, their abundances, and hatchery input.

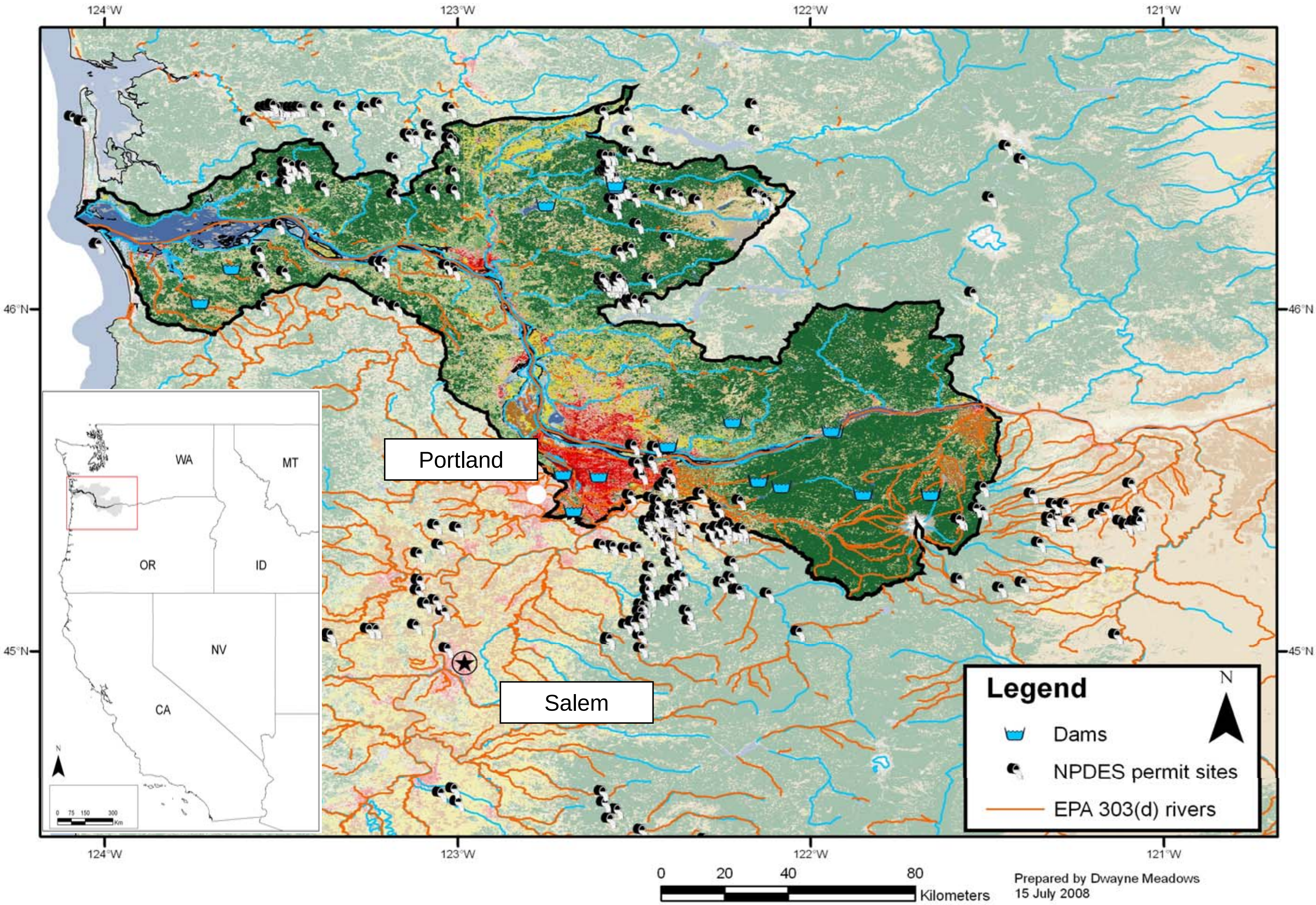


Figure 9. Lower Columbia River Chinook salmon distribution. The Legend for the Land Cover Class categories is found in Figure 7 .

Table 5. Lower Columbia River Chinook salmon - preliminary population structure, abundances, and hatchery contributions (Good 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)**	

Life History
LCR Chinook salmon display three life history types including early fall runs ("tules"), late fall runs ("brights"), 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 subyearling (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. (2006) 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 (WLCRTRT) 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. (2006), 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 lower Columbia River, (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 groupings (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 6). Table 6 identifies populations within the Upper Columbia River Chinook salmon ESU, their abundances, and hatchery input.

Table 6. Upper Columbia River Chinook salmon - preliminary population structure, abundances, and hatchery contributions (Good 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)**		

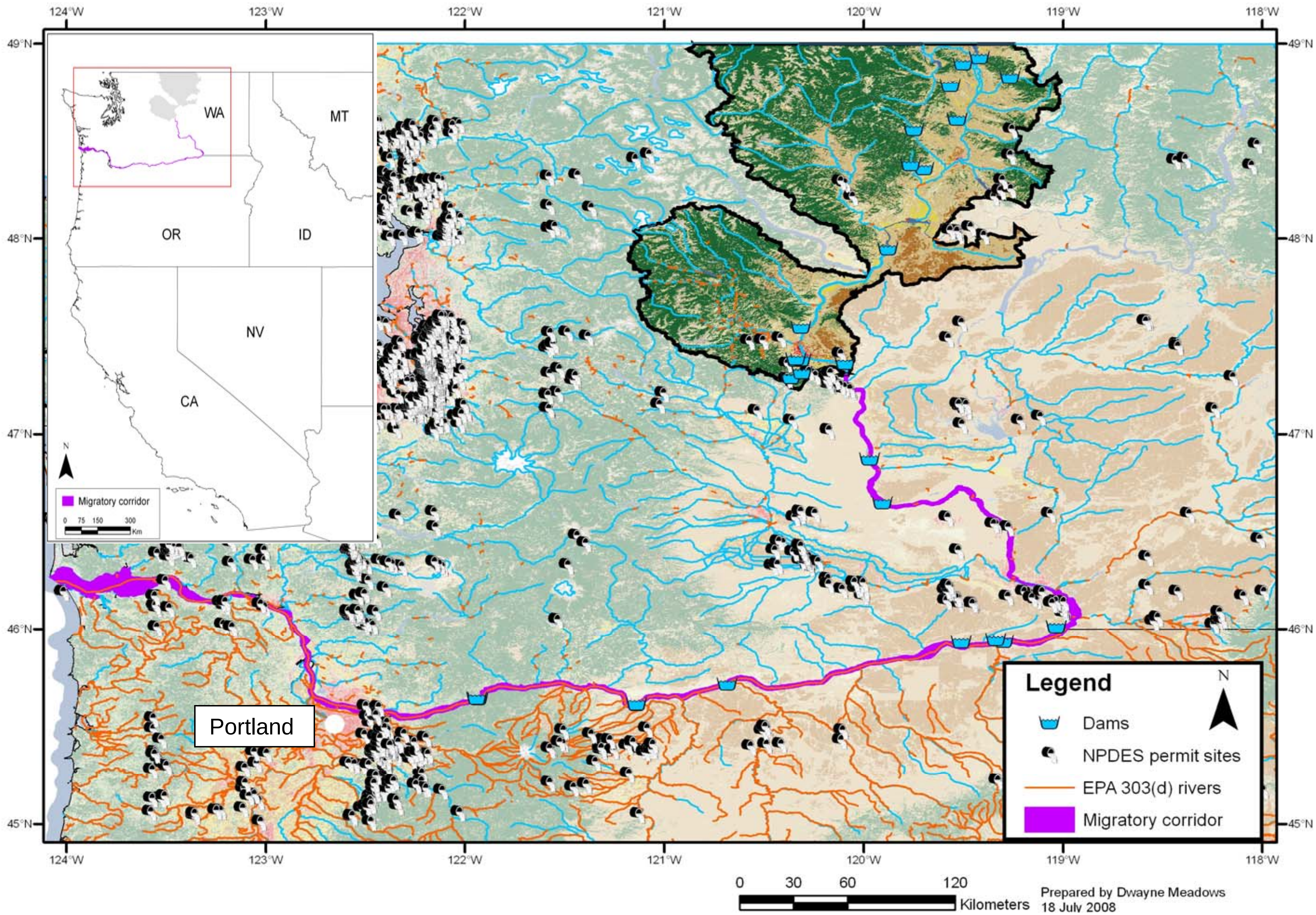


Figure 10. Upper Columbia River Spring-run Chinook salmon distribution. The Legend for the Land Cover Class categories is found in Figure 7 .

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.

In the most recent 5-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 (using the Dennis Model) 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 7). Table 7 identifies populations within the Puget Sound Chinook salmon ESU, their abundances, and hatchery input.

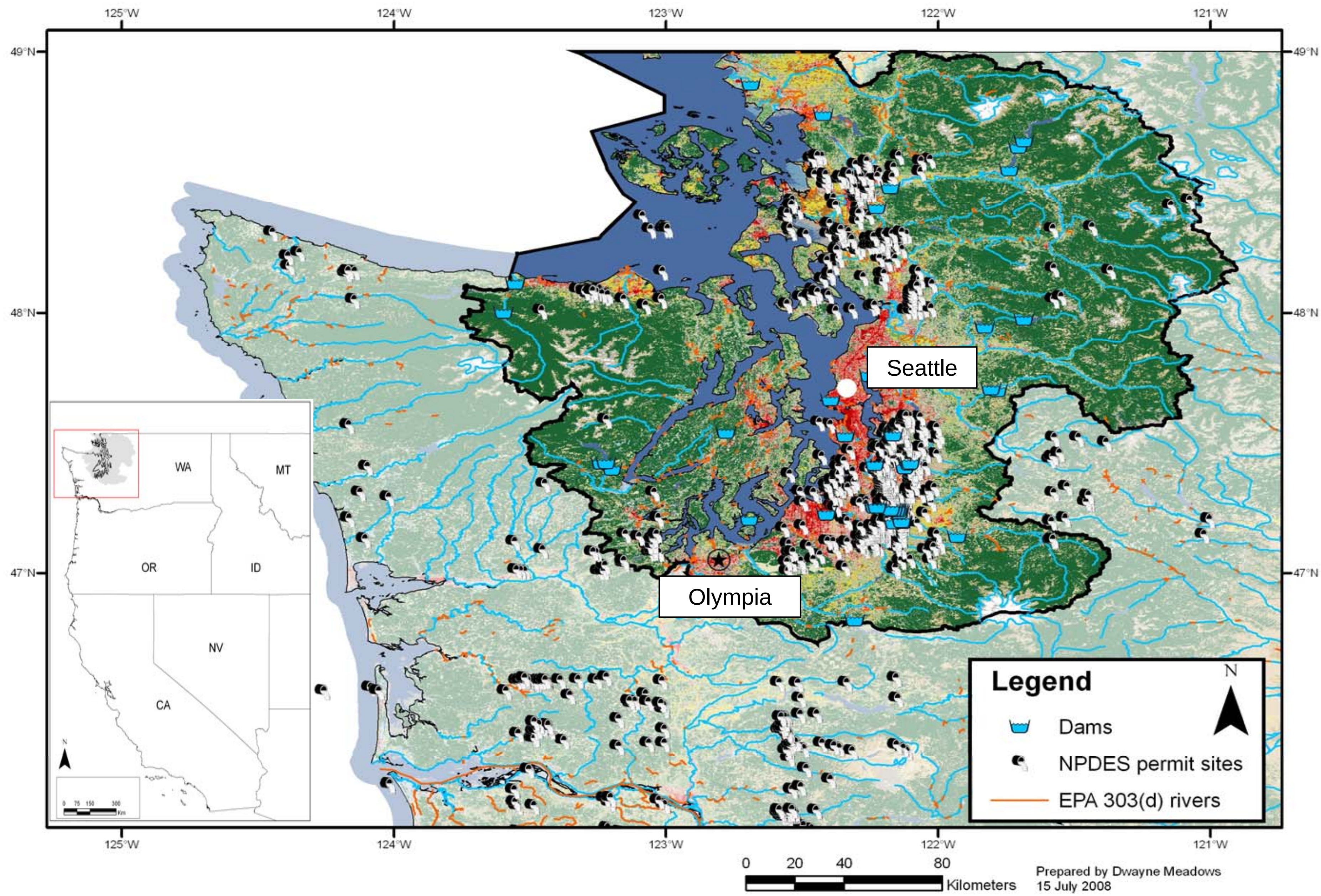


Figure 11. Puget Sound Chinook distribution. The Legend for the Land Cover Class categories is found in Figure 7 .

Table 7. Puget Sound Chinook salmon - preliminary population structure, abundances, and hatchery contributions (Good 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 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 et al. 2005). Nine of the 15 extinct spawning aggregations were early-run type Chinook salmon (Good 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 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 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 (Myers 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 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.

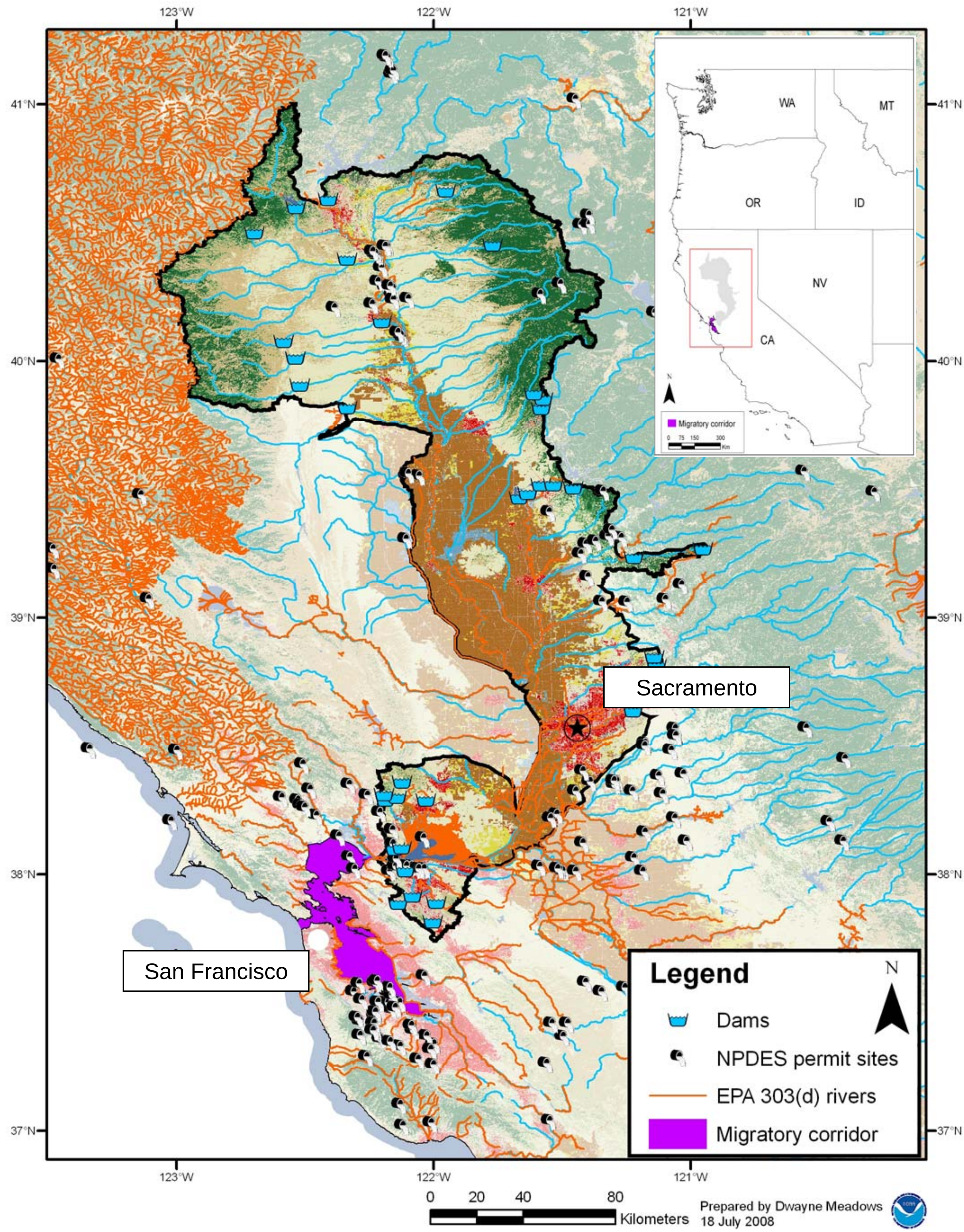


Figure 12. Sacramento River Winter-run Chinook salmon distribution. The Legend for the Land Cover Class categories is found in Figure 7.

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 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 et al. 2005). Most recent estimates indicate that the short-term trend is 0.26, and the population growth rate is less than 1.

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

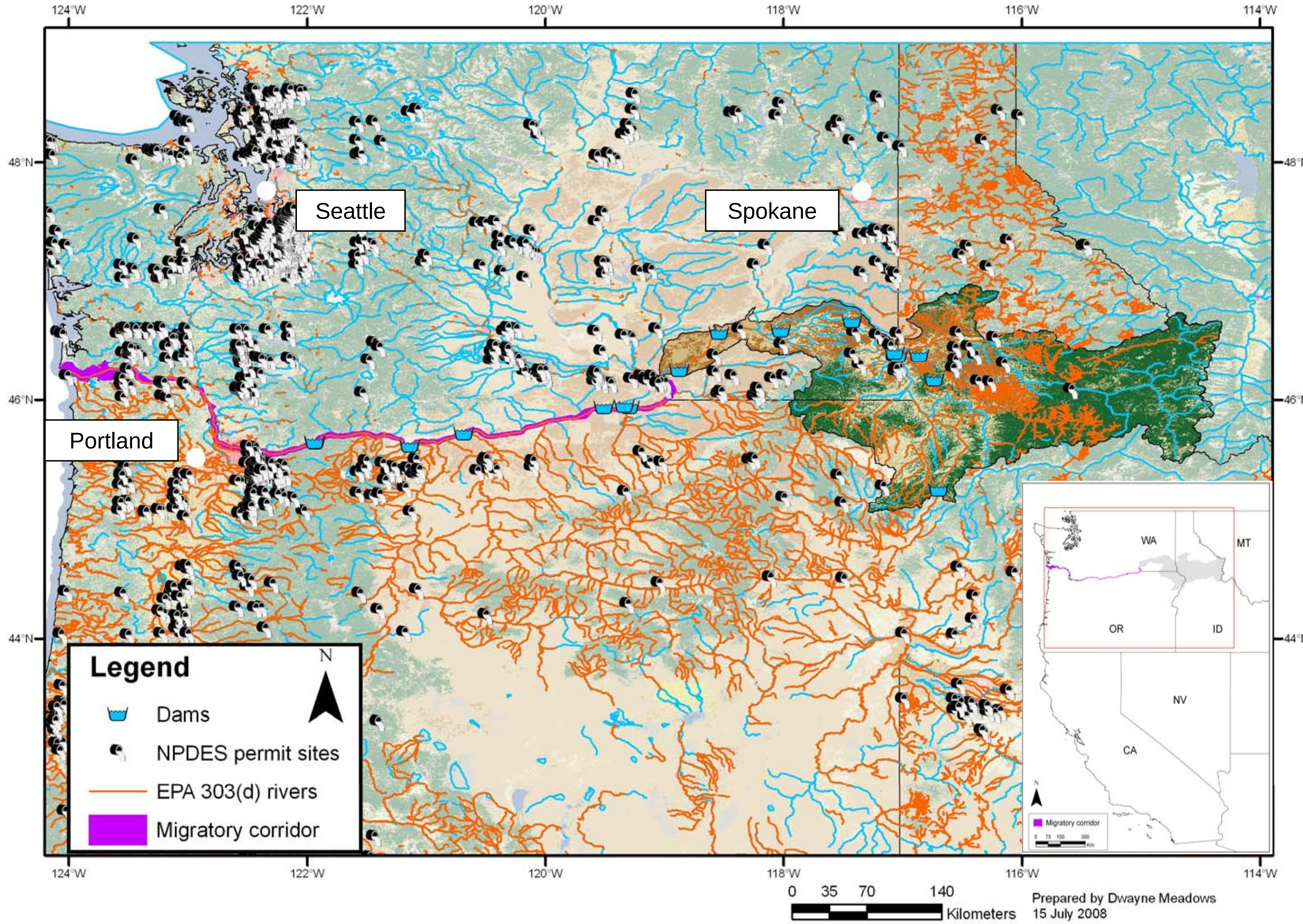


Figure 13. Snake River Fall-run Chinook salmon distribution. The Legend for the Land Cover Class categories is found in Figure 7.

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

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

Life History

Prior to alteration of the Snake River basin by dams, fall Chinook salmon exhibited a largely ocean-type life history, where they migrated downstream and reared in the mainstem Snake River during their first year. Today, fall Chinook salmon in the Snake River Basin exhibit one of two life histories: ocean type and reservoir-type (Connor 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 Snake River. 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 Snake River.

Adult Snake River 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.

Status and Trends

Snake River 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 Snake River 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 Snake River dams (1961 to 1975). Counts of natural-origin adult Snake River 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 et al. 2005). Numbers of natural-origin Snake River 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.

Snake River 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 Snake River 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 Snake River 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 Snake River fall-run Chinook salmon is relatively low, but has been increasing in the last decade (Good et al. 2006). 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 Snake River 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 Snake River fish from the Lyons Ferry Hatchery program, and strays from hatchery programs outside the Snake River. Nevertheless, the Snake River fall-run Chinook salmon remains genetically distinct for similar fish in other basins. Phenotypic characteristics have shifted in apparent response to environmental changes from hydroelectric dams (Connor et al. 2005).

The ICBTRT has defined only one extant population for the Snake River fall-run Chinook salmon, the lower Snake River mainstem population. This population occupies the Snake River 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 Snake River 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 Snake River, and all Snake River reaches upstream to Hells Canyon Dam. Critical habitat also includes several river reaches presently or historically accessible to Snake River fall-run Chinook salmon. Limiting factors identified for Snake River 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 Snake River 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 2005b). 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

Snake River spring/summer-run Chinook salmon are primarily limited to the Salmon, Grande Ronde, Imnaha, and Tucannon Rivers in the Snake River basin (Figure 14). The Snake River 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,

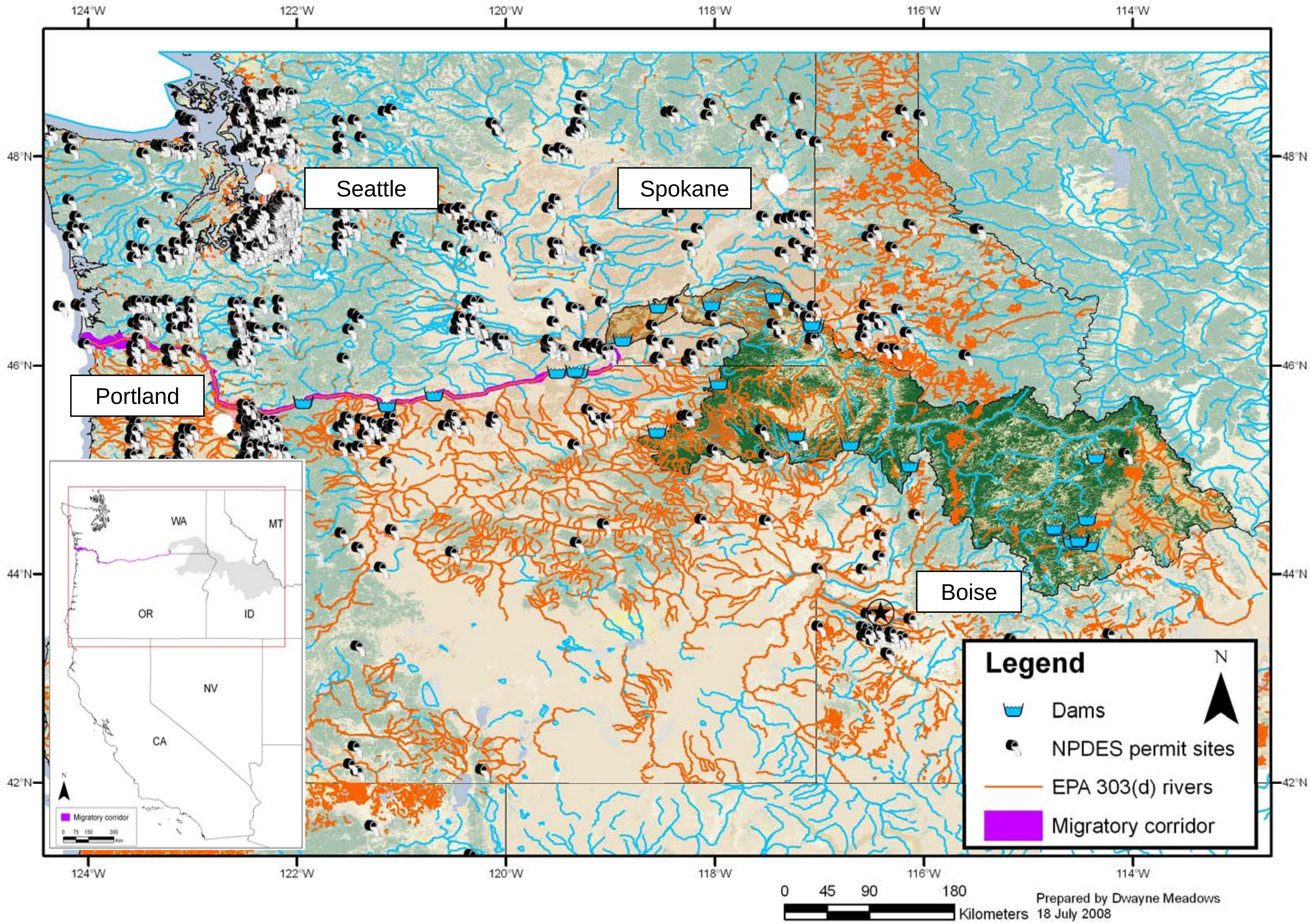


Figure 14. Snake River Spring/Summer-run Chinook salmon distribution. The Legend for the Land Cover Class categories is found in Figure 7.

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 8 identifies populations within the Snake River spring/summer Chinook salmon ESU, their abundances, and hatchery input.

Table 8. Snake River Spring/Summer Chinook salmon populations, abundances, and hatchery contributions (Good 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

Snake River 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. Snake River spring/summer-run Chinook salmon return from the ocean to spawn primarily as four and five year-old fish, after two to three years in the ocean. A small fraction of the fish return as three year-old "jacks", heavily predominated by males.

Status and Trends

Snake River 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 Snake River 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 Snake River 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 et al. 2005). Good et al. (2006) 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 Snake River 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. Snake River spring/summer Chinook salmon remains likely to become endangered (Good 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 Snake River 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 Snake River, and all Snake River reaches upstream to Hells Canyon Dam; the Palouse River from its confluence with the Snake River upstream to Palouse Falls, the Clearwater

River from its confluence with the Snake River upstream to its confluence with Lolo Creek; 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 et al. 2006). 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 9 identifies populations within the UWR Chinook salmon ESU, their abundances, and hatchery input.

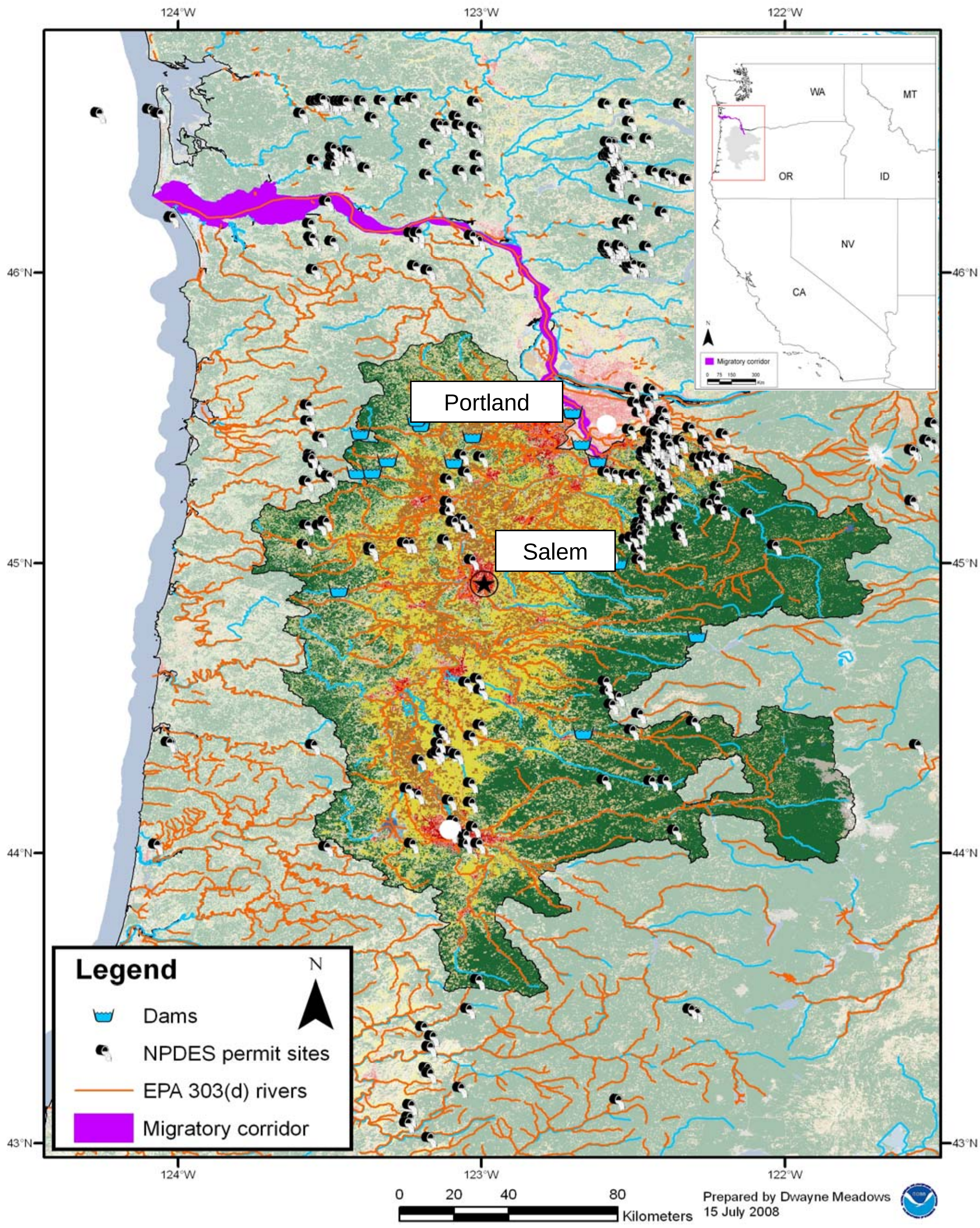


Figure 15. Upper Willamette River Chinook salmon distribution. The Legend for the Land Cover Class categories is found in Figure 7.

Table 9. Upper Willamette River Chinook salmon populations, abundances, and hatchery contributions (Good 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

Life History

UWR Chinook salmon exhibit an earlier time of entry into the Columbia River and estuary than other spring Chinook salmon ESUs (Meyers 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 et al. 2004).

Status and Trends

Upper Willamette River 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 (Good et al. 2005; Myers et al. 1998). 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 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 Upper Willamette River 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 Upper Willamette River 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 et al. 1997b). 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 10. Columbia River Chum salmon populations, abundances, and hatchery contributions (Good et al. 2005).

Current Populations	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Youngs Bay	Unknown	0	0
Gray's 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**	

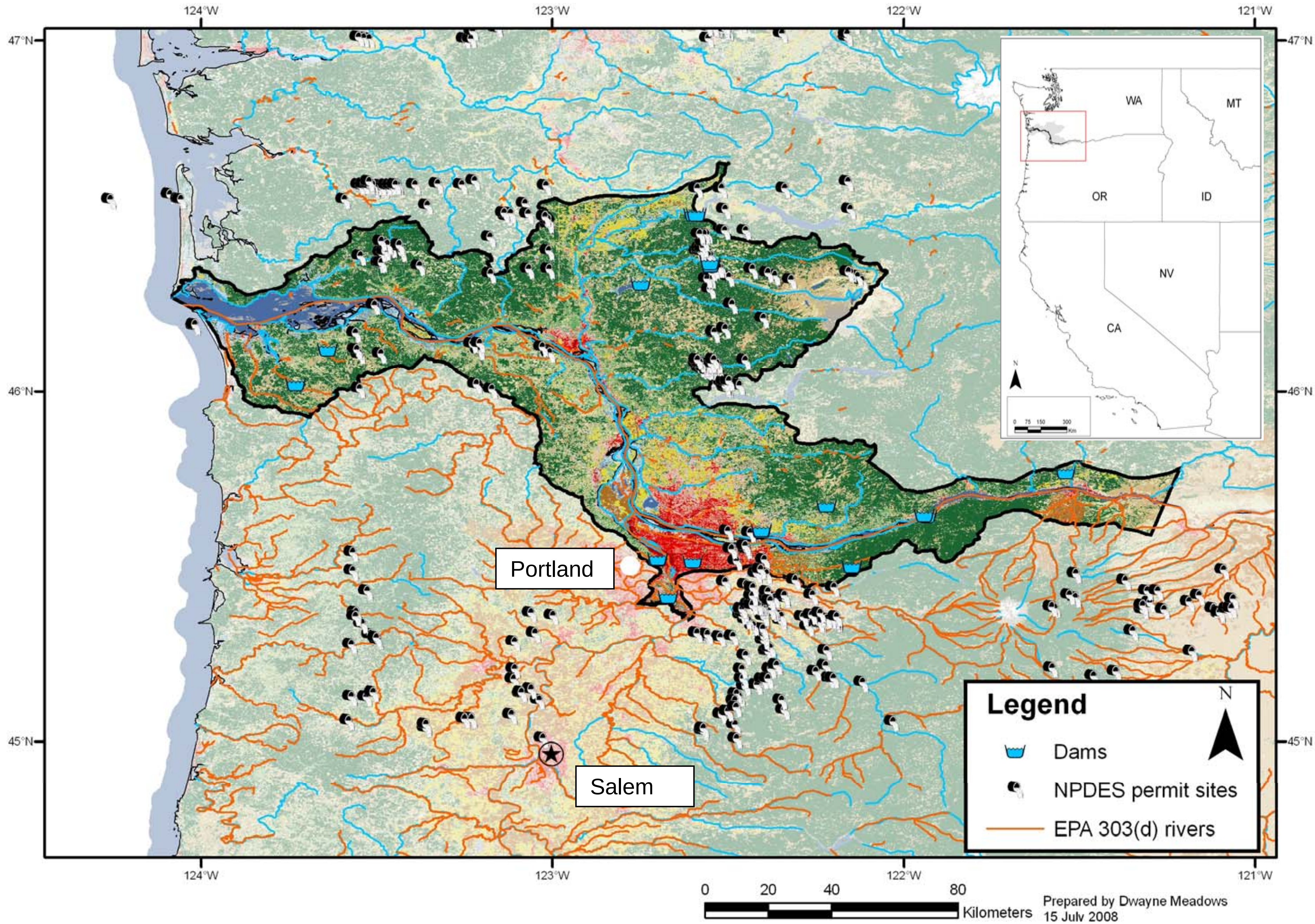


Figure 16. Columbia River Chum salmon distribution. The Legend for the Land Cover Class categories is found in Figure 7.

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 Lower Columbia River basin. However, by the 1950s most runs disappeared (Fulton 1968; Marr 1943; Rich 1942). 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 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 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. However in the Cascade MPG, chum sampled from each tributary recently were shown to be the remnants of genetically distinct populations (Greco et al. 2007). 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 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 11 identifies populations within the Hood Canal summer-run Chum salmon ESU, their abundances, and hatchery input.

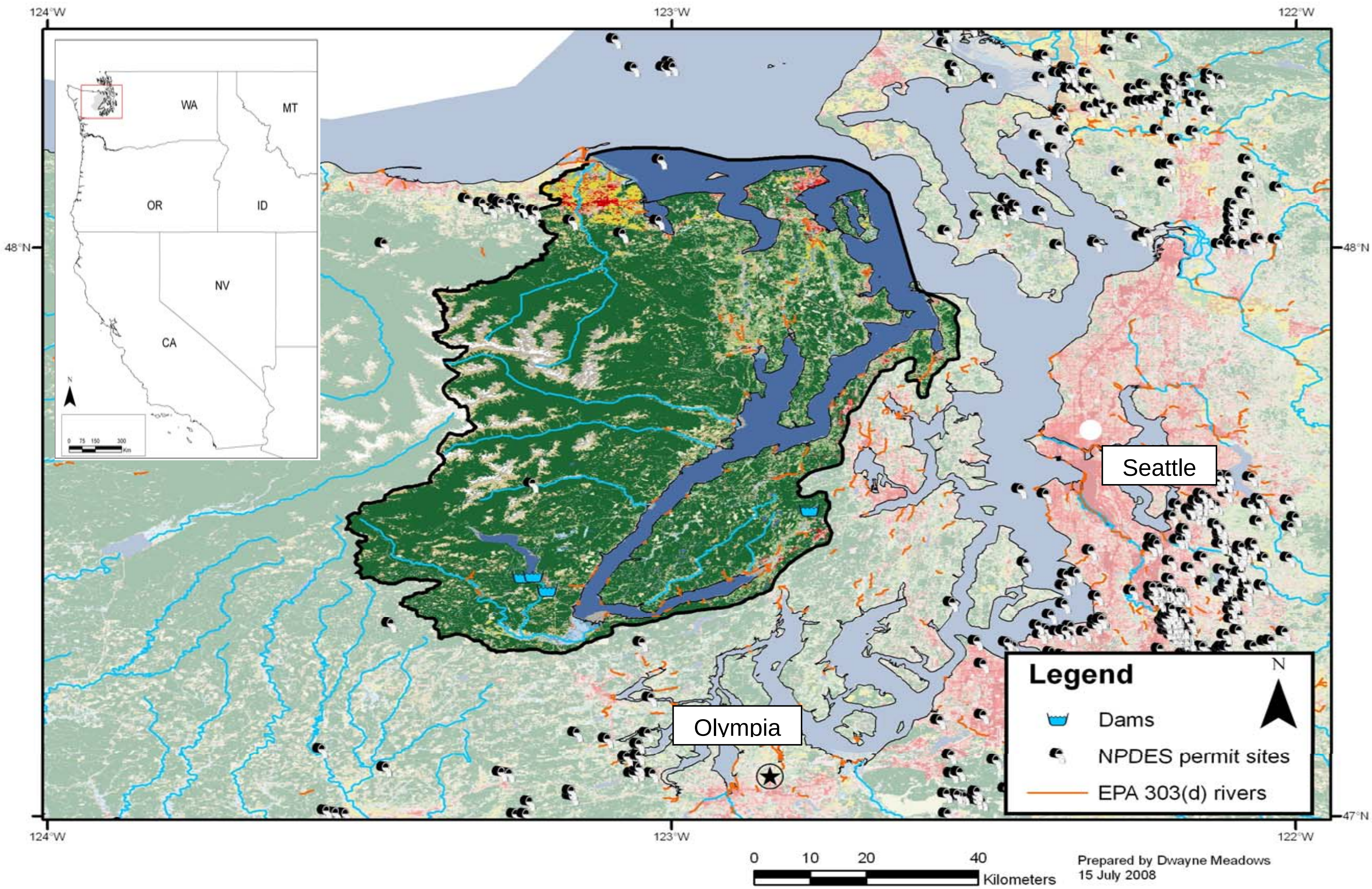


Figure 17. Hood Canal Summer-run Chum salmon distribution. The Legend for the Land Cover Class categories is found in Figure 7.

Table 11. Hood Canal summer-run Chum salmon populations, abundances, and hatchery contributions (Good 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**	

Life History

The Hood Canal summer-run Chum salmon are defined in the Salmon and Steelhead Stock Inventory (WDFet 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 et al. 1997b). 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; Salo 1991; Schroder 1977; Schroder et al. 1974)]. 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 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 Central California Coast coho salmon ESU extends from Punta Gorda in northern California south to and including the San Lorenzo River in central California (Weitkampet al. 1995). Table 12 identifies populations within the Central California Coast Coho salmon ESU, their abundances, and hatchery input (Figure 18).

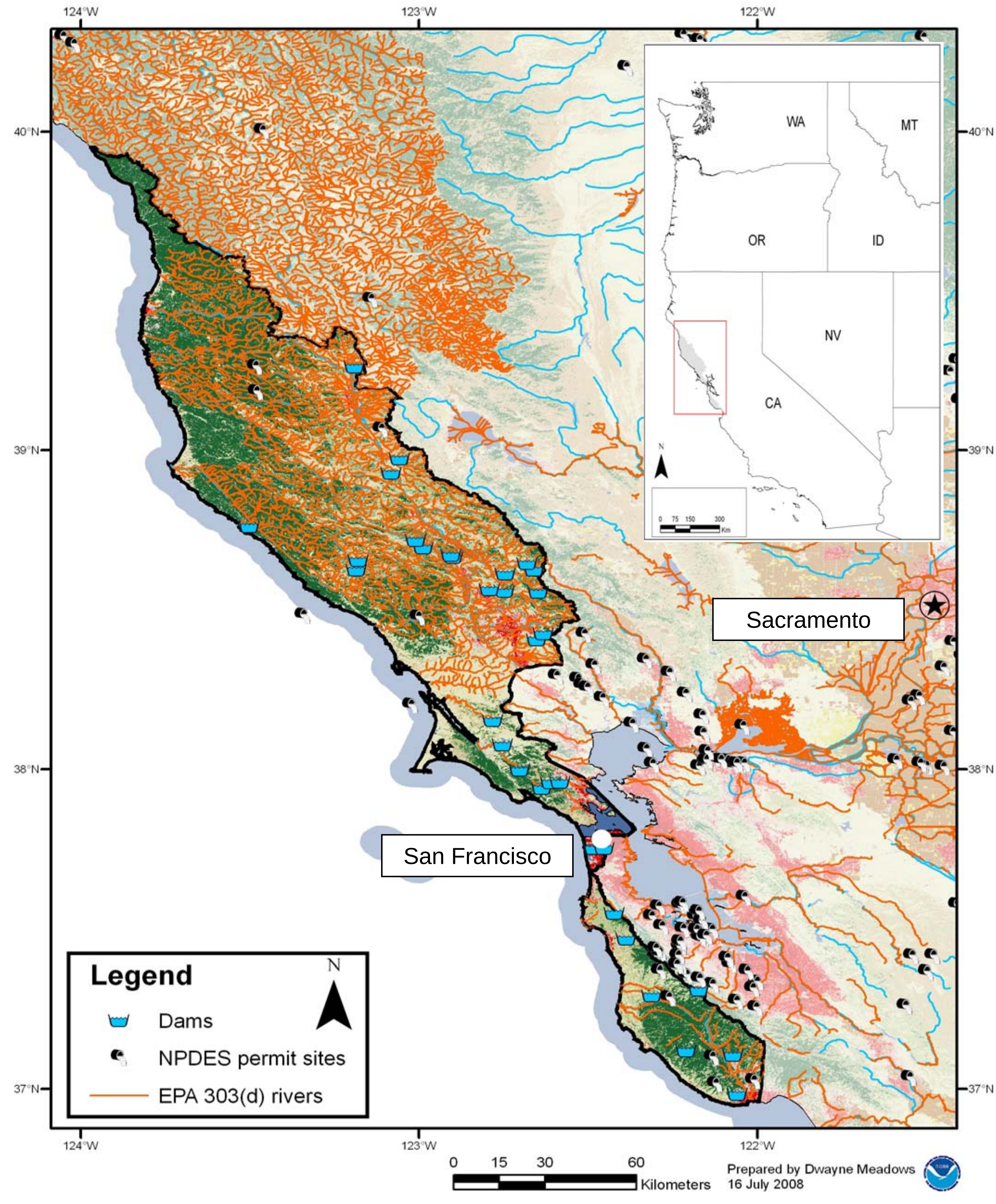


Figure 18. Central California Coast Coho salmon distribution. The Legend for the Land Cover Class categories is found in Figure 7.

Table 12. Central California Coast Coho salmon populations, abundances, and hatchery contributions (Good 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 Central California Coast 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 Central California Coast 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 Central California Coast 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.

Central California Coast 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 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. Informed by the VSP risk assessment and the associated uncertainty, the strong majority opinion of the BRT was that the naturally spawned component of the Central California Coast coho salmon ESU was "in danger of extinction." The minority opinion was that this ESU is "likely to become endangered within the foreseeable future" (70 FR 37160). Based on these conclusions, NMFS granted endangered status for this ESU on June 28, 2005 (70 FR 37160).

Critical Habitat

Critical habitat for the Central California Coast 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

Lower Columbia River (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.

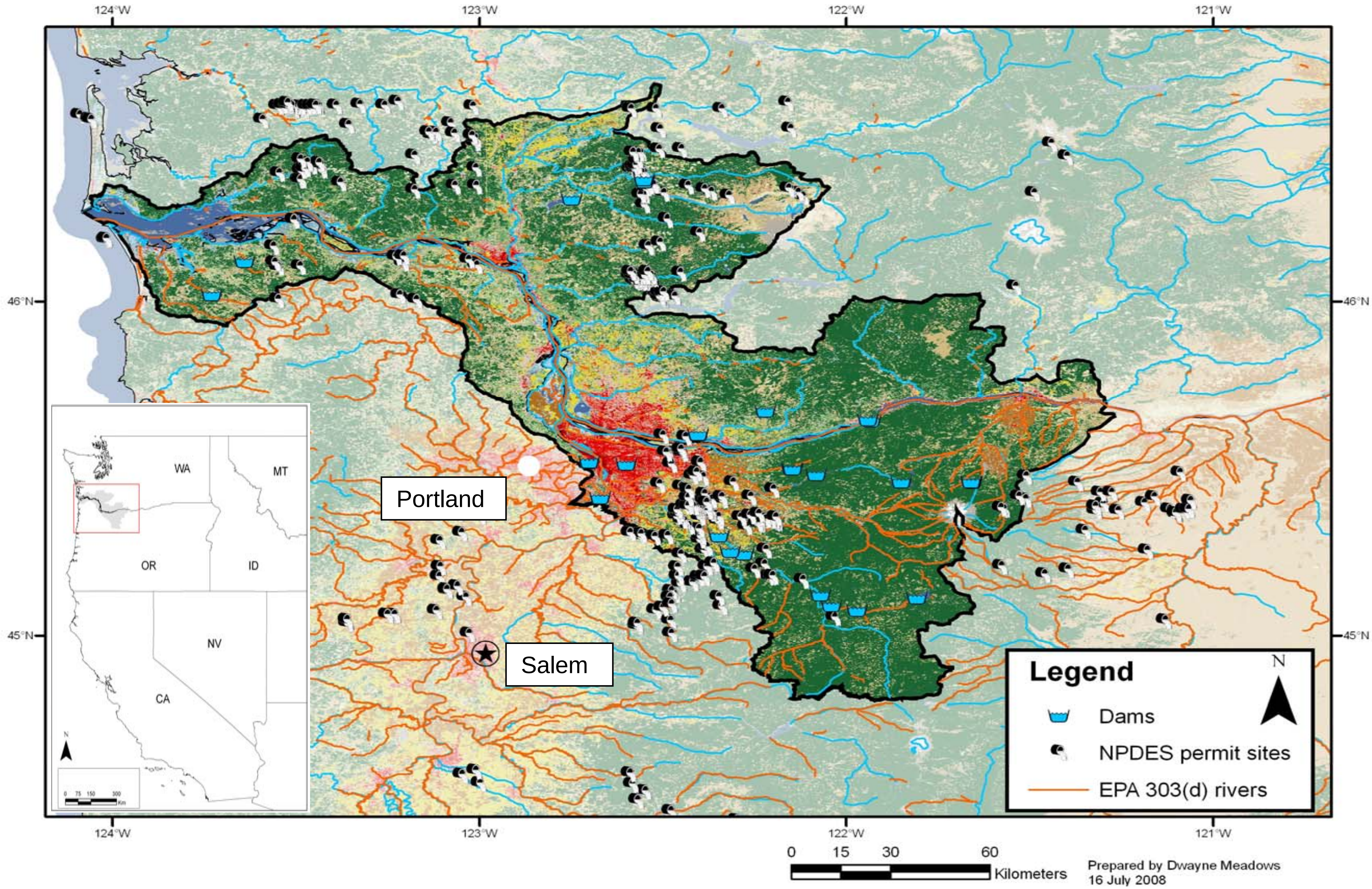


Figure 19 . Lower Columbia River coho salmon distribution. The Legend for the Land Cover Class categories is found in Figure 7.

Table 13 identifies populations within the Lower Columbia River Coho salmon ESU, their abundances, and hatchery input.

Table 13. Lower Columbia River Coho salmon populations, abundances, and hatchery contributions (Good 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
Lower Columbia River 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)**	

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 Lower Columbia River 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 WLCTRT 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 Lower Columbia River coho salmon.

Southern Oregon/Northern California Coast Coho Salmon

Distribution
Southern Oregon/Northern California Coast 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).

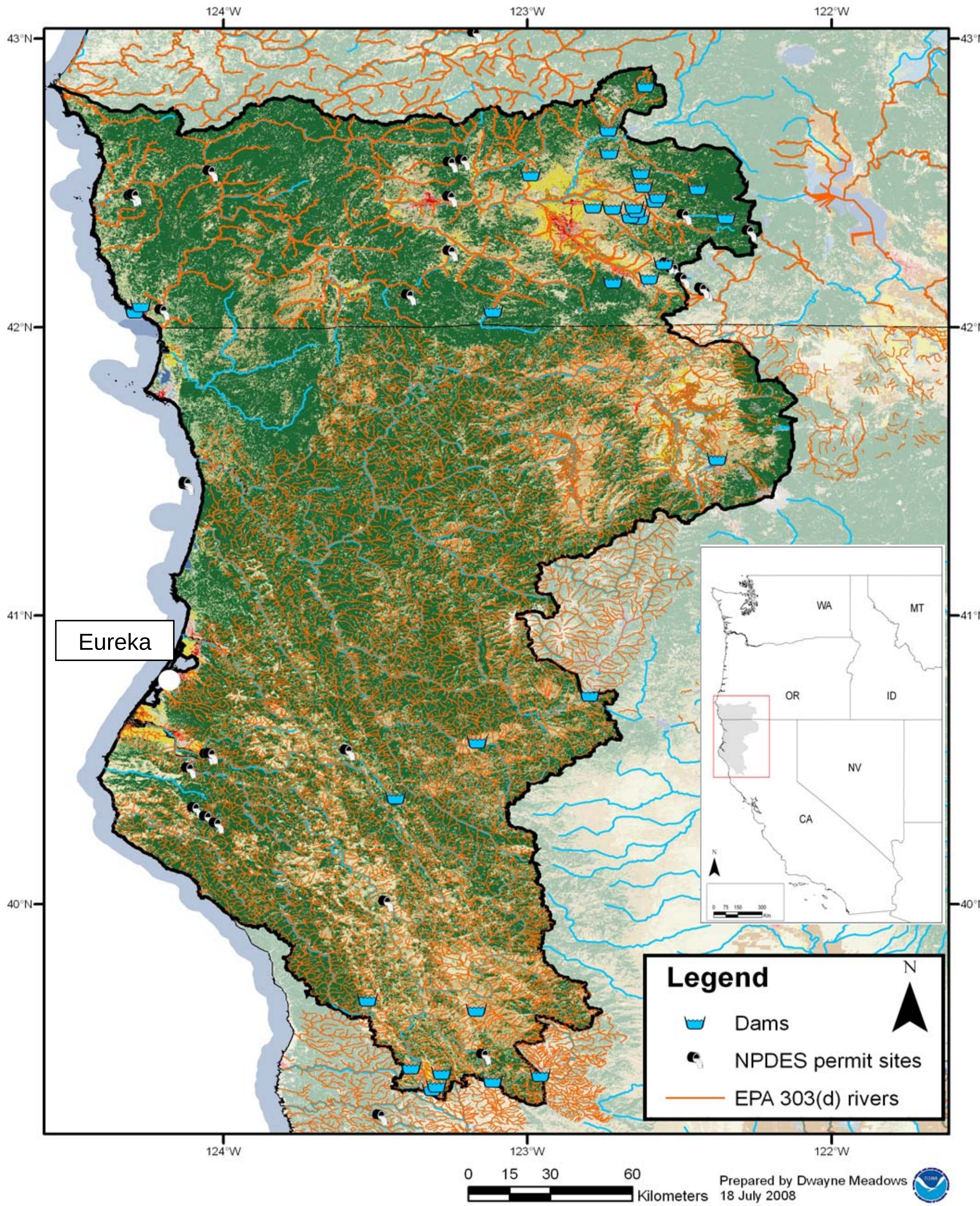


Figure 20. Southern Oregon/Northern California Coast coho salmon distribution. figure. The Legend for the Land Cover Class categories is found in Figure 7.

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

Southern Oregon/Northern California Coast 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

Southern Oregon/Northern California Coast 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 Southern Oregon/Northern California Coast coho salmon ESU, was formally assessed in 1995 (Weitkamp et al. 1995). Two subsequent status review updates have been published by NMFS. One review update addressed all West Coast coho salmon ESUs (Busby et al. 1996). The second update specifically addressed the Oregon Coast and Southern Oregon/Northern California Coast coho salmon ESUs (Gustafson 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.

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 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 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 Southern Oregon/Northern California Coast 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 Southern Oregon/Northern California Coast 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 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 14 identifies populations within the Oregon Coast coho salmon ESU, their abundances, and hatchery input.

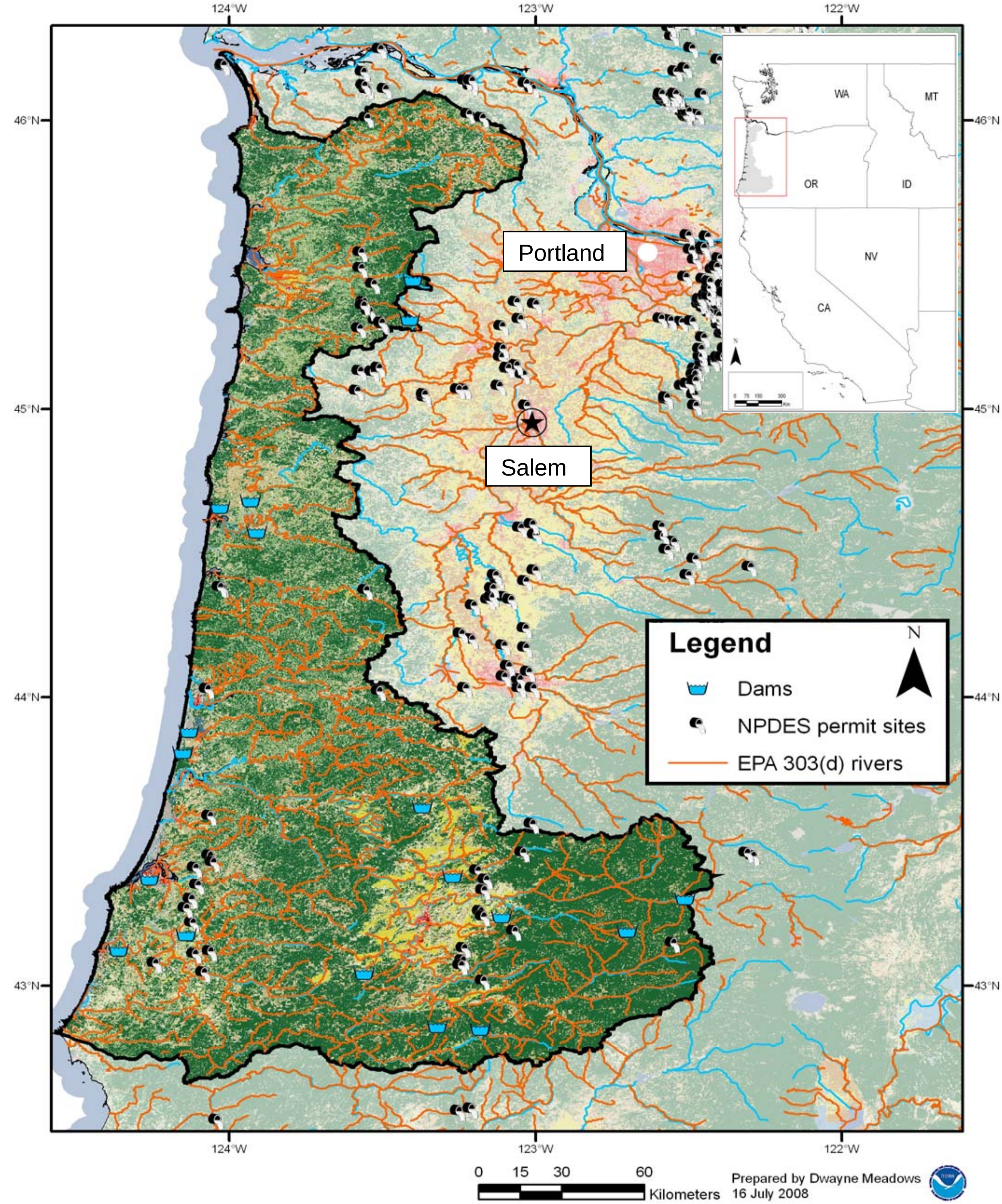


Figure 21. Oregon Coast Coho salmon distribution. The Legend for the Land Cover Class categories is found in Figure 7.

Table 14. Oregon Coast Coho salmon populations, abundances, and hatchery contributions (Good 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**	

Status and Trends

The Oregon Coast coho salmon ESU was listed as a threatened species on February 11, 2008 (73 FR 7816). The most recent NMFS status review for the Oregon Coast coho salmon ESU was conducted by the BRT in 2003, which assessed data through 2002. The abundance and productivity of Oregon Coast coho salmon since the previous status review (Gustafson 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 Oregon Coast 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 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 Oregon Coast 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.

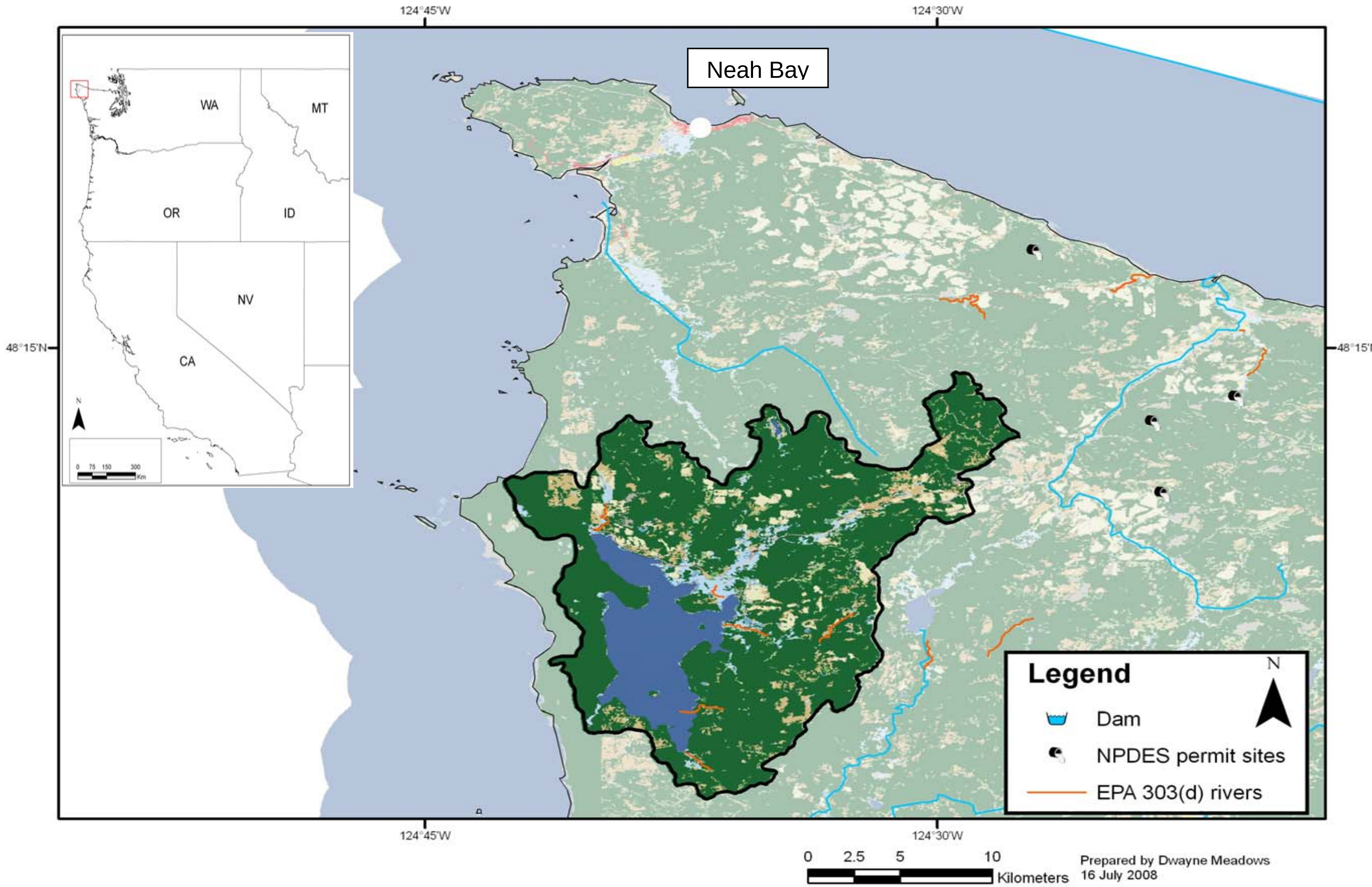


Figure 22. Ozette Lake Sockeye salmon distribution. The Legend for the Land Cover Class categories is found in Figure 7.

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 et al. 1996).

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 (2006) 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 (Myers et al. 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 (Myers et al. 1998), 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 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 (Haggerty et al. 2007; Hard et al. 1992). For return years 2000 to 2003, the four year average abundance estimate was slightly over 4,600 sockeye (Haggerty 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 et al. 2005). It is estimated that between 35,500 and 121,000 spawners could be normally carried after full recovery (Hard 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. (2006) 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 (Boomer 1995; Good et al. 2005; Kemmerich 1945).

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

Life History

Snake River sockeye salmon are unique compared to other sockeye salmon populations. Sockeye salmon returning to Redfish Lake in Idaho’s Stanley Basin travel a greater distance from the sea (approximately 900 miles) to a higher elevation (6,500 ft) than any other sockeye salmon population and are the southern-most population of sockeye salmon in the world (Bjornn et al. 1968). Stanley Basin sockeye salmon are separated by 700 or more river miles from two other extant upper Columbia River populations in the

Wenatchee River and Okanogan River drainages. These latter populations return to lakes at substantially lower elevations (Wenatchee at 1,870 ft, Okanagon at 912 ft) and occupy different ecoregions.

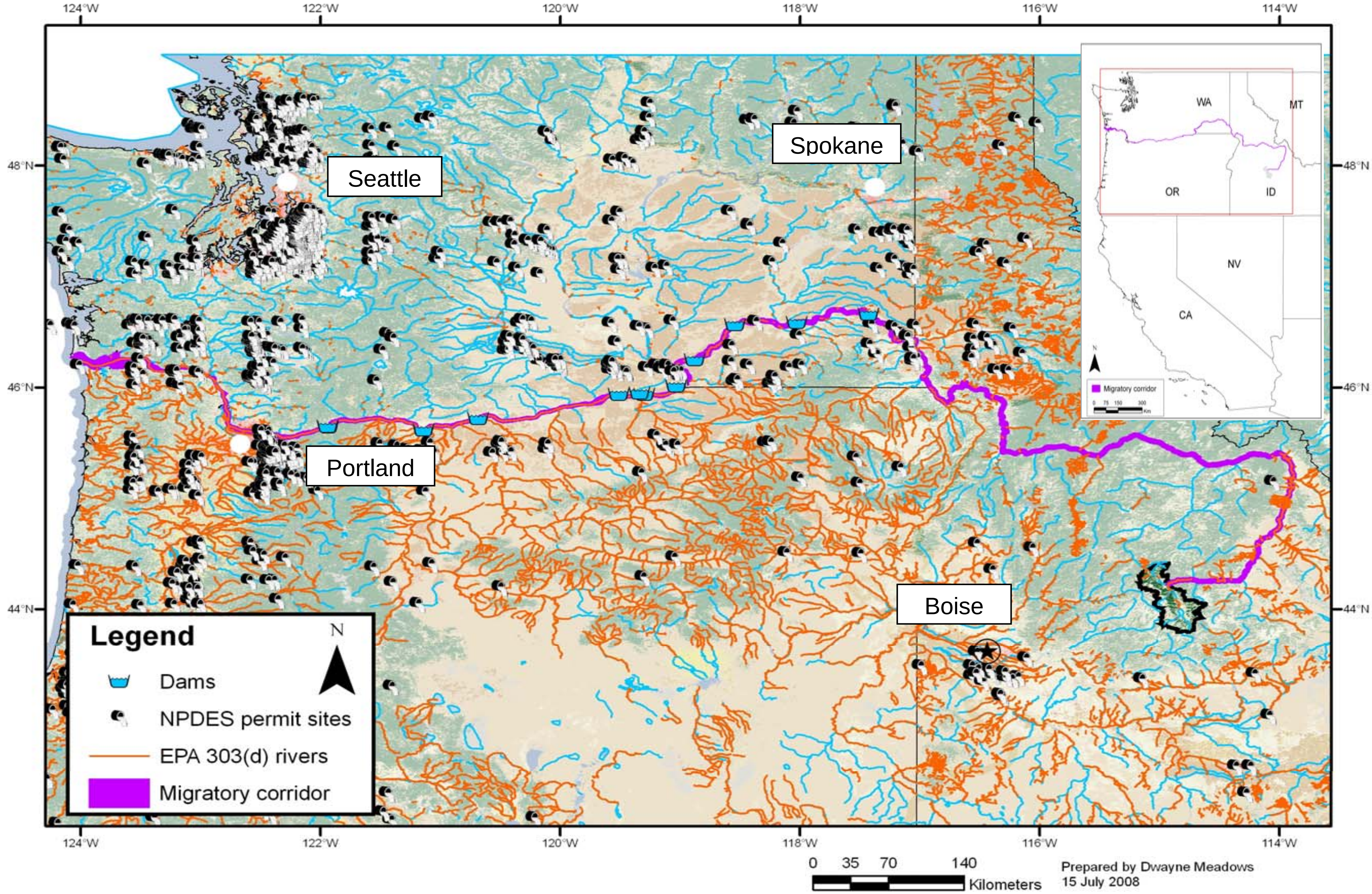


Figure 23. Snake River Sockeye Salmon distribution. The Legend for the Land Cover Class categories is found in Figure 7.

Status and Trends

Snake River 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 Snake River 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 Snake River 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 et al. 1997). These populations are now considered extinct. Although kokanee, a resident form of *O. nerka*, occur in numerous lakes in the Snake River 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 Snake River 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 et al. 2005). Five lakes in the Stanley Basin historically contained sockeye salmon: Alturas, Pettit, Redfish, Stanley and Yellowbelly (Bjornn 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 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 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 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 et al. 2004). Based on current abundance and productivity information, the Snake River 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 Snake River 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 Snake River, and all Snake River reaches upstream to the confluence of the Salmon River; all Salmon River reaches to Alturas Lake Creek; Stanley, Redfish, yellow Belly, Pettit, and Alturas Lakes (including their inlet and outlet creeks); Alturas Lake Creek and that portion of Valley Creek between Stanley Lake Creek; and the Salmon River. Limiting factors identified for Snake River 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 (Good et al. 2005; Gustafson et al. 1997; Moyle 1976). 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 (Busby et al. 1996; Nickelsen et al. 1992). They require cool, deep holding pools during summer and fall, prior to spawning (Nickelsen 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 et al. 1992) in January and February (Barnhart 1986). Winter steelhead enter freshwater between November and April in the Pacific Northwest (Busby et al. 1996; Nickelsen et al. 1992), 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 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 et al. 1996), although steelhead rarely spawn more than twice before dying; most that do so are females (Nickelsen et al. 1992). Iteroparity is more common among southern steelhead populations than northern populations (Busby 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 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 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 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 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 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 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 15 identifies populations within the Central California Coast Steelhead salmon ESU, their abundances, and hatchery input.

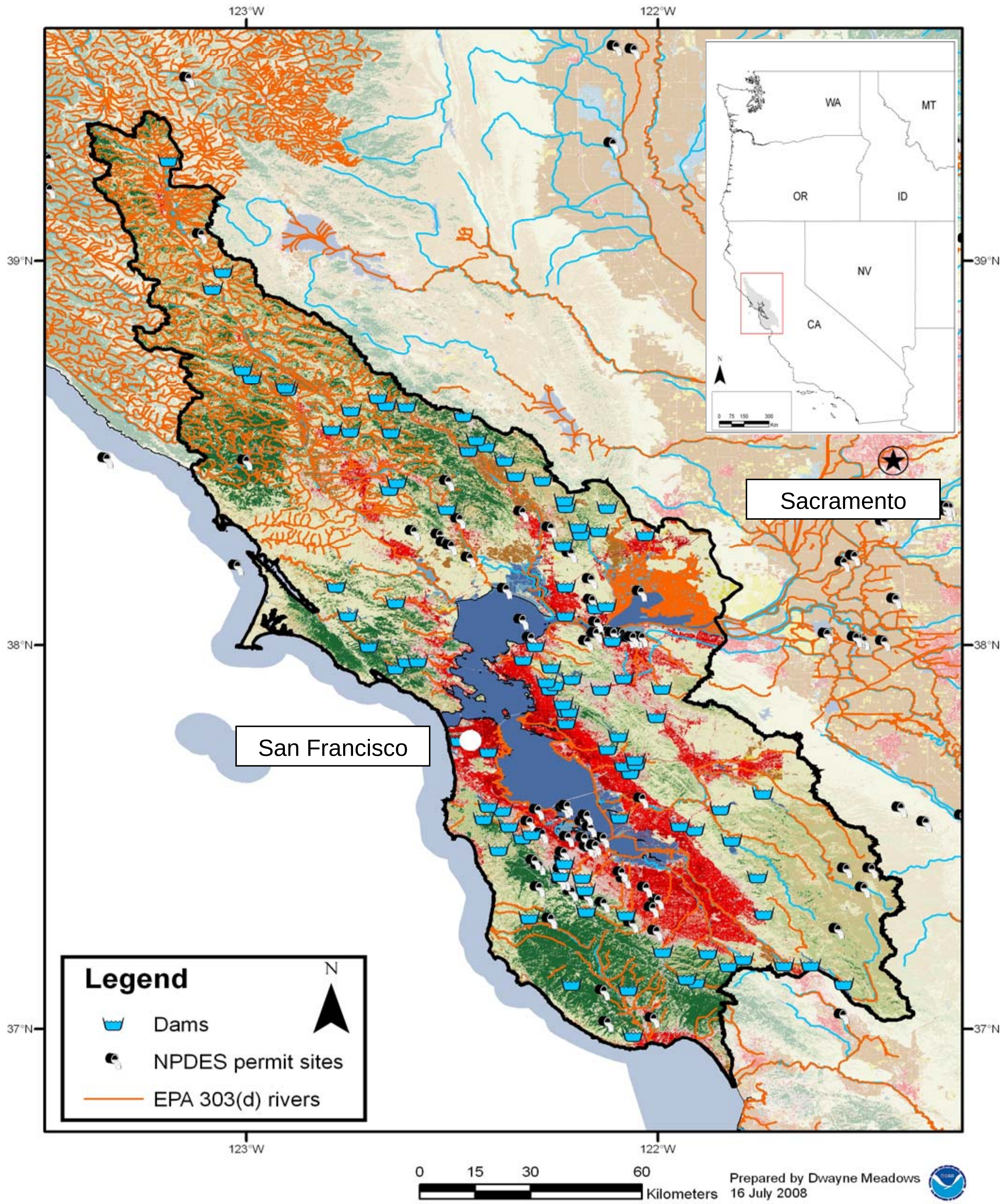


Figure 24. Central California Coast steelhead. The Legend for the Land Cover Class categories is found in Figure 7.

Table 15. Central California Coast Steelhead salmon populations, abundances, and hatchery contributions (Good 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 ESU 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 ESU.

Status and Trends

The Central California Coast 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 (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 ESU, 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 (CDFG 1994; Reavis 1991; 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 Central California Coast steelhead ESU.

A status review update in 1997 (Gustafson 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 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 Central California Coast steelhead ESU. 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 ESU’s range—notably those in Santa Cruz County and the South Bay area (Good et al. 2005).

Critical Habitat

Critical habitat was designated for the Central California Coast 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 Central Valley steelhead occupy the Sacramento and San Joaquin Rivers and its tributaries (Figure 25).

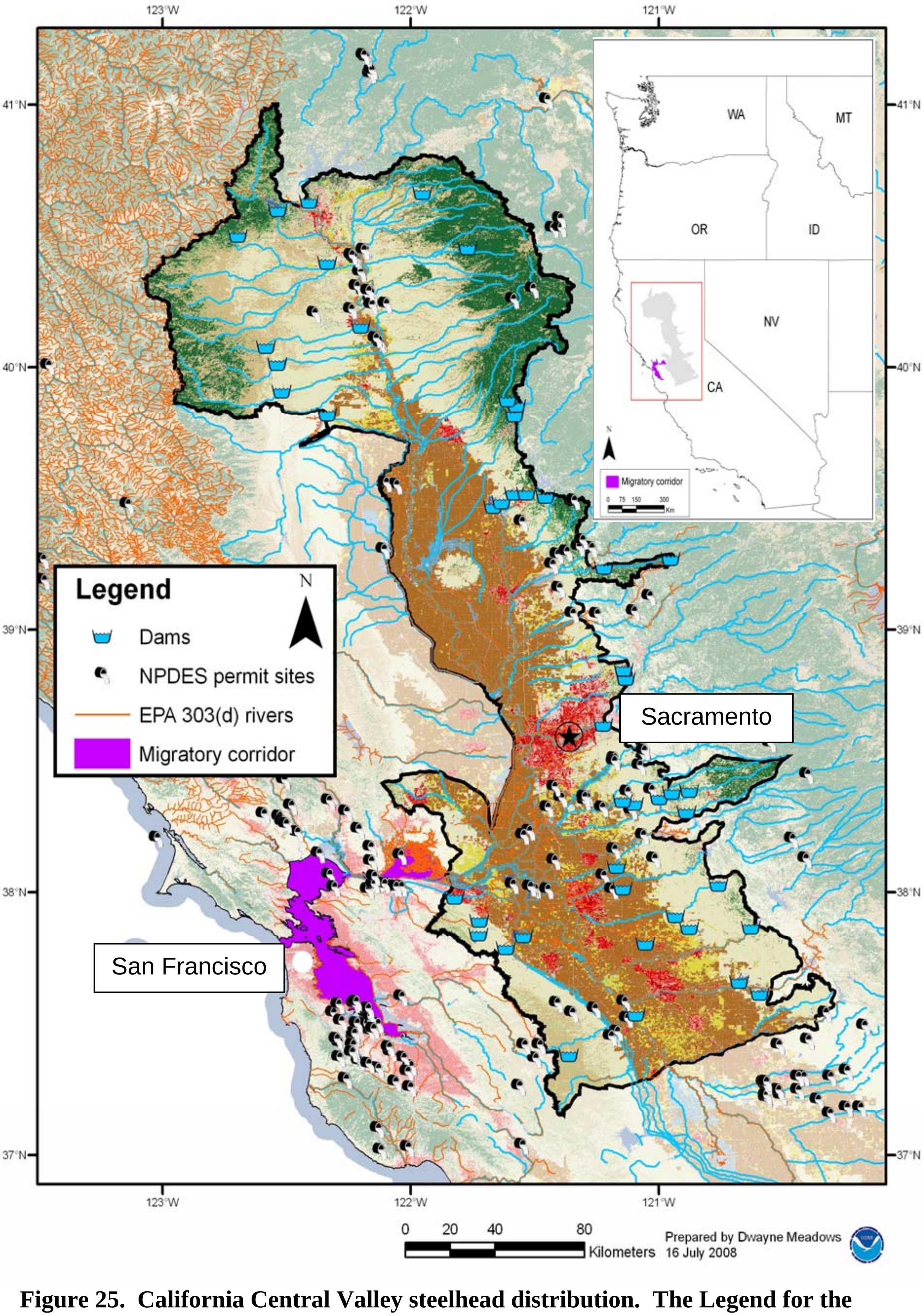


Figure 25. California Central Valley steelhead distribution. The Legend for the Land Cover Class categories is found in Figure 7.

Life History

California Central Valley 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 ESU 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).

Status and Trends

California Central Valley 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 Central Valley. Steelhead historically were well distributed throughout the Sacramento and San Joaquin Rivers (Busby 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 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 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 et al. 1996).

Historic Central Valley 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 Central Valley are mostly confined to the upper Sacramento River and its tributaries, including Antelope, Deer, and Mill Creeks and the Yuba River. Populations may exist in Big Chico and Butte Creeks. A few wild steelhead are produced in the American and Feather Rivers (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. (2006). 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 Central Valley 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 16 identifies populations within the LCR Steelhead salmon ESU, their abundances, and hatchery input.

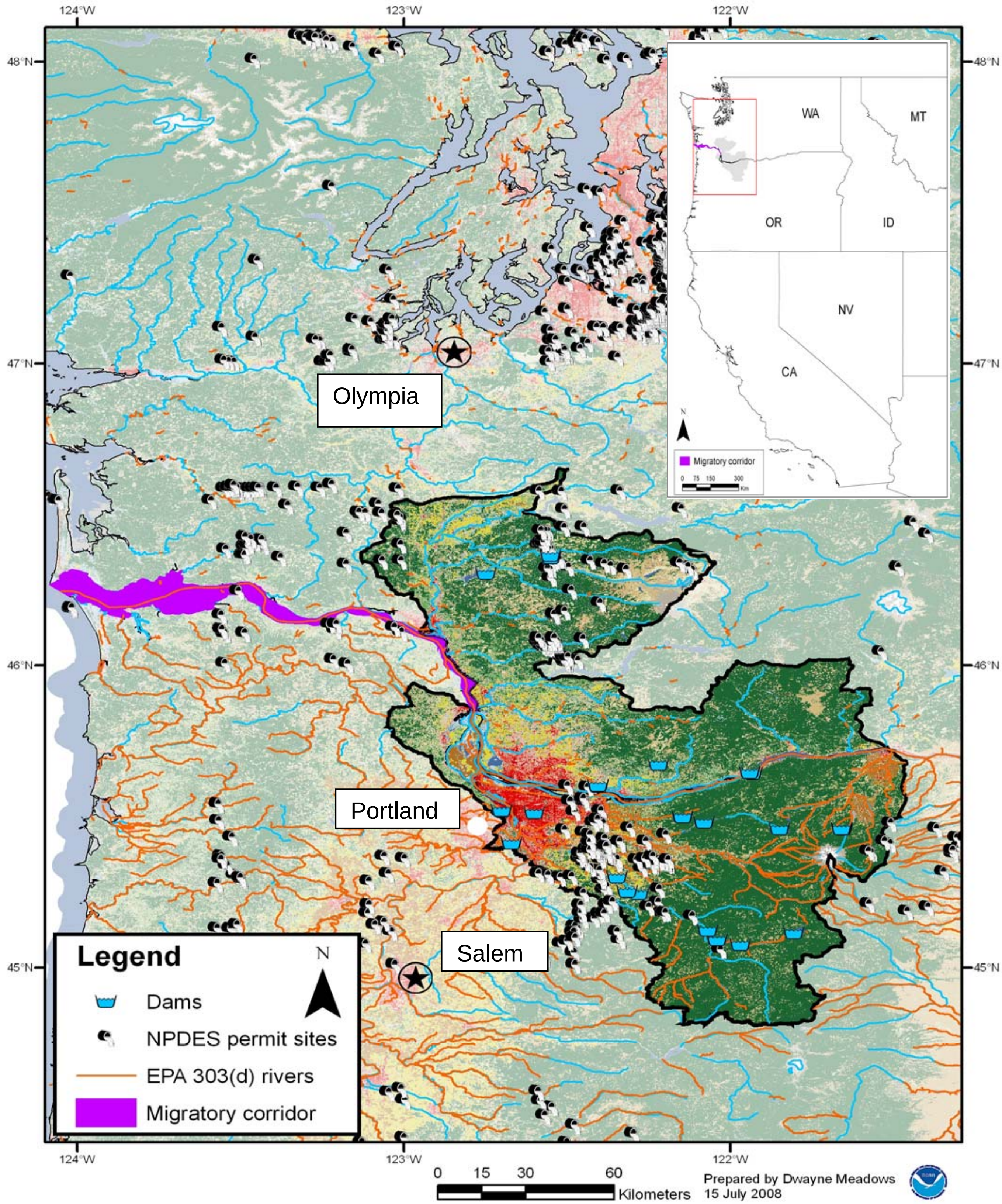


Figure 26. Lower Columbia River Steelhead distribution. The Legend for the Land Cover Class categories is found in Figure 7.

Table 16. Lower Columbia River Steelhead salmon populations, abundances, and hatchery contributions (Good 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. (2006), 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 Snake River 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 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 17 identifies populations within the MCR Steelhead salmon ESU, their abundances, and hatchery input.

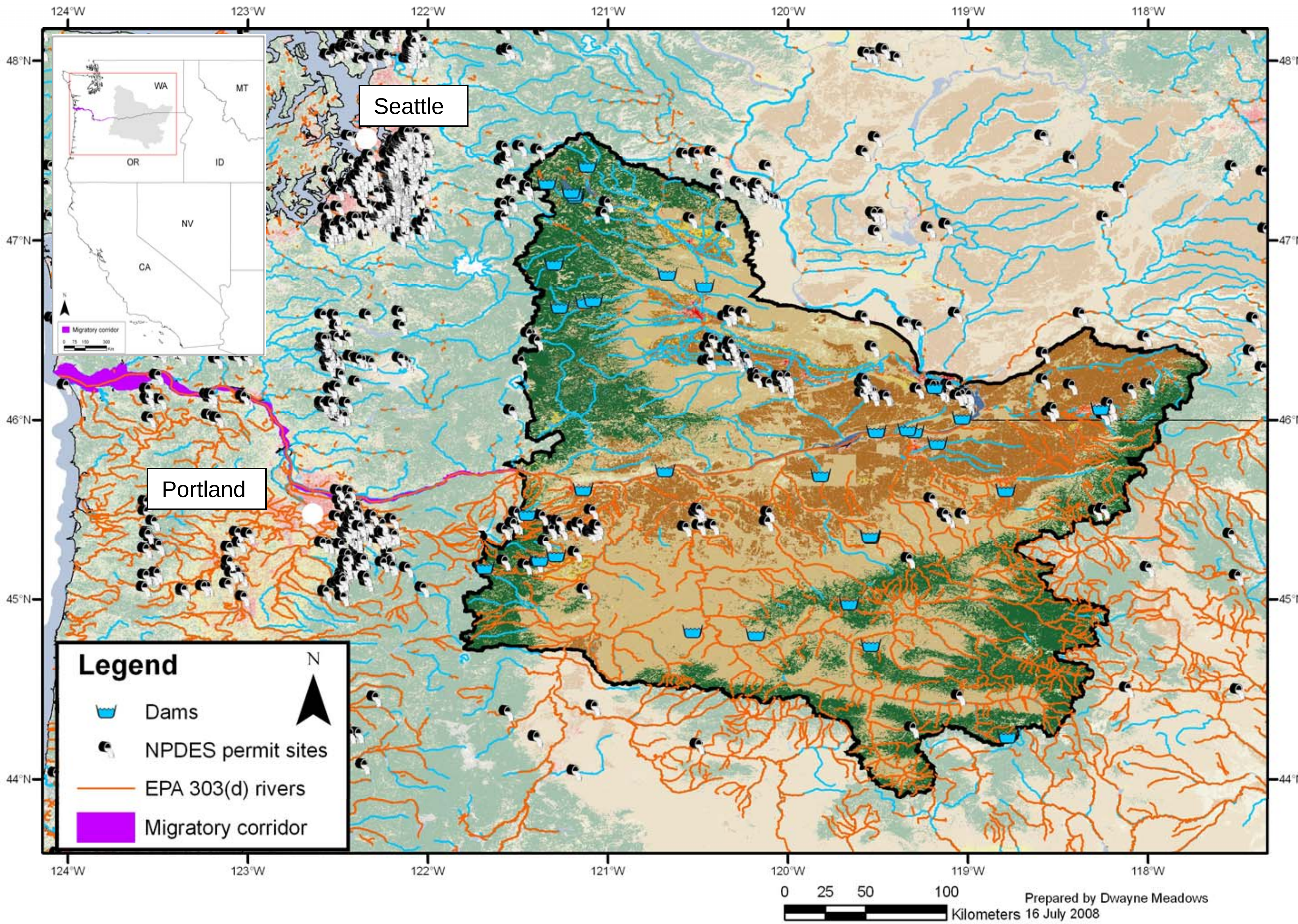


Figure 27. Middle Columbia River Steelhead distribution. The Legend for the Land Cover Class categories is found in Figure 7.

Table 17. Middle Columbia River Steelhead salmon populations, abundances, and hatchery contributions (Good 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		

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 (Howellet 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 (2003a) 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 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 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 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 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 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 California coastal river basins from Redwood Creek south to the Gualala River, inclusive (Figure 28). Table 18 identifies populations within the Northern California Steelhead salmon ESU, their abundances, and hatchery input.

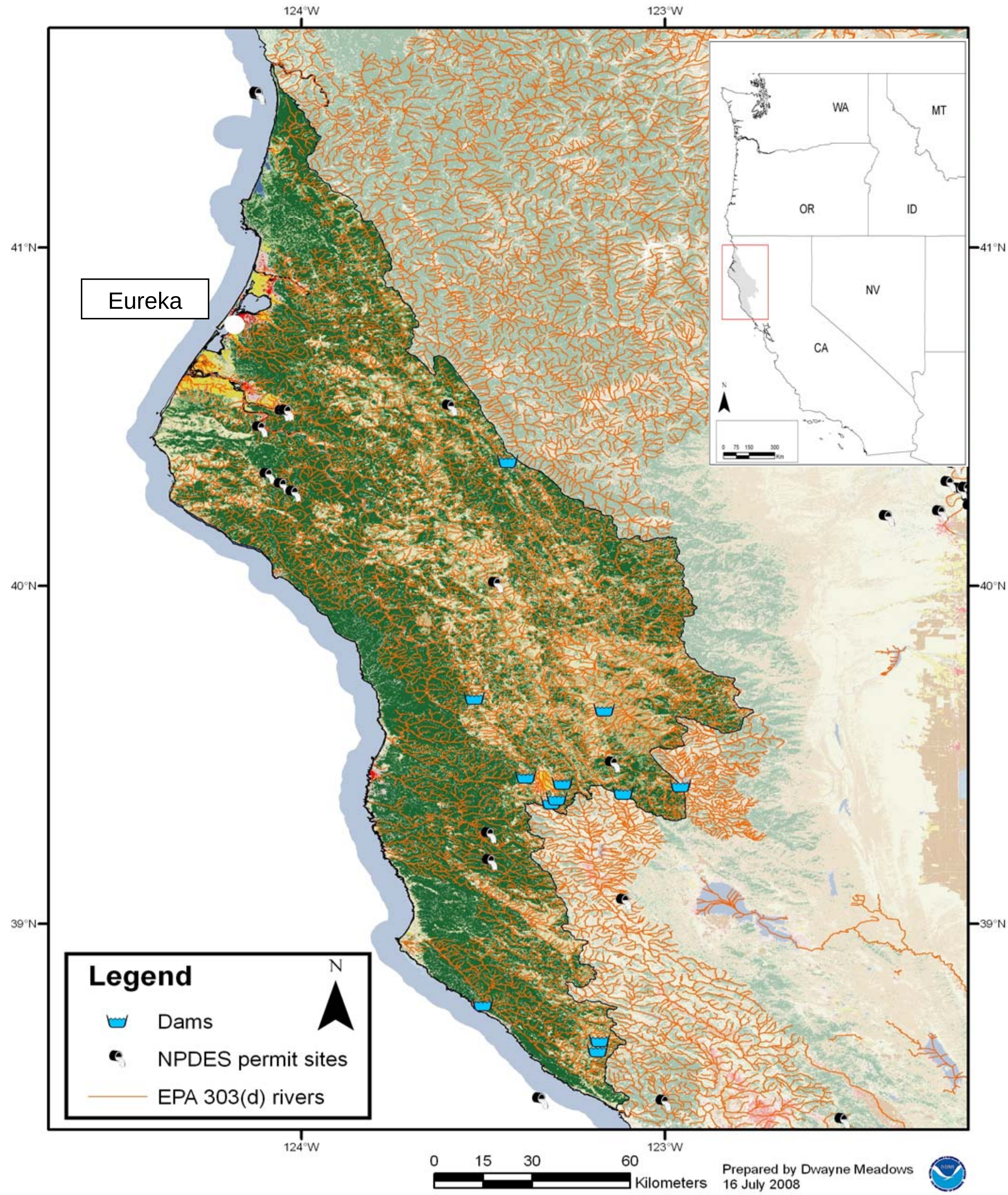


Figure 28. Northern California Steelhead distribution. The Legend for the Land Cover Class categories is found in Figure 7.

Table 18. Northern California Steelhead salmon populations, abundances, and hatchery contributions (Good 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 ESU 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

Northern California 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 Northern California 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.

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 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. (2006) estimated lambda at 0.98 with a 95% confidence interval of 0.93 and 1.04. The result is an overall downward trend in both the long- and short- term. Juvenile data were also recently examined. Both upward and downward trends were apparent (Good et al. 2005). 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 Northern California 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

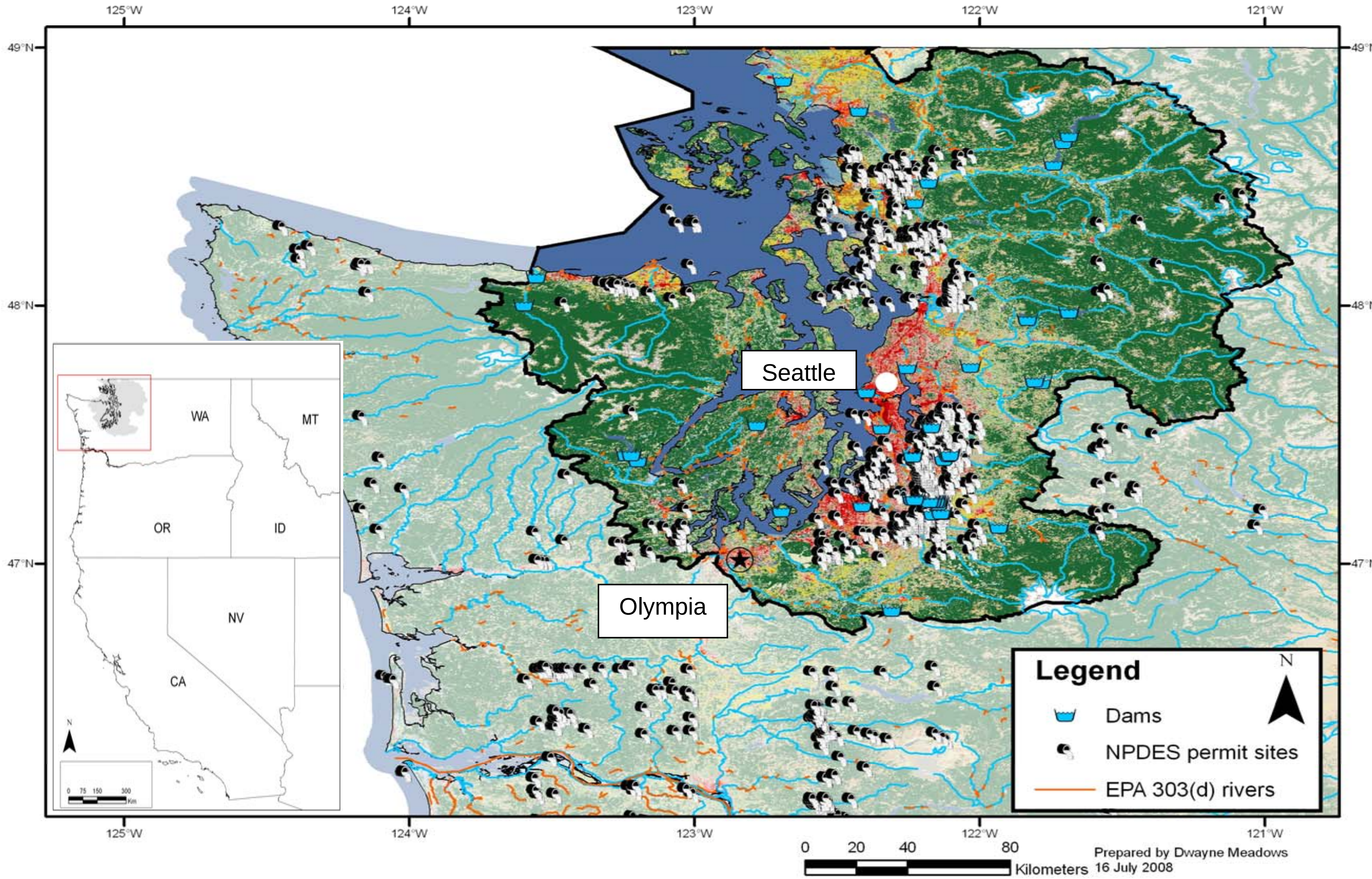


Figure 29. Puget Sound steelhead distribution. The Legend for the Land Cover Class categories is found in Figure 7.

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. (1997) 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.

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
Snake River Basin steelhead is an inland species that occupies the Snake River basin of Idaho, northeast Oregon, and southeast Washington. The Snake River Basin steelhead species includes all naturally spawned populations of steelhead (and their progeny) in streams in the Snake River Basin of Idaho, northeast Oregon, and southeast Washington Snake River 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 Snake River'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.

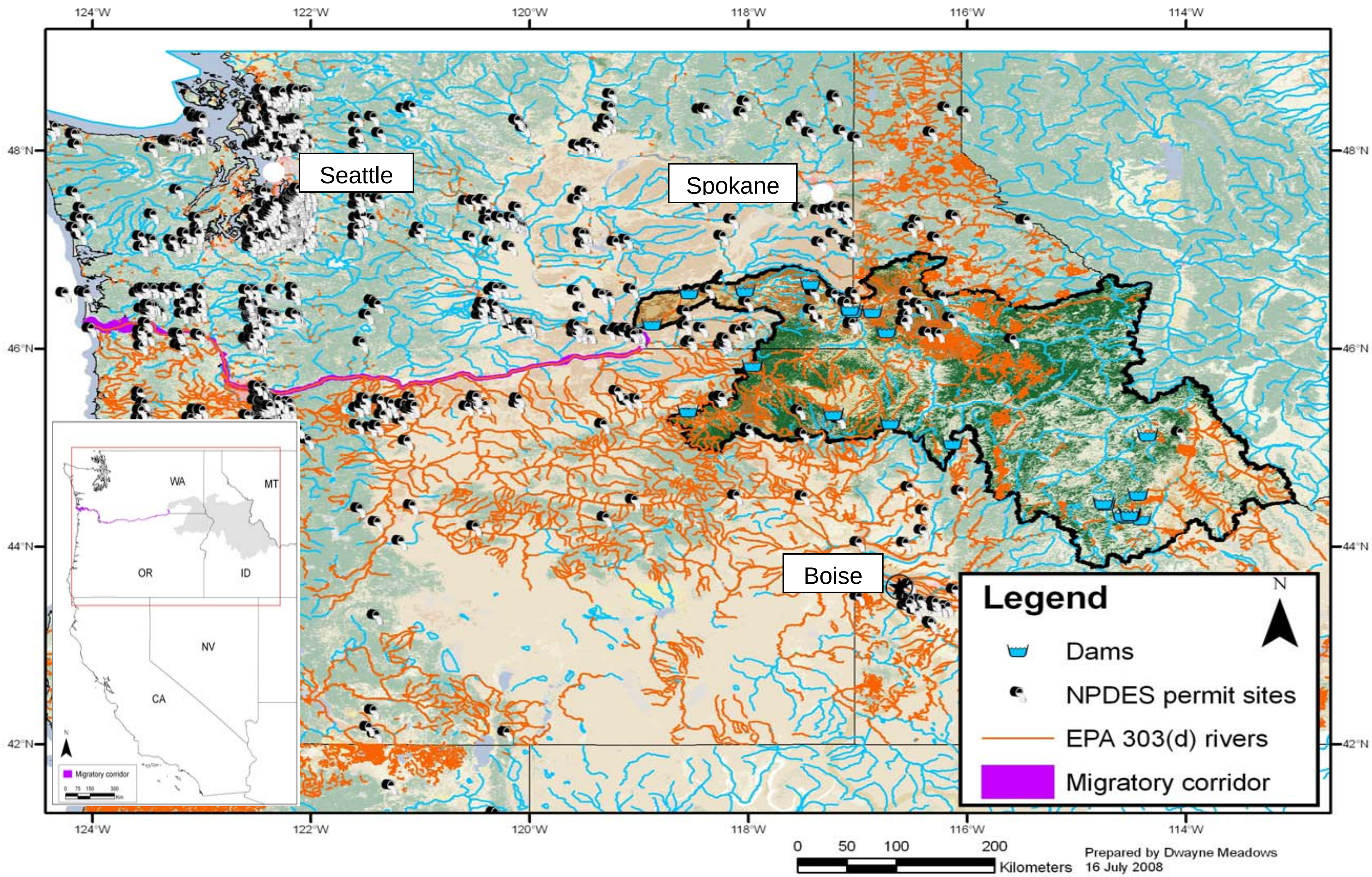


Figure 30. Snake River Basin Steelhead distribution. The Legend for the Land Cover Class categories is found in Figure 7.

Table 19 identifies populations within the Snake River Basin Steelhead salmon ESU, their abundances, and hatchery input.

Table 19. Snake River Basin Steelhead salmon populations, abundances, and hatchery contributions (Good 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)**	**?**	

Life History

Snake River Basin steelhead occupy habitat that is considerably warmer and drier (on an annual basis) than other steelhead DPSs. Snake River Basin steelhead are generally classified as summer run, based on their adult run timing pattern. Sexually immature adult Snake River Basin summer steelheads enter the Columbia River from late June to October. Snake River 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 Snake River Basin steelhead typically spend two to three years in fresh water before they smolt and migrate to the ocean.

Status and Trends

Snake River 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 (2003a) identified 23 populations in the following six MPGs: Clearwater River, Grande Ronde River, Hells Canyon, Imnaha River, Lower Snake River, and Salmon River. Snake River Basin steelhead remain spatially well distributed in each of the six major geographic areas in the Snake River basin (Good 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. Snake River Basin steelhead were blocked from portions of the upper Snake River beginning in the late 1800s and culminating with the construction of Hells Canyon Dam in the 1960s. The Snake River Basin steelhead "B run" population levels remain particularly depressed. The ICBTRT has not completed a viability assessment for Snake River Basin steelhead.

Limited information on adult spawning escapement for specific tributary production areas for Snake River 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 5-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 Snake River supports approximately 63% of the total natural-origin production of steelhead in the Columbia River Basin. The current condition of Snake River Basin steelhead (Good et al. 2005) is summarized below:

There is uncertainty for wild populations given limited data for adult spawners in individual populations. Dam counts are currently 28% of interim recovery target for the Snake River 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 Snake River 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 Snake River Basin steelhead 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 et al. 2006).

South-Central California Coast Steelhead

Distribution

The South-Central California 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 ESU. Migration and spawn timing are similar to adjacent steelhead populations. There is little other life history information for steelhead in this ESU.

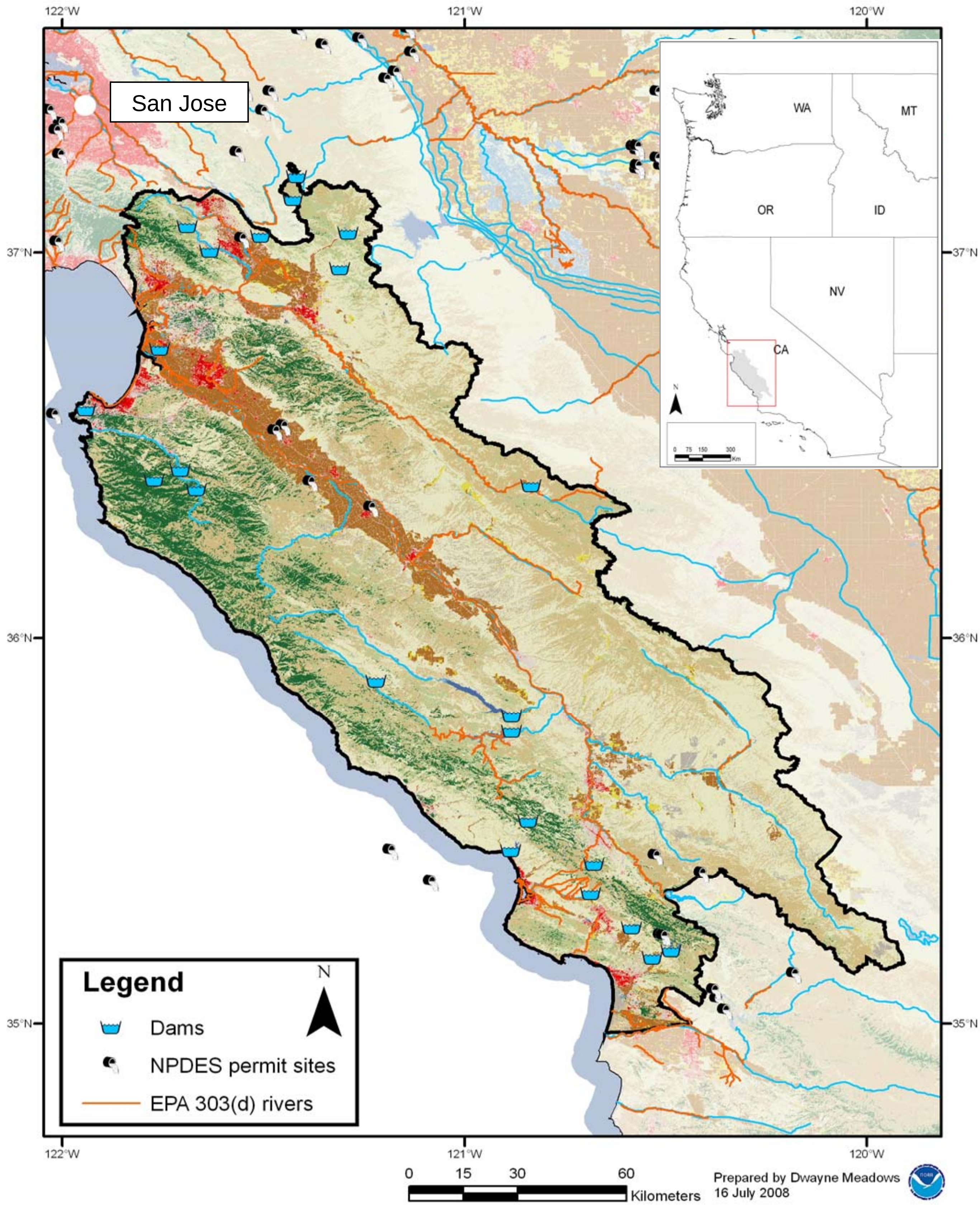


Figure 31. South Central California Coast steelhead distribution. The Legend for the Land Cover Class categories is found in Figure 7.

Status and Trends

South-Central California Coast 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 South-Central California Coast 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 et al. 2005). From 1991 the population increased from one adult, to 775 adults at San Clemente Dam. Good et al. (2006) 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 steelhead occupy rivers from the Santa Maria River to the U.S. – Mexico border (Figure 32).

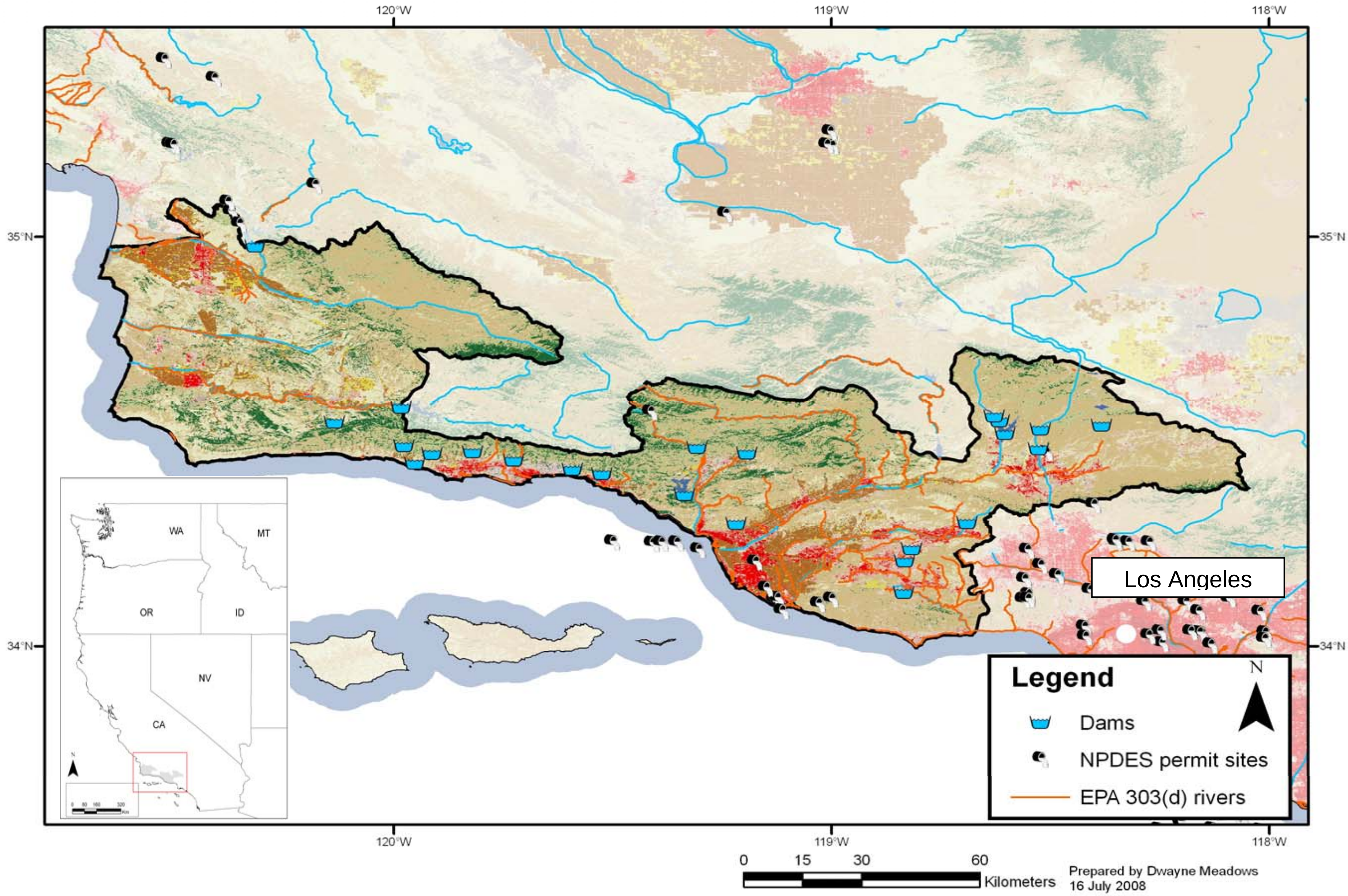


Figure 32. Southern California steelhead distribution. The Legend for the Land Cover Class categories is found in Figure 7.

Table 20 identifies populations within the Southern California Steelhead salmon ESU, their abundances, and hatchery input.

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

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

Life History

Migration and life history patterns of southern California steelhead are dependent on rainfall and streamflow (Moore 1980). Steelhead within this ESU 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 ESU.

Status and Trends

Southern California 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 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 Southern California steelhead ESU, including uncertainty on the metapopulation dynamics in the southern part of the ESU'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 21 identifies populations within the UCR Steelhead salmon ESU, their abundances, and hatchery input.

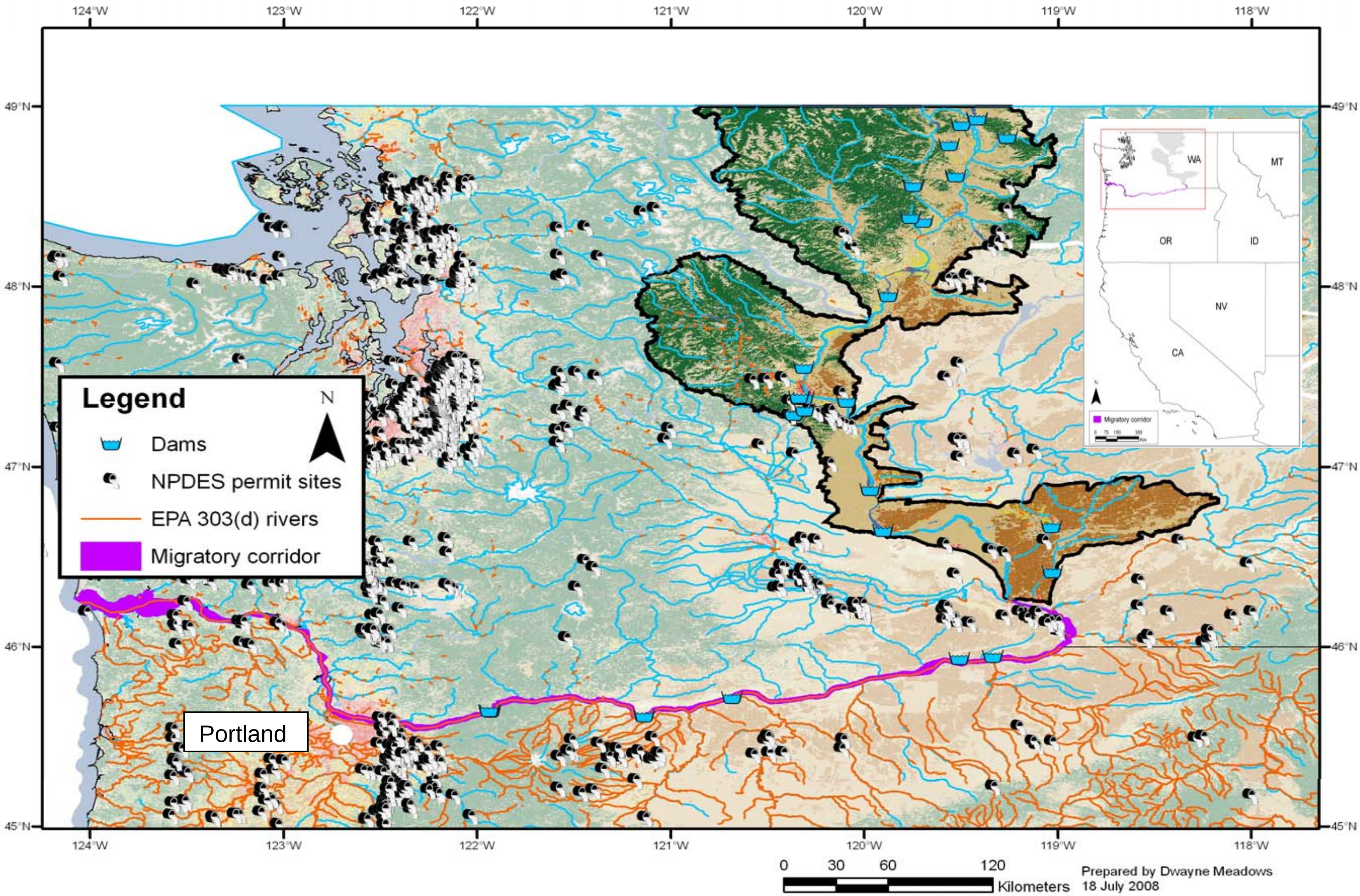


Figure 33. Upper Columbia River Steelhead distribution. The Legend for the Land Cover Class categories is found in Figure 7.

Table 21. Upper Columbia River Steelhead salmon populations, abundances, and hatchery contributions (Good et al. 2005).

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

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.

Status and Trends

UCR steelhead were originally listed as endangered in 1997 (62 FR 43937). Following a status review, they were reclassified to threatened on January 5, 2006 and then reinstated to endangered status per U.S. District Court decision in June 2007 (62 FR 43937). This DPS includes all naturally spawned anadromous steelhead populations below natural and manmade impassable barriers in streams in the Columbia River Basin upstream from the Yakima River, Washington, to the U.S.-Canada border, as well six artificial propagation programs: the Wenatchee River, Wells Hatchery (in the Methow and Okanogan Rivers), Winthrop NFH, Omak Creek, and the Ringold steelhead hatchery programs. The ICBTRT has identified five populations within this DPS: the Wenatchee River, Entiat River, Methow River, Okanogan Basin, and Crab Creek.

Returns of both hatchery and naturally produced steelhead to the upper Columbia River have increased in recent years. The average 1997 to 2001 return counted through the Priest Rapids fish ladder was approximately 12,900 fish. The average for the previous five years (1992 to 1996) was 7,800 fish. Abundance estimates of returning naturally produced UCR steelhead 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 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

Upper Willamette River 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 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 22 identifies populations within the Upper Willamette River Steelhead salmon ESU, their abundances, and hatchery input.

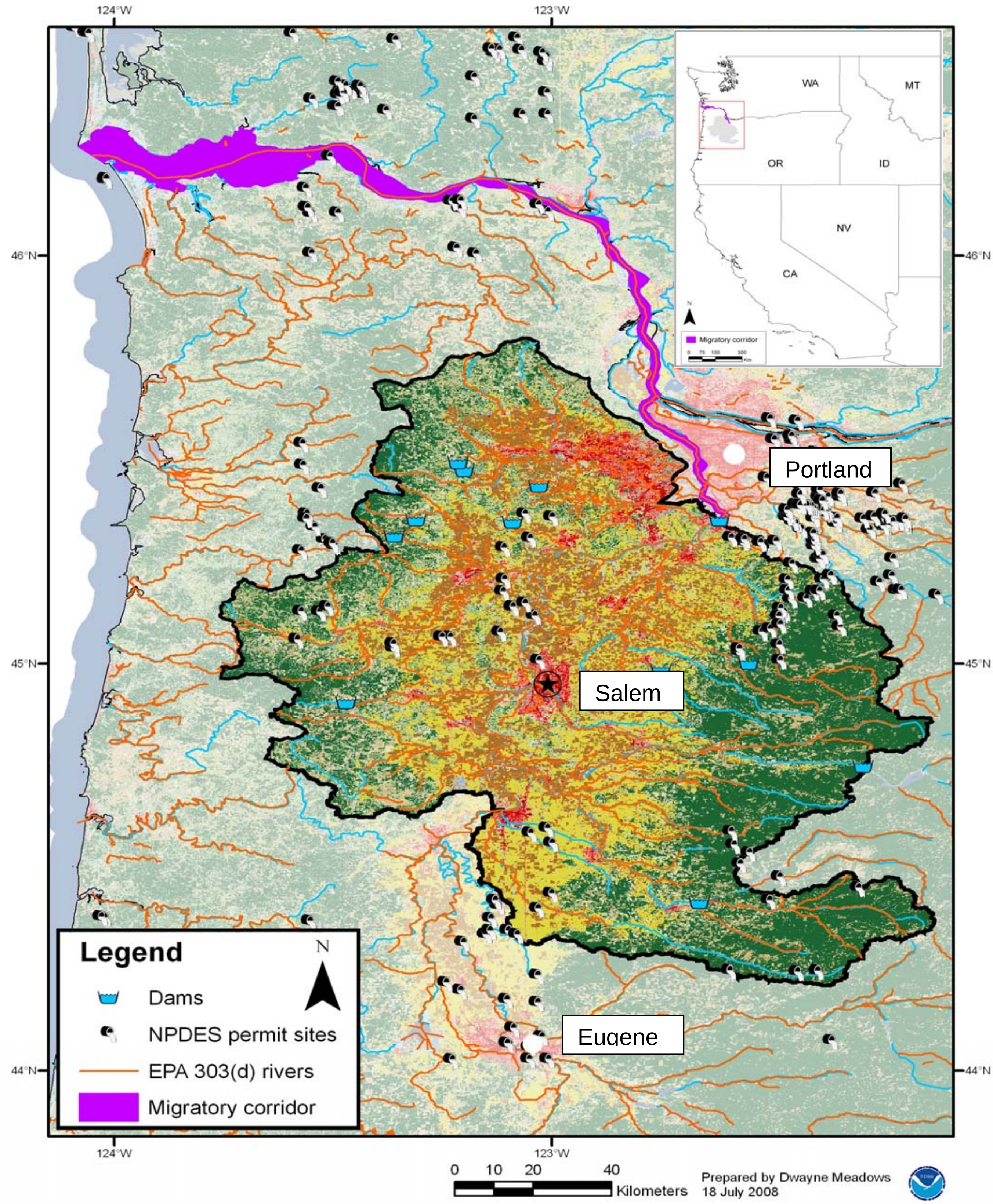


Figure 34. Upper Willamette River Steelhead distribution. The Legend for the Land Cover Class categories is found in Figure 7.

Table 22. Upper Willamette River Steelhead salmon populations, abundances, and hatchery contributions (Good 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 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 et al. 1996). Steelhead in the Upper Willamette River 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 et al. 1996).

Status and Trends

Upper Willamette River 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 Upper Willamette River 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 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 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 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 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 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 Upper Willamette River 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 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's 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 (1995), salmonid mortality rates range from 90 to 99%, depending on the species, the size at ocean entry, and the length of time spent in the ocean. Predation, inter- and intraspecific competition, food availability, smolt quality and health, and physical ocean conditions likely influence the survival of salmon in the marine environment (Brodeur et al. 2004). In freshwater rearing habitats, the natural mortality rate averages about 70% for all salmonid species (Bradford 1995b). Past studies in the Pacific Northwest suggest that the average freshwater survival rate (from egg to smolt) is 2 to 3% throughout the region (Bradford 1995b; Marshall and Britton 1990). A number of suspected causes contributing to natural mortality include parasites and/or disease, predation, water temperature, low water flow, wildland fire, and oceanographic features and climatic variability.

Parasites and/or Disease

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

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

IHN is a viral disease in many wild and farmed salmonid stocks in the Pacific Northwest. This disease affects rainbow/steelhead trout, cutthroat trout (*Salmo clarki*), brown trout (*Salmo trutta*), Atlantic salmon (*Salmo salar*), and Pacific salmon including Chinook, sockeye, chum, and coho. 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. On the Pacific coast of Canada, the louse-induced mortality of pink salmon is over 80% (Kroksek et al. 2007). *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). Based on the available information, the consequences of disease and parasitism are a concern. However, they do not appear as significant impediments to recovery of listed Pacific salmonids at this time.

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 (Ford and Ellis 2006; Hanson et al. 2005; Hard et al. 1992). Generally, harbor seals do not feed on salmonids as frequently as California sea lions (Pearcy 1997). California sea lions from the Ballard Locks in Seattle, Washington have been estimated to consume about 40% of the steelhead runs since 1985/1986 (Gustafson et al. 1997). In the Columbia River, salmonids may contribute substantially to sea lion diet at specific times and locations (Pearcy 1997). Spring Chinook and steelhead are subject to pinniped predation when they return to the estuary as adults [see NMFS 2006 *in* FCRPS (2008)]. Adult Chinook 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 (Gorge Winter Run MPG) 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 completed section 7 consultations on granting permits to the States of Idaho, Oregon, and Washington. under section 120 of the Marine Mammal Protection Act, for the lethal removal of certain individually identified California sea lions that prey on adult spring-run Chinook in the tail race of Bonneville Dam [see NMFS 2008d *in* FCRPS (2008)]. This action may increase the survival of adult Chinook salmon and steelhead.

Avian Predation

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

Antolos et al. (2006) 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 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 [see Roby et al. 2008 *in* FCRPS (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 and Neville 1995; Beamish et al. 1992; Pearcy 1992).

Wildland Fire

Wildland fires that are allowed to burn naturally in riparian or upland areas may benefit or harm aquatic species, depending on the degree of departure from natural fire regimes. Although most fires are small in size, large size fires increase the chances of adverse effects on aquatic species. Large fires that burn near the shores of streams and rivers can have biologically significant short-term effects. They include increased water temperatures, ash, nutrients, pH, sediment, toxic chemicals, and large woody debris (Buchwalter et al. 2004; Rinne 2004). Nevertheless, fire is also one of the dominant habitat-forming processes in mountain streams (Bisson et al. 2003). As a result, many large fires burning near streams can result in fish kills with the survivors actively moving downstream to avoid poor water quality conditions (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 et al. 2003), (Buchwalter et al. 2004; Minshall et al. 2001). Larger fires can also indirectly affect fish by altering water quality. Ash and smoke contribute to elevated ammonium, nitrate, phosphorous, potassium, and pH, which can remain elevated for up to four months after forest fires (Buchwalter et al. 2003).

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 (Hare et al. 1999; Mantua et al. 1997). 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 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 1995b). 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 (Hard et al. 1992). 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 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. 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 et al. 1992).

Climate change poses significant hazards to the survival and recovery of salmonids along the west coast. Changes in water temperature can change 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 et al. 2005; NMFS 2007). Earlier spring runoff and lower summer flows may 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 (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 across the action area. Two major issues, pesticide contamination and water temperature, are discussed. 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 chlorpyrifos, diazinon, and malathion 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. 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 2007d). We summarize the principal anthropogenic factors occurring in the environment that influence the current status of listed species within each region.

Baseline Pesticide Detections in Aquatic Environments

According to Gilliom et al. (2007), 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 U.S. Geological Survey National Water-Quality Assessment Program (NAWQA) 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 (Belden et al. 2007).

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 which 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 (Belden et al. 2007).

Additionally, more frequent detections and higher concentrations of insecticides occur in sampled urban streams (Belden et al. 2007). 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 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. Nonagricultural uses of chlorpyrifos and diazinon totaled about 5 million and 4 million lbs per year in 2001, respectively (Belden et al. 2007). 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 (Belden et al. 2007).

Another dimension of pesticides and degradates in the aquatic environment is their simultaneous occurrence as mixtures (Belden et al. 2007). Mixtures result from the use of different pesticides for multiple purposes within a watershed or groundwater recharge area. Pesticides generally occur more often 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 2 or more pesticides or degradates. About 70% and 20% of the time, streams had 5 or more and 10 or more pesticides or degradates, respectively (Belden et al. 2007). 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 2 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 (Belden et al. 2007).

NAWQA analysis of all detections indicates that more than 6,000 unique mixtures of 5 pesticides were detected in agricultural streams (Belden et al. 2007). 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 was present 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. Insecticides are typical constituents in mixtures are commonly found in 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 concentration 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 the 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 above for NPDES permits co-located within listed salmonid ESUs within the States of California, Idaho, Oregon, and Washington.

Baseline Water Temperature

Clean Water Act

Elevated temperature is considered a pollutant in most states with approved Water Quality Standards under the Federal Clean Water Act (CWA) of 1972. 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. Water bodies listed under 303(d) are those that are considered impaired or threatened by pollution.

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

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

Temperature is significant for the health of aquatic life. Water temperatures affect the distribution, health, and survival of native cold-blooded salmonids in the Pacific Northwest. These fish will experience adverse health effects when exposed to temperatures outside their optimal range. For listed Pacific salmonids, water temperature tolerance varies between species and life stages. Optimal temperatures for rearing salmonids range from 10ºC and 16ºC. In general, the increased exposure to stressful water temperatures and the reduction of suitable habitat caused by drought conditions reduce the abundance of salmon. Warm temperatures can reduce fecundity, 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 (McCullough 1999; Spence et al. 1996). Migrating adult salmonids and upstream migration can be delayed by excessively warm stream temperatures. Excessive stream temperatures may also negatively affect incubating and rearing salmonids (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 23).

Table 23. 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 1-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 24). Because the 303(d) list is limited to the subset of rivers tested, the chart values should be regarded as underestimates.

Table 24. 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	–	–	–	39
	Lower Columbia River	–	57	230	–	286
	Puget Sound	–	–	705	–	705
	Snake River Fall - Run	–	610	247	400	1257
	Upper Williamette River	–	2468	–	–	2468
Chum Salmon	Columbia River	–	57	225	–	282
	Hood Canal Summer - Run	–	–	90	–	90
Coho Salmon	Central California Coast	39	–	–	–	39
	Lower Columbia River	–	292	234	–	525
	Oregon Coast	–	3716	–	–	3716
Sockeye Salmon	Ozette Lake	–	–	5	–	5
	Snake River	–	–	–	0	0
Steelhead Trout	Central California Coast	0	–	–	–	0
	California Central Valley	0	–	–	–	0
	Lower Columbia River	–	201	169	–	371
	Middle Columbia River	–	3519	386	–	3905
	Northern California	39	–	–	–	39
	Puget Sound	–	–	705	–	705
	Snake River	–	991	247	738	1975
	Southern California	0	–	–	–	0
	Upper Columbia River	–	–	282	–	282
	Upper Williamette River	–	1668	–	–	1668

*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).

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 can be the result of a variety of factors, including wastewater discharge, decreased water flow, minimal shading by riparian areas, and climatic variation.

Southwest Coast Region

The basins in this section occur in the State of California and the southern parts of the State of Oregon. Tables 25 and 26 show land area in km² for each ESU /DPS located in the Southwest Coast Region.

Table 25. Area of land use categories within the range Chinook 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) (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
Herbaceuous	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 Herbaceuous 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 26. Area of Land Use Categories within the Range of Steelhead DPSs (km²). Land cover image data were taken from Multi-Resolution Land Characteristics (MRLC) Consortium, a consortium of nine federal agencies (USGS, EPA, USFS, NOAA, NASA, BLM, NPS, NRCS and USFWS) (National Land Cover Data 2001). Land cover class definitions are available at: http://www.mrlc.gov/nlcd_definitions.php

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
Herbaceuous	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 Herbaceuous 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. Essentially, the Southwest Coast region encompasses all Pacific Coast rivers south of Cape Blanco, Oregon through southern California. The Cape Blanco area marks a major biogeographic boundary. NMFS has identified the Cape Blanco area as an ESU/DPS 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 27).

Table 27. 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 28). Overall, the most developed watersheds are the Santa Ana, Russian, and Santa Margarita rivers. The Santa Ana watershed encompasses portions of San Bernardino, Los Angeles, Riverside, and Orange counties. About 50% of the coastal subbasin in the Santa Ana watershed is dominated by urban land uses and the population density is about 1,500 people per square mile. When steep and undevelopable lands are excluded from this area, the population density in the watershed is about 3,000 people per square mile. However, the most densely populated portion of the basin is near the City of Santa Ana. Here, the population density reaches 20,000 people per square mile (Belitz et al. 2004; Burton et al. 1998). The basin is home to nearly 5 million people. However, this population is projected to increase two-fold in the next 50 years (Belitz et al. 2004; Burton et al. 1998).

Table 28. Land uses and population density in several southwest coast watersheds (Carter and Resh 2005).

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

The Santa Ana watershed is the most heavily developed watershed in the region. As a watershed becomes urbanized, population increases and changes occur in stream habitat, water chemistry, and the biota (plants and animals) that live there. The most obvious effect of urbanization is the loss of natural vegetation which results in an increase in impervious cover and dramatic changes to the natural hydrology of urban streams. Urbanization generally results in land clearing, soil compaction, modification and/or loss of riparian buffers, and modifications to natural drainage features (Richter 2002). The increased impervious cover in urban areas leads to increased volumes of runoff, increased peak flows and flow duration, and greater stream velocity during storm events. Other impacts include loss of large woody debris, increased bank erosion and bed scour, changes in sediment loadings, increased stream temperature, and decreased base flow. Thus, decreased quantity and quality of large woody debris and modified hydrology reduce and degrade salmonid rearing habitat.

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 to treat greater quantities of human waste and combined sewer /stormwater overflows (CSOs). 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 2007a). 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 2007a). The 2006 303(d) list identifies water bodies listed due to the presence of specific pollutants, including chlorpyrifos, diazinon, malathion, and elevated temperature (Table 29). See species ESU/DPS maps for NPDES permits and 303(d) waters co-located within listed salmonid ESUs in California.

Table 29. California's 2006 Section 303(d) List of Water Quality Limited Segments: segments listed for exceeding temperature, chlorpyrifos, diazinon, and malathion limits (CEPA 2007b).

Pollutant	Estuary Acres Affected	River / Stream Miles Affected	# Water Bodies
Temperature	-	16,907.2	41
Chlorpyrifos	43,614.0	610.3	44
Diazinon	44,738.0	1,299.2	94
Malathion	-	49.0	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 polychlorinated biphenols (PCBs), nickel, selenium, cadmium, pesticides, 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 et al. 2004). Stormwater and agricultural runoff frequently contain pesticides, fertilizers, sediments, nutrients, pathogenic bacteria, and other chemical pollutants to waterways and degrade water quality. The above inputs have resulted in elevated concentrations of nitrates and pesticides in surface waters of the basin. Nitrates and pesticides were more frequently detected here than in other national NAWQA sites (Belitz et al. 2004).

Additionally, Belitz et al. (2004) found that pesticides and volatile organic compounds (VOCs) were frequently detected in surface and ground water in the Santa Ana Basin. Of the 103 pesticides and degradates routinely analyzed for in surface and ground water, 58 were detected. Pesticides included diuron, diazinon, carbaryl, chlorpyrifos, lindane, malathion, and chlorothalonil. 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 conducted by the USGS in the mid-1990s on water quality within the San Joaquin-Tulare basins detected 49 pesticides in the mainstem and three subbasins. Pesticides included the herbicides simazine, dacthal, metolachlor, and EPTC (Eptam), and the insecticides diazinon and chlorpyrifos. Specifically, 22 pesticides were detected in 20% of the samples and concentrations of 7 pesticides exceeded criteria for aquatic life (Dubrovsky et al. 1998). These pesticides include diuron, trifluralin, azinphos-methyl, carbaryl, chlorpyrifos, diazinon, and malathion. Forty percent of these exceedances were attributed solely to diazinon. Organochlorine insecticides in bed sediment and tissues of fish or clams were also detected. They include DDT and toxaphene. Levels at some sites were among the highest in the nation. Concentrations of trace elements in bed sediment generally were higher than concentrations found in other NAWQA study units (Dubrovsky et al. 1998).

Sacramento River Basin: NAWQA analysis
Another study conducted by the USGS from 1996-1998 within the Sacramento River Basin detected up to 24 out of 47 pesticides in surface waters (Domagalski 2000). Pesticides included thiobencarb, carbofuran, molinate, simazine, metolachlor, and dacthal, chlorpyrifos, carbaryl, and diazinon. These pesticides were applied in agricultural and urban settings. 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.

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. Of these, roughly 1% 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 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). While about 75% of the runoff occurs in basins in the northern half of California, 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 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 12 million plus. 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 are also harvested. Illegal poaching of abalone, including endangered white abalone, continues to be of concern. Illegal poaching is influenced by the demand for abalone in local restaurants, seafood markets, and international businesses (Daniels and Floren 1998). 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 et al. 1989). Wilcove 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 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 discharch 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 (Markleet al. 2005). These options included: 1) join a Coalition Group approved by the CVRWQCB, 2) file for an Individual Discharger Conditional Waiver, and 3) comply with zero discharge regulation (Markle et al. 2005). Many growers opted to join a Coalition as the other options were more costly. Coalition Groups were charged with completing two reports – a Watershed Evaluation Report and a Monitoring and Reporting Plan. The Watershed Evaluation Report 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, resulting 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 as well as other products (pyrethroids and sulfur).

In 2006 California Department of Pesticide Regulation's (CDPR) put limitations on dormant spay application of certain insecticides in orchards, including diazinon and chlorpyrifos, to adequately protect aquatic life.

The CDPR published voluntary interim measures for mitigating the potential impacts of pesticide useage to listed species. Measures that apply to chlorpyrifos, diazinon and malathion use in salmonid habitat are:

1) Do not use in currently occupied habitat
2) 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.
3) Conduct irrigations efficiently to prevent excessive loss of irrigation waters through run-off. Schedule irrigations and pesticide applications to maximize the interval of time between the pesticide application and the first subsequent irrigation. Allow at least 24 hours between application of pesticides listed in this bulletin and any irrigation that results in surface run-off into natural waters. Time applications to allow sprays to dry prior to rain or sprinkler irrigations. Do not make aerial applications while irrigation water is on the field unless surface runoff is contained for 72 hours following the application.
4) 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 30, Table31, and Table 32 show the types and areas of land use within each salmonid ESU/DPS.

Table 30. Area of land use categories within Chinook 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

		Chinook Salmon					
Landcover Type	code	Lower Columbia River	Upper Columbia River Spring Run	Puget Sound	Snake River Fall Run	Snake River Spring/ Summer Run	Upper Williamette 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
Herbaceuous	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 Herbaceuous 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 31. 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

		Chum Salmon		Coho Salmon	
Landcover Type	code	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
Herbaceuous	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 Herbaceuous 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 32. 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 Williamette 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
Herbaceuous	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 Herbaceuous 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 33) for a description of select Columbia River tributaries). The Snake River is the largest tributary at more than 1,000 miles long. The headwaters of the Snake River originate in Yellowstone National Park, Wyoming. The second largest tributary is the Willamette River in Oregon (Hinck et al. 2004; Kammerer 1990). The Willamette River is also the 19th largest river in the nation in terms of average annual discharge (Kammerer 1990). The basins drain portions of the Rocky Mountains, Bitteroot Range, and the Cascade Range.

Table 33. 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 (Stanfordet al. 2005). Salmonids within the basin include Chinook, chum, coho, 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 et al. 2004). See
Table 34 for a summary of land uses and population densities in several subbasins within the Columbia River watershed (data from Stanford et al. 2005).

Table 34 . Land uses and population density in select tributaries of the Columbia River (Stanford et al. 2005)

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

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. Critical ecological connectivity (mainstem to tributaries and riparian floodplains) has been disconnected by dams and associated activities such as floodplain deforestation and urbanization. The most productive floodplains of the watershed are either flooded by hydropower dams or dewatered by irrigation diversions. Portions of the basin are also subject to impacts from cattle grazing and irrigation withdrawals. 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 meters 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 et al. 2005).

Agriculture and Ranching

Agriculture, ranching, and related services in the Pacific Northwest provide employment for 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 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 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, crop-type, and basin location are some of the variables that influence the distribution and frequency of pesticides within a tributary. Detection frequencies for a particular pesticide can vary widely. One study conducted by the USGS between May 1999 and January 2000 in the surface waters of Yakima Basin detected 25 pesticide compounds (Ebbert and Embry 2001). Atrazine was the most widely detected herbicide and azinphos-methyl was the most widely detected insecticide. Other detected compounds include simazine, terbacil, trifluralin; deethylatrazine, carbaryl, diazinon, malathion, and DDE. In addition to current use-chemicals legacy chemicals continue to pose a serious problem to water quality and fish communities despite their ban in the 1970s and 1980s (Hinck et al. 2004).

Fish and macroinvertebrate communities exhibit an almost linear decline in condition as the level of agriculture intensity increases within a basin (Cuffney et al. 1997; Fuhrer et al. 2004). A study conducted in the late 1990s examined 11 species of fish, including anadromous and resident fish collected throughout the basin, for a suite of 132 contaminants. They included 51 semi-volatile chemicals, 26 pesticides, 18 metals, 7 PCBs, 20 dioxins, and 10 furans. Sampled fish tissues revealed PCBs, metals, chlorinated dioxins and furans (products of wood pulp bleaching operations), and other contaminants.

Yakima River Basin: NAWQA analysis

The Yakima River Basin is one of the most agriculturally productive areas in the U.S. (Fuhrer et al. 2004). Croplands within the Yakima Basin account for about 16% of the total basin area of which 77% is irrigated. The extensive irrigation-water delivery and drainage system in the Yakima River Basin greatly controls water quality conditions and aquatic health in agricultural streams, drains, and the Yakima River (Fuhrer et al. 2004). From 1999 to 2000, the USGS conducted a NAWQA study in the Yakima River Basin. Fuhrer et al. (2004) reported that nitrate and orthophosphate were the dominant forms of nitrogen and phosphorus found in the Yakima River and its agricultural tributaries. Arsenic, a known human carcinogen, was also detected in agricultural drains at elevated concentrations 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. The median and maximum number of chemicals in a mixture was 8 and 26, respectively (Fuhrer et al. 2004). The herbicide 2,4-D, occurred most often in the mixtures, along with azinphos-methyl, the most heavily applied pesticide, and atrazine, one of the most aquatic mobile pesticides (Fuhrer et al. 2004).

However, the most frequently detected pesticides in the Yakima River Basin are total DDT, and its breakdown products DDE, dichloro diphenyl dichloroethane (DDD), and dieldrin (Fuhrer et al. 2004; Johnson and Newman 1983; Joy 2002). 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 50 pesticides were detected in streams and 10 pesticides exceeded criteria established by the EPA for the protection of freshwater aquatic life from chronic toxicity. Atrazine, simazine, metolachlor, deethylatrazine, diuron, and diazinon were detected in more than one-half of stream samples. 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 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.

Urban and Industrial Development
The largest urban area in the basin is the greater Portland metropolitan area, located at the mouth of the Willamette River. Portland's population exceeds 500,000 (Hinck et al. 2004). Although the basin's land cover is about 8% of the U.S. total land mass, its' human population is one-third the national average (about 1.2% of the U.S. population) (Hinck et al. 2004).

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

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

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 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 (Anderson et al. 2007; Stanford et al. 2005). According to the U.S. Bureau of Mines, there are about 14,000 inactive or abandoned mines within the Columbia River Basin. Of these, nearly 200 pose a potential hazard to the environment (Quigley 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 Bureau of Reclamation 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.

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

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

Development of the Pacific Northwest regional hydroelectric power system, dating to the early 20th century, has had profound effects on the ecosystems of the Columbia River Basin (ISG 1996). These effects have been especially adverse to the survival of anadromous salmonids. The construction of the FCRPS modified migratory habitat of adult and juvenile salmonids. In many cases, the FCRPS presented a complete barrier to habitat access for salmonids. 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. Dams have also 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 of habitat from migrating salmon and steelhead (Wydoski and Whitney 1979). 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 large woody debris 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. Consequently, estuary dynamics have changed substantially. 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. They 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 et al. 2007b, 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 with the construction and operation of surface bypass routes at Lower Granite, Ice Harbor, and Bonneville dams and other configuration improvements (see Corps et al. 2007a *in* FCRPS (2008).

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. (2007b *in* FCRPS 2008) estimated that hydropower configuration and operational improvements implemented in 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 et al. 2005).

The impact of artificial propagation on the total production of Pacific salmon and steelhead has been extensive (Hard et al. 1992). Hatchery practices, among other factors, are a contributing factor to the 90% reduction in natural coho salmon runs in the lower Columbia River over the past 30 years (Flagg et al. 1995). Past hatchery and stocking practices have resulted in the transplantation of salmon and steelhead from non-native basins. The impacts of these hatchery practices are largely unknown. Adverse effects of these practices likely included: loss of genetic variability within and among populations (Busack 1990; Hard et al. 1992; Reisenbichler 1997; Riggs 1990), disease transfer, increased competition for food, habitat, or mates, increased predation, altered migration, and the displacement of natural fish (Fresh 1997; Hard et al. 1992; Steward and Bjornn 1990). Species with extended freshwater residence are likely to face higher risk of domestication, predation, or altered migration than are species that spend only a brief time in fresh water (Hard et al. 1992). Nonetheless, artificial propagation may also contribute to the conservation of listed salmon and steelhead. However, it is unclear whether or how much artificial propagation during the recovery process will compromise the distinctiveness of natural populations (Hard et al. 1992).

The states of Oregon and 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 their work on LCR populations and provided their 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 (ODFW 2007; WDFW 2005). The program is designed to achieve HSRG objectives related to controlling the number of hatchery-origin fish on the spawning grounds and in the hatchery broodstock.

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

Commercial, Recreational, and Subsistence Fishing

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

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

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

Non-Indian fisheries in the lower Columbia River are limited to a harvest rate of 1%. Treaty Indian fisheries are limited to a harvest rate of 5 to 7%, depending on the run size of upriver Snake River sockeye stocks. Actual harvest rates over the last 10 years have ranged from 0 to 0.9%, and 2.8 to 6.1%, respectively (see TAC 2008, Table 15 *in* 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 et al. 2004). It is clear that the introduction of non-native species has changed the environment, though whether these changes will impact salmonid populations is uncertain (Sytsma et al. 2004).

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, subestuaries (tidally influenced portions of river and stream mouths), sand spits, beaches and backshore, banks and bluffs, and marine riparian vegetation. These habitats provide critical functions such as primary food production and support habitat for invertebrates, fish, birds, and other wildlife.

Major rivers draining to Puget Sound from the Cascade Mountains include the Skagit, Snohomish, Nooksack, Puyallup, and Green rivers, as well as the Lake Washington/Cedar River watershed. Major rivers from the Olympic Mountains include the Hamma Hamma, the Duckabush, the Quilcene, and the Skokomish rivers. Numerous other smaller rivers drain to the Sound, many of which are significant salmonid production areas despite their small size.

The Puget Sound basin is home to more than 200 fish and 140 mammalian species. Salmonids within the region include coho, Chinook, sockeye, chum, and pink salmon, kokanee, steelhead, rainbow, cutthroat, and bull trout (Kruckeberg 1991; Wydoski and Whitney 1979). Important commercial fishes include the five Pacific salmon and several rockfish species. A number of introduced species occur within the region, including brown and brook trout, Atlantic salmon, bass, tunicates (sea squirts), and a saltmarsh grass (*Spartina* spp.). Estimates suggest that over 90 species have been intentionally or accidentally introduced in the region (Ruckelshaus and McClure 2007). At present, over 40 species in the region are listed as threatened and endangered under the ESA.

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

An indication of this sensitivity occurs in Pacific herring, one of Puget Sound's keystone forage fish species (Collier et al. 2006). These fish spend almost all of their lives in pelagic waters and feed at the lower end of the food chain. Pacific herring should be among the least contaminated of fish species. However, monitoring has shown that herring from the main basins of Puget Sound have higher body burdens of persistent chemicals (e.g., PCBs) compared to herring from the severely contaminated Baltic Sea. Thus, the pelagic food web of Puget Sound appears to be more seriously contaminated than previously anticipated.

Chinook salmon that are resident in Puget Sound (a result of hatchery practices and natural migration patterns) are several times more contaminated with persistent bioaccumulative contaminants than other salmon populations along the West Coast (Collier et al. 2006). Because of associated human health concerns, fish consumption guidelines for Puget Sound salmon are under review by the Washington State Department of Health.

Extremely high levels of chemical contaminants are also found in Puget Sound's top predators, including harbor seals and ESA-listed southern resident killer whales (Collier et al. 2006). In addition to carrying elevated loads of toxic chemicals in their tissues, Puget Sound's biota are also showing 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 to 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, pharmaceuticals, nutrients (phosphorus and nitrogen), and sediment (Table 35). 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 35. 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	Natural and synthetic hormones	Ethinyl estradiol	hospitals, dental facilities,

personal care products	soaps and detergents	Nonylphenol	residences, municipal and industrial waste water discharges
Polyaromatic hydrocarbons (PAHs)	Tricylic PAHs	Phenanthrene	fossil fuel combustion, oil and gasoline leaks, highway runoff, creosote-treated wood
Industrial chemicals	PCBs PBDEs Dioxins	Penta-PBDE	utility infrastructure, flame retardants, electronic equipment

Many other metals have been found in elevated concentrations in urban stream sediments including arsenic, iron, boron, cobalt, silver, strontium, rubidium, antimony, scandium, molybdenum, lithium and tin (Wheeler et al. 2005). The concentration, storage, and transport of metals in urban streams are connected to particulate organic matter content and sediment characteristics. Organic matter has a high binding capacity for metals and both bed and suspended sediments with high organic matter content frequently exhibit 50-7,500 times higher concentrations of zinc, lead, chromium, copper, mercury, and cadmium than sediments with lower organic matter content.

Although urban areas occupy only 2% of the Pacific Northwest land base, the impacts of urbanization on aquatic ecosystems are severe and long lasting (Spence et al. 1996). O'Neill et al. (2006) found that Chinook salmon returning to Puget Sound had significantly higher concentrations of PCBs and PBDEs compared to other Pacific coast salmon populations. Furthermore, Chinook salmon that resided in Puget Sound in the winter rather than migrate to the Pacific Ocean (residents) had the highest concentrations of POPs, followed by Puget Sound fish populations believed to be more ocean-reared. Fall Chinook from Puget Sound have a more localized marine distribution in Puget Sound and the Georgia Basin than other populations of Chinook 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.

In addition to POPs, endocrine disruptors (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 are 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 et al. 2005). Urban stream fish communities have lower overall abundance, diversity, taxa richness and are dominated by pollution tolerant species. Lead content in fish tissue is higher in urban areas. Furthermore, the proximity of urban streams to humans increases the risk of non-native species introduction and establishment. Thirty-nine non-native species were collected in Puget Sound during the 1998 Puget Sound Expedition Rapid Assessment Survey (Brennan et al. 2004). Lake Washington, located within a highly urban area, has 15 non-native species identified (Ajawani 1956).

PAH compounds also have distinct and specific effects on fish at early life history stages (Incardona et al. 2004). PAHs tend to adsorb to organic or inorganic matter in sediments, where they can be trapped in long-term reservoirs (Johnson et al. 2002). Only a portion of sediment-adsorbed PAHs are readily bioavailable to marine organisms, but there is substantial uptake of these compounds by resident benthic fish through the diet, through exposure to contaminated water in the benthic boundary layer, and through direct contact with sediment. Benthic invertebrate prey are a particularly important source of PAH exposure for marine fishes, as PAHs are bioaccumulated in many invertebrate species (Meador et al. 1995; Varanasi et al. 1989; Varanasi et al. 1992).

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

Habitat Loss

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 are also experiencing increasing effects from industrial and urban causes. By 1980, an estimated 27,180 acres of intertidal or shore wetlands had been lost at 11 deltas in Puget Sound (Bortleson et al. 1980). Tidal wetlands in Puget Sound amount to roughly 18% of their historical extent (Collins and Sheikh 2005). Coastal marshes close to seaports and population centers have been especially vulnerable to conversion with losses of 50-90%. By 1980, an estimated 27,180 acres of intertidal or shore wetlands had been lost at eleven deltas in Puget Sound (Bortleson et al. 1980). More recently, tidal wetlands in Puget Sound amount to about 17-19% of their historical extent (Collins and Sheikh 2005). Coastal marshes close to seaports and population centers have been especially vulnerable to conversion with losses of 50-90% common for individual estuaries. Salmon use freshwater and estuarine wetlands for physiological transition to and from saltwater and rearing habitat, the conversions and losses of Pacific Northwest wetlands constitute a major impact. Salmon use marine nearshore areas for rearing and migration, with juveniles using shallow shoreline habitats (Brennan et al. 2004).

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 36.

Table 36. Pollutants of Concern in Puget Sound (PSAT 2005)

Pollutant	Sources
Heavy Metals: Pb, Hg, Cu, and others	vehicles, batteries, paints, dyes, stormwater runoff, spills, pipes.
Organic Compounds: Polycyclic aromatic hydrocarbons (PAHs)	Burning of petroleum, coal, oil spills, leaking underground fuel tanks, creosote, asphalt.
Polychlorinated biphenyls (PCBs)	Solvents electrical coolants and lubricants, pesticides, herbicides, treated wood.
Dioxins, Furans	Byproducts of industrial processes.
Dichloro-diphenyl-trichloroethane (DDTs)	Chlorinated pesticides.
Phthalates	Plastic materials, soaps, and other personal care products. Many of these compounds are in wastewater from sewage treatment plants.
Polybrominated diphenyl ethers (PBDEs)	PBDEs are added to a wide range of textiles and plastics as a flame retardant. They easily leach from these materials and have been found throughout the environment and in human breast milk.

Puget Sound Basin: NAWQA analysis

The USGS sampled waters in the Puget Sound Basin between 1996 and 1998. Ebbert et al. (2006) 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. The insecticide diazinon was also frequently detected in urban streams at concentrations that exceeded EPA guidelines for protecting aquatic life (Bortleson and Ebbert 2000). However, 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, however, are also 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 (NMA 2007; Palmisano et al. 1993). Metal mining for all metals (zinc, copper, lead, silver, and gold) peaked between 1940 and 1970 (Palmisano et al. 1993). Today, construction sand and gravel, Portland cement, and crushed stone are the predominant materials mined. Where sand and gravel is mined from riverbeds (gravel bars and floodplains) it may result in changes in channel elevations and patterns, instream sediment loads, and seriously alter instream habitat. In some cases, instream or floodplain mining has resulted in large scale river avulsions. The effect of mining in a stream or reach depends upon the rate of harvest and the natural rate of replenishment, as well as flood and precipitation conditions during or after the mining operations.

Artificial Propagation

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

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

Overall, the use of Green River stock throughout much of the extensive hatchery network in this ESU may reduce the genetic diversity and fitness of naturally spawning populations (Good et al. 2005).

Hydromodification Projects

More than 20 dams occur within the region's rivers and overlap with the distribution of salmonids. A number of basins contain water withdrawal projects or small impoundments that can impede migrating salmon. The resultant impact of these and land use changes (forest cover loss and impervious surface increases) has been a significant modification in the seasonal flow patterns of area rivers and streams, and the volume and quality of water delivered to Puget Sound waters. Several rivers have been 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 (Ruckelshaus and McClure 2007; Wunderlich et al. 1994). Estimates suggest that nearly 400,000 salmon could begin using the basin within 30 years after the dams are removed (PSAT 2007).

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% 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.

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.

Harvest impacts on Puget Sound Chinook salmon populations average 75% in the earliest five years of data availability and have dropped to an average of 44% in the most recent five year period (Good et al. 2005). Populations in Puget Sound have not experienced the strong increases in numbers seen in the late 1990s in many other ESUs. Although more populations have increased than decreased since the last BRT assessment, after adjusting for changes in harvest rates, trends in productivity are less favorable. Most populations are relatively small, and recent abundance within the ESU is only a small fraction of estimated historic run size.

Atmospheric deposition
Pesticides were detected in wet deposition (rain) (Capel et al. 1998), and snow samples from Mount Rainier National Park, Washington (Hageman 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 (Belitz et al. 2004; Carter and Resh 2005; Kagan et al. 1999).

Land Use
The rugged topography of the western Olympic Peninsula and the Oregon Coastal Range has limited the development of dense population centers. For instance, the Nehalem

River and the Umpqua River basins consist of less than 1% urban land uses. Most basins in this region have long been exploited for timber production, and are still dominated by forestlands. In Washington State, roughly 90% of the coastal region is forested (Palmisano et al. 1993). Approximately 92% of the Nehalem River basin is forested, with only 4% considered agricultural (Belitz et al. 2004). Similarly, in the Umpqua River basin, about 86% is forested land, 5% agriculture, and 0.5% is considered urban lands. Roughly half the basin is under Federal management (Carter and Resh 2005).

Clackamas River Basin: NAWQA assessment
A study conducted by the USGS from 2000-2005 on water quality in the lower Clackamas River basin detected 63 compounds (Carpenter et al. 2008). A total of 119 samples were collected from 30 sites over a six-year period. Detected compounds include 33 herbicides, 15 insecticides, 6 fungicides, and 9 pesticide degradation products. Atrazine and simazine were detected in about half of the samples. Other high-use herbicides such as glyphosphate, triclopyr, 2,4,-D, and metolachlor were also frequently detected, particularly in the lower-basin tributaries.

Pesticides were detected in all eight of the lower basin tributaries sampled. The highest pesticide (loads) amounts (for 15-18 pesticides) were found in Deep and Rock Creeks. These medium-sized streams drain a mix of agricultural land (row crops and nurseries), pastureland, and rural residential areas. Other sites having relatively high pesticide yields included middle Rock Creek and upper Noyer Creek. Both sites drain basins having nurseries, pasture, and rural residential land (Carpenter et al. 2008).

According to Carpenter et al. (2008), concentrations of diazinon, chlorpyrifos, and azinphos-methyl, and *p,p'*-DDE exceeded EPA aquatic-life benchmarks in six creeks. Additionally, some of the pesticides detected do not have benchmarks for evaluation including benomyl, metalaxyl, imidacloprid, 3,4 dichloroaniline (a diuron degradate), and AMPA (a glyphosate degradate). These pesticides were occasionally detected at concentrations ranging from 1.5 to 5.7 ug/L.

Twenty-six pesticides and degradates were detected in 39 samples collected from the Clackamas River mainstem. At least one pesticide was detected in 65% of samples, with an average of two to three pesticides per sample. These compounds typically occurred at much lower concentrations than those detected in the lower-basin tributaries.

While most of the 51 current use pesticides detected have multiple uses, 94% can be used on agricultural crops. About 92% can be used on nursery or floriculture crops. About one-half are commonly used on either lawns and landscaping in urban areas (57%), on golf courses (49%), and along roads and right-of-ways (45%). Some pesticides can also be used on forestland (7%).

Agriculture. According to Carpenter et al. (2008), Clackamas County has about 100,000 acres of agricultural land. In the Clackamas River basin, agricultural land is concentrated on the high plateau between the Clackamas and Sandy Rivers. Some agricultural land is also located next to or within the floodplain of the Clackamas River. Clackamas County is one of the top Christmas tree producing counties in the U.S. About 18 herbicides, 12 insecticides, and 4 fungicides are used on Christmas trees in Oregon. Although a great diversity of crops are grown, pastureland, hay fields (mostly alfalfa), nurseries, and greenhouses make up 65% of the agricultural land in the basin.

In 2002, there were over 13,000 acres of nursery and floriculture land in Clackamas County (NASS 2002). A survey of nursery and floriculture operations reported pesticide useage in six states: California, Florida, Michigan, Orgeon, Pennsylvania, and Texas. About 275 herbicides, insecticides, and fungicides were applied to nursery and floriculture crops during 2003 (NASS 2004). The number of unique active ingredients used in these states increased to 374 by 2006 (NASS 2007). Pesticide applications occur in open areas and inside greenhouses.

Urban uses. About 55% of pesticides detected in the Clackamas River basin have urban uses. Several herbicides are applied along fences utility lines, roads and other right-of-ways in urban areas. Many urban-use pesticides were detected in the Clackamas River basin, including atrazine, metolachlor, simazine, prometon, diuron, and 2,4-D. These were the most common herbicides detected in urban steams nationwide (Gilliom et al. 2006; USGS 1999).

Golf courses. The extent of pesticide use on golf courses in the Clackamas River basin is unknown. Six golf courses are located within the drainage basin, and turf are treated for various fungal, insect, and weed pests. About 50% of the pesticides detected in the Clackamas River basin have been reported for golf courses.

Hood River Basin

The Hood River Basin ranks fourth in the state of Oregon in total agricultural pesticide usage (Jenkins et al. 2004). About 61 active ingredients, totaling 1.1 million lbs, are applied annually to roughly 21,000 acres. Of the top 10, three are organophosphate insecticides. Over 14,000 lbs of chlorpyrifos are applied to crops within Hood River basin annually. Lime sulfur and oil account for nearly ¾ of the annual pesticide usage. The land in Hood River basin is used to grow five crops: alfalfa, apples, cherries, grapes and pears.

The Hood 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 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 in one or more surface-water samples exceed freshwater-chronic criteria for the protection of aquatic life in some samples. They include the herbicide triallate and five insecticides (azinphos-methyl, chlorpyrifos, diazinon, *gamma*-HCH, and parathion). Chlorpyrifos was detected in 9% of samples, exceeding freshwater-chronic criteria in 4 samples. Diazinon was detected in 4% if samples, but only exceeded freshwater-chronic criteria once. Malathion never exceeded the concentration of 0.1 μg/L, but was detected in 2% of samples.

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 7 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.

Stream habitat degradation in Columbia Central Plateau is relatively high. A total of 16 sites were evaluated, all of which showed signs of degradation. Streams in this area have an average of 20% canopy cover and 70% bank erosion. Fish communities can be influenced by multiple factors, including pesticides, increased aquatic plant growth due to nutrients, reduced riparian habitat, and sediment runoff form agricultural practices. The two sites with the most impacted fish communities were a wastewater-dominated urban stream and a large dryland farming stream. Small dryland streams associated with spring systems contained the most trout. Only six sites were included in the fish community analysis, one of which was a highly degraded stream site. The remaining five were ranked between the 25^{th} and 50^{th} percentile of national NAWQA data.

Mining

Oregon is ranked 35^{th} nationally in total nonfuel mineral production value in 2004. In that same year, Washington was ranked 13^{th} nationally in total nonfuel mineral production value and 17^{th} in coal production (NMA 2007; Palmisano et al. 1993). Metal mining for all metals (e.g., zinc, copper, lead, silver, and gold) peaked in Washington between 1940 and 1970 (Palmisano et al. 1993). Today, construction sand, gravel, Portland cement, and crushed stone are the predominant materials mined in both Oregon and Washington. Where sand and gravel is mined from riverbeds (gravel bars and floodplains) changes in channel elevations and patterns, 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.

Field studies in southwest Oregon streams found that coho salmon , cutthroat, and yearling steelhead rearing densities decreased linearly as temperatures exceeded 17ºC (Frissell 1992). Coho salmon juveniles were absent in waters that reached 21-23ºC, except where thermal refugia were available. Juvenile salmonids will not persist in streams where temperature stress exceeds some threshold that can be defined by species and duration of high temperatures.

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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

Idaho, Oregon, and Washington have some protective measures in place to prevent harm to aquatic species from pesticides. In 2002/2003 EPA published IREDs for chlorpyrifos, diazinon, and malathion in order to protect human and environmental health. These documents include mandatory usage restrictions that will be in place until reregistration is complete.

In addition to the IREDs, growers must also adhere to the court-ordered injunctive relief. A Seattle court, in January 2004, imposed mandatory buffers for the three active ingredients for salmon-bearing streams within the listed ESUs. Buffers are 20 yards for ground application and 100 yards for any aerial application. These measures are mandatory in all four states.

California and Oregon both have Pesticide Use Reporting (PUR) legislation. California PUR 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, is for a government agency, or is in a public place.

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 run-off 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 OP 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 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 Best Management Practices (BMP) handbook for OPs, including information on alternative methods of pest control. The mid-Columbia area is of particular concern, as many orchards are in close proximity to streams.

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. 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 may be affected by the proposed registration of chlorpyrifos, diazinon, and malathion 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 likely have some level of effect on all 28 ESUs in the proposed action area. We expect the combined consequences of those effects, including impaired water quality and temperature, 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. The combined analysis evaluated effectsto listed Pacific salmonids and their designated critical habitat as outlined in the *Approach to the Assessment* (Figure 2).

Exposure Analysis

In this section, we identify and evaluate exposure information from the stressors of the action (Figure 35). We begin by presenting a general discussion of the physical and chemical properties of chlorpyrifos, diazinon, and malathion that influence the distribution and persistence of action stressors in the environment and exposure of listed species and designated critical habitat. 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 is intended to ensure that the agency action is not likely to jeopardize listed species or destroy or modify critical habitat, NMFS considers worst case scenario in addition to averages and more routine circumstances.

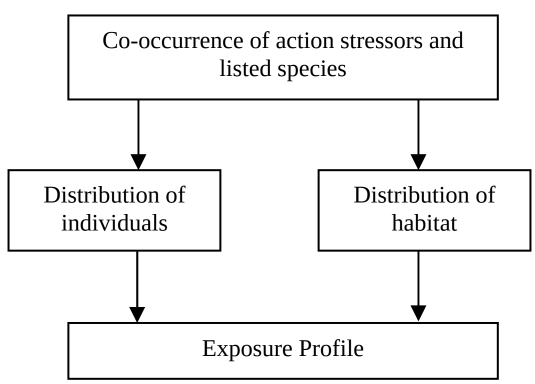
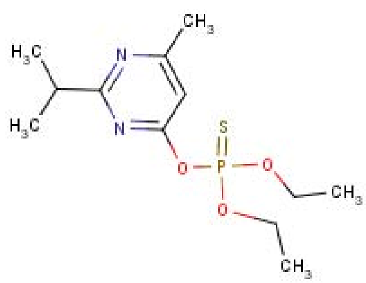


Figure 35. Exposure analysis

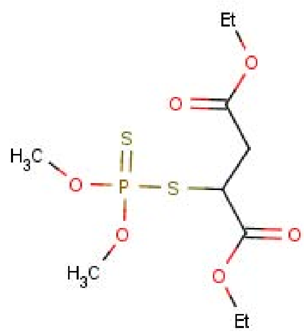
Summary of Chemical Fate of Active Ingredients

Chlorpyrifos

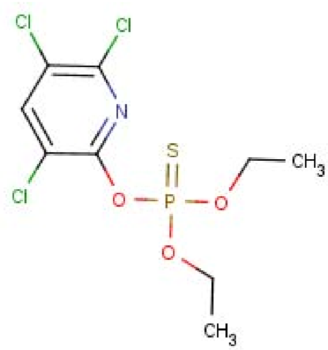
The major route of dissipation of chlorpyrifos in the environment appears to be aerobic and anaerobic metabolism. Chlorpyrifos degrades slowly in soils. Half-lives are variable depending on soil type, environmental conditions, and application rates. Soil persistence can vary with half-lives from a few days to well over 100 days (EPA 2000a). Chlorpyrifos is relatively immobile in soils given its low water solubility and high soil binding capacity. However, there is the potential for parent chlorpyrifos sorbed to soil to runoff into surface water via erosion. Spray applied chlorpyrifos may also enter surface waters through spray drift. The persistence of chlorpyrifos in surface waters varies with water chemistry. In neutral and acidic conditions chlorpyrifos half-lives are comparable (e.g., 72 and 73 days, respectively) (EPA 2000a). Hydrolysis increases under alkaline conditions (e.g., half-life 16 days at pH 9). The rate of hydrolysis also increases with increasing temperature (EPA 2000a). Chlorpyrifos has the potential to bioaccumulate in fish and other aquatic organisms and enter the aquatic food web (EPA 2003).



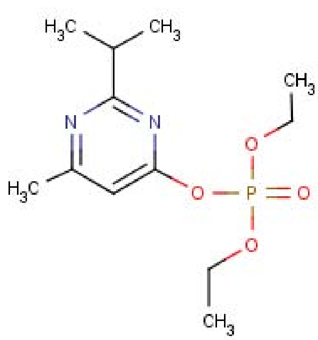
diazinon



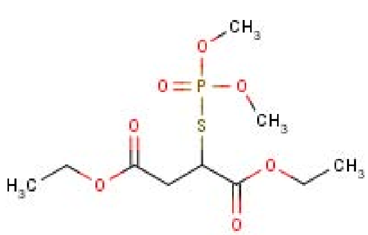
malathion



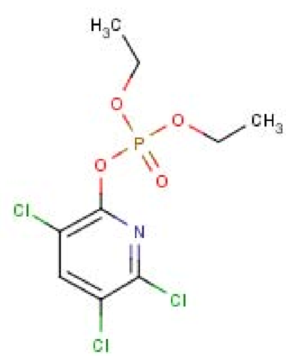
chlorpyrifos



diazinon oxon(diazoxon)



malathion oxon (malaxon)



chlorpyrifos oxon

Figure 36. Chemical structure of chlorpyrifos, diazinon, malathion, and their oxon metabolites.

Diazinon

Degradation of diazinon in the environment occurs through hydrolysis in water and through photolysis and metabolism in water and soils. In water, hydrolysis increases under acidic conditions and degrades more slowly under neutral and basic conditions. Reported half-lives at pH 5, pH 7, and pH 9 were 12, 138, and 77 days, respectively (EPA 2000b). The major route of degradation of diazinon in soils is through metabolism with first-order aerobic soil half-lives of 37 and 39 days for sandy loam soils (pH 5.4 and 7.8, respectively). Bioconcentration of diazinon does not occur to a significant extent in aquatic organisms. Diazinon is not expected to adsorb to soils to a significant degree. It is characterized as slightly mobile in 80% of soils tested and immobile in 20% of soils. Diazinon leaches in light textured soils that are saturated and have low organic content (EPA 2000b).

Malathion

The primary routes of degradation of malathion include microbial-mediated soil metabolism and hydrolysis under neutral or basic conditions in soil and water. Degradation occurs rapidly under neutral and alkaline soil conditions (half-life of 6.21 days) and aquatic environments (half-life 2.5 days with sediment pH 7.8, water pH 8.7). However, malathion is stable to hydrolysis in acidic environments (half-life 107 days). Malathion is also generally stable to photolysis but degradation is rapid in soils with microorganisms (half-lives hrs to 11 days). Persistence is extended in less microbially active soils, particularly soils that are dry, sandy, low nitrogen, low carbon, and acidic. As malathion mobility is high in many soils (e.g., sandy loam, loam, silt loam), it may contaminate surface waters through runoff or leaching. Additionally, drift, especially for approved ultra low volume (ULV) applications is a prominent pathway for exposure to aquatic habitats (EPA 2001).

Pathways and routes of exposure to chlorpyrifos, diazinon, and malathion

Chlorpyrifos, diazinon, and malathion can contaminate designated critical habitat and other aquatic habitats utilized by listed salmonids through runoff, leaching, drift, and deposition from precipitation. All life stages of salmonids may be exposed to these pesticides through direct contact with contaminated surface water or pore water. Additionally, dietary consumption of the three active ingredients is a likely route of exposure in salmonids and their prey. The dietary route of exposure may be most significant for chlorpyrifos given its greater tendency to accumulate in the tissues of aquatic organisms (EPA 2003). However, exposure from consumption of dead or dying aquatic and terrestrial insects also represents a potential route of exposure for all three pesticides. Chlorpyrifos, diazinon, and malathion are typically applied to control terrestrial insects which often make up a substantial portion of salmonids' diets (Baxter et al. 2007).

Metabolites and degradates

Chlorpyrifos, diazinon, and malathion are thionophosphorus organophosphate insecticides (OP) that are relatively weak inhibitors of acetylcholinesterase (AChE) in comparison to the oxygen analogs (oxons) of these contaminants. Transformation of the parent compounds to the oxon occurs through metabolism by vertebrates and invertebrates. Abiotic degradation can also transform the parent compounds to the more toxic oxon forms. For example, chlorpyrifos is rapidly transformed to chlorpyrifos oxon in chlorinated waters (Wu and Laird 2003). Diazoxon is the primary degradate of diazinon formed by hydrolysis in water (EPA 2000b). Time course studies on malaoxon production on sand and soil show malaoxon concentration relative to initial malathion were 1.4% after 10 days on sand and 10.7% after 21 days on soil (EPA 2001). Chlorpyrifos, diazinon, malathion, and their metabolites and degradation products are common surface water contaminants found in agricultural and residential watersheds (Anderson et al. 2007; Burke et al. 2006; CDPR 1995; Gilliom et al. 2006; Kozlowski et al. 2004). Pathways for surface water contamination following treatment of terrestrial habitats with these compounds include drift, runoff, and leaching. These pathways are most likely in situations where the applications occur in close proximity to surface water. However, longer range transport is also possible. A recent study reported that 100% of rainwater samples collected from an agricultural watershed in California contained diazinon concentrations as high as 1.2 ug/L despite no use of diazinon within the watershed (Vogel et al. 2008). The oxon degradates of chlorpyrifos, diazinon, and malathion were also detected in rainwater samples at a frequency of 79%, 76%, and 22%, respectively. Maximum concentrations were 0.100 ug/L chlorpyrifos, 0.118 ug/L diazinon, and 0.041ug/L malathion ug/L (Vogel et al. 2008). Other pathways may also result in detectable concentrations of oxons in surface water. Malaoxon has been detected in surface water runoff in concentrations that exceed one hundred ppb (CDPR 1995; EPA 2000c).

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 depend on the species and the individuals' life history and lifestage (Table 37). Many migrate hundreds or thousands of miles during their lifetime. Monitoring studies indicate detection of chlorpyrifos, diazinon, and malathion occurs frequently throughout the action area in freshwater and nearshore environments associated with urban, agricultural, or mixed land use watersheds (Anderson et al. 2007; Burke et al. 2006; CDPR 1995; CDPR 2008b; Gilliom et al. 2006). Given that all listed Pacific salmonid ESUs use watersheds where the use of chlorpyrifos, diazinon, and malathion products are authorized, and these compounds are frequently detected in watersheds where they are used, we expect all listed Pacific salmonid ESUs will be exposed to these compounds and other stressors of the action.

Table 37. General life histories of Pacific salmonids.

Species (number of listed ESUs)	General Life History Descriptions		
	Spawning Migration	Spawning Habitat	Juvenile Rearing and Migration
Chinook (9)	Mature adults (usually four to five years old) enter rivers (spring through fall, depending on run). Adults migrate and spawn in river reaches extending from above the tidewater to as far as 1200 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.
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 through-out 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 about 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.

General life histories of Pacific salmonids (continued)			
(number of listed ESUs)	Spawning Migration	Spawning Habitat	Juvenile Rearing and Migration
Sockeye (2)	Mature adults (usually four to five years old) begin entering rivers from May to October. Sockeye are semelparous (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.
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 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 Chlorpyrifos, Diazinon, and Malathion

Exposure estimates for non-crop pesticide applications

The BEs indicate that chlorpyrifos, diazinon, and malathion have many registered uses. Many of the uses identified in the BEs, particularly non-crop uses of chlorpyrifos and diazinon, have been voluntarily canceled, modified, or phased out. A number of uses were approved through EPA reregistration activities for chlorpyrifos, diazinon, and malathion for a variety of crop and non-crop applications (Table 38). Relatively few exposure estimates were provided for the "non-crop" uses of the three active ingredients.

Table 38. Examples of registered uses of chlorpyrifos, diazinon, and malathion and the exposure method used by EPA in BEs (EPA 2002; EPA 2003; EPA 2004b).

Active Ingredient	Registered Use	Exposure Characterization in BE
Chlorpyrifos	Agricultural Uses: More than 60 crops	PRZM-EXAMS Estimates for 11 crops
	Adult mosquito control	Assumed 10% drift
	Golf course applications	Based on Florida monitoring study
	Fire ant control; Road median strips; Industrial plant sites; Nonstructual wood treatments including fence posts, utility poles, railroad ties, landscape timbers, logs pallets, wood containers, and processed wood products; Residential use of containerized baits; Indoor areas including ship holds, railroad boxcars, industrial plants, manufacturing plants, and food processing plants; Cattle ear tags; Christmas trees; Woodlands.	No estimates provided
Diazinon	Agricultural Uses: More than 40 crops	PRZM-EXAMS Estimates for 7 crops
	Special local needs registration (24(c)) in California for drenching residential fruit trees for control of the Mediterranean fruit fly (CA960016); tree trunk wraps for commercial agriculture and horticulture; outdoor applications to ornamental plants in commercial nurseries; cattle ear tags.	No estimates provided
Malathion	Agricultural Uses: More than 100 crops	PRZM-EXAMS Estimates for 11 crops
	Public health (mosquito and fly control)	EPA interim rice model and AgDisp Model
	Uncultivated agricultural sites; non-agricultural uncultivated areas/soil; Christmas tree plantations, cull piles; drainage systems; fence rows/hedge rows; grain/cereal /flour bins and elevators; greenhouse; outdoor perimeter of household/domestic dwellings; intermittently flooded areas; non-agricultural outdoor structures; non-agricultural rights of way; ornamental and shade trees; ornamental herbaceous plants; ornamental non-flowering plants; ornamental woody shrubs and vines; pine seed orchards; outdoor refuse/solid waste containers; outdoor refuse/solid waste sites; swamps/marshes/stagnant water; wide-area public health use.	No estimates provided

Chlorpyrifos mosquito control

EPA derived Estimated Environmental Concentrations (EECs) for authorized mosquito control applications with chlorpyrifos and malathion using differing techniques. For chlorpyrifos, EPA assumed 10% of applied rate may drift to surface water. Therefore an application rate of 0.025 lbs chlorpyrifos per acre would result in concentrations of 1.5 – 18.5 ppb (ug/L) chlorpyrifos in surface water at depths of six inches to six ft. EPA also provided an estimate for permethrin as it had authorized the use of a formulated product for mosquito control that contains both chlorpyrifos and permethrin. The resulting EECs of permethrin ranged from 0.04 – 0.5 ppb in surface water of 6 inches to 6 ft deep. The potential risk posed by permethrin to salmonids or their habitat was not further explored despite the likelihood that these concentrations may be acutely toxic to aquatic invertebrates and fish. EPA reported EC50 and LC50 values of 0.1 and 0.8 ug/L for aquatic invertebrates and fish (EPA 2007b).

We expect that the EPA estimates of exposure based on 10% drift are under-predictive of the drift that may occur in aquatic habitats utilized by listed salmon. Drift estimates derived by NMFS using AgDrift [version 2.0.05; (Teske 2001)], a spray drift model developed by a consortium of pesticide registrants under a cooperative research agreement with EPA, suggest higher drift rates would be expected for spray droplet size distributions typically applied for control of adult mosquitoes (Table 39). For example, point deposition estimates using the fine-medium droplet size distribution (EPA default assumption) predict deposition of approximately 10% of the applied application rate 100 ft downwind of the application site. At the edge of the treatment area drift is much higher. However, mosquito adulticides are applied in very fine droplet distributions (fog applications) so that they remain suspended for longer periods increasing their effectiveness in controlling mosquitoes. AgDrift point-deposition estimates using very fine droplet size distribution predict deposition of approximately 22% the application rate at 100 ft downwind, and 17% at 150 ft downwind.

Table 39. AgDrift estimates for downwind deposition of chlorpyrifos expressed as a percentage of the application rate.

Aerial Application/ Droplet size distribution	Percent of application rate deposited downwind at various distances downwind from application				
	edge of field	25 ft	50 ft	100 ft	150 f
Fine-medium	50	22	17	10	6
Very fine-fine	50	36	30	22	17

Malathion mosquito control

Malathion is registered for terrestrial applications to control adult mosquitoes and aquatic applications to control mosquito larvae. EPA used two exposure models to estimate concentrations of malathion in salmonid habitats resulting from applications to control mosquitoes. EPA derived an EEC of 306 ug/L malathion for static water bodies approximately 0.10 m in depth using the "interim rice model." An EEC of 120 ug/L malathion was derived for flowing water bodies assumed to be approximately 0.5 m deep using AGDISP, a model that predicts drift of pesticides during application. Both models assumed an application rate of 0.5 lbs malathion/ acre (EPA 2001).

The interim rice model assumed a direct application to water and instantaneous partitioning of malathion to the sediment. Although it would be expected to take some time before malathion reaches equilibrium in the aquatic environment, this model appears to provide a relatively protective estimate for acute exposure for mosquito control given the shallow depth of water assumed (4 inches). The AgDisp stream assessment model was used to assess drift to small streams associated with terrestrial applications of malathion. The model incorporated a 5 ft buffer, a 1.64 ft deep stream moving at 2.24 mph. These assumptions are consistent with some of the habitats utilized by listed salmonids. However, the size distribution assumed for spray droplets (ASAE medium/coarse) is inconsistent with terrestrial applications for adult mosquitoes and would result in less drift than would be predicted with finer droplet size distributions typical of applications to control adult mosquitoes. Additionally, we recognize that dissipation rates may be less than or greater than those predicted using the AgDisp stream model depending on site-specific characteristics of the aquatic habitat (e.g., recharge rates and flow rates). Consequently, actual concentrations in aquatic habitats adjacent to treated areas are expected to be less than or greater than the EPA estimates depending on site-specific conditions.

Other non-crop uses of chlorpyrifos, diazinon, and malathion

No other exposure estimates were provided to evaluate non-crop uses of diazinon or malathion. Several non-crop uses of chlorpyrifos were discussed, but information to assess potential exposure was generally lacking. For example:

- Nursery use on ornamentals- EPA indicated it cannot estimate potential aquatic exposure of chlorpyrifos from the approved uses on ornamentals. Exposure to these uses remains a significant source of uncertainty.

- Golf courses- EPA did not provide EECs but indicated concentrations of 1.69 and 2.55 ppb were found in water where chlorpyrifos was likely the cause of a fish kill in Florida. The study included two applications at a rate of 4 lbs per acre. EPA indicated that golf course applications of chlorpyrifos are now limited to 1 lb per acre (although the number of applications does not appear to be restricted). It was suggested that the four-fold reduction in application rates would result in corresponding reductions in exposure. NMFS agrees that reduced rates are likely to result in corresponding reductions in exposure. However, it is unlikely that the concentrations measured in this study (incidental observations associated with a terrestrial field study) represent maximum concentrations that might be observed with golf course applications. Regardless, EPA recognized that concerns remain for direct effects to fish with a four-fold reduction in observed surface water concentrations. The chlorpyrifos BE indicated that a 25 ft buffer zone suggested for crop applications of chlorpyrifos likely would not apply to golf course application. The BE also stated that, "it is difficult to consider an elimination of all direct risk for golf course areas immediately next to salmon bearing streams" (EPA 2003). NMFS agrees that the information suggests golf course applications of 1 lb chlorpyrifos/acre and more may be sufficient to cause adverse effects to listed salmonids and their habitat. Additional information would be helpful to assess risk at lower application rates.

- Cattle ear tags- EPA indicated that salmonid exposure to chlorpyrifos from this approved use was discountable. NMFS agrees that significant contamination of designated critical habitat or significant exposure to listed salmonids from cattle ear-tags falling off of animals and into surface waters is extremely unlikely.

- Road median strips and industrial plant surfaces- EPA stated the use of chlorpyrifos for these purposes would be minimal and dispersed. Therefore, there would be no effect on listed fish. However, EPA did not provide adequate information that would allow us to concur with such a conclusion i.e., use statistics and/or EPA restrictions that would eliminate potential exposure to chlorpyrifos, etc. Consequently, exposure from this use remains a significant source of uncertainty.

- Termite use- No exposure estimates were provided. EPA indicated that multiple fish kills have occurred from this use. All sales and use of termiticide products were scheduled to be discontinued as of December 31, 2005 (EPA 2004a). However, use may be allowed beyond 2005 if data are submitted that show that residential post application risks from this use are not a concern (EPA 2004a).

As indicated above, there are many registered uses of chlorpyrifos, diazinon, and malathion that were not evaluated in EPA's BEs including applications to non-crop agricultural sites, residential sites, commercial sites, and animal and structural treatments. Additionally, some of these uses allow applications at rates that exceed those allowed in agricultural crops (Table 41). Many of the uses, application rates, and intervals evaluated in the BEs have been adjusted since EPA completed the REDs for these products in 2006. However, many existing non-crop uses are currently authorized for the three active ingredients. Malathion, in particular, includes an extensive list of uses with a maximum single application rate of 7.5 lbs/acre in crops, and rates exceeding 10 lb/acre for several other uses. Non-crop uses may pose equivalent or greater risk to listed species than the relatively few crop scenarios assessed in the BE. For example, “monitoring data suggest that urban malathion use poses the highest risk of contaminating surface water (EPA 2000c).” The absence of information on potential exposure of listed salmonids to non-crop uses of chlorpyrifos, diazinon, and malathion contribute a significant amount of uncertainty with the proposed action.

Exposure estimates for crop applications

The BEs provide EECs predicted for several examples of registered uses of chlorpyrifos, diazinon, and malathion (Table 40). These exposure estimates were generated using the PRZM-EXAMS model (EPA 2004c). 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” utilized by listed salmon (EPA 2003). 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 non-main channel habitats that are critical to rearing. All listed salmonids use shallow, low flow habitats at some point in their life cycle (Table 37). 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 40. PRZM-EXAMS exposure estimates from EPA’s BEs (EPA 2002; EPA 2003; EPA 2004b).

Scenario: crop, state	Application: rate (lbs a.i./A)/ method/ number of applications	Acute EEC (ppb)	Chronic EEC 60-d average (ppb)
CHLORPYRIRFOS			
Sugarbeets, CA	1.0/ground/1	0.94	0.27
Alfalfa, CA	1.0/aerial/4	4.5	2.4
Alfalfa, CA	1.0/ground/1	0.61	0.17
Almonds, CA	2.0/airblast/3	9.8	4.7
Cotton, CA	1.0/aerial/6	6.6	4.5
Apples, OR	3.0/airblast/1	9.2	2.8
Christmas trees, OR	1.0/aerial/1	3.1	0.84
Christmas trees, OR	1.0/aerial/2	4.5	1.7

DIAZINON			
Almonds, CA	1.5/aerial/3	8.9	6.4
Apples/pears, NY	2.0/aerial/3	25.1	15.4
Blueberries MI	2.0/aerial/5	75.4	44.8
Potatoes ME	10/ground/1	182	114
Strawberries FL	1.0/aerial /4	112	83
Stone fruits, GA	2.0/aerial/3	25.1	15.4
Cucumber FL	1.0/ground/4	429	258
MALATHION			
Alfalfa , CA	1.24/ULV[1]/ 2	39.1	3.9
Alfalfa, CA	2.46/NR[2]/ 2	7.8	0.8
Strawberries, CA	10 /NR/ 4	36.2	8.9
Lettuce, CA	2.46/NR/ 2	8.5	1.1
Walnuts, CA	15.33/NR/ 2	48.9	5.2
Citrus, CA	25.37/NR/ 4	77.4	13.4
Dates, CA	4.25/NR/ 6	15.1	4.6
Cherries, OR	8.0/ULV/4	42.7	9.6

[1]Ultra Low Volume droplet distribution assumed. Method of application assumed not reported.
[2]Method of application assumed not reported.

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. In the BEs, EPA characterizes the PRZM-EXAMS estimates as “worst-case” or even “unrealistic” for listed Pacific salmonids. In order to ensure no likely jeopardy, we consider the highest exposure any individuals of the population are likely to be exposed to. Therefore, to assess risk to individuals, we must consider the highest exposure any individuals of the population may be exposed to. Several lines of evidence discussed below suggest that EECs in the BEs may underestimate exposure of some listed organisms and designated critical habitat.

Monitoring data indicate that measured concentrations in aquatic habitats sometime exceed PRZM-EXAMS estimates. Although EPA characterized these exposure estimates as “worst case” in the BEs, it has also acknowledged that measured concentrations in the environment sometimes exceed PRZM-EXAMs EECs (EPA 2007a). Rather than worst case, EPA has clarified that PRZM-EXAMS estimates are protective for the vast majority of applications and aquatic habitats (EPA 2007a). NMFS agrees that the model is designed to produce generally protective estimates of exposure. However, monitoring data suggest that some individuals are likely to be exposed to concentrations greater than predicted with the PRZM-EXAMS estimates.

Recent reviews of EPA informal consultations by the USFWS and NMFS found that concentrations measured in surface water sometimes exceed peak concentrations predicted with PRZM/EXAMS modeling (NMFS 2007; USFWS 2008). NMFS also found examples where measurement of chlorpyrifos, diazinon, and malathion in surface waters exceeded EPA's peak concentration estimates predicted by PRZM-EXAMS modeling (EPA 2000a). EPA characterized the diazinon EECs provided in the BE as "quite unrealistic for use with Pacific salmon and steelhead" because these simulations "were modeled for areas that will have far more runoff than will occur in the Pacific states." EPA indicated that the California almonds simulation was the only exception, and it might be unrealistically high as well given that "all aerial uses will be canceled." However, monitoring for diazinon in the Salinas Valley, California and within the distribution of the threatened South Central Coast steelhead ESU included a peak detection of 67 ppb diazinon versus a maximum of 8.9 ug/L estimated in the California almond simulation (EPA 2002; Kozlowski et al. 2004) (Kozlowski et al. 2004). These findings demonstrate that EECs generated using PRZM-EXAMS can underestimate peak concentrations that actually occur in some aquatic habitats. Therefore, peak exposure experienced by some individuals of listed species may be underestimated.

Model assumptions and output suggest listed salmonid exposure to chlorpyrifos, diazinon, and malathion may exceed those concentrations predicted using PRZM-EXAMS. Two assumptions are discussed below that show salmonids may be exposed to higher concentrations than predicted with PRZM-EXAMS modeling:

Assumption 1: Model output 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-days, and 60-days. These concentrations represent the upper 10th percentile of the estimates derived using PRZM-EXAMS (Lin 1998).

Assumption 2: Model inputs used the highest use rates and greatest number of applications. The BEs lacked a definitive and comprehensive list of pesticide use restrictions authorized through product labeling. Critical information missing from the exposure assessment included maximum use rates permitted (single and seasonal), number of applications allowed, minimum application intervals required, and allowable application methods (EPA 2002; EPA 2003). EPA stated that it will not provide a comprehensive list of all label restrictions for consultation because it is not feasible for the agency to compile the information from all of the existing product labels. EPA does not maintain a master label that is inclusive of all registered uses (EPA 2007a).

EPA indicated that the pertinent information regarding use restrictions is presented in the most recent RED and IRED documents (EPA 2008a; EPA 2008c; EPA 2008e). However, a great deal of uncertainty remains as these documents present only summary information and do not include all relevant information found on pesticide labels. Additionally, it is unclear when restrictions outlined in the RED and IRED documents will be fully implemented. For example, as of October 16, 2008, EPA had not formally required malathion registrants to submit new end-use product labels to reflect changes outlined in the 2006 malathion RED. EPA had not confirmed when this would occur (EPA 2008d). Consequently, there is a great deal of uncertainty on whether PRZM-EXAMS scenarios encompass the full range of use rates, number of applications, etc. currently authorized for chlorpyrifos, diazinon, and malathion containing pesticide products.

There are hundreds if not thousands of pesticide product labels that contain chlorpyrifos, diazinon, or malathion. We received and reviewed a small subset of existing product labels (less than 20). EPA based PRZM-EXAMS estimates on a few examples of labeled uses. The PRZM-EXAMS scenarios frequently did not match up well with current use restrictions. In some cases the assumed application rate was less than, and in other cases it was more than those currently allowed as per the most recent IRED and RED documents.

For example, the chlorpyrifos estimates were derived by simulating applications to crops at rates of 1-3 lbs a.i./acre and Christmas trees at 1 lb a.i./acre. While the range is consistent with IRED restrictions for most crops, the documents do not specifiy approved application rates for Christmas trees or forest habitats. Other tree crops are approved for much higher applications, such as 4-6 lbs a.i./acre for citrus crops, which implies that Christmas tree application may be higher as well. The absence of this information is noteworthy, as the Use Profile section specifies that chlorpyrifos is approved for Christmas trees and woodlands, and the occupational exposure section provides a Margin of Exposure (MOE) for applications to pine seedlings. The diazinon IREDs/REDs raise similar questions. Six of the seven diazinon estimates were based on several applications by ground or air at rates of 1-2 lbs a.i./acre. These estimates do not cover the entire breadth of diazinon use as the IRED allows single applications by ground of up to 4 lbs a.i./acre in more than 25 crops. Potential exposure of listed salmonids to chlorpyrifos, diazinon, and malathion may be underestimated for some uses given EPA's authorization for greater use of these pesticides than was assessed with PRZM-EXAMS modeling.

There are examples of overestimates as well. The 10 lbs a.i./acre simulated in potatoes is 2.5 fold the rate specified in the IRED. The majority of malathion simulations assumed applications rates that are much greater than those allowed according to the RED. Simulations assumed use rates that ranged from 1.24 – 25.37 lbs a.i./acre. The RED specifies limits of 1-2 lbs a.i./acre for most crops with a maximum of 7.5 lbs a.i./acre in citrus. The simulated use rates that exceed existing RED/IRED limitations may represent labeled uses and products that have been canceled or phased out. However, use of those products is expected to substantially decline as RED/IRED restrictions are implemented and as the existing stocks are exhausted. PRZM-EXAMS simulations that assume use rates that are substantially greater than existing restrictions likely overestimate the concentrations that would occur in farm pond habitats.

Table 41. Use sites and application information approved on malathion product labels (adapted from Table 3, malathion BE (EPA 2004b).

Use Sites	Application Rate (lbs a.i./Acre)	Application interval	Maximum # of applications/ year
Vineyards	0.94 - 2.79	7 - 10 days	as needed
Orchards (i.e., apple, cherry, plum, prune)	0.63 - 14.4	7 - 10 days	NS
Tree nut (i.e., walnut, Macadamia nut, pecan)	0.31 - 15.33	7 - 10 days	NS
Fruits (i.e., citrus, bramble, melon, fig, date)	0.63 - 25.37	7 - 12 days	NS
Vegetables (i.e., squash, bean, lettuce, broccoli, spinach, onion)	0.19 - 4.3	7 - 10 days	NS
Grains (i.e., sorghum, rice, hops, barley, rye)	0.63 - 2.46	3 - 10 days	NS
Cotton	1.88 - 4.91	3 - 10 days	as needed
Homeowner (i.e., vegetable and flower garden, trees, indoor and outdoor pest control)	0.006-2.23	NS	NS
Open space (pasture land, range land, hay)	0.94-1.41	NS	NS
Turf (i.e., lawn, golf course, ornamental)	0.51-54.54	NS	NS
Public health (mosquito, fly)	0.001-0.74	NS	NS

Ornamental (i.e., flower, tree, nursery stock)	1.28-2.91	7-10 days	repeat as necessary
Tree farms (i.e., Christmas tree plantations)	6.4	NS	NS
Outdoor dwelling (commercial and domestic)	0.51-54.45		NS
Livestock	0.04-10	10 days-8 weeks	repeat as necessary
Outdoor surfaces (painted)	8.54-696.96	NS	NS

*NS = not specified

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 chlorpyrifos, diazinon, and malathion. For example, estimates of chlorpyrifos exposure were provided for 11 agricultural crops. An evaluation of currently registered uses of a single chlorpyrifos product label (Lorsban 4E) revealed chlorpyrifos can be applied to more than 60 agricultural crops in California alone (CDPR 2008a). Similarly, the product Diazinon 50W can be applied to over 80 crops in California while exposure estimates for only 7 agricultural crops were provided for all diazinon containing products. The BE indicated malathion-containing products are approved for use on more than 100 crops, whereas EPA provided exposure estimates on 11 crops for malathion products. There are logistic considerations that limit the number of scenarios that can be evaluated. However, information to suggest that the simulations run would be representative of other registered uses was not included in the BEs.

Crop scenarios are likely not representative of the entire action area. The regional scale that the modeled scenarios are intended to represent is unclear. Scenarios were identified by crop and state. However, many of the scenarios were conducted for states outside the distribution of listed salmonids. For example, of the seven crop scenarios presented in the diazinon assessment, only one used input parameters intended to represent a western state (California almonds). The assumed rainfall and other site-specific input assumptions can have large impacts on predicted exposure. For example, the chlorpyrifos BE provided EECs for application in cotton based on a Mississippi and a California scenario (EPA 2003). The EECs developed for the two scenarios differed by a factor of 4 despite simulating the same application rate and number of applications. 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 active ingredients. 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 chlorpyrifos, diazinon, and malathion. To evaluate potential exposure to environmental mixtures of chlorpyrifos, diazinon, and malathion, we considered cumulative exposure based on generated 60 day time-weighted average concentrations to simulate situations where pesticide products containing these active ingredients were applied at separate times during the growing season (Table 40). To address potential variability between sites, we generated exposure values for a few labeled uses based on restrictions specified in RED/IRED documents using the GENEEC model which is intended to provide screening estimates over large geographic regions (Table 42)[2]. The input parameters utilized were consistent with previous EPA model inputs (EPA 2000a; EPA 2000b; EPA 2000c; EPA 2001; EPA 2002; EPA 2003).

Table 42. GENEEC estimated concentrations of chlorpyrifos, diazinon, and malathion in surface water adjacent to cherries, onions, and strawberries.

Chemical use	Rate	No[1]	Interval	Buffer	EEC (ug/L)				
Ground application	lbs/ acre		days	feet	Peak	4-d avg	21-d avg	60-d avg	90-d avg
CHERRIES									
Chlorpyrifos dormant spray	2	1	NA	50	14.86	14.56	12.35	8.92	7.19
Diazinon foliar spray	2	1	NA	0	70.81	70.15	66.57	59.27	54.41
Malathion foliar spray	1.75	4	3	0	93.56	79.46	37.19	14.24	9.51
ONIONS									
Chlorpyrifos foliar spray	1	2	7	25	14.02	13.64	11.64	8.41	6.77
Diazinon In-furrow	4	1	NA	0	51.13	50.68	48.11	42.87	39.37

[2] EPA characterizes GENEEC as a tier-1 screening model EPA. 2004c. Overview of the Ecological Risk Assessment Process in the Office of Pesticide Programs, U.S. Environmental Protection Agency - Endangered and Threatened Species Effects Determinations. In: Resources OoP, editor. It 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 above for PRZM-EXAMS.

Malathion foliar spray	1.56	2	7	0	54.31	46.19	21.63	8.28	5.51
STRAWBERRIES									
Chlorpyrifos foliar spray	1	2	10	25	13.94	13.56	11.57	8.36	6.73
Diazinon foliar spray	1	1	NA	0	36.13	35.80	33.98	30.26	27.78
Malathion foliar spray	2	4	7	0	72.88	62.03	29.05	11.13	7.42

[1]Number of applications

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 active ingredient (solubility, soil adsorption coefficient, soil metabolisms rate, etc.,), assumed meteorological conditions (amount of rainfall), and other site-specific assumptions [soil type, slope, etc., (EPA 2004c)]. 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 non-main channel habitats that would be expected to yield higher pecticide concentrations than would be predicted with the "farm pond" based PRZM/EXAMS model. Examples of off-channel habitats include alcoves, channel edge sloughs, overflow channels, backwaters, terrace tributaries, off-channel dredge ponds, off channel ponds, and braids (Anderson 1999; Swift III 1979). 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 and Bolton 1999; Beechie et al. 2005; Caffrey 1996; Henning 2006; Montgomery 1999; Morley et al. 2005; Opperman and Merenlender 2004; Roni 2002).

Small streams and some off-channel habitats represent examples of habitats utilized 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 drainage area is treated and the aquatic habitat is assumed to be static (no inflow or outflow). Pesticide treatment areas of 10-hectares (approximately 25 acres) and larger occur frequently in agricultural crops, particularly under pest eradication programs. Additionally, aquatic habitats utilized 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 off-channel habitats utilized by salmonids

Direct over-spray

To estimate potential exposure of salmon to pesticides in shallow-water habitats we first determined the initial average concentrations that will result from a direct overspray of shallow surface water. Malthion use is permitted in swamps, marshes, stagnant water, and intermittently flooded areas (EPA 2006b). Direct overspray of standing water is permitted for control of mosquito larvae using malathion. The Malathion 8-E Insecticide label (EPA Reg. No. 34704-452) recommends applying malathion at a rate of 0.5 lbs a.i./acre to intermittently flooded areas. The resulting initial concentrations are a function of the application rate and the depth of the water body (Table 43). Malathion applied at a rate of 0.5 lbs a.i./acre would result in an average initial surface water concentrations in excess of 100 ug/L where depths are less than 0.5 m. The label specifies that applications may not be made around bodies of water where fish or shellfish are grown and/or harvested commercially. However, that statement does not appear to prohibit applications of malathion to areas where commercial interests do not apply including intermittently flooded freshwater habitats used by listed juvenile salmonids for rearing. NMFS is unaware of circumstances under which EPA has authorized direct application of chlorpyrifos or diazinon products to surface water.

Table 43. Average initial concentration of any active ingredient in surface water resulting from an overspray of aquatic habitat.

Application Rate (lbs active ingredient / acre)	Water Depth (meters)	Active Ingredient Concentration in Surface Water (ug/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
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 44). 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 et al. 2002). No-spray buffer zones (or setbacks) may significantly reduce pesticide exposure to salmon by reducing runoff and drift inputs.

The RED/IRED documents specify setback or buffer requirements for some uses. For example, chlorpyrifos spray restrictions for outdoor product spray application include setbacks to some aquatic habitats of 25, 50, and 150 ft for ground boom, airblast, and aerial applications, respectively. However, the chlorpyrifos BE suggests that these buffers do not apply to noncrop uses (e.g,. applications to golf courses) or granular formulations. Additionally, some chlorpyrifos labels we reviewed (e.g., Lorsban-4E, EPA Reg. No. 62719-220, revised 09-07-04) specified that the buffers are specific to "permanent water bodies." Therefore, buffers do not appear to apply to many important off-channel habitat types such as intermittent streams or manmade watercourses that either contain listed species or drain to such habitats. The malathion RED indicates buffers of 25 and 50 ft are required to all aquatic areas for aerial non-ULV and ULV agricultural applications, respectively. The IRED for diazinon requires no buffers to aquatic areas.

Chemical-specific buffer zones according to RED/IRED restrictions were assessed below. 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 et al. 2005; Roni 2002). Average initial concentration estimates derived from the simulations ranged from 0.6-333 ppb for each lb of active ingredient applied. These simulations indicate that applications of several lbs active ingredient 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.

The chlorpyrifos IRED indicates rates of 1-2 lbs/acre are common for ground boom and aerial spray applications. Estimated initial concentrations at those rates predict initial concentration ranges of 1-22 ug/L for ground application with a 25 ft buffer and of 3-128 ug/L for aerial applications with a 150 ft buffer. Simulations of airblast sprays of chlorpyrifos with a buffer of 50 ft at application rates of 1-2 lbs per/acre predict initial concentrations of 1-54 ug/L. However, the maximum application rate in citrus of 4-6 lbs/acre would produce a substantially higher exposure range.

Diazinon can be applied to most crops at rates of 1-4 lbs/acre by ground boom. Simulations with no buffer at those rates predict initial average concentration in the modeled habitat at 4-304 ug/L.

Ground application of malathion in non-ULV droplet size distributions most frequently allow application rates of 1-2 lbs/acre with no buffer requirements. Ground application simulations predict a concentration range of 4-152 ug/L. Maximum application rates for orchard crops generally ranged from 1-3 lbs/acre. Simulations with no buffer at those rates predict initial average malathion concentrations of 11-642 ug/L. Maximum aerial applications rates for most non-ULV application range from 1-2 lbs/acre and require a 25 ft buffer for aquatic habitats. Simulations based on those considerations predict initial average concentrations of 10-414 ug/L. However, maximum applications rates of non-

ULV aerial applications in citrus are substantially higher (4.5 – 7.5 lbs/acre) and predict aquatic concentrations exceeding 1 mg/L for some off-channel habitats. Most of the ULV aerial applications are applied at 0.61-1.22 lbs/acre. ULV simulations with a 50 ft buffer at 0.61-1.22 lbs/acre predict initial average concentrations of 9-376 ug/L.

Table 44. 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 (feet)	Average Initial Concentration in Surface Water (ug/L)
Aerial Applications, EPA default (ASAE fine-medium droplet size distribution)		
2	0	17
1	0	34
0.5	0	67
0.1	0	333
2	25	10
1	25	21
0.5	25	41
0.1	25	207
2	150	3
1	150	6
0.5	150	13
0.1	150	64
Aerial Applications, (ASAE very fine – fine droplet distribution)		
2	2	2
1	1	1
0.5	0.5	0.5
0.1	0.1	0.1
2	2	2
1	1	1
0.5	0.5	0.5
0.1	0.1	0.1
Air Blast Applications, Dormant Spray		
2	0	11
1	0	21
0.5	0	43
0.1	0	214
2	50	1
1	50	3
0.5	50	5
0.1	50	27
Ground Application, Low Boom, ASAE very fine-fine distribution, 50th percentile		
2	0	4
1	0	8
0.5	0	15
0.1	0	76
2	25	1
1	25	1
0.5	25	2
0.1	25	11

Monitoring: Measured Concentrations of Chlorpyrifos, Diazinon, and Malathion

The BEs summarized surface water monitoring data available for chlorpyrifos, diazinon, and malathion from USGS and CDPR water quality programs. Data from the USGS' NAWQA program was summarized by EPA (Table 45). The NAWQA program was designed to describe the status and trends of a representative portion of the nation's water and to provide a scientific understanding of the primary natural and human factors affecting water quality (Hirsch 1988). The NAWQA summaries used by EPA were designed to give a broad, national-level perspective of water quality (EPA 2000b). The NAWQA program is an aggregation of some 60 regional study units, which are monitored on a rotating schedule for long-term variations in water quality. EPA summarized monitoring results for 20 of the study units. The NAWQA design does not result in an unbiased representation of surface waters. For example, some agricultural activities and related pesticide use that may be very important in a particular region may not be represented in the locations sampled.

Table 45. Maximum concentrations observed in NAWQA surface water monitoring presented in EPA BEs (EPA 2002; EPA 2003; EPA 2004b).

Active Ingredient	Maximum concentration (ppb) observed in 20 NAWQA study units		
	Agricultural areas	Urban streams	Mixed-use streams
Chlorpyrifos	0.4	0.19	0.13
Diazinon	3.80	2.90	Not Reported
Malathion	1.14	9.58	Not Reported

EPA also presented data from some surface water monitoring studies conducted in the state of California (Table 46). Although the data are not directly comparable because they are categorized differently, maximum concentrations observed in the California studies are slightly higher, but were generally within an order of magnitude of those reported for NAWQA monitoring (Table 45). Maximum concentrations reported for both the NAWQA and California monitoring studies were generally below, or at the lower end of peak (acute) EECs predicted in modeled scenarios (Table 46).

Table 46. Maximum concentrations reported in California monitoring results presented in EPA BEs(EPA 2002; EPA 2003; EPA 2004b).

Active Ingredient	Maximum concentration observed (ug/L)	
	Rivers	Tributary streams
Chlorpyrifos	0.35	2.28
Diazinon	36.8	2.89
Malathion	6.0 (type of surface water not identified)	

We performed additional database queries to evaluate the occurrence of chlorpyrifos, diazinon, and malathion in monitored surfaces waters in California, Idaho, Oregon, and Washington. Data were obtained from the USGS NAWQA database for the three active ingredients listed salmon habitat. Specific data were from NAWQA study basins during 1992-2006 (USGS 2008). Malathion was detected in approximately 6% of the samples analyzed. Chlorpyrifos and diazinon were detected more frequently (26% and 40%, respectively). Additional summary information from the query is presented in Table 47.

Table 47. Summary of detections of chlorpyrifos, diazinon, and malathion in filtered stream samples collected in California, Idaho, Oregon, and Washington streams, USGS NAWQA program (1992-2006).

Chemical	Chlorpyrifos	Diazinon	Malathion
Number of detections	1,131	1,767	272
Minimum (ug/L)	0.004	0.002	0.005
Maximum (ug/L)	0.401	3.800	1.350
Arithmetic Mean (ug/L)	0.022	0.084	0.049
Standard Deviation (ug/L)	0.037	0.230	0.121

We also reviewed data obtained from CDPR's Surface Water Database (CDPR 2008b). This database provides results from 51 pesticide monitoring studies conducted by Federal, state, and local agencies, private industry, and environmental groups. The samples were obtained from California rivers, creeks, urban streams, agricultural drains, the San Francisco Bay delta region, and urban stormwater runoff. Many of the sites are also salmonid habitat (August 1990-June 2005). As with the Regional NAWQA data, malathion was detected at a frequency of 6%, and chlorpyrifos and diazinon were detected at much greater frequencies (49% and 67%, respectively). Summary statistics for the California database are provided below (Table 48).

Table 48. Summary of detections of chlorpyrifos, diazinon, and malathion in California Department of Pesticide Regulation's Surface Water Database.

Chemical	Chlorpyrifos	Diazinon	Malathion
Number of detections	1290	1652	82
Minimum (ug/L)	0.001	0.001	0.005
Maximum (ug/L)	2.420	29.371	0.420
Arithmetic Mean (ug/L)	0.062	0.159	0.054
Standard Deviation (ug/L)	0.168	1.035	0.070

We reviewed several surface water monitoring studies of chlorpyrifos, diazinon, and malathion available in the open literature, or discussed in EPA documents for reregistration evaluations (EPA 2000a; EPA 2000b; EPA 2000c). These results are summarized below for the potential exposure to listed species from EPA approved uses of chlorpyrifos, diazinon, and malathion.

Runoff of diazinon and esfenvalerate was evaluated in two studies of similar design (Werner et al. 2002; Werner et al. 2004). In both studies the pesticides were applied to a prune orchard in Glenn County, California and runoff concentrations were monitored following rain events. These concentrations indicate the degree of pesticide loading that may occur in aquatic habitats due to runoff. Concentrations of diazinon were generally one to two orders of magnitude greater than esfenvalerate (Table 49). The co-occurrence of these two chemicals in runoff is likely as both are widely used in orchards crops common in California and the Pacific Northwest (CDPR 2007).

A separate study conducted in southern California characterized diazinon and chlorpyrifos concentrations from different urban land uses (Schiff and Sutula 2004). Of the 128 runoff samples from different land uses over five storm events, diazinon was detected in 93% of the samples while chlorpyrifos was detected in 12% of the samples. The mixed agricultural land use areas had a diazinon flow-weighted mean concentration of 4 ug /L, higher than any other land use by one to two orders of magnitude. There was high variability in replicate sites and replicate storm events, which highlighted the difficulty in modeling these systems (Schiff and Sutula 2004).

Table 49. Concentrations of diazinon and esfenvalerate detected in runoff samples from Glenn County, California.

	Diazinon (ug/L)		**Esfenvalerate (ug/L)**	
Ground cover	**2000** (Werner et al. 2002)	**2001** (Werner et al. 2004)	**2000** (Werner et al. 2002)	**2001** (Werner et al. 2004)
Bare soil	210.4	11.10-339.7	3.6	0.81-1.96
Sod	135.9	10.70-207.2	6.3	0.79-2.25
Resident vegetation	155.2	19.50-290.2	3.9	0.73-2.04
Clover	118.2	13.60-277.1	2.9	1.20-3.47

Another study evaluated concentrations of diazinon and chlorpyrifos in urban waterways in northern California (Bailey et al. 2000). Water samples were collected from streams, sumps, and sloughs in the cities of Sacramento and Stockton during 1994 and 1995. Concentrations found during this study are presented in Table 50.

Table 50. Concentrations of chlorpyrifos and diazinon detected in surface water in northern California (Bailey et al. 2000).

Chemical	Number of samples	Maximum ug/L	Median Ug/L
Chlorpyrifos	90	0.19	0.05
Diazinon	230	1.50	0.21

Kozlowski and others monitored chlorpyrifos and diazinon in surface waters listed as impaired in the lower Salinas Valley during dry and wet seasons of 2002 and 2003 (Kozlowski et al. 2004). The study found that accumulation of these chemicals in ditch, canal, and slough sediments during the dry season provided a source for later remobilization during the wet season. This study was particularly relevant as some of the sample sites provide habitat for the South-Central California coast steelhead ESU. The study included a range of relevant aquatic habitats with nine sample sites on river, lagoon, lake, and agricultural canal and drain locations. Peak surface water detections of 5.8 and 67.2 ug/L were observed for chlorpyrifos and diazinon, respectively (

Table 51). This study also reported concentrations of chlorpyrifos and diazinon in sediments (Table 52). The highest concentrations of both chemicals were observed in agricultural drains. Listed salmonids are known to utilize agricultural drains where they are accessible (e.g., Middle Columbia River steelhead use of Marion Drain). The particular agricultural drains where the highest chlorpyrifos and diazinon concentrations were observed (sites #3 and #8) may not be used by listed South-Central California coast steelhead. However, these agricultural drains may represent conditions in similar habitats that are used by listed salmonids. This study also reported concentrations of chlorpyrifos and diazinon in sediments (Table 52). Sediments are recognized sources of OP exposure to aquatic biota and can cause toxic responses in aquatic invertebrates (Anderson et al. 2003; Anderson et al. 2006; Philips et al. 2004).

Table 51. Average and maximum concentration of chlorpyrifos and diazinon monitored in filtered samples collected in surface waters of the lower Salinas Valley (2002-2003, (Kozlowski et al. 2004)].

Site	Chlorpyrifos (ug/L)			Diazinon (ug/L)		
	2002 mean	2003 mean	Max	2002 mean	2003 mean	Max
#1 Salinas river	0.067	0.078	0.222	0.114	0.057	0.387
#2 Salinas lagoon	0.051	0.056	0.107	0.093	0.029	0.203
#3 agricultural drain	0.058	0.069	5.786	0.173	0.099	4.343
#4 agricultural drain	0.056	0.060	0.123	0.508	0.089	1.869
#5 agricultural canal	0.082	0.093	0.283	0.627	0.419	1.620
#6 Old Salinas river	0.069	0.071	0.222	0.109	0.144	0.192
#7 Moss landing harbor	0.074	0.078	0.145	0.043	0.095	0.073
#8 agricultural drain	0.356	0.380	0.938	21.61	0.709	67.24
#9 Espinosa slough (lake)	0.069	0.062	0.091	0.063	0.060	0.103

Table 52. Sediment concentrations (ng/kg-dry weight) of chlorpyrifos and diazinon detected in the lower Salinas Valley (2002-2003, (Kozlowski et al. 2004)).

Site	Chlorpyrifos ng/kg-dry weight		Diazinon ng/kg-dry weight	
	2002 mean	2003 mean	2002 mean	2003 mean
#1 Salinas river	46,591	17,373	24,759	8,482
#2 Salinas lagoon	10,195	23,278	2,909	2,090
#3 agricultural drain	75,150	22,628	7,576	4,510
#4 agricultural drain	2,905	4,427	3,488	4,140
#5 agricultural canal	270,081	109,013	122,550	34,232
#6 Old Salinas river	4,840	10,236	13,338	15,207
#7 Moss landing harbor	1,762	2,845	1,206	2,901
#8 agricultural drain	124,651	455,560	469,693	3,916,689
#9 Espinosa slough (lake)	no detects	3,046	3,834	2,808

A pesticide loading simulation model that focuses on runoff was used to predict concentrations of numerous pesticides in the upper San Joaquin watershed of California (Luo et al. 2008). The Soil and Water Assessment Tool (SWAT) model was used to simulate spatially distributed hydrological information under different temporal conditions. The model results showed that 55% of diazinon concentrations were above the detection limit (0.005 ug/L); of those 3% exceeded 0.100 ug/L, a value shown to be toxic to aquatic life (Luo et al. 2008). The model was used to predict areas with high risk of runoff for diazinon and chlorpyrifos. In general, the model showed multiple areas within the San Joaquin watershed that are highly vulnerable to runoff with average pesticide yields for diazinon of greater than 10 grams/kilometer^2, and greater than 15 grams/ kilometer^2 for chlorpyrifos (Luo et al. 2008). ESA-listed salmonids exist throughout the San Joaquin watershed, including the runoff sites.

We also reviewed summaries of monitoring data presented in EPA's assessment for the reregistration of malathion (EPA 2000c). These summaries included monitoring results from several large-scale malathion control programs. Concentrations reported were much higher than the NAWQA monitoring data presented in the malathion BE (EPA 2004b). All malathion detections reported in the NAWQA database were less than 10 ug/L. However, the monitoring results presented in Table 53 shows all 11 monitoring studies reported malathion detections greater than 10 ug/L. One study reported a detection of malathion greater than 1,000 ug/L.

Table 53. EPA report of malathion detections in surface water associated with several large scale control programs (EPA 2000c).

APHIS Program	samples	Frequency of detection	Concentration range (ppb)[1]	Concentration mean (ppb)[1]
Medfly applications in Florida 1985-1990	128	55%	0.2 - 51	9.4
Grasshopper control in 13 western states 1984-1989	NR	NR	0.11 - 85	NR
Southeast boll weevil control 1996-1997	NR	NR	Runoff: 0 – 93.5 Drift into creek:0 – 10.89	NR
South rolling plains boll weevil control 1995	NR	NR	Stream: 0.503 – 86.9 River: 0.589 – 7.45	NR
Bollweevil control 1985-1990				
(Alabama)	82	59%	0.10 - 25	NR
(Florida)	15	53%	6 - 49	NR
(Georgia)	NR	NR	NR	12.9 (day 0) 5.18 (day 1-5) 1.78 (day 6-10) 1.86 (day 11-71)
Medfly eradication Santa Clara county California (1981)	NR	NR	summer: 0 – 152 winter: 0 – 1000[2]	NR
Medfly eradication Santa Cruz county California (1981)	NR	NR	<0.1 - 41	NR
Medfly eradication San Mateo county California (1981)	NR	NR	Up to 103 ppb in creek[2]	NR
Field studies for mosquito control				
Pensacola, FL 1974	NR	NR	Saltmarsh: <0.1 - 5.2	NR
West Galveston, TX 1975	NR	NR	Saltmarsh: 1 – 69	NR
San Francisco Medfly Monitoring Program 1981				
23 inland creeks	NR	NR	0 - 157	0.2 – 57.4
4 creeks near drainage culvert	NR	NR	NR	**Malathion:** 37.5 - 569 **Malaoxon:** 13.5 – 384
8 locations in SF Bay Estuary	NR	NR	0 - 18	0 – 7
Ventura County, California medfly monitoring program 1997	NR	NR	**Malathion:** 787 creek 11.2 lagoon **Malaoxon:** 160 creek 2.62 lagoon	**Malathion:** 44.2 **Malaoxon:** 0.05

[1]Concentration of malathion unless otherwise specified, [2]Fish kills coincided with one or more applications

Monitoring data considerations

Schulz summarized general trends in monitoring data by evaluating field studies on insecticides in surface waters due to agricultural practices, published after 1982 (Schulz 2004). Several of the studies summarized discussed the detection of chlorpyrifos and diazinon (Table 54). There were no case studies provided for malathion in the United States. Of the chlorpyrifos studies evaluated, most assessed runoff. Concentrations ranges were most frequently less than 1 ug/L. The peak concentration of chlorpyrifos observed within the listed species distribution was 3.2 ug/L in California. All of the diazinon studies summarized from the U.S. were from California. The concentration ranges exceeded one ug/L in 50% of the studies with a peak of 7 ug/L reported for a sample from San Joaquin tributaries. This value was sampled from selected storm events and was part of the NAWQA monitoring program discussed above.

Table 54. Summary of field case studies on insecticides in surface waters due to agricultural practices published since 1982 (adapted from Table 1 in Schulz 2004).

Concentration ug/L	Source	Detections	Sampling interval	Location
Chlorpyrifos				
0.004-0.12	Leaching (irrigation)	15	Weekly	Royal Lake, WA
0.06-0.52	Nonpoint sources	8	Monthly	Sacramento-San Joaquin catchment, CA
0.004-0.86	Runoff	-	Event	Streams in Midwest U.S.
0.13	Runoff	1	14 d	White River, IN
0.01-0.26	Runoff	17	Event	San Joaquin catchment, CA
0.03-3.2	Runoff	52	Event	creek channel, central California coast
0.02-3.8	Runoff	7	8 hour	Sandusky River, OH
0.2-2.8	Runoff, assumed	7	1 d (peak)	Sandusky River, OH
0.67	Runoff, assumed	7	1 d (peak)	Turlock Irrigation Ditch, CA
Diazinon				
0.05-1.06	Leaching, runoff	7	Monthly	Pajaro River estuary, CA
0.4	Nonpoint sources	1	Monthly	Sacramento-San Joaquin catchment, CA
0.05-0.4	Nonpoint sources	5	Seasonal	Sacramento-San Joaquin estuary, CA
0.02-0.62	Nonpoint sources	7	Weekly	San Joaquin catchment, CA
0.12- 7	Runoff	17	Event	San Joaquin tributaries, CA
0.02-1.03	Runoff	~60	Daily	Sacramento-San Joaquin catchment, CA

Surface water monitoring can provide useful information regarding real-time exposure and the occurrence of environmental mixtures. The available monitoring studies were conducted under a variety of protocols and for varying purposes. Very few have been designed for the purpose of evaluating exposure in listed Pacific salmonid habitats. One exception is a monitoring effort which targeted agricultural areas within Washington state (Anderson et al. 2007; Burke et al. 2006). 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 ESU) and high diversity and intensity of agriculture (Johnson and Cowles 2003). The sample design did not target specific applications of pesticides nor did they 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 favored the detection of multiple pesticides, rather than peak concentrations in some habitats used by Middle Columbia River steelhead. Consequently, we discuss this monitoring set in the *Environmental Mixtures* section below.

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. Sampling from the NAWQA studies and other studies reviewed was typically not conducted in coordination with specific applications of chlorpyrifos, diazinon, and malathion. Similarly, sampling was not designed 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. Recent data show a decrease in use of chlorpyrifos and diazinon in California that may be associated with restrictions on residential uses of those active ingredients. However, pesticide use patterns change and may result in either increases or decreases in use of pesticide products. There is considerable uncertainty regarding the representativeness of monitoring conditions to forecast future use of products containing the three active ingredients.

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 active ingredients. 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 frequently 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 active ingredients, 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 chlorpyrifos, diazinon, and malathion

The oxon forms of chlorpyrifos, diazinon, and malathion are metabolites and degradates known to be strong inhibitors of acetylcholinesterase relative to the parent compounds. However, the BEs provided no exposure estimates for these compounds. The chlorpyrifos BE did not discuss the conversion of chlorpyrifos to the oxon metabolite. The diazinon BE discussed a field study in the Sacramento Basin that found 2.5% of diazinon as diazoxon and concluded the formation of diazoxon was at a rate that did not warrant concern. The malathion BE indicates conversion of malathion to malaoxon ranges from 1.8% to 10.7% of the parent depending on environmental conditions. Other information also suggests malaoxon can occur at very high concentrations in the environment. Monitoring results of the Mediterranean fruit fly eradication program in California detected malaoxon concentrations as high as 384 ug/L in a creek and as high as 2.62 ug/L in an estuarine lagoon (EPA 2000c). Estimates of cumulative exposure to cholinesterase inhibiting degradates are needed because we expect salmonid exposure to the parent compounds and degradates simultaneously.

The BEs identified “major degradates” of the parent compounds and presented the acute toxicity of some of these intermediates. EPA defines major degradates as degradation products of the active ingredient identified in environmental fate studies whose field concentrations exceed 10% of the applied active ingredient. The BE did not identify "minor degradates" (found at concentrations <10% of a.i.). However, other “minor degradates” (found at concentrations <10% of a.i.) may be toxicologically significant. One major degradate of chlorpyrifos is 3,5,6-trichloro-2-pyridinol (TCP). EPA concluded that TCP was less toxic to fish and invertebrates than chlorpyrifos based on standardized acute toxicity tests. Therefore, the occurrence of TCP in the environment would not contribute to the salmonids’ risk. However, exposure to TCP is expected to be much greater than exposure to chlorpyrifos. Substantial fractions of applied chlorpyrifos can persist in fields for weeks after application based on environmental fate characteristics (EPA 2003). TCP is expected to be more persistent and mobile in soils compared to chlorpyrifos. Additionally, TCP is expected to be more persistent in water and sediment with concentrations expected to be comparable in the two matrices based on partitioning coefficients (EPA 2003). Therefore, we expect listed salmonids will experience acute and chronic exposure to TCP. Estimates of acute and chronic exposure to TCP were not provided in the BE (EPA 2003).

Exposure estimates for the major soil and water degradate for diazinon, oxypyrimidine, were also lacking in the BE. Oxypyrimidine is more stable and mobile in soils than diazinon suggesting a high likelihood that aquatic species will be exposed (EPA 2002). Nevertheless, the risk of oxypyrimidine was assumed to be negligible to aquatic species based on lethality toxicity tests in rats (EPA 2002). However, it is highly questionable that rats are an appropriate surrogate for aquatic species.

Isomalathion and malathion monocarboxylic acid (MCA) were identified as degradates of malathion (EPA 2001). The BE discussed that the presence of isomalathion would increase the toxicity of malathion. However, no relevant environmental fate discussions or exposure estimates were provided for isomalathion. MCA was characterized as a substantial residue in fish tissue suggesting bioaccumulation by salmon. However, assessment of the exposure and risk of these compounds was not provided and remains an uncertainty.

Other ingredients in formulated products

Registered pesticide products containing chlorpyrifos, diazinon, and malathion always include other ingredients such as carriers and surfactants. Pesticide products may also include other registered active ingredients (Table 55). EPA indicated that a product containing both chlorpyrifos and permethrin is registered for mosquito control. Exposure estimates were provided for both active ingredients in this formulation. However, exposure to other product ingredients within chlorpyrifos, diazinon, and malathion containing formulations were not evaluated.

Table 55. Example of listed ingredients on labels of some products containing chlorpyrifos, diazinon, and malathion.

EPA Product Registration Number	Active Ingredients	Other Ingredients
499-405	chlorpyrifos 8%, cyfluthrin 1.6%	90.4%
4329-36	chlorpyrifos 12% permethrin 4%	84%
39039-6	chlorpyrifos 12% diazinon 4%	60%
655-441	chlorpyrifos 13%, dichlorvos 4.82%	82.18%
66222-19	chlorpyrifos 42.5%	57.5%
7501-112-5905	diazinon 15%, lindane 25%, carboxin 14%	46%
11556-123	diazinon 20%, coumaphos 20%	60%
270-260	diazinon 18%, piperonyl butoxide 2%	80%
61483-92	diazinon 40%, tetrachlorvinphos 10%	50%
4-122	malathion 6%, carbaryl 0.3%, captan 11.8%	81.9%
4-59	malathion 3%, carbaryl 0.5%, captan 5.87%	90.63%
4-355	malathion 6%, sulfur 25%, captan 6.03%	62.97%
4-157	malathion 13.5%, captan 13.5%	73%
7401-163	malathion 7.5%, PCNB 12.5%	80%
11474-96	malathion 2%, piperonyl butoxide 0.12%, pyrethrins 0.05%	97.83%
5481-275	malathion 2%,carbaryl 2%	96%
8329-29	malathion 30.6%, piperonyl butoxide 4.96 %, resmethrin 1.88%	62.66%
769-646	malathion 5.5%, petroleum distillates and mineral oil 89.0%	5.5%

Nonylphenol (NP) and nonylphenol polyethoxylates are inert ingredients that may be formulated in pesticide products and are common adjuvant ingredients added during pesticide applications. NP and nonylphenol polyethoxylates are common ingredients in detergents, cosmetics, and other industrial products. These compounds are also common wastewater contaminants from industrial and municipal sources. A national survey of streams found that NP was among the most common organic wastewater contaminants in the U.S. and was detected in more than 50% of the samples tested. The median concentration of NP in streams was 0.8 ug/L and the maximum concentration detected was 40.0 ug/L (Table 56). Related compounds were also detected at a relatively high frequency (Koplin et al. 2002).

Table 56. Detection of nonionic detergent degradates in streams of the United States (Koplin et al. 2002)

Chemical	Frequency Detected	Maximum (ug/L)	Median (ug/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 occur in chlorpyrifos, diazinon, and malathion product formulations. EPA did not provide the inert profile of the end-use products. 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 2008b). Many of these inerts are also known to be hazardous. For example, xylene is a neurotoxin and coal tar is a known carcinogen. Additionally, several permitted inerts are also registered active ingredients (e.g., copper, zinc, chloropictrin, chlorothalonil). Inerts often make up more than 50% of the mass of pesticide products and millions of lbs of products containing chlorpyrifos, diazinon, and malathion are applied to the landscape each year (CDPR 2007). This may equate to very large contaminant loads of inerts that may adversely affect salmon or their habitat. The uncertainty regarding exposure to these ingredients must therefore 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, the Lorsban 4-E Insecticide label (EPA Reg. No. 62719-220) recommends the product be applied in a petroleum spray oil and provides recommendations for tank mixtures with other insecticides (e.g., pyrethroids and fenamiphos, another organophosphate), herbicides (e.g., paraquat, glyphosate), fertilizers, and surfactants. Another chlorpyrifos label (EPA Reg. No. 66222-19) recommends tank mixtures with multiple pesticides to control invertebrate pests. The label’s suggestions include products containing other organophosphates, avermectin, an organochlorine, and an organotin. 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. Yet, exposure and consequently risk of these ingredients 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 use a population of a listed species's base condition to evaluate 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 lifecycle. 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 active ingredients were registered in the US for use in more than 20,000 different pesticide products (Aspelin and Grube 1999). Typically 10 to 20 new active ingredients are registered each year (Aspelin and Grube 1999). In a typical year in the United States, pesticides are applied at a rate of approximately five billion lbs of active ingredient per year (Kiely et al. 2004). Pesticide contamination in the nation's freshwater habitats is ubiquitous and pesticides usually occur in the environment as mixtures (Gilliom et al. 2006). "More than 90 percent 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 percent of the time they had detections of 10 or more (Gilliom et al. 2006)." The likelihood of exposure to multiple pesticides throughout a listed salmonids' lifetime is great considering their migration routes and habitats occupied for spawning and rearing. In a three year monitoring study conducted by the Washington DOE, pesticide mixtures were found to be common in both urban and agricultural watersheds (Burke et al. 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 et al. 2006).

Atrazine is frequently detected in streams throughout the U.S. It was detected in over 50% of samples taken from urban and agricultural streams in a national monitoring program and was the most common ingredient in pesticide mixtures (Gilliom et al. 2006). The insecticides diazinon, chlorpyrifos, carbaryl, and malathion were common in mixtures found in urban streams (Gilliom et al. 2006). The co-occurrence of atrazine with chlorpyrifos, diazinon, malathion, and other OP pesticides in aquatic habitats increases the likelihood of adverse responses in salmonids and their aquatic prey. Atrazine is known to potentiate the toxicity of OPs in aquatic invertebrates by inducing metabolic enzymes (cytochrome P450 monooxgenases) that are responsible for converting parent OP to much more toxic oxons (Miota 2000). Aquatic invertebrates are important prey items for rearing Pacific salmonids. Reduced populations of prey may affect growth and development at critical life stage transitions (e.g., alevin-fry). Surface water monitoring in several streams that support listed salmon in Washington State reveal atrazine detection at relatively high rates in some streams (Anderson et al. 2007). Atrazine was the most frequently detected pesticide in agricultural streams in the lower Yakima watershed of eastern Washington with detection rates generally ranging from 50 – 75% of analyzed samples (Anderson et al. 2007; Burke et al. 2006). A comparison to NAWQA monitoring in the Granger drainage of the lower Yakima showed even greater frequency, with atrazine being detected in 99% of the samples collected from 1999-2004 (Burke et al. 2006). Simazine, another triazine herbicide was also commonly detected at frequencies ranging from 38-74% (Burke et al. 2006). Chlorpyrifos, diazinon, and malathion were among the most frequent insecticides detected with annual detection frequencies as high as 31%, 16%, and 7% of the samples, respectively (Anderson et al. 2007; Burke et al. 2006). Several other cholinesterase-inhibiting insecticides were also detected in samples from the lower Yakima monitoring. They include the organophosphates azinphos methyl, dimethoate, ethoprop, and disulfoton, and the carbamates aldicarb, aldicarb sulfone, and carbaryl (Anderson et al. 2007; Burke 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 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 active ingredients 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 lifecycle. Chlorpyrifos, diazinon, and malathion are widely used pesticides and their detection is common in freshwater habitats within the four western states where listed Pacific salmonids are distributed. Therefore, we expect some individuals within all the listed Pacific salmon and steelhead ESUs will be exposed to these chemicals and other stressors of the action. Concentrations of chlorpyrifos, diazinon, and malathion can occur at levels well over 100 ug/l and upwards of 1,000 ug/l based on measured environmental concentrations and exposure models. 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 active ingredients. More specifically:

- Although RED and IRED 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.

Several authors have utilized monitoring data to predict exposure distributions to aquatic species (Geisy et al. 1999; Giddings et al. 2000; Hall 2002a; Hall 2002b; Hall and Anderson 2000; Poletika et al. 2002). A major limitation of these assessments is that the monitoring data utilized were not designed to determine exposure to listed salmonids. 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 chlorpyrifos, diazinon, and malathion 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 37, 429, and 77 ug/L predicted for registered uses of chlorpyrifos, diazinon, and malathion, respectively. We generated additional exposure estimates for shallow off-channel habitats with predicted concentrations exceeding 1,000 ug/L for all three compounds. Additionally, we considered monitoring data presented by EPA and from other sources which indicate comparable concentrations of chlorpyrifos, diazinon, and malathion have been detected in surface waters within the four states where the listed salmon and steelhead are distributed (6, 67, and over 1,000 ug/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 surveys indicate chum salmon fry are distributed extremely close to the shoreline and concentrated in the top 6 inches of water. Chum salmon fry are less likely to be exposed to high concentrations of pesticides than other salmonids given the habitat they occupy and the duration of time spent in the shallow water habitats. 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. Thehighest concentrations detected in surface waters were consistently those associated with large-scale spray programs. 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 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 (
Figure **1**) to individual salmonids and their supporting habitats. We constructed a visual conceptual model to guide development of risk hypotheses and assessment endpoints to highlight potential uncertainties uncovered by literature searches and evaluations. We begin the response analysis by describing the toxic mode and mechanism of action of chlorpyrifos, diazinon, and malathion. Next we briefly summarize the toxicity data presented in the three BEs and assign the information to applicable assessment endpoints. We then evaluate toxicity information from other sources related to each assessment endpoint. 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 indentified as applicants, independent reports, and comments and information provided by other to EPA on the draft Opinion as described in the *Approach to the Assessment* section. The most relevant study results are those that directly address 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.

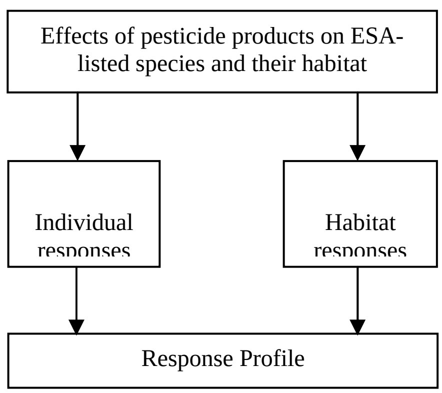


Figure 37. Response analysis

Mode and Mechanism of Action

Chlorpyrifos, diazinon, and malathion share a similar mode and mechanism of toxic action. The three insecticides share a similar chemical structure and act as neurotoxicants by impairing nerve cell transmission in vertebrates and invertebrates. They inhibit the enzyme acetylcholinesterase (AChE) present in cholinergic synapses. Fish and invertebrates metabolize OPs into oxon metabolites which are significantly more potent inhibitors of AChE than the parent compounds. Abiotic transformation in the environment also can lead to oxon formation (Wu and Laird 2003). The normal function of AChE is to breakdown (hydrolyze) the neurotransmitter, acetylcholine, thereby serving as an off switch to the electrochemical signal along nerve cells. Acetylcholinesterase 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 OPs may affect a wide array of physiological systems in fish (Figure 4).

The mechanism of action of OPs and oxons (inhibition of AChE), involves a series of enzyme-mediated reactions (Kennedy 1991). Briefly, in an irreversible[3] reaction OPs phosphorylate AChE thereby inhibiting AChE's normal activity to hydrolyze the neurotransmitter acetylcholine at nerve synapses. This reaction is similar to carbamate insecticides with the main exception being a carbamylation of AChE instead of a phosphorylation. Additionally, carbamates are typically referred to as reversible because they have a much slower rate of alkylation. The key result of AChE inhibition by OPs and carbamate insecticides is accumulation of acetylcholine in a nerve 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, including 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 (Antwi 1985; Coppage and Matthews 1974; Haines 1981; Holland et al. 1967; Rabeni and Stanley 1975; Williams and Sova 1966) as well as multiple reviews, text books (Geisy et al. 1999; Mineau 1991; Smith 1993), and wildlife poisoning cases document inhibition of AChE activity in exposed invertebrates (Detra and Collins 1986; Detra and Collins 1991) and vertebrates including

[3] The inhibition may not be completely "irreversible" as phosphorylated ACHE can spontaneously dephosphorylate to its active form. However spontaneous de-alkylation of one of the alkyl groups can occur which results in permanent inactivation known as aging, reviewed in Eto M. 1979. Organophosphorus Pesticides: Organic and Biological Chemistry. Boca Raton: CRC Press. 387 p, Fest C, Schmidt KJ. 1973. The Chemistry of Organophosphorus Pesticides. New York: Springer-Verlag. 339 p.

salmonids following exposures to OPs (Eder et al. 2007; Grange 2002; Hoy et al. 1991; Sandahl et al. 2004; Sandahl et al. 2005; Scholz et al. 2006; St. Aubin 2004; Tierney et al. 2007).

Studies with mixtures of AChE inhibiting insecticides

Because OPs share a common mode and mechanism of action, are used and applied in the same watersheds, and have demonstrated additive and synergistic effects (see below) in aquatic organisms, we discuss relevant mixture studies with fish. We also evaluate the response of salmonids and their habitat not just to single OPs, but also to common mixtures of OPs. Our analysis also includes an analysis of combinations of malathion, diazinon, and chlorpyrifos (see *Risk Characterization* section).

One of the earliest mixture studies evaluated bluegill survival following a range of exposure durations (24, 48, 72, or 96 h) to binary combinations of 19 insecticide mixtures (Macek 1975). The equation used to calculate mixture toxicity was, AB/ (A+B) = X; where AB was the number of dead fish from a mixture of pesticides A and B, and A + B was the sum of dead fish from A and B alone. The resulting ratios, X, were designated as 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. Malathion containing mixtures resulted in additive (DDT, toxaphene), synergistic (with Baytex [OP], parathion [OP], carbaryl [carbamate], perthane) and antagonistic (with copper sulfate). Antagonism is when the cumulative toxicity of a mixture is less than additive. Caution should be placed on the difference between additive and synergistic designations as the threshold was arbitrarily set at 1.5 and mixture results with DDT and toxaphene were at 1.31 and 1.14, respectively. Diazinon and parathion were synergistic to bluegill survival, i.e., more fish died than predicted. Validation of chemical concentrations with analytical chemistry was not conducted. Although the lack of raw data makes it difficult to determine exact concentrations tested, the study showed that both additive and synergistic responses occurred with OPs and particularly in combinations containing malathion.

Additive toxicity of binary combinations of OPs and carbamates was demonstrated from *in vitro* experiments with Chinook salmon (Scholz et al. 2006). The oxons of diazinon, chlorpyrifos, and malathion in addition to the carbamates carbaryl and carbofuran caused additive toxicity as measured by AChE inhibition in salmonid brain tissue (Scholz 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 sets of experiments to investigate whether additive toxicity as measured by AChE inhibition also occurred when live fishes were exposed for 96 h to the parent compounds, i.e., *in vivo* exposures.

The results of the second set of experiments were unexpected. 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 et al. *In Press*). The results have been presented at several scientific meetings and the raw data were made available to us. As with the *in vitro* study, brain AChE inhibition in juvenile coho salmon (*O. kisutch*) exposed to sublethal concentrations of chlorpyrifos, diazinon, and malathion as well as the carbamates carbaryl and carbofuran were measured (Laetz et al. *In Press*). Dose-response data for individual chemicals were normalized to their respective EC_{50} concentrations and collectively fit to a non-linear regression. The regression line was used to determine whether toxicological responses to binary mixtures were antagonistic, additive, or synergistic. No binary mixtures resulted in antagonism. Additivity and synergism were both observed, with a greater degree of synergism at higher exposure concentrations. Moreover, certain combinations of OPs were lethal at concentrations that were sublethal in single chemical trials. Concentrations of each insecticide are listed in Table 57 and combinations that resulted in mortality can be found in Table 58. 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 57. Concentrations (ug/L) of insecticides used in mixture exposures. EC50s were calculated from dose-response data using non-linear regression. Coho salmon exposed to 1.0, 0.4, or 0.1 EC50 treatments had an equipotent amount of each OP within the treatment e.g., to attain the 1.0 EC50 treatment for diazinon and chlorpyrifos, 1.0 ug/L of chlorpyrifos (0.5 the EC50) was combined with 72.5 ug/L (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
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
Carbaryl	145.8	72.9	29.2	7.3
Carbofuran	58.4	29.2	11.7	2.9

As determined by the regression, these levels of enzyme inhibition would result from exposure to 0.1, 0.4, and 1.0 EC_{50} 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. 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 EC_{50} (malathion at 37.3, chlorpyrifos at 2, diazinon at 72.5 ug/L), binary combinations produced synergistic toxicity. Many fish species die following high rates of acute brain AChE inhibition, i.e., 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. Fish exposed to these OP mixtures showed toxic signs 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 with some combinations. Thus, for binary combinations of malathion, diazinon, and chlorpyrifos synergism is likely to occur at exposure concentrations that were below the lowest used in (Laetz et al. *In Press*), i.e,. chlorpyrifos less than 0.1 ug/L; diazinon less than 7.3 ug/L; malathion less than 3.7 ug/L. The mechanism for synergistic toxicity in salmonids is unknown.

Table 58. Mixture concentrations resulting in 100% mortality of juvenile coho following 96 h exposures (Laetz et al. *In Press*).

OP mixture	Concentration, ug/L
diazinon + malathion	72.5 diazinon, 37.3 malathion 29.0 diazinon, 14.9 malathion
chlorpyrifos + malathion	1.0 chlorpyrifos, 37.3 malathion

We expect that juvenile salmonids exposed to these effect concentrations in the environment will respond similarly. Thus in some cases, juvenile salmonids will die. Unfortunately, we are unable to create a predictive model of synergistic toxicity as dose-response relationships with multiple ratios of pesticides are not available and the mechanism remains to be determined. That said, we conducted a mixture analysis with chlorpyrifos, diazinon, and malathion 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 (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" from published, peer-reviewed scientific publications (books and journals) or submitted by pesticide registrants during the registration process. 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 2004c). Population-level endpoints and analyses were generally absent in the BEs, other than a few measurements of fish and aquatic invertebrate reproduction. The BEs also presented some information on multispecies microcosm and mesocosm studies. The individual organismal endpoints were not translated into consequences to populations. NMFS translated effects to individual salmonids into potential population level consequences in the *Risk Characterization* portion of the *Effects of the Proposed Action* section, and ultimately made a conclusion on the likely risk to listed species.

Survival of individuals is typically measured by incidences of death following 96 hour (h) exposures (acute test) and incidences of death following 21 day (d), 30 d, 32 d, and "full life cycle" exposures (chronic tests) to a subset of freshwater and marine fish species reared in laboratories under controlled conditions (temperature, pH, light, salinity, etc.,) (EPA 2004c). 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. It is derived from the number of surviving individuals at each concentration tested following a 96 h exposure and is usually estimated by probit or logit analysis and recently by statistical curve fitting techniques. 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 lifestage of the listed species or a representative surrogate. In the case of ESA-listed Pacific salmonids, there are several surrogates including hatchery reared coho salmon, Chinook salmon, steelhead, and chum salmon, as well as rainbow trout[4]. Unfortunately, slopes,

[4] Rainbow trout and steelhead are the same genus species (*Oncorhynchus mykiss*), with the key differentiation that steelhead migrate to the ocean while rainbow trout remain in freshwaters. Rainbow

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. 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 studies. We evaluate the likelihood of concentrations that are expected to kill fish and apply qualitative and quantitative methods to infer population-level responses of ESA-listed salmonids within the *Risk Characterization* section (Figure 2).

Growth of individual organisms is an assessment endpoint derived from chronic fish, invertebrate, and algae toxicity tests summarized in the BEs. However, invertebrate and fish population responses to reductions in individual growth were not described in the BEs. This is a data gap as we are required to assess population-level consequences from reductions of an individual's fitness (e.g., growth).

Reproduction, at the scale of an individual, can be measured by the number of offspring per female (fecundity) while at the scale of a population by measuring the number of offspring per female in a population over multiple generations. The BEs summarized reproductive endpoints at the individual scale from chronic, freshwater fish experiments. Other assessment measures of reproduction include egg size, spawning success, sperm and egg viability, gonadal development, and hormone levels- most of which are not usually measured in standardized toxicity experiments.

Some of the BEs estimated sublethal effects to Pacific salmonids from short-term, acute lethality tests when chronic data were unavailable (e.g., within the chlorpyrifos BE). Qualitative observations of sublethal effects were summarized from 96 h lethality dose-response bioassays. These observations generally were limited, and when noted, pertained to unusual swimming behaviors. None of these behaviors were rigorously measured and therefore are of limited value to assessing the effects of these OP 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. Formulation toxicity information was presented but generally not discussed or used in EPA's estimates of risk.

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

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.

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.

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.

Chlorpyrifos

Chlorpyrifos
Assessment endpoint: Fish survival
Assessment measure: 96 h survival from laboratory bioassays reported as an LC50.
Freshwater fish 96 h LC50s ranged from 0.8 – 2,200 ug/L for formulated products, technical grade formulations, and active ingredient (Table 3 *in* (EPA 2003)[5]. For tests with the active ingredient, the LC50 range is 1.3 – 595 ug/L. Salmonid LC50s ranged from <1.0 – 2,200 ug/L (ten reported studies on rainbow trout [*O. mykiss*]). Eight of the ten LC50s were below 8.3 ug/L, while the remaining two were 51 ug/L and 2,200 ug/L (EPA 2003). Under EPA's toxicity classification system chlorpyrifos is "very highly toxic to moderately toxic" (LC50 < 100 ug/L is rated as "very highly toxic"). Eighty percent of the reported LC50s fall in the "very highly toxic" category (EPA 2003). Formulation and technical grade exposures of chlorpyrifos resulted in similar 96 h LC50s. Rainbow trout and bluegill sunfish were equally sensitive to acute concentrations of chlorpyrifos. Fathead minnows appeared much less sensitive to the same concentrations.

Temperature influenced chlorpyrifos' toxicity in freshwater fish. In a rainbow trout study, LC50s decreased as temperatures increased in a dose dependant manner; at 2 °C LC50 = 51 ug/L, at 7 °C LC50 = 15 ug/L, at 13 °C LC50 = 7.1, and at 18 °C LC50 < 1.0

[5] From the list of LC50s in Table 3, EPA selected 1.8 ug/L as the effect concentration for the risk quotient analysis. Operationally EPA divided 1.8 ug/L by 20 to determine the threshold concentration at which no direct effects to individual ESA-listed salmonids are expected. This value is 0.09 ug/L chlorpyrifos. It is not explained why 0.8 ug/L was not selected as the lowest LC50. If it was selected, EPA's no effect threshold would be reduced to 0.04 ug/L.

ug/L. According to these results, chlorpyrifos is approximately 51 times more toxic at 18 °C than 2 °C. The temperature effect was also observed in Bluegill sunfish (*Lepomis macrochirus*), although less pronounced, where at 13 °C LC50 = 4.2 ug/L, at 18 °C LC50 = 1.8 ug/L, at 24 °C LC50 = 2.5 ug/L, and at 29 °C LC50 = 1.7 ug/L (Macek 1975). These data suggest a pronounced temperature effect on the acute toxicity of chlorpyrifos to salmonids and emphasize the necessity of evaluating chlorpyrifos' and other OP's effects in combination with elevated temperatures. Other water quality parameters such as pH and water hardness were tested to determine their potential effect on chlorpyrifos' toxicity to fish survival. Results were not definitive for water hardness, but as pH increased acute toxicity increased. This was demonstrated in three species, lake trout (*Salvelinus namaycush*), bluegill sunfish (*Lepomis macrochirus*), and cutthroat trout (*Salmo clarki*). Formulated product toxicity closely mirrored LC50 results of chlorpyrifos exposure, ranging from 0.8 ug/L – 2200 ug/L. It is noteworthy that the lowest LC50 reported, 0.80 ug/L was conducted with a formulation, Dursban 6. The lowest chlorpyrifos LC50 was 1.8 ug/L, 55% less toxic than Dursban 6. No information was provided on individual ingredient toxicity of Dursban 6. However, it is a reasonable deduction that the increased toxicity is due to the aggregated toxic effect of the mixture within the formulation. Nevertheless, a definitive result on toxic potency is not possible at this time without parallel tests comparing chlorpyrifos' and chlorpyrifos-containing formulations' acute lethality. The BE concluded that chlorpyrifos is "very highly toxic to fish" and EPA's screening level risk assessment noted concerns for direct, lethal effects to fish.

Chlorpyrifos
Assessment endpoint: Reproduction
Assessment measure: Number of offspring, number of fish that attained sexual maturity by 136 d, number of spawns per spawning pair

Results from two life-cycle tests with fathead minnows were reported, one with technical grade chlorpyrifos (i.e., active ingredient only) and one with a formulated product, Dursban CR (Jarvinen et al. 1983). At 1.09 ug/L, both survival of adult fathead minnows was reduced by 14% on Day 12, and number of offspring was reduced by 35% at Day 5. In experiments with a formulated product (Dursban CR), there was a statistically significant effect on weight of adults, biomass of offspring, and a 25% reduction in maturation of offspring at 0.12 ug/L (Jarvinen et al. 1983). A significant reduction in the number of sexually mature fish at 136 d was observed at all test concentrations compared to control fish with a strong correlation between chlorpyrifos concentration and percent of sexually mature fathead minnows, (r) coefficient = 0.71. The mean number of eggs produced by females was reduced at all exposure concentrations with statistically significant reductions occurring at 0.63 and 2.68 ug/L. At 0.12 and 0.27 ug/L, egg production was reduced by 44 %. At 0.63 and 1.21 ug/L egg production was reduced by 60%, and at 2.68 ug/L egg production was reduced by 89%. Embryo hatchability was significantly reduced at 2.68 ug/L and only 2 of the 8 pairs of spawners spawned effectively enough to produce embryos for the hatchability experiments. The BE concluded that these two studies indicated that adverse effects occur in both generations tested and that the second generation is more sensitive than the first generation (EPA 2000a). It is noted that in acute toxicity tests, fathead minnows were significantly less sensitive (by two orders of magnitude) to chlorpyrifos than salmonids. This makes it difficult to translate these chronic fathead minnow data to how salmonids would respond. However, these results indicate that fish reproduction is significantly impaired at concentrations from 0.12 – 2.68 ug/L chlorpyrifos and possibly at lower concentrations for listed salmonids.

Chlorpyrifos
Assessment endpoint: Fish growth
Assessment Measure: Growth rate, weight, length, or biomass of second generation as measured in chronic toxicity tests
The BE identified three of five chronic test results that reported growth effects to fathead minnows (*P. promelas*). Two of the experiments were classified as freshwater fish early lifestage toxicity tests and the third was classified as a freshwater fish lifecycle test. Growth was significantly affected at 3.2 ug/L (16% reduced body weight) and at 4.8 ug/L (32% reduced body weight) following 32 d exposures in separate experiments. In the lifecycle test, body weight of fathead minnows was reduced by 9%, and a 53% reduction in biomass of eggs was measured at 0.12 ug/L. Although juvenile fathead minnows' growth is affected at 0.12 ug/L, it is difficult to extrapolate the degree to which juvenile salmonids growth would be affected at these concentrations. EPA concluded that "chronic risks to freshwater fish are likely to be considerably greater than the risk quotients estimated for chlorpyrifos" because fathead minnows are much less sensitive than other cold water fish such as salmonids (EPA 2000a).

Chlorpyrifos
Assessment endpoint: Habitat- salmonid prey
Assessment measure: Aquatic invertebrate survival, growth, reproduction from acute and chronic laboratory toxicity tests
Many freshwater acute toxicity tests on aquatic invertebrates have been conducted with chlorpyrifos, its primary degradate TCP, and multiple chlorpyrifos-containing product formulations. The BE summarized acute studies by stating that "technical grade chlorpyrifos is very highly toxic to several freshwater invertebrates including adult life stages" (EPA 2003). Acute LC50s of four species that salmonids typically eat ranged from 0.1 - 50 ug/L chlorpyrifos. The BE reported several acute LC50 study results

ranging from 0.05 - 0.8 ug/L[6] for the daphnid/water flea *Ceriodaphnia dubia*, a species consumed by salmonid fry and juvenile. Table 5 in the BE provided acute 96 h LC50/EC50 values for a variety of aquatic insects and other invertebrates from 0.039 - > 600 ug/L. Caddisflies, mayflies, midges, stoneflies, daphnids, amphipods, and copepods (all commonly consumed by ESA-listed salmonids) were highly sensitive to chlorpyrifos reflected by LC50s well below 1 ug/L (EPA 2003).

EPA reported on a single 21 d chronic study with another daphnid, *D. magna* (EPA 2003). *Daphnia magna's* survival and reproduction, measured by number of offspring, were significantly reduced at 0.08 ug/L with a reported reproductive LOEC of 0.04 ug/L. No other sublethal endpoints from chronic studies were reported for other salmonid prey items. EPA concluded that, "The high toxicity to organisms that serve as food items for threatened and endangered Pacific salmon and steelhead are also of significant concern in areas where there is considerable chlorpyrifos use" (EPA 2003). We concur with this statement. We add that based on the acute toxicity to aquatic invertebrates, significant concern exists where chlorpyrifos is applied and expected to enter salmonid aquatic habitats.

Degradates of chlorpyrifos
Assessment endpoint: Fish survival
Assessment measure: Fish and aquatic invertebrate survival, growth, reproduction from acute and chronic laboratory toxicity tests
No information was presented on the toxicity of chlorpyrifos-oxon, a known degradate of chlorpyrifos (Jarvinen and Tanner 1982; EPA 2000a). 3,5,6-trichloro-2-pyridinol (TCP) was identified by EPA as a primary degradate of chlorpyrifos. Twelve 96 h LC50s were reported in Table 9 for TCP and ranged from 1.5 – 83 mg/L (EPA 2003). Seven of the species were salmonids and LC50s ranged from 1.5 – 12.6 mg/L TCP. These values suggest that TCP is significantly less acutely toxic to salmonids than chlorpyrifos. No sublethal endpoints or chronic tests were discussed for TCP or any other degradates, therefore it is difficult to make any definitive comparisons of toxicity between chlorpyrifos and degradates of chlorpyrifos. We do consider chlorpyrifos-oxon as more toxic than parent chlorpyrifos, however this information was not reviewed in the BE.

Formulations and other (inert) ingredients found in chlorpyrifos' formulations
Assessment endpoint: Fish survival, aquatic invertebrate survival, primary production

[6] The references to these study results could not be identified or located by EPA. EPA. 2003. Chlorpyrifos Analysis of Risks to Endangered and Threatened Salmon and Steelhead. Office of Pesticide Programs. p 134.

Assessment measure: Aquatic invertebrate survival, growth, reproduction from acute and chronic laboratory toxicity tests

The acute toxicity (48 or 96 h LC50s) of three formulations of chlorpyrifos (Lorsban 15G, 75 WG, and 4E) to fish, aquatic invertebrate, and an alga were presented in Appendix 2 (EPA 2003). However, the species tested were not identified. One of the surfactants in Lorsban 4E (1.5 % by weight of formulation) exhibited high acute toxicity with each species (135 ug/L LC50 fish, 43 ug/L LC50 invertebrates, and 27 ug/L LC50 algae). The source information for this ingredient appears to be based on a study with NP which is also used as a pesticide adjuvant. We discuss the toxicity of NP later in the document as it is commonly added to formulated products as an adjuvant. Some of the ingredients found in chlorpyrifos formulations had no reported toxicity data. One ingredient with no data is labeled as a carrier in Lorsban 15G and represents 82.5% by formulation weight. Dow Agro Sciences, manufacturer of Lorsban products, reported that this carrier is clay which is not expected to be acutely toxic. An emulsifier in Lorsban 75WG had no toxicity data reported for it although it made up more than 20.62 % by weight of the formulated product. Several of the ingredients exhibited low acute toxicity (in the high mg/L range) and were not major components of the formulation. NMFS is certain that these three formulations do not represent all the formulations currently registered that contain chlorpyrifos[7]. It is important to note that more than 4,000 other/inert ingredients are currently registered for use across the U.S (EPA 2008b).

Identified data gaps and uncertainties of chlorpyrifos' toxicity information present in BE:

- Reported LC50s not accompanied by slopes, experimental design (number of treatments and replicates, lifestage of organism, concentrations tested), CIs;

- No sublethal data discussed for salmonids;

- Chlorpyrifos oxon toxicity data not presented or summarized;

- Few toxicity data on formulations, other ingredients within formulations;

- Sensitivity of surrogate lab strains compared to wild fish with different environmental stressors;

- No data summarized for mixture toxicity including tank mixtures and environmental mixtures.

[7] The BE referenced eight labels in an attachment, however possibly hundreds are currently registered.

Diazinon

Diazinon:
Assessment endpoint: Fish survival
Assessment measure: 96 h survival from laboratory bioassays reported as an LC50
Numerous LC50s were reported in Tables 3, 5, 8 (EPA 2002)[8]. Freshwater fish 96 h LC50s ranged from 90 – 7,800 ug/L for formulated products, technical grade formulations, and the active ingredient [Table 3; (EPA 2002)]. Reported LC50s ranged from 90 – 7,800 ug/L diazinon [Table 3; (EPA 2002)]. The range of salmonid LC50s was 90 – 2,760 ug/L and included the salmonid species *O. mykiss* (n = 4), *O. Clarki* (n = 2), *Salvelinus fontinalis* (n = 1), and *S. namaycush* (n = 1). The range of values indicates a high degree of variability in the sensitivity of salmonid species. Two species of *Oncorhynchus* were tested. Rainbow trout (*O. mykiss*) were sensitive, yet 96 h LC50s varied over two orders of magnitude (90 – 1,650 ug/L). Comparatively, diazinon is less acutely toxic to salmonids than chlorpyrifos (lowest LC50 of <1.0 ug/L). The salmonids, *O. clarki*, *S. namaycush*, and *S. fontinalis*, showed high variability in LC50s as well (602-2,760 ug/L). Fathead minnows were the least sensitive of the fish LC50s reported wherein 50% of the individuals died at 7,800 ug/L diazinon. No life stage or dose-response slope information was provided for any of the tests. The BE also summarized other reported LC50s from EPA's AQUIRE database (EPA 2002). Rainbow trout LC50s ranged from 400 to 6200 ug/L (n = 5). A further analysis of individual studies referenced in the BE is not possible as primary sources of information were not provided. Fifty percent of tested marine and estuarine fishes died at similar concentrations of diazinon compared with freshwater fishes; the LC50 range was 10 - 1470 ug/L. The marine species, *Chasmichthys dolichognathus*, was the most sensitive fish tested with an LC50 of < 0.1 ug/L. No tests were reported that evaluated diazinon-induced salmonid mortalities in salt water. Although no studies were reported that addressed the influence of temperature on diazinon's acute lethality, we expect incidences of death to increase when salmonids are jointly exposed to diazinon and elevated temperatures given this response was observed for chlorpyrifos.

Diazinon:
Assessment endpoint: Growth
Assessment measure: Weight
Following 274 d of exposure to 2.4 ug/L diazinon brook trout were smaller, and died at 9.6 ug/L (Allison and Hernandez 1977). At 0.8 ug/L, progeny of exposed trout were significantly smaller than progeny of unexposed trout. EPA concluded that brook trout

[8] EPA indicated that caution should be exercised in assessing LC50 values from older studies due to the presence of a degradate/impuirty called sulfotep which is apparently more toxic than diazinon. However, the suspect values were not identified in the BE, so we included all reported LC50s.

were significantly more sensitive than fathead minnows which illustrates that fathead minnows are an imperfect surrogate.

Diazinon:
Assessment endpoint: Early lifestage development
Assessment measure: Hatching success of progeny, qualitative observations of spinal shape

Progeny of fathead minnows continuously exposed for 274 d had reduced hatchability at 3.2 ug/L (Allison and Hermanutz 1977). Additionally, scoliosis in parental fathead minnows occurred at concentrations as low as 3.2 ug/L after 274 d. However, scoliosis was not observed after 19 weeks of exposure and scoliosis was not observed in the progeny after 60 d of exposure (Allison and Hermanutz 1977). Statistical results for occurrence of scoliosis were not reported for these observations.

Diazinon:
Assessment endpoint: Fish olfaction and olfactory-mediated behaviors
Assessment measure: Homing of adult salmon, feeding behavior

Olfaction is an ecologically relevant sensory system that mediates a suite of fish behaviors involved in feeding, predator avoidance, kin recognition, spawning, homing, and migration. Two studies were briefly discussed regarding the effects of diazinon on olfactory-mediated behaviors. One study indicated statistically significant effects to juvenile coho swimming and feeding behaviors in the presence of an alarm cue following 24 h exposures of 1 and 10 ug/L compared to control fish, and reduced homing at 0.1 ug/L (Scholz et al. 2000). The other study, tested Atlantic salmon's olfactory response to diazinon and was dismissed by EPA because "the nature of their test system, direct exposure of olfactory rosettes, could not be quantitatively related to exposures in the natural environment" (Moore and Waring 1996). We found both studies to be highly relevant and discuss them in greater detail later in this section (*Summary of Toxicity Information from Other Sources; Assessment endpoints: Olfaction and olfactory-mediated behaviors.)* .

Diazinon:
Assessment endpoint: Habitat: Salmonid prey
Assessment measure: acute and chronic laboratory toxicity tests

Aquatic invertebrate LC50s (0.2 - 25 ug/L, n = 7) indicate that diazinon is acutely toxic at low ug/L concentrations [Table 3, (EPA 2000b)]. The BE also summarized other reported LC50s from EPA's AQUIRE database (EPA 2000b). The majority of the LC50s were derived from experiments with aquatic invertebrates that are common prey items for juvenile salmonids such as amphipods, mayflies, caddisflies, stoneflies, midges, copepods, and water fleas/daphnids. A range of acute exposures (3 h, 24 h, 48 h, and 96 h) were tested in dose-response experiments with salmonid prey items. Reported LC50s varied considerably for aquatic invertebrates (0.03-2,500 ug/L). Although many of the experiments exposed test species to formulations, specific names of formulations were not reported, hampering a comparison of current-use labels. No data were presented in the BE on aquatic macrophyte toxicity, however two tests were summarized with freshwater algae. LC50s ranged from 3.7 mg/L to more than 10 mg/L indicating that the algae tested were much less sensitive to diazinon than aquatic invertebrates and fish. Concentrations of diazinon needed to affect growth of algae would result in death of most fish and invertebrates tested. Therefore, effects of diazinon to algae are less meaningful.

Identified data gaps and uncertainties of diazinon's toxicity information present in BE:

- No information was presented on the toxicity of ingredients within pesticide formulations containing diazinon;

- No information was presented on the toxicity to aquatic species for two of the known degradates, oxypryimidine and diazoxon;

- No study results were reported for diazinon's toxic effects to fish reproduction;

- No information was presented on mixture toxicity of diazinon with other similar and co-occurring organophosphates.

Malathion

Malathion-
Assessment endpoint: Fish survival
Assessment measure: 96-h survival from laboratory bioassays reported as an LC50.
The acute toxicity studies reported indicate that freshwater fishes exposed to malathion or formulations containing malathion die following 96 h exposures in the low ug/L range, which is comparable to chlorpyrifos and is more toxic than diazinon. The lowest fish LC50 was 1.5 ug/L for Indian catfish. Survival values included LC50s from salmonids [n = 7; 4.1 - 174 ug/L LC50; (EPA 2000b)]. The 4.1 ug/L value was not incorporated into the analysis because of experimental flaws. Survival estimates (LC50s) from EPA's AQUIRE database were reported for rainbow trout (*O. mykiss*; n = 14), coho salmon (*O. kisutch*; n =1), Chinook salmon (*O. tshawytscha*; n = 3), cutthroat trout (*O. clarki*; n = 2), brown trout (S*almo trutta*; n = 1), brook trout (*S. fontinalis*; n = 2), the lowest reported LC50 was 2.8 ug/L and the highest LC50 was 234 ug/L for salmonids. The abundance of LC50s for salmonids significantly reduces acute survival as a data gap. Uncertainty remains as to which LC50 is the most accurate given the large variability in reported salmonid LC50s, 2.8 – 234 ug/L. We cannot comment on lifestage sensitivity as no age information was provided in the reported LC50s. Additionally, we cannot predict

toxicity of concentrations below or above the LC50 as slope or concentration ranges tested were not provided. Although no studies were reported that addressed the influence of temperature on malathion's acute lethality to salmonids, a study with Bluegill showed a statistically significant inverse relationship between acute toxicity (96 h LC50) and temperature (Mayer and Ellersieck 1986). We expect incidences of death to increase when salmonids are jointly exposed to malathion and elevated temperatures as was observed with chlorpyrifos.

Malathion:
Assessment endpoint: Reproduction and growth
Assessment Measure: Chronic toxicity tests, no specific toxicity information provided

The BE reported results from two fish experiments (rainbow trout and fathead minnow) that when combined addressed growth and reproduction endpoints. However, EPA did not discern which effect was attributed to a particular study. Therefore, we can only comment on the reported LOEC and NOEC from each study. Following a 97 d exposure, *O. mykiss* had a significant effect to either growth or reproduction, LOEC = 44 and a NOEC = 21 ug/L. In *P. promelas*, a 350 d exposure had a significant effect to either growth or reproduction, LOEC = 350 ug/L and NOEC = not determined. The information reported by EPA indicates a data gap on sublethal assessment endpoints in the BE. The fathead minnow study provides relevant information to the effects of malathion on sublethal assessment endpoints of growth and reproduction as *P. promelas* are much less sensitive (at least acutely) than salmonids to malathion.

Malathion:
Assessment endpoint: Habitat: Salmonid prey
Assessment measure: Aquatic invertebrate survival, growth, reproduction

Malathion is acutely toxic to a wide array of aquatic invertebrates, many of which are documented salmonid prey items as reported in Table 25 (EPA 2004b). An abundance of studies indicate that malathion kills salmonid prey at < 1 ug/L. The lower range of acute toxicity values (48 h and 96 h LC50s) reported for prey items begins at 0.5 ug/L for an amphipod and 0.69 ug/L for a stonefly. Prey taxa tested included stoneflies, caddis flies, amphipods, copepods, midges, mayflies, and daphnids. A 21 day chronic test with daphnids showed that survival and number of progeny per adult were significantly lower than unexposed daphnids, reported in Table 22 (EPA 2004b). Concentrations of malathion as low as 0.1 ug/L affected reproduction, growth, and survival of daphnids (EPA 2004b).

Degradate of malathion: Malaoxon (malathion-oxon)-
Assessment endpoint: Survival
Assessment measure: 2, 24, 48 h LC50s

Five test results were discussed from acute exposures to medaka, pumpkinseed, perch, black bullhead, and a midge. Survival was reported as LC50s and ranged from 5.4 - 450 ug/L, however comparisons to other fish LC50s is complicated by the differences in exposure duration and species. None of the tests were run for 96 h. Tests were run at 2, 24, or 48 h. The assessment endpoint was not reported for 2 h exposures although the lowest effect concentration was 0.25 ug/L for pumpkinseed fish.

Other ingredients within malathion-containing formulations:
Assessment endpoint: Multiple
Assessment measure: Multiple
No fish toxicity data on malathion products that contain other active pesticide ingredients were reported (EPA 2004b). However, acute and chronic toxicity data for some of the other ingredients found in formulated products were discussed. These other ingredients are briefly described below and include piperonyl butoxide, methoxychlor, resmethrin, captan, and carbaryl[9].

Piperonyl butoxide is a chemical that inhibits the biotransformation of OPs to their oxon metabolites, thereby decreasing the toxicity of the insecticide (Amweg and Weston 2007). According to the BE, it is a common constituent of insecticide containing formulations. It is also very highly toxic to aquatic invertebrates and fish (Table 27, EPA 2004). Two LC50s were reported for rainbow trout, 2.4 and 6.1 ug/L following 96 h exposures to a formulation containing piperonyl butoxide (Label not provided). *Daphnia magna* exposed for 48 h to piperonyl butoxide were very sensitive with reported EC50s of 0.51 and 1.7 ug/L. Other aquatic species were also tested and highly sensitive [see Table 27,(EPA 2004b)]. In longer term exposures piperonyl butoxide affects fish and aquatic invertebrates at concentrations as low as 0.11 and 0.12 ug/L, respectively. Assessment endpoints were not reported for LOEC or NOEC values presented (EPA 2004b).

Methoxychlor is an organo-chlorine insecticide that is very highly toxic to fish and aquatic invertebrates. It is a co-constituent in formulations with malathion, piperonyl butoxide, and others as reported by EPA. Reported LC50s and EC50s ranged from 0.78 – 3.32 ug/L. Formulated products appeared more toxic than methoxychlor alone (Table 29, (EPA 2004b)). One 96 h LC50 (1.7 ug/L) was reported for fish (Atlantic salmon [*Salmo salar*]) from an exposure to a formulation. No other fish studies were identified in the BE and no toxicity information was presented from longer term exposures to fish or aquatic invertebrates.

Resmethrin is a synthetic pyrethroid insecticide that is used to control flying insects in homes, greenhouses, etc, and for mosquito control. Resmethrin is very highly toxic to fish and aquatic invertebrates. Coho salmon and brown trout were also acutely sensitive (LC50s of 0.277, 1.5, and 1.77 ug/L in coho and 0.75 ug/L in brown

[9] NMFS and EPA are consulting on the effects of captan and carbaryl registered products on ESA-listed Pacific salmon and steelhead in a separate Opinion.

trout). *Daphnia magna* appeared to have less acute sensitivity compared to the fish with a reported LC50 of 3.1 ug/L. Chronic exposures to *Daphnia magna*, sheepshead minnow, fathead minnow, and rainbow trout indicate that adverse effects to aquatic organisms are likely at concentrations less than one ug/L. In the case of rainbow trout after a 52 d exposure, the LOEC was 0.59 ug/L with a reported NOEC of 0.32 ug/L.

Captan is a non-systemic fungicide used on fruit trees, ornamentals, and vegetables. It is very highly toxic to fish and aquatic invertebrates. Acute LC50 and EC50 values range from 0.056 (coho and Chinook salmon) – 8.4 ug/L (shrimp), some of the most toxic values reported in this Opinion. No aquatic insect data were reported in the BE. The toxicity to coho and Chinook salmon from captan indicates that salmonids exposed in the environment will kill fish, warranting measures to keep this material out of salmonid habitats. No toxicity information was reported for longer term exposures i.e., longer than 96 hours.

Carbaryl is a carbamate insecticide used on crops, livestock, poultry, pets, and estuarine mudflats to kill mud and ghost shrimp in Washington State. It is acutely toxic to fish in the low ug/L range, and moderately toxic to aquatic invertebrates according to EPA toxicity criteria. Cabaryl does bioaccumulate in aquatic species, including plants. Acute toxicity values range from 0.35 to 7.2 ug/L for freshwater fish (see Table 31; EPA 2004).

In most formulated products containing malathion and other active ingredients, malathion is the predominant active ingredient. However, one fruit tree spray contains 3.00 % malathion, 5.87 % captan, and 90.5 % carbaryl. The toxicity of carbaryl and captan is roughly equivalent to the acute toxicity of malathion in fish. Another product, a home fruit spray, contains 7.5% of malathion and 9.78% of captan. An agricultural alfalfa spray contains 13.787 % of methoxychlor and 23.807 % of malathion. Methoxychlor is very highly toxic to aquatic invertebrates, and its toxic effects are comparable to malathion.

These active ingredients appear comparable in toxicity to fish and aquatic invertebrates as malathion itself. Endpoint values (LC_{50}) ranged from 0.056 ug/L for captan (Chinook and coho salmon) to 8.8 ug/L for piperonyl butoxide (Sheepshead minnow). Piperonyl butoxide, which is used sometimes in combination with malathion to control mosquitoes, seems to be very highly toxic to mussels and appears more toxic to such organisms than malathion. Collectively, this information emphasizes the importance of addressing risk to all constituents within OP formulations to listed salmonids and their prey. However, the BEs did not provide a complete summary of currently registered labels. Thus, it is difficult if not impossible to determine what other active ingredients are in the formulations.

Identified data gaps and uncertainties of malathion toxicity information present in BE:

- Reported LC50s not accompanied by slopes, experimental design (number of treatments and replicates, lifestage of organism, concentrations tested), confidence intervals;

- Large range in reported salmonid LC50s (2.8 – 234 ug/L)

- Few sublethal data discussed for salmonids;

- Malathion oxon toxicity data limited to survival;

- Few toxicity data on formulations;

- No data summarized for mixture toxicity including tank mixtures and environmental mixtures.

Summary of Toxicity Information from Other Sources

Recall that assessment endpoints are biological attributes of salmonids and their habitat that are susceptible to the stressors of the action (Table 1). 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. 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. We 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; and 4) had no substantial flaws in the experimental design. When a study did not meet these components, 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, spontaneous swimming activity

Swimming is a critical function for anadromous salmonids that is necessary to complete their lifecycle. 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 chlorpyrifos, diazinon, and malathion. Swimming capacity is a measure of orientation to flow as well as the physical capacity to swim against it (Dodson and Mayfield 1979; Howard 1975). Swimming activity includes measurements of frequency and duration of movements, speed and distance traveled frequency and angle of turns, position in the water column, and form and pattern of swimming. A review paper published in 1990 summarized many of the experimental swimming behavioral studies and concluded that effects to swimming activity generally occur at lower concentrations than effects to swimming capacity (Little and Finger 1990). Therefore, measurements of swimming activity are usually more sensitive than measurements of swimming capacity. A likely reason is that fishes that have 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[10], 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 chlorpyrifos, diazinon, and malathion. Several of the studies also measured AChE inhibition 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.

Chlorpyrifos-

Spontaneous swimming speed, feeding swimming speed, feeding behaviors (number of food strikes, time period before first food strike), and brain and muscle AChE levels of juvenile coho salmon were evaluated following 96 h exposures (Sandahl et al. 2005). At 0.6, 1.2, 1.8, and 2.5 ug/L statistically significant effects were reported for all endpoints measured. A bench mark concentration analysis indicated that chlorpyrifos

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[10] The current hazard quotient-derived threshold for effects to threatened and endangered species utilized by EPA is 5 % (1/20[th]) 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.

concentrations of 0.4 ug/L are sufficient to inhibit brain AChE and feeding behavior by 10% (BMC10). Chlorpyrifos at 0.3 ug/L is sufficient to reduce the spontaneous swimming rate of individual coho by 10%. A statistically significant correlation existed between brain AChE activity and swimming behaviors indicating a putative relationship between AChE inhibition and swimming behaviors (Sandahl et al. 2005). We ranked this study as a highly relevant result to address effects of chlorpyrifos on salmonid swimming behaviors.

Chlorpyrifos inhibited AChE activity in a concentration-dependent manner relative to unexposed juvenile coho (control treatment) following 96 h exposures (at 5 ug/L = 18.2%, 10 ug/L= 47.8%, 20 ug/L =72.7%, and 40 ug/L = 78.7% relative to controls) (Tierney et al. 2007). Significant differences in AChE activity from the control occurred with exposures of 10 ug/L or greater ($p < 0.05$). Two types of swimming behaviors were measured, critical swimming performance and acceleration. Neither behavior differed significantly in unexposed fish or within chlorpyrifos-exposed treatments. Therefore, neither was more or less sensitive as an indicator of swimming impairment. At 20 and 40 ug/L, both critical swimming performance and acceleration were affected compared to controls (p values of 0.018 and 0.001, respectively). We ranked this study as highly relevant because it was conducted with juvenile coho and quantified impacts to swimming behavior.

Diazinon-

Juvenile rainbow trout exposed for 96 h to diazinon swam slower, covered less distance, turned less, turned more slowly, and had reduced AChE activity compared to unexposed fish (Brewer et al. 2001). During the exposure period, juvenile swimming activity was measured at 24 h and 96 h to 250, 500, and 1,000 ug/L. Following a recovery period of 48 h, swimming activity was measured to determine if recovery occurred. Reductions in distance traveled and speed of movement were apparent by 24 h in 500 and 1,000 ug/L. Fish exposed to 500 ug/L traveled less distance than control fish. Interestingly, at 500 ug/L juveniles showed no statistical difference in swimming speed at 96 h compared to unexposed fish. However following the 48 h recovery period, fish swam significantly slower than controls. A possible explanation provided by the authors was that salmon somehow compensated for this effect. Behavioral parameters were correlated with AChE activity in fish exposed to diazinon. Inhibition of AChE accounted for 44% and 41% of the variation measured in distance traveled and speed ($p = 0.02$), respectively, and as AChE activity increased so did distance traveled and speed. Tortuosity was not affected from any of the diazinon exposures. The number and binding affinity of muscarinic cholinergic receptors (MChR) were evaluated to investigate the potential for salmonids to adapt to diazinon. No statistically significant reductions were observed when compared to unexposed fish which highlighted a lack of adaptation (Beauvais and Jones 2000). We ranked this study as relevant because it was conducted with rainbow trout (a surrogate for steelhead and Pacific salmon) and quantified impacts to swimming behavior. Nevertheless, concentrations used were high compared to other study results. A highly relevant ranking was not given because validation of chemical concentrations was not performed. However, these study results provide support for a correlation between AChE inhibition and impaired swimming behavior, and show that swimming behavior is adversely affected by diazinon at concentration below reported LC50s.

Malathion-

Juvenile rainbow trout swimming activity was measured at 24 h, 96 h, and following a 48 h recovery period to 0, 20, and 40 ug/L malathion (Beauvais and Jones 2000; Brewer et al. 2001). Juveniles exposed for 24 h to malathion swam more slowly, covered less distance, turned less, turned slower, and had reduced ACHE activity compared to unexposed fish (Brewer et al. 2001). By 96 h, fish remained affected, swimming slower and covering less distance than control fish. Full recovery of affected swimming behaviors occurred after 48 h. The number and binding affinity of muscarinic cholinergic receptors (MChR) were evaluated to investigate the potential for salmonids to adapt to malathion. No statistically significant reductions in MChR were observed when compared to unexposed fish (Beauvais and Jones 2000). We ranked this study as relevant because it was conducted with rainbow trout (a surrogate for steelhead and Pacific salmon) and quantified impacts to swimming behavior. A highly relevant ranking was not assigned because validation of chemical concentrations was not performed. However, these study results provide support for a correlation between AChE inhibition and impaired swimming behavior, and show that swimming behavior is adversely affected by malathion following 24 and 96 h exposures.

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 ug/L in brook trout; 0, 55, 112, 175 ug/L in rainbow trout; and 0, 100, 200, 300 ug/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 ug/l (brook trout), 55 ug/L (rainbow trout), and 100 ug/L (coho). Coho required at least twice the concentration of malathion compared to brook and rainbow trout to inhibit AChE activity. 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 et al. 1990). Recovery of AChE in exposed salmonids took 25 d for brook trout, 35 d for rainbow trout, and 42 days for coho. There was no difference in recovery time based on concentrations tested within species. Post and Leasure (1974) concluded, "these figures are significant in that they point out the need for spacing malathion insecticide usage in ecosystems where this insecticide is used at intervals during a growing season." Additionally, Post and Leasure (1974) emphasized that where OP insecticides are used, "their effect must also be taken into consideration". We ranked this experiment as relevant as several salmonid species were tested using a rigorous experimental design, although validation of malathion concentrations was not performed.

Other AChE inhibiting insecticides effects on swimming and related behaviors- We also reviewed study results conducted with other OP and carbamate insecticides because both classes of compounds share a toxic mode of action, inhibition of AChE. Fenitrothion, carbaryl, parathion, and methyl parathion adversely affected a suite of swimming behaviors reviewed in (Little and Finger 1990). One noteworthy study investigated the effects of six pesticides including methyl-parathion (OP), DEF (OP), and carbaryl (carbamate) on rainbow trout swimming behavior (Little et al. 1990). All insecticides adversely affected spontaneous swimming activity while carbaryl and DEF also reduced swimming capacity in juvenile rainbow trout (Little et al. 1990). Experiments with carbaryl have shown that Cutthroat trout's swimming abilities are compromised by sublethal exposures (750 and 1,000 ug/L) resulting in increased predation (Labenia et al. 2007). Carbofuran, a carbamate insecticide, adversely affected swimming behaviors in goldfish (*Carassius auratus*) following 24 h and 48 h exposures to the lowest concentration tested, 5 ug/L (Bretaud et al. 2002). Swimming activity (fish swimming from one zone to another), the least sensitive endpoint, was significantly affected at 500 ug/L carbofuran, while burst swimming, the most sensitive endpoint, was significantly affected at 5 ug/L following 24 h exposure (Bretaud et al. 2002). Burst swimming behavior in goldfish was also significantly reduced from exposure to 1 ug/L carbofuran following a 4 h exposure (Saglio et al. 1996). In bluegill methyl-parathion adversely affected burst swimming behavior at 300 ug/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 ug/L. This suggests that these social behaviors are very sensitive to AChE inhibition (Henry and Atchison 1984). Although we found no studies that measured social behaviors of salmonids following OP or carbamate exposures, it is probable that behaviors predicated on swimming are sensitive to chlorpyrifos, diazinon, and malathion. In summary these results provide weight of evidence that OPs and carbamates adversely affect swimming behaviors at sublethal concentrations which can reduce individual survival (e.g., reduced predator avoidance).

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, 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

The olfactory sensory system in salmonids is particularly sensitive to toxic effects of metals and other 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 lifecycles, 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).

Chlorpyrifos-

Juvenile coho salmon lost 25, 50 and 50% of olfactory function following 7 d exposures to 0.625, 1.25, and 2.50 ug/L, respectively (Sandahl et al. 2004). AChE activity in coho salmon olfactory rosettes was inhibited by 25% at the highest exposure level tested, 2.5 ug/L. However no significant correlation between AChE inhibition and olfactory impairment was found. These results indicate that olfaction is impaired by chlorpyrifos exposures below 1 ug/L, and olfactory AChE activity is reduced at 2.5 ug/L. This study measured olfactory response of a listed salmonid species, coho, exposed to chlorpyrifos using a well-executed experimental design and therefore is ranked as highly relevant.

Diazinon-

We located two studies that investigated effects of diazinon on salmonid olfaction and olfactory –mediated behaviors; both were briefly discussed in the BE (Moore and Waring 1996; Scholz et al. 2000).

The first study investigated two aspects of diazinon's effect on olfaction in Atlantic salmon parr (Moore and Waring 1996). First, male parr were exposed to diazinon concentrations (0, 0.1, 1.0, 2.0, 5.0, 10, and 20 ug/L) for 30 minutes and EOG recordings were analyzed to determine parr's ability to detect female-released priming odorant $PGF_{2\alpha}$, a prostaglandin involved in spawning synchronization that also has a role as a primer on male plasma steroids and gonadotropin production. At 1.0 ug/L, diazinon significantly reduced the capacity for parr to detect $PGF_{2\alpha}$ by 22% compared to controls. At 20 ug/L, diazinon inhibited olfaction by 79%. Olfaction remained affected for up to 4-5 hrs post exposure, however the recovery time of longer term exposures were not tested. Second, diazinon's affect following 120 d exposures on male parr's plasma reproductive steroid levels was assessed following exposure to ovulating female's urine. Female urine, detected by males via olfaction, is important for a variety of male salmon reproductive priming behaviors including attraction detection of an ovulating female, and eliciting orientation behavior. Four male hormones (17, 20β-dihydroxy-4-pregnen-3-one [17,20βP], testosterone, 11-ketotestosterone [11-KT], and gonadotropin II [GtH II]) and milt were measured following diazinon exposures. Diazinon concentrations of 0.3 – 45 ug/L abolished the induction of 17, 20βP and 0.8-45 ug/L abolished the induction of GtH II. Testosterone and 11-KT levels were not significantly affected by diazinon. Milt production in parr was significantly reduced (~ 28%) at all concentrations of diazinon, 0.3 - 45 ug/L. In summary, the impairment of Atlantic salmon's ability to detect and respond to reproductive scents may lead to missed spawning opportunities. We infer that ESA-listed salmonids would likely have a similar impairment from exposure to diazinon.

The second study addressed two olfactory-mediated behaviors: predator avoidance behavior as measured by alarm response of juveniles, and homing ability of adults as measured by number of returning adults (Scholz et al. 2000). Both of these endpoints are ecologically relevant behaviors and were assessed in Chinook salmon after acute exposures. Following 2 h exposures to nominal concentrations (0.1, 1, and 10 ug/L diazinon), juvenile Chinook salmon showed reduced alarm response (as measured by pre and post swimming and feeding behaviors) at 1 and 10 ug/L ($p = 0.05$). Compared with unexposed juveniles, diazinon-treated Chinook salmon remained more active and fed more frequently when exposed to the predator alarm signal, skin extract from another Chinook. The lack of response to the alarm cue indicates that olfaction was impaired, leaving Chinook salmon oblivious to a predator's presence, thereby increasing the likelihood of being eaten. Swimming and feeding (food strikes/ minute) in the absence of the alarm cue were not affected by diazinon exposures as would be expected as maximal AChE inhibition generally takes many hours (Scholz et al. 2000). Homing of adult Chinook salmon was significantly affected at 10 ug/L diazinon where 6 of 40 fish returned compared with 16 of 40 fish in control treatment. At 0.1 and 1.0 ug/L, fewer fish returned (12 of 40) compared to controls (16 of 40) although the effect was not statistically significant. In summary, diazinon significantly impaired responses by juvenile Chinook salmon (*O. tshawytscha*) to alarm scents, thereby increasing their susceptibility to predation and also decreasing adult Chinook homing which may reduce their ability to locate their natal streams.

Collectivly, these two studies show that exposure to diazinon in the low ug/L range impairs predator avoidance behavior in juvenile Chinook salmon, homing in adult Chinook salmon, and reproductive priming and milt production in adult Atlantic salmon. Both studies' results are highly relevant to addressing the effects of diazinon on olfaction.

Malathion-
Olfaction may be impaired by malathion and other organophosphates given observations with chlorpyrifos and diazinon. However, we found no studies that measured fish olfaction or olfactory-mediated behaviors following exposures to malathion. This is a significant data gap.

Other OPs and carbamates-
Coho salmon exposed for 30 minutes to three carbamates (carbofuran, antisapstain IPBC, mancozeb) had reduced olfactory ability and affected AChE activity (Jarrard et al. 2004). Carbofuran reduced olfaction by 50% (EC50) at 10.4 ug/L, IPBC at an EC50 concentration of 1.28 ug/L, and mancozeb at an EC50 concentration of 2.05 mg/L. All three carbamates also affected AChE activity with highly variable results. This study shows that coho salmon's olfactory systems are very sensitive to carbamates over short (< 30 minutes) exposure periods.

Mixtures containing chlorpyrifos, diazinon, and malathion-
In a recent study, olfactory measurements were recorded from juvenile steelhead exposed for 96 h to an environmentally relevant pesticide mixture (Tierney et al. 2008). Three treatment concentrations of a mixture containing 10 pesticides were tested. Treatments of 0.1x (low), 1x (realistic), and 10x (high) of the 10 most prevalent pesticides detected in the Nicomekl River, a salmon producing river in British Columbia, Canada, were used. Within the three treatments, measured concentrations of chlorpyrifos were 1.7, 13.4, 114 ng/L; for diazinon 15.7, 157, 1820 ng/L; and for malathion 0, 46.3, and 926 ng/L. Juvenile steelhead exposed to these mixtures showed no significant reductions in olfactory response to a single odor (L-serine) presented against a background with no L-serine. However when steelhead were exposed to an increase in odor intensity from 10^{-5} to 10^{-3} l-serine, olfactory responses were significantly reduced by the realistic (1x) and high (10x) treatments (Tierney et al. 2008). These results indicate that at environmentally realistic concentrations of a mixture that includes chlorpyrifos, diazinon, and malathion, juvenile steelhead's ability to detect changes in odorant concentrations is compromised. Without properly functioning olfaction, behaviors that rely on smell such as homing and migration may be impaired. We ranked this study as highly relevant because it was conducted with juvenile steelhead, measured an ecologically relevant endpoint, used environmentally relevant concentrations detected in salmonid watersheds, and followed a rigorous experimental design. The degree to which salmonids' olfaction

is affected by OPs remains uncertain, however the evidence supports that olfaction is impaired following exposures to OPs.

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 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. 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, contaminated diet-induced AChE levels regained some of their activity. 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 chlorpyrifos, diazinon, and malathion 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, recovery of aquatic communities following OP exposures
Assessment measures: 24, 48, and 96 h survival of prey items from laboratory bioassays reported as LC50s; sublethal effects to prey items; field studies on community abundance; indices of biological integrity (IBI); community richness; community diversity;

Death of aquatic invertebrates in laboratory toxicity tests was summarized in each of the BEs. In summary, salmonid aquatic and terrestrial prey are highly sensitive to the three OP insecticides. Death of individuals and reductions in individual taxa and prey communities have been documented and are expected following exposures to OPs that achieve effect concentrations- some as low as ng/L levels. Complete or partial elimination of aquatic invertebrates from streams contaminated by insecticides has been documented for fenitrothion (OP), carbaryl (carbamate), and methoxychlor (another ingredient in malathion formulations) (Muirhead-Thomson 1987). A review of more than

60 field studies 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). Importantly, chlorpyrifos was one of the top three (azinphos-methyl (OP) and endosulfan were the other two) most frequently detected at levels expected to result in toxicity (Schulz 2004).

Drift, feeding behavior, swimming activity, and growth are sublethal endpoints of aquatic prey negatively affected by OP exposures (Davies and Cook 1993; Schulz 2004). Drift of aquatic invertebrates is an evolutionary response to aquatic stressors. However, insecticides, particularly OPs, can trigger catastrophic drift of salmonid prey items (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 ug/L cypermethrin. It is difficult to compare these effect concentrations to OP insecticides. However, it is illustrative of how insecticides can damage multiple endpoints of an aquatic community (Davies and Cook 1993). We expect that concentrations of chlorpyrifos, diazinon, and malathion sufficient to kill aquatic invertebrates will trigger catastrophic drift.

In one study, two instars of a midge that are common fish prey items, *Chironomus riparius*, and a caddisfly, *Hydropsyche angustipennis*, were assessed for their survival, activity, and growth following diazinon exposures (Stuijfzand et al. 2000). First instars died at lower concentrations (96 h LC50 = 1.3 ug/L, *H. angustipennis* and 22.8 ug/L, *C. riparius*) than older instars (96 h LC50 = 29.4 ug/L, *H. angustipennis* and 167 ug/L, *C. riparius*) and reductions in activity were more pronounced in the late instars (EC50 = 3.7 ug/L, *H. angustipennis*, 48 h) compared to the early instars (EC50 =14.5 ug/L, *H. angustipennis*, 48 h), highlighting differential life stage toxicity (Stuijfzand et al. 2000). These results suggest that developmental stage plays an important role in species sensitivity and careful comparisons of lifestage are warranted when ranking species sensitivity.

Several scientific peer-reviewed publications (Barron and Woodburn 1995; Leeuwangh 1994; Van Wijngaarden et al. 2005), registrant-submitted reports (Giesy et al. 1998), and EPA documents have reviewed multi-organism microcosm and mesocosm test results for the three OPs. Van Wijngaarden et al. (2005) conducted a literature review that listed ecological threshold values (e.g., $NOEC_{eco}$ and $LOEC_{eco}$) for chlorpyrifos and diazinon 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 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. Van Wijngaarden et al. (2005) found reductions in population densities from relatively low AChE-inhibiting insecticide concentrations (including chlorpyrifos and diazinon) of many salmonid prey organisms including the taxonomic groups Amphipoda, Cladocera, Copepoda, Isopoda, Ostracada, Trichoptera, Ephemeroptera, and Diptera. Adverse effects to these groups occurred well 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 or mesocosm experiments that measured responses of aquatic communities that contained salmonids and salmonid prey items simulataneously; a recognized data gap. Many studies evaluated aquatic invertebrate responses to the three OPs in static systems. Below we discuss some of these studies organized by OP.

Chlorpyrifos –
Several sources have reviewed the available chlorpyrifos mesocosm information (Giesy et al. 1998; Van Wijngaarden et al. 2005). Van Wijngaarden et al. (2005) utilized the results of 12 studies, two of which were conducted in running waters and 10 in static systems. The majority of exposure doses applied a single application and a minority applied multiple or continuous doses. For static systems, the reported $NOEC_{ecos}$ were highly consistent with the exact same value for the three independent studies, 0.1 ug/L. The reported $LOEC_{ecos}$ were 0.3 and 0.5 ug/L for slight effects and 0.1, 0.5, 0.9, 1.0, 1.0, 5, 5, 10, and 35 ug/L for more severe effects (Van Wijngaarden et al. 2005). For running water systems, the reported chlorpyrifos $NOEC_{eco}$ was 0.1 ug/L and 5 ug/L $LOEC_{ecos}$. A recent publication found significant changes to macroinvertbrate assemblages of artificial stream systems following a six hour exposure to chlorpyrifos at 1.2 ug/L; the lowest concentration tested (Colville et al. 2008). The addition of chlorpyrifos to the artificial streams resulted in a rapid (6-h) change in the macroinvertebrate assemblages of the streams, which persisted for at least 124 days after dosing (Colville et al. 2008). The chlorpyrifos dissipated from the system within 48 hours (Pablo et al. 2008), however the macroinvertebrate community did not recover rapidly. Several species similar to salmonid prey items were significantly affected. These data suggest that at concentrations of less than 1 ug/L adverse effects to modeled ecosystems occur. How

these data compare to actual ecologocal effects in salmonid habitats found in California, Idaho, Oregon, and Washington is unknown.

Diazinon
Van Wijngaarden et al. (2005) reported the results of one study conducted in a static system which showed an $LOEC_{eco}$ of 2.4 ug/L (Giddings et al. 1996). Multiple mesocosm and microcosm studies indicated adverse responses of tested organisms from a variety of test designs (review *in* Giddings et al. 2000). Zooplankton and insect taxa appeared the most sensitive from these studies and in particular salmonid prey taxa from trichoptera, diptera, and cladocera were highly sensitive, adverse effects in the low ug/L range diazinon.

Malathion
A registrant submitted mesocosm study evaluated the effects of a single application of a European malathion formulation to aquatic orangisms in 1 m^3 outdoor enclosures (Ebke 2004). The study concluded a NOEC of 5 ug/L based on reported transient impacts to Cladocerans in the Daphniidae and Chydoridae families. Data for emergent insects showed temporal decreases in organisms evaluated (Diptera, Insecta, and Chironomidae) but differences were not dose responsive. Statistical differences were rarely observed between treatments. It is difficult to draw conclusions from this study given the high degree of variability observed in control and malathion treatments both pre- and post-application.

The available literature from field experiments indicates that populations of insects and crustaceans are likely the first aquatic organisms damaged by exposures to chlorpyrifos, diazinon, and malathion contamination. For example, in listed steelhead habitat in the Salinas River, California, abundances of the salmonid prey items including mayfly taxa, daphnids and *Hyalella azteca* (an amphipod) were significantly reduced downstream of an irrigation return drain compared to upstream (Anderson et al. 2003a; Anderson et al. 2003b; Anderson 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 which were present at 0.925 ug/L, a value that is 10 times greater than the 10-d *H. azteca* LC50 for chlorpyrifos (Anderson et al. 2003b). Other pesticides were likely present and responsible for some of the toxicity in the Salinas River. In a subsequent study on the Salinas River, Toxicity Identification Evaluations (TIE) demonstrated that chlorpyrifos and diazinon were responsible for the observed death of *Cerodaphnia dubia* (a daphnid) (Hunt et al. 2003). These data support the line of evidence that field concentrations of OPs can adversely affect aquatic invertebrates in salmonid habitats.

Benthic community shifts from sensitive mayfly, stonefly and caddisfly taxa to worms and midges occur in areas with degraded water quality including from contaminants such as pesticides (Cuffney et al. 1997; Hall et al. 2006). Reduced salmonid prey availability correlated to OP use in salmonid bearing watersheds (Hall et al. 2006). Subsequent effects to salmonid's growth from reduced prey availability and quality remain untested and are a current data gap.

We located one highly relevant study that focused on fish growth following a single exposure of chlorpyrifos. 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 ug/L (chemical analysis of water concentrations provided at 0, 12, 24, 96, 384, 768 hrs) 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 day -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 enclusures 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 OP insecticides can result in reduced juvenile salmonid growth and ultimately reduced survival and productivity. The exact levels of prey reduction necessary to cause subsequent reductions in salmonid growth remain a recognized data gap.

Although the cause is unknown, recent declines in aquatic species in the Sacramento-San Joaquin River Delta in California have been attributed to toxic pollutants, including pesticides (Werner et al. 2000). Significant mortality or reproductive toxicity in *C. dubia* was detected in water samples collected at 24 sites in the Sacramento-San Joaquin River Delta in California. Ecologically important back sloughs had the largest percentage of toxic samples (14 - 19%). Toxicity Identification Evaluations (TIE) identified chlorpyrifos, diazinon, malathion, and two other cholinesterase-inhibiting insecticides (carbofuran and carbaryl) as the primary toxicants in these samples responsible for the adverse effects.

We did not locate any information evaluating changes in nutritional quality of salmonid prey items associated with pesticide-induced changes in prey abundance. This remains a current data gap.

Recovery of salmonid prey communities following acute and chronic exposures from chlorpyrifos, diazinon, and malathion depends on the organism's sensitivity, lifestage, length of lifecycle, 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 lifecycles 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 as well. 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 (Morley and Karr 2002; Paul and Meyer 2001). Similarly, yet due to a different set of circumstances, watersheds with intensive agriculture land uses show compromised invertebrate communities (Cuffney et al. 1997). Indices of biological integrity (IBI) and other invertebrate community metrics are useful measures of the health of an aquatic community 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).

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 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 (Cuffney 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.

Adjuvant toxicity

Assessment endpoints: Survival of fish and aquatic prey items, endocrine disruption in fish

Assessment measures: 24, 48, 96 h LC50s, vitellogenin levels in fish plasma

Although no data were provided in the BEs related to adjuvant toxicity, an abundance of toxicity information is available on the effects of the alkylphenol polyethoxylates, a family of non-ionic surfactants used extensively in combination with pesticides as dispersing agents, detergents, emulsifiers, adjuvants, and solubilizers (Xie et al. 2005). Two types of alkylphenol polyethoxylates, nonylphenol ethoxylates and octylphenol ethoxylates degrade in aquatic environments to the more persistent, toxic, and bioaccumulative degradates, octylphenol and nonylphenol, respectively. We discuss nonylphenol's toxicity as an example of potential adjuvant toxicity since we received no information on adjuvant use or toxicity within the BEs.

We queried EPA's ECOTOX online database and retrieved 707 records of NP's acute toxicity to freshwater and saltwater species. The lowest reported LC50 for a salmonid was 130 ug/L for Atlantic salmon. Aquatic invertebrates, particularly crustaceans, were killed at low concentrations of nonylphenol, lowest reported LC50 = 1 ug/L for *H. azteca*. These data indicate that a wide array of aquatic species is killed by NP at ug/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 ug/L for fathead minnow reproduction. Numerous fish studies reported LOECs at or below 10 ug/L. Additionally, salmonid prey species are also sensitive to sublethal effects of nonylphenol. The amphipod, *Corophium volutator*, grew less and had disrupted sexual differentiation (Brown et al. 1999). Multiple studies with fish indicated that nonylphenol disrupts fish endocrine systems by mimicking the female hormone 17B-estradiol (Arsenault et al. 2004; Brown and Fairchild 2003; Hutchinson et al. 2006; Jardine et al. 2005; Lerner et al. 2007a; Lerner et al. 2007b; Luo et al. 2005; Madsen et al. 2004; McCormick et al. 2005; Segner 2005). NP induced the production of vitellogenin in fish at concentrations ranging from 5-100 ug/L (Arukwe and Roe 2008; Hemmer et al. 2002; Ishibashi et al. 2006; Schoenfuss et al. 2008). Vitellogenin is an egg yolk protein produced by mature females in response to 17-β estradiol, however immature male fish contain the capacity to produce vitellogenin if exposed to estrogenic compounds. As such, vitellogenin is a robust biomarker of exposure. A retrospective analysis of an Atlantic salmon population crash suggested the crash was due to NP applied as an adjuvant in a series of pesticide applications in Canada (Brown and Fairchild 2003; Fairchild et al. 1999). Additionally, processes involved in sea water adaptation of salmonid smolts are impaired by NP (Jardine et al. 2005; Lerner et al. 2007a; Lerner et al. 2007b; Luo et al. 2005; Madsen et al. 2004; McCormick et al. 2005).

These results show that nonylphenol is of concern to aquatic life, particularly salmonid endocrine systems involved in reproduction and smoltification. We summarized data for one of the more than 4,000 inerts/other ingredients and adjuvants currently registered for use in pesticide formulations. Unfortunately we received minimal information on the constituents found in chlorpyrifos-, diazinon-, and malathion-containing formulations. Consequently, the effects that these ingredients may have on listed salmonids and designated critical habitat remain an uncertainty and are a recognized data gap of EPA's action under this consultation.

Summary of Response Analysis:

We summarize the available toxicity information by assessment endpoints in Table 59. Data and information reviewed for each assessment endpoint was assigned a generally 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. 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 active ingredients 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 active ingredients to support registration. We did locate a significant amount of data on one group of adjuvants/surfactants, the nonylphenol ethoxylates. However, we located minimal information for the majority of tank mixes and other ingredients within formulations.

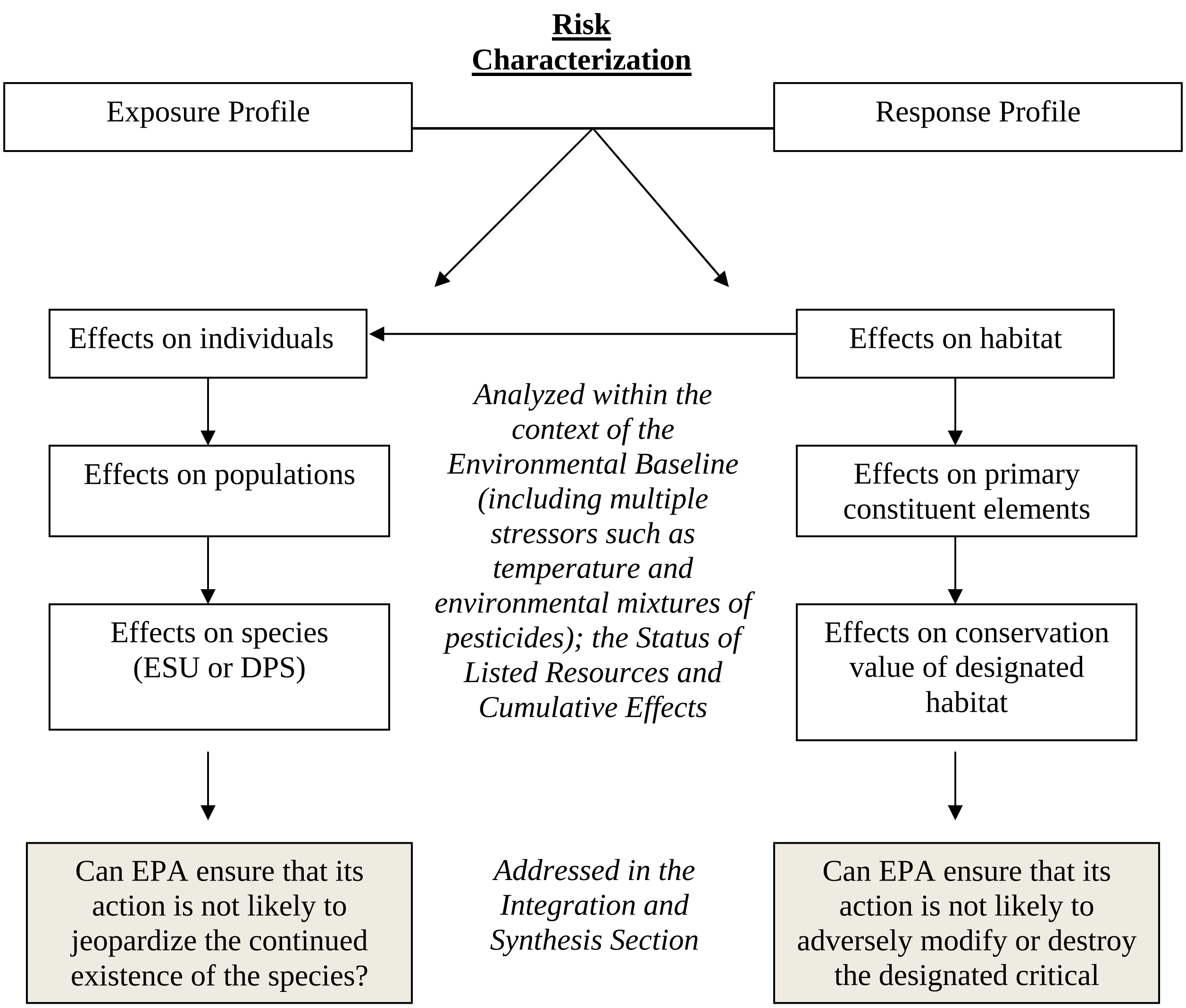
Table 59. Summary of assessment endpoints and effect concentrations

Assessment Endpoint	**Evidence of adverse responses**	**Concentration ranges of observed effect (ug/L)**	**Degree of confidence in effects (Low, Medium, High)**
Chlorpyrifos			
Fish:			
-survival (LC50)	Yes	0.8 - 2200	High
-growth	Yes	0.12 - 4.8	High
-reproduction	Yes	1.09 - 1.21	High
-swimming	Yes	0.3 - 40	High
-olfactory-mediated behaviors	Yes	0.625 - 2.5	High
Habitat:			
-prey survival (LC50)	Yes	0.05 - 600	High
Diazinon			
Fish:			
-survival (LC50)	Yes	90 - 7800	High
-growth	Yes	0.8	High
-reproduction	Yes	0.35 - 3.2	High
-swimming	Yes	500	High
-olfactory-mediated behaviors	Yes	0.1 – 1.0	Medium
Habitat:			
-prey survival (LC50)	Yes	0.03 - 2500	High
Malathion			
Fish:			
-salmonid survival (LC50)	Yes	2.8 - 234	High
-growth	Yes	NS	Low
-reproduction	Yes	NS	Low
-swimming	Yes	40 - 175	High
-olfactory-mediated behaviors	No	-	-
Habitat:			
-prey survival (LC50)	Yes	0.5 - 100	High
Other ingredients			
Nonylphenol			
Fish:			
-survival	Yes	130 - >1000	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- >1000	High
Additive toxicity of OPs	Yes	multiple	High
Synergistic toxicity OPs	Yes	multiple	High

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. 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. Further, we 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, 40, and 41, we show the overlap between exposure estimates for the three OPs 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 BE that represent crop uses; and NMFS' modeling estimates for off-channel habitats. None of the exposure estimates were derived for non-crop use. However, some of the monitoring data targeted mosquito and Medfly control programs. 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. However, we cannot accurately predict at what concentrations death first occurs because no slope information was presented in the information reviewed. We do however incorporate survival using a default slope in a population modeling exercise discussed below. This slope is recommended by EPA where more relevant information is unavailable (EPA 2004c). 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. However, it does allow us to systematically address which assessment endpoints are affected from chlorpyrifos, diazinon, and malathion exposure. Where significant uncertainty arises, NMFS highlights the information and discuss its influence on our inferences and conclusions. Several of the assessment endpoints we evaluated in the response analysis are not amenable to this type of comparison because we lack either exposure or response information. We discuss the uncertainties related to this information under each of the risk hypotheses.

Chlorpyrifos

The ranges of chlorpyrifos concentrations from the three sources of exposure information overlap the assessment endpoints presented in Figure 39. Therefore, we expect that chlorpyrifos will impair swimming and olfaction, and reduce reproduction and growth in listed salmonids when exposed for sufficient durations. Furthermore given the very low LC50 values for salmonids following 96 h exposures, we expect many immature salmonids will die, as well as some adults, if exposed to chlorpyrifos at concentrations greater than 1 ug/L. This does not account for the potential enhanced toxicity of chlorpyrifos to salmonids in aquatic habitats where elevated temperatures occur.

Abundance of salmonid prey items is expected to be significantly reduced, especially highly sensitive species, some with LC50s less than 0.1 ug/L. We discuss these effects in more detail under the risk hypotheses.

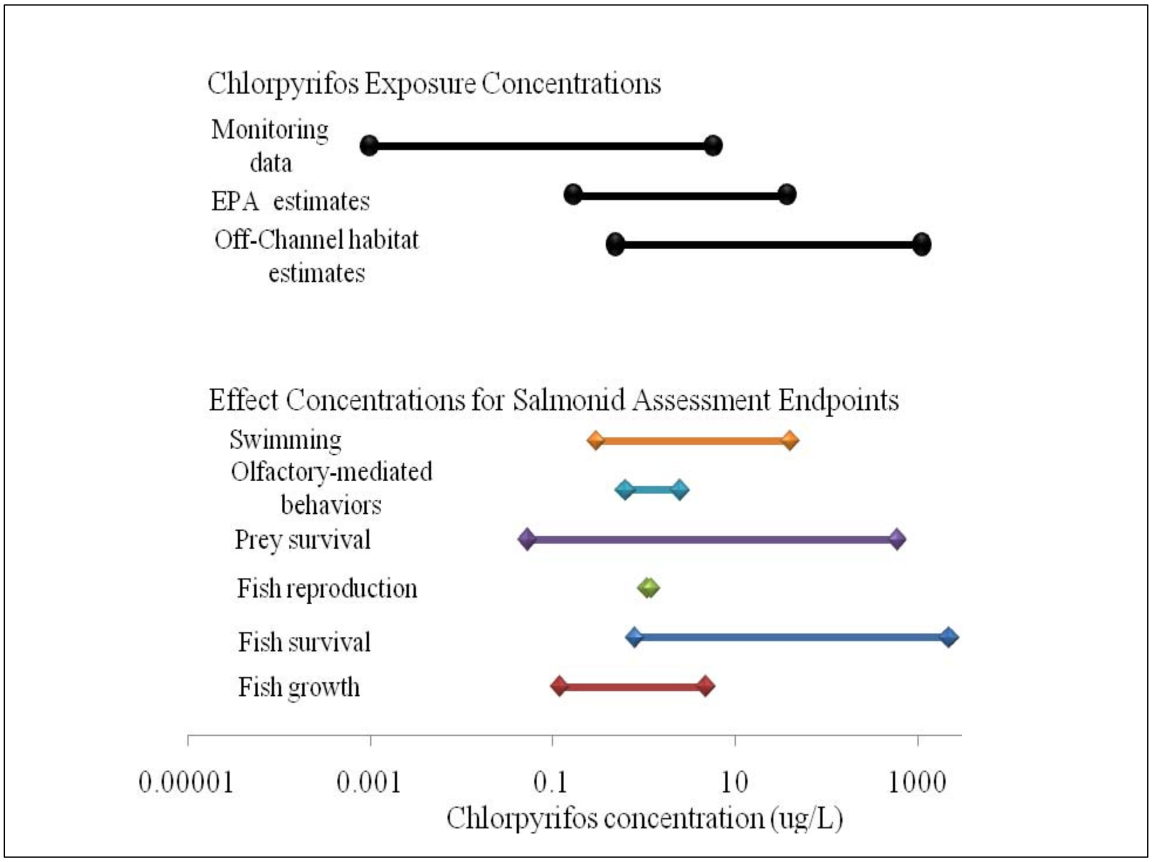


Figure 39. Chlorpyrifos exposure concentrations and salmonid assessment endpoints' effect concentrations in ug/L.

Diazinon

Concentration ranges overlap with the majority of the assessment endpoints indicating that adverse effects are expected in salmonids if exposed for a sufficient duration (Figure 40). Diazinon is less toxic than chlorpyrifos when comparing salmonid LC50s. However, salmonid prey appear just as sensitive to diazinon as to chlorpyrifos. Salmonid reproduction, olfactory-mediated behaviors, and growth effect concentrations are encompassed or exceeded by all three exposure ranges. Swimming was the least sensitive response reviewed, although there was little information available to fully assess this endpoint. Death of salmonids is predicted at the higher end of concentration ranges from the monitoring data and at the middle of the concentration range for EPA's crop estimates and NMFS' off-channel habitat estimates. As with chlorpyrifos, elevated temperatures are expected to enhance toxicity and lead to death and other effects at lower diazinon concentrations.

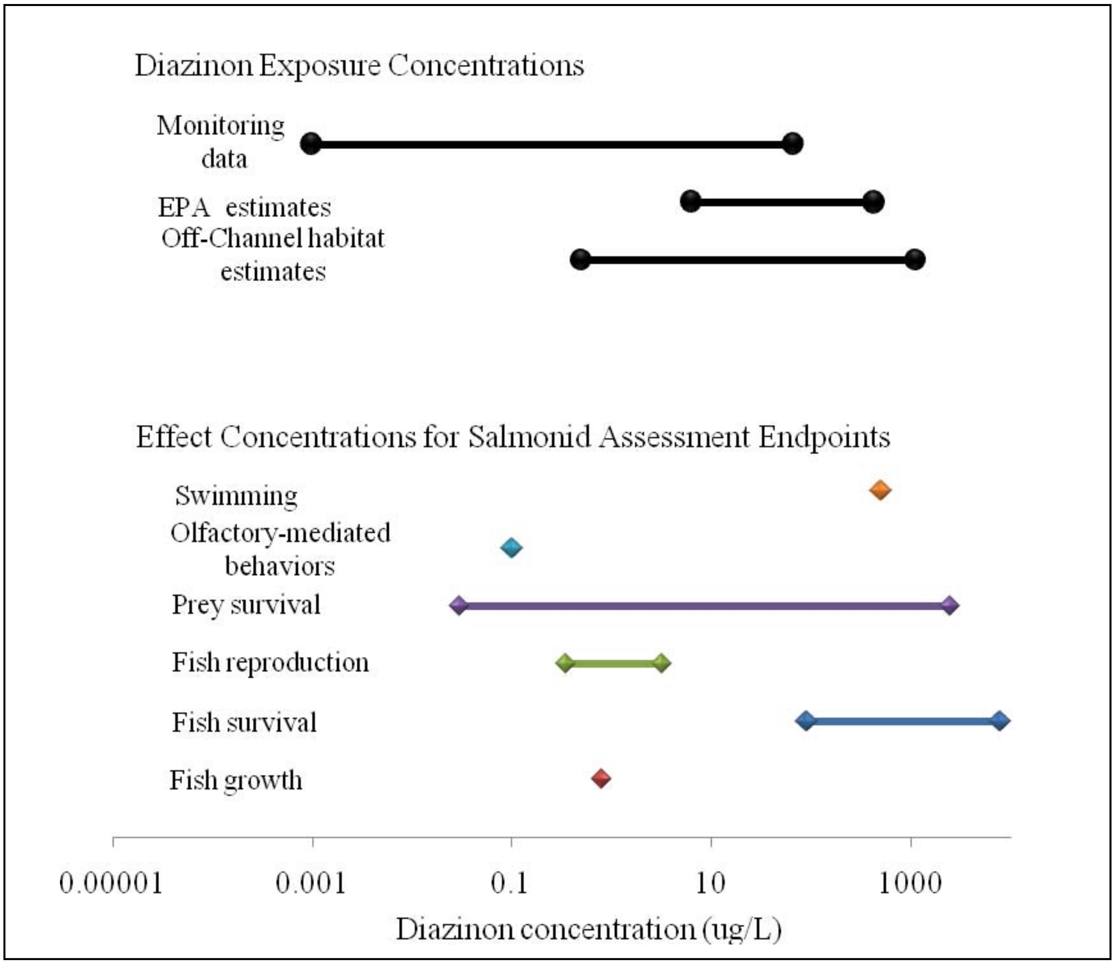


Figure 40. Diazinon exposure concentrations and salmonid assessment endpoints' effect concentrations in ug/L.

Malathion

Ranges of exposure concentrations for malathion are at or exceed the effect concentrations for the various assessment endpoints presented in Figure 41. Salmonid LC50s range from 2.8 – 234 ug/L which are likely achieved in some habitats given the modeling estimates, particularly for off-channel habitats as well as at the higher end of monitoring data. Salmonid prey items are very sensitive and at risk to malathion's toxicity as shown by the low effect concentrations and the exceedances in exposure estimates. At the lower end of both EPA's estimates and NMFS' off-channel habitat estimates, many salmonid prey items are likely killed and population abundance of prey reduced. If this occurs during the first feeding of fry following absorption of the yolksac, starvation is likely. The magnitude of reduction in prey abundance will depend on which taxa are present and the actual concentrations and exposure durations. We discuss this in greater detail in the risk hypotheses below. Swimming was the least sensitive endpoint according to the data. However, we only located two studies that measured swimming responses in three species of salmonid following acute exposures (24- 96 h). Growth and reproduction assessment endpoints were combined because effect concentrations were not differentiated in the BE between the two studies. Thus, the actual effect concentrations on reproduction and growth remain an uncertainty. Nevertheless, the evidence suggests that both are affected by the concentration ranges presented if exposure durations are achieved (97 d and 340 d). A notable data gap is the absence of information on malathion's toxicity to olfactory-mediated behaviors. Given the effects of the other two OPs, we expect that malathion can impair olfaction, but have no information on its potency.

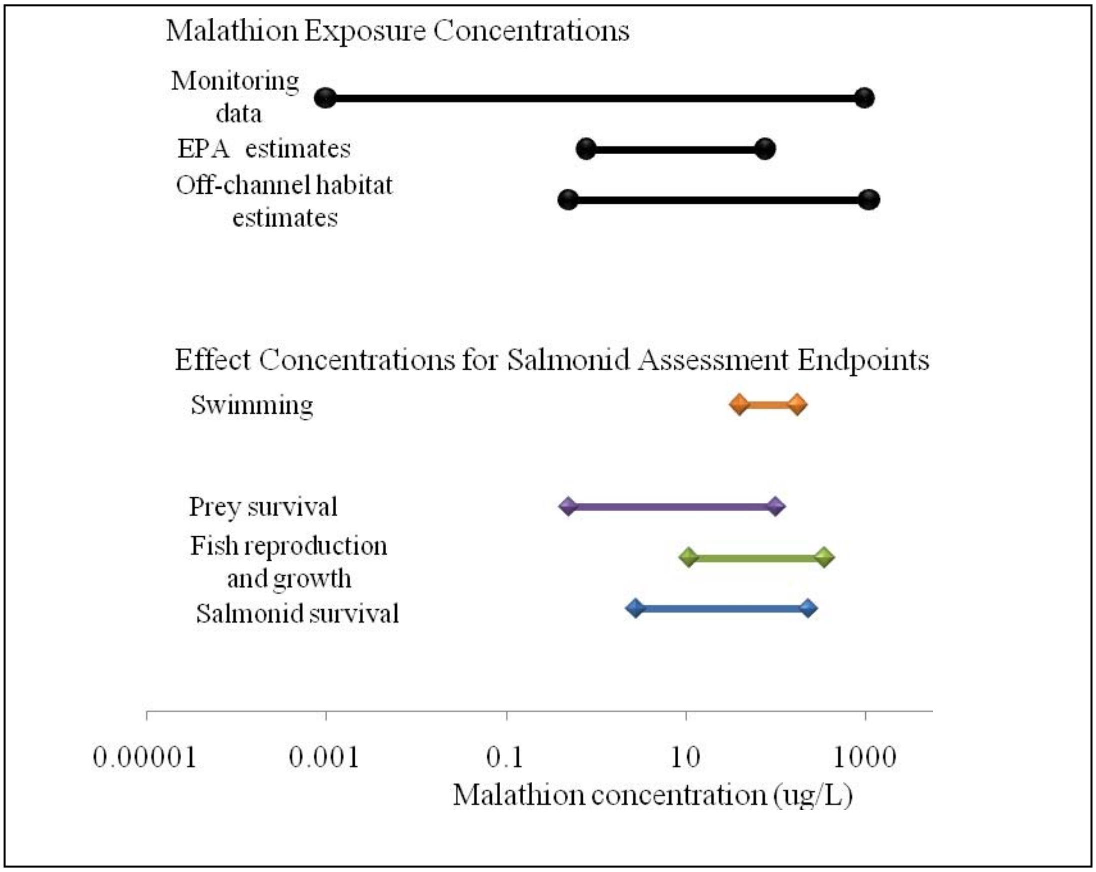


Figure 41. Malathion exposure concentrations and salmonid assessment endpoints' effect concentrations in ug/L.

Relationship of Pesticide use to Effects in the Field

Schulz reviewed 45 field and *in situ* studies published in peer-reviewed journals that evaluated relationships between insecticide contamination and biological effects in aquatic ecosystems (Schulz 2004). The relationship of exposure to effect was classified 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. It should be noted that these studies were not designed to establish effect thresholds and in our review is 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. Chlorpyrifos and malathion studies evaluated are summarized in Table 60. No studies involving diazinon were evaluated. One study resulted in a "likely relationship" between brain acetylcholinesterase in carp and a measured concentration of 0.12 ug chlorpyrifos/L in a pond in the Central Columbia Plateau (Pacific Northwest).

Schulz (2004) found that eight published studies since 1999 have shown a strong connection between agricultural insecticide contamination and adverse effects to abundance dynamics or community composition of macroinvertebrates. For example, three studies were characterized as having a "clear relationship" between chlorpyrifos and reduced invertebrate survival (1.3, 89.4, and 300-720 ug/L) or community composition (344 ug/L). Less information was available for malathion. One study examined the potential effects of malathion use on insect taxa in streams receiving discharge water from treated rice fields. Although there was an "assumed relationship" to reduced abundance of odanate species, a "clear relationship" was not established to the observed malathion concentrations (0.26-0.69 ug/L).

Another study summarized documented *in situ* mortality of amphipods (*Gammarus pulex)* associated with application of malathion to watercress although no exposure quantification was conducted. Schulz 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 measured and real exposure to be 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 real exposure or if a general difference between the field and laboratory reactions of aquatic invertebrates is responsible. The review by Schulz shows a large body of evidence that natural aquatic ecosystems can be adversely affected by OP insecticides including chlorpyrifos and malathion. We expect a similar relationship for diazinon.

Table 60. Published field and in situ studies designed to establish a relationship between the insecticide contamination of surface waters due to agricultural practices (adapted from Table 2 in Schulz 2004).

Source	Concentration ug/L	Duration	Endpoint	Species	Relationship of exposure and effect
chlorpyrifos					
Leaching (irrigation)	0.12	~ 24 hours	Brain cholinesterase	Carp (*Cyprinus carpio*)	Likely
Runoff, spray drift	344	1.3 hours	Community composition	Ephemeroptera, other insects	Clear
Runoff	1.3, 89.4	4 hours	Mortality (in situ bioassy)	dipteran (*Chironomus spp.)*	Clear
Runoff	300-720	Few hours	Mortality (in situ bioassy)	amphipod (*Paramelita nigroculus)*	Clear
malathion					
Application to rice	0.26-0.69	Few days	Abundance	Various odanate species	Assumed
Application to watercress	No	Few hours	Mortality (in situ bioassy)	Amphipod (*Gammarus pulex*)	Assumed

Field studies in ESA-listed salmonid habitats: Hood River Oregon

A group of field studies evaluated macroinvertebrate community responses in the orchard-dominated Hood River Basin, Oregon and correlated results with chlorpyrifos and azinphos-methyl use and detections (St. Aubin 2004; Van der Linde 2005; Grange 2002). 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 second 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 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 eight day period showed chlorpyrifos ranging from 0.032 -0.183 ug/L). There were more pollutant tolerant taxa and less intolerant taxa at the agricultural sites (Van der Linde 2005). Collector –gather species, many of which are salmonid prey items, declined rapidly at agricultural sites compared to abundances at the reference sites. 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). Hood River Basin contains several listed anadromous salmonids, including lower Columbia River steelhead. 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 indicated that OP-insecticides inhibited AChE activity in steelhead held in cages in the Hood River Basin which correlated to chlorpyrifos and azinphosmethyl detections and to a lessor 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 ug/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 steelehead. Study results show that, steelhead in these systems exposed to OP insecticides lose AChE activity (up to 33%) and, 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 orchard- dominated streams during chlorpyrifos and azinphosmethyl applications. Additionally, the macroinvertebrate communities in these systems were compromised to such an extent that salmonid prey abundance were reduced.

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. Below we summarize the most pertinent incident reports to EPA's proposed actions.

There have been several fish kill incidents associated with the reported use or detection of diazinon since 2002. All reports of fish kills associated with diazinon use in recent years occurred in California. A total of five fish kill incidents were reported since 2002. Of the incidents reviewed for diazinon, one is particularly relevant given its location. In June 2002, a fish kill involving over 2000 fish was reported to the CDFG. The dead fish were found in Monterey County in the Tembladera Slough and the Old Salinas River channel. These waters are within the South-Central California coast steelhead ESU and within its designated critical habitat. Monterey County Agricultural Commissioner staff indicated that a small number of applications of diazinon had been made in the general area when the fish kill occurred. Water samples collected from the sites detected diazinon in four of six samples with concentrations ranging from 0.095 – 0.183 ug/L. Gill samples from all five fish showed recent exposure to chlorpyrifos with concentrations ranging from 5 - 40 ug/kg. Methidathion, another OP, was also detected at low concentrations in the water but was absent in gill tissue. It was estimated that the kill occurred a few days prior to sampling. EPA classified this incident as "probable" that diazinon use caused the fish kill. NMFS agrees with EPA's conclusion. Although concentrations in the water column were well below median lethal concentrations for fish observed in the laboratory, it is likely that peak concentrations were not detected and that diazinon dissipation was likely significant in the few days between the occurrence of the fish kill and sampling.

EPA classified several fish kill incidents as "probable" results of malathion exposure. The majority of the fish kill incidents with malathion were associated with boll weevil control or mosquito control. Several of these incidents reported aquatic concentrations exceeding 100 ug/L. The more frequent occurrence of incidents associated with these applications may suggest greater risk than other approved uses. Alternatively, the frequency of these incidents may also reflect greater monitoring efforts associated with wide-area applications of malathion.

EPA classified several fish kill incidents as "probable" results of chlorpyrifos exposure. The majority of those incidents were the result of termiticide applications. EPA and applicants have indicated the use of chlorpyrifos as a termiticide has been completely phased out. Therefore, this use is expected to no longer be an issue. However, other incidents provide useful insight regarding the risk of other chlorpyrifos uses. One 2003 incident classified as "probable" involved the aerial application of a tank mixture of chlorpyrifos and cyfluthrin (a pyrethroid insecticide) to an agricultural field. Application of tank mixtures, and resulting environmental mixtures is a concern and is discussed throughout the Opinion. This fish kill involved hundreds of fish and occurred in Imperial County, California. Water samples at the kill site detected chlorpyrifos at 11.7 ug /L, one of the highest concentrations measured in California in recent years and well above other state monitoring data. Cyfluthrin concentrations at the kill site were 0.33 ug/L.

A second incident involved the community application of a chlorpyrifos product to control adult mosquitos in 2003. A resident reported a fish kill in a private home pond. Apparently no samples were collected to definitively determine the cause of the kill. EPA classified the likelihood that this fish kill was caused by chlorpyrifos as "possible." This incident is of interest because mosquito applications with chlorpyrifos may occur over many land use categories and substantial portions of many of the ESU's ranges.

Mixture Analysis of Chlorpyrifos, Diazinon, and Malathion

As noted earlier, pesticides most often occur in the aquatic environment as mixtures and chlorpyrifos, diazinon, and malathion are among the most common insecticides found in mixtures. EPA assesses human risk of mixtures containing these OPs assuming dose-addition because they share a common mechanism of action (EPA 2006). 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. The assumption of dose-addition for mixtures of anticholinesterase pesticides has also been extended to aquatic life (Belden et al. 2007). In salmon, dose-additive inhibition of brain AChE activity by mixtures of OPs and carbamates was demonstrated *in vitro* (Scholz et al. 2006). More recently, it has been found that salmonid responses to OP and carbamate mixtures vary *in vivo*; responses observed were either additive or synergistic (Laetz et al. *In Press*). NMFS used the dose-addition method to predict responses utilizing the modeling estimates and measured concentrations of chlorpyrifos, diazinon, and malathion presented in the *Exposure Analysis*. In Figure 42, we show an example of mixture toxicity based on additivity. The result of additivity for AChE inhibition (Figure 13 A.) and survival (Figure 13 B.) for the three OPs show an increased response from mixture toxicity compared to responses from each OP individually. Due to the very steep slopes of the two dose-response curves, and especially the mortality slope, small changes in concentrations elicit large changes in observed toxicity. This model utilized a probit slope of 4.5 for the mortality graph in Figure 42B. Empirical slopes derived from standard acute toxicity studies with aquatic organisms show a range of probit slopes that bracket 4.5 for each of the three insecticides[11]. Exposure values represent maximum concentrations of the three constituents detected in California surface waters based on the CDPR Database (CDPR 2008b, Table 61). We recognize that this approach is likely to under-predict toxicity for some mixtures, particularly those containing malathion that likely produce synergistic rather than additive responses (Laetz et al. *In Press*).

[11] http://www.ipmcenters.org/Ecotox/index.cfm

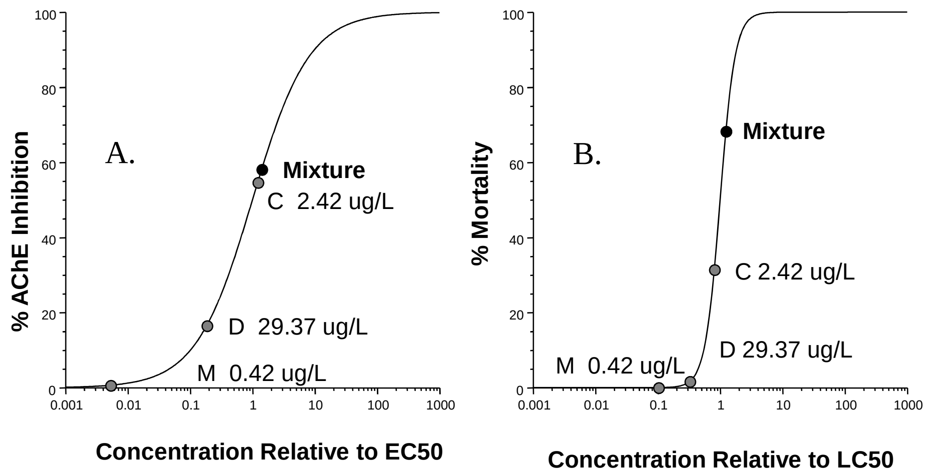


Figure 42. Percent AChE inhibition (A.) and percent mortality (B.) expected from exposure to chlorpyrifos (C), diazinon (D), and malathion (M) as separate constituents and as mixtures (C 2.42 ug/L, D 29.37 ug/L, and M 0.42 ug/L)[12].

We utilized a variety of exposure estimates and monitoring data to evaluate responses to different mixtures of chlorpyrifos, diazinon, and malathion (Table 61). The predicted additive responses from these mixtures ranged from 20-78% inhibition of AChE and 8-99% 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 common use rates (1 – 1.5 lbs a.i./acre) and numbers of applications (1-3). These application rates are on the lower end of allowable uses (up to 6 lbs a.i./acre or more are allowed for all active ingredients). Additionally, we used 60-day, time-weighted averages estimates 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.

Table 61. Predicted AChE inhibition and mortality from estimated and measured exposure to chlorpyrifos, diazinon, and malathion.

	Concentration (ug/L)	% AChE Inhibition	% Mortality
Modeling: PRZM-EXAMS 60-day averages[1] (from Table 40)			

[12] EPA's standard 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, In Press #386}.

Chlorpyrifos	0.84	30.41	0.97
Diazinon	6.40	4.39	0.01
Malathion	3.90	5.25	0.06
Additive response		**34.45**	**6.56**
Modeling: GENEEC 90-day averages (from Table 42**)**			
Chlorpyrifos	6.77	76.41	95.05
Diazinon	39.37	20.80	4.74
Malathion	5.51	5.51	0.18
Additive response		**77.84**	**97.89**
Monitoring: NAWQA maxima in 4 states (from Table 47**)**			
Chlorpyrifos	0.40	17.68	0.07
Diazinon	3.80	2.71	0.04
Malathion	1.35	1.96	0.00
Additive response		**20.42**	**0.41**
Monitoring: CDPR database maxima (from Table 48**)**			
Chlorpyrifos	2.42	54.68	31.43
Diazinon	29.37	16.54	1.69
Malathion	0.42	0.65	0.01
Additive response		**58.12**	**68.30**
Monitoring: Lower Salinas maxima (from Table 51**)**			
Chlorpyrifos	5.79	73.59	91.56
Diazinon	67.24	30.51	25.76
Additive response		**76.05**	**97.27**
Monitoring: Lower Salinas means (from Table 51**)**			
Chlorpyrifos	0.36	16.08	0.04
Diazinon	21.61	12.87	0.56
Additive response		**24.82**	**2.36**

[1]PRZM-EXAMS estimates for chlorpyrifos in Oregon Christmas trees (l lb a.i./acre), diazinon in California almonds (1.5 lb a.i./acre, 3 applications), and malathion in California alfalfa (1.24 lb a.i./acre, 2 applications).

The GENEEC estimates are 90-day, time-weighted averages that were based on labeled uses of the three compounds in a single crop, onions. We found no restrictions that would prevent co-application or sequential applications of chlorpyrifos, malathion, or diazinon. We assumed four lbs of diazinon were applied in-furrow, with two foliar applications of chlorpyrifos (1 lb/acre) and seven foliar applications of malathion (1.25 lbs/acre). These are common use rates found on several labels.

The NAWQA and CDPR monitoring values represent the maximum concentrations found in the respective databases. These databases included over 2,000 and 4,000 samples tested for the three insecticides in the CDPR (1990-2006) and NAWQA (1992-2006) datasets, respectively. Most of the detections in these and other monitoring studies occurred at or below the ppb level. We expect these concentrations to be representative of similar aquatic habitats where these OPs are used. We also expect that concentrations of the three OPs will be much higher in off-channel habitats compared to the maximum values reported in the CDPR and NAWQA databases.

Finally, we evaluated mixture exposure using maximum and mean monitoring values for chlorpyrifos and diazinon from sampling conducted in the Lower Salinas Valley, California. The maximum values were selected from a dataset of 177 samples collected over two years from nine sites. The mean values represent the average concentration detected at a single site during 2002 (N=5). We expect that comparable concentrations of chlorpyrifos, diazinon, and malathion occur in other watersheds where use of these compounds is similar.

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 chlorpyrifos, diazinon, and malathion 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. We expect concentrations levels of chlorpyrifos, diazinon, and malathion in salmonid habitats will reach lethal levels based on exposure concentrations derived from monitoring data, EPA's modeling estimates, and our own modeling estimates. The youngest, swimming salmonids appear to be the most likely to die from short-term, acutely toxic exposures. However, adults are also susceptible at higher concentrations. Although we found no information on egg survival following acute exposures, we do not expect death of eggs from these insecticides as insecticides entry into the eggs via the water column is unlikely. Further support for this hypothesis is found in field incidences of death attributed to chlorpyrifos, diazinon, and malathion that EPA ranked as "probable" or "highly probable". We located multiple cases in California showing death of fish. Other incident data are discussed in EPA's BEs and Science Chapters of recorded deaths of fish following applications (EPA 2000a; EPA 2000b; EPA 2000c; EPA 2002; EPA 2003; EPA 2004b).

We expect all swimming life stages of listed salmonids to be at risk of death, primarily in freshwater off-channel and edge habitats, and secondarily in marine and estuarine nearshore habitats. In conclusion, there is an abundance of evidence in support of this hypothesis. We therefore carry this endpoint into our population analysis and translate the reduced survival of individuals to potential population level consequences.

B. Reduce salmonid survival through impacts to growth.

Fish growth is reduced following long-term exposures to chlorpyrifos, diazinon, and malathion. Studies with fathead minnows and two salmonids, brook trout and rainbow trout, showed reduced growth following chronic exposure upwards of 274 d. The effect concentrations were as low as 0.12 for chlorpyrifos and 0.8 ug/L for diazinon and most were less than 5 ug/L. No information was available that assessed growth effects of malathion in fish. 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 diazinon and chlorpyrifos for acute durations causes reduced feeding success which likely results in impacts to growth (Sandahl et al. 2005; Scholz et al. 2000). Reduction in feeding is a consequence of impaired AChE resulting in reductions in normal swimming and impairment of olfactory mediated behaviors, both of which are discussed under separate hypotheses below. We expect that juvenile fish exposed to chlorpyrifos, diazinon, and malathion to both acute and chronic exposures during their freshwater residency would feed less successfully resulting in reduced size and growth rates. 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 low ug/L concentrations of OPs. 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 OPs. Therefore, we address the potential for population level repercussions due to reduced growth using a population model below.

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 OPs. These primarily involved evaluating laboratory experimental results that reported on incidences of death or sublethal effects. 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 real-world exposures to each of the insecticides and mixtures of the pesticides.

The second line of evidence is whether field level reductions in aquatic invertebrates correlate to OP insecticide use and/or concentrations in salmonids habitats. We found numerous reports on the condition of aquatic invertebrate communities in areas with OP use (urban and agricultural). Aquatic habitats that are routinely exposed to OP insecticides showed reduced abundances of salmonid prey (Cuffney et al. 1997; Hall et al. 2006; St. Aubin 2004; Van der Linde 2005). 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 et al. 2006). 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 eight day period showed chlorpyrifos ranging from 0.032 -0.183 ug/L) (Van der Linde 2005). There were more pollutant tolerant taxa and less intolerant taxa at the agricultural sites (Van der Linde 2005).

The third line of evidence we evaluated was whether salmonids showed reduced growth in areas of low prey availability, particularly those areas that coincide with use of chlorpyrifos, diazinon, and malathion. 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 that attributed 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 (Baxter et al. 2007; Brazner and Kline 1990; Metcalfe et al. 1999) or document fish growth rates below maximal potential growth rates when prey are limited (Dineen et al. 2007).

One study in particular, tested the hypothesis that single applications of chlorpyrifos (0.5, 5, 20 ug/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 days 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 enclusores and unexposed larvae grew significantly more compared to chlorpyrifos-treated enclosures. Furthermore, the reductions in prey items in diets mirrored the reduction in prey items in the enclosures. This further supports the hypothesis that reduction in prey abundances translates to reductions subsequent ration as well as individual growth. The authors concluded that "low levels of contaminants that induce slower growth in young-of-the-yearfish through food chain effects of 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 with this endpoint in the next section.

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 whether swimming behaviors are affected following exposures to chlorpyrifos, diazinon, and malathion that would occur in salmonid habitats. We discussed compelling evidence that the three OPs can impair salmonid swimming behaviors (discussed in the *Response Analysis*). Further, the concentrations that impair swimming overlapped with concentrations expected in salmonid habitats especially during occupation of off channel habitats. The three OPs generally had different toxic potencies to swimming behavior. However, these differences appear primarily attributed to the specific swimming behavior tested. The most sensitive swimming endpoints are those associated with swimming activity compared to those that measure swimming capacity (Little and Finger 1990; Little et al. 1990). Irrespective, there is robust information that showed reductions in swimming speed, distance swam, acceleration, as well as other swimming activities from the three OPs. The next line of evidence we evaluated is whether experimental evidence suggests that an individual's feeding, migration, reproduction, or survival is compromised due to impaired swimming behaviors. The ecological consequences to salmonids from aberrant swimming behaviors are implied primarily through the impairment of feeding, translating to reduced growth; migratory pattern; survival; and reproduction. These are more difficult assessment endpoints to measure in the laboratory and particularly in the field. However, laboratory evidence showed reductions in survival due to impaired swimming (Labenia et al. 2007). Cutthroat trout exposed to sublethal concentrations of the AChE-inhibiting carbamate, carbaryl, showed significantly reduced swimming abilities and were consumed at higher rates by a predator compared to unexposed fish. Impaired swimming behavior correlated with both AChE inhibition and increased depredation rates (Labenia et al. 2007). Statistically significant correlations were found between brain AChE activity and swimming behaviors indicating a putative relationship between AChE inhibition and swimming behaviors (Beauvais and Jones 2000; Kumar and Chapman 1998; Post and Leasure 1974; Sandahl et al. 2005). Although NMFS was unable to locate results from field 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.

E. Reduce olfactory-mediated behaviors resulting in consequences to survival, migration, and reproduction.

The first line of evidence we evaluated is whether olfaction is impaired by the three OPs. Definitive evidence supports that olfaction is impaired by concentrations we expect to occur in salmonid habitats for chlorpyrifos and diazinon. No studies were identified that measured the effects of malathion on olfaction or olfactory-mediated behaviors. However, given that diazinon and chlorpyrifos as well as other OPs and carbamates impair olfaction, we expect that malathion may also impair olfaction at concentrations summarized in the exposure analysis. The second line of evidence we address is whether salmonids that experience impaired olfaction show subsequent impacts to their survival, migration, and reproduction. We located two studies that together measured these individual level consequences and we discussed them in detail within the response analysis. Increased rates of predation are expected for salmonids exposed to the three insecticides. Direct evidence shows reduced alarm behavior in Chinook salmon following 2 h diazinon exposures (Scholz et al. 2000). In the field, juvenile salmonids could miss the alarm scents and have an increased probability of predation. The evidence also supports that adult migration (homing) is likely affected by low ug/L concentrations of diazinon following a 24 h exposure in salmonid habitats. Atlantic salmon showed reduced hormone levels in males following exposure to 0.3 - 45 ug/L diazinon, suggesting that males may not be able to detect a spawning female (Moore and Waring 1996). Evidence of impaired olfaction from other OPs and carbamates was also located. Collectively, the available evidence supports this hypothesis and we assess the potential for population-level consequences below.

2. Exposure to mixtures of chlorpyrifos, diazinon, and malathion 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. Evidence of additive and synergistic effects on survival and AChE inhibition in salmonids and their prey 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 an increased, reduced survival predicated on simple additively (mixture analysis section). 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 OP 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 well supported by the available information and we assess the potential for population level consequences below.

3. Exposure to other stressors of the action including oxon degradates, adjuvants, tank mixtures, and other active and other ingredients in pesticide products containing chlorpyrifos, diazinon, and malathion cause adverse effects to salmonids and their habitat.

We found evidence that strongly supports this hypothesis for some of the stressors of the action. Although there is a wealth of exposure and toxicity information available for the three OPs, much less information was available on other stressors of the action. Oxon degradates are more potent than that of the parent OPs. However, we found few experiments that tested the toxicity of oxons to aquatic species. The BEs provided minimal information on the relative potency of oxons compared to parent OPs. The one study result comparing the parent and degradate indicated that diazoxon was 20 times more toxic than diazinon based on death of killifish (Tsuda 1997). It is hypothesized that differences in species sensitivity to OPs is largely a result of the rates of biotransformation of the parent OPs to the oxon metabolites (Fuji and Asaka 1982). We infer that this would also be the case for salmonids and aquatic invertebrates and with the other oxons. By dividing the effect concentrations by 20 for the three OPs, we expect adverse effects to listed salmonids and their habitat in the ng/L range. In Table 17, we show that monitoring data from the spraying for Medflys detected maloxon (malathion's oxon) as high as 384 ug/L; a concentration that would kill much of the aquatic fauna based on acute toxicity values. The primary data gap regarding risk to the oxons is the concentrations in the environment and the actual concentrations that lead to adverse impacts to listed resources.

Several formulations of the three OPs contain other pesticides. Acute and chronic toxicity data for several of these ingredients are either more or equally toxic as the three OPs. For example, malathion is present in formulations that contain methoxychlor, resmethrin, captan, and carbaryl insecticides. We expect fitness consequences in salmonids and their prey following exposure to ng/L and low ug/l concentrations of these insecticides. We were provided no information on the occurrence of these other insecticides within the BEs; however, some of them have been detected in salmonid-supporting watersheds.

We did not receive a complete list of the currently registered formulations containing chlorpyrifos, diazinon, and malathion. 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 pose a significant risk to salmonids and their habitat. We selected NP ethoxylates and nonylphenol 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 et al. 2004; Brown and Fairchild 2003; Hutchinson et al. 2006; Jardine et al. 2005; Lerner et al. 2007a; Lerner et al. 2007b; Luo et al. 2005; Madsen et al. 2004; McCormick et al. 2005; Segner 2005). Importantly, we found studies that linked Atlantic salmon population crashes in Canada to use of nonylphenol 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 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.

4. Exposure to other pesticides present in the action area can act in combination with chlorpyrifos, diazinon, and malathion to increase effects to salmonids and their habitat.

The available toxicity and exposure data support the hypothesis. Other OPs and carbamates found in the action area likely result in additive or synergistic effects to exposed salmonids and aquatic invertebrates. The magnitude of effects will depend on the duration and concentrations of exposed fauna. We therefore frame our conclusions in the context of the likelihood of other AChE-inhibiting insecticides within aquatic habitats. More than 50 OPs are currently registered and an unknown number of carbamates are registered. The triazine, atrazine, potentiates the effect of OPs within invertebrates. This does not seem to be the case with fish. However, atrazine is one of the most commonly detected pesticides in U.S. waters and frequently is detected in water samples containing OPs including chlorpyrifos, diazinon, and malathion. We expect that where atrazine co-occurs with the three OPs at concentrations of 100 ug/L, aquatic invertebrates will die at lower concentrations compared to single OP exposure. This level of atrazine is fairly high, although targeted sampling has shown higher concentrations in aquatic habitats. We therefore caveat our conclusions with the assumption that if atrazine and possibly other triazines co-occur with one of the three insecticides then we expect enhanced toxicity to invertebrates.

5. Exposure to elevated temperatures can enhance the toxicity of the stressors of the action.

We found a substantive dataset that supports this hypothesis for several cold water fish species including salmonids. As the water temperature increases, salmonid LC50s decrease – that is more fish died at elevated temperatures. We expect elevated temperatures across the freshwater habitat of listed salmonids to co-occur with chlorpyrifos, diazinon, and malathion concentrations. Many salmonid populations reside in watersheds which have been listed by the four western states as impaired due to temperature exceedances. We expect that salmonids exposed to both elevated temperatures and the three insecticides in the environment will die at relatively lower concentrations compared to salmonids exposed to the three insecticides at non-elevated temperatures in laboratory assays. We therefore discuss qualitatively temperature 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 individuals based on 4 d acute lethality to four generalized populations of salmonids. 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 OPs 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. We also discuss in general terms the likelihood of exposure to the range of pesticide concentrations that occur in salmonid habitats.

We also 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 that affect growth 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 and expected environmental concentrations. Following our analysis of the model results, we discuss the population-level responses to other effects 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. We also discuss population-level effects in the context of elevated temperatures and other OPs, and carbamates present in the environmental baseline of the action area.

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 growth effects over more than 140 days 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 juveniles

An acute toxicity model was constructed that estimated the population-level impacts of juvenile mortality resulting from exposure to lethal concentrations of chlorpyrifos, diazinon, and malathion. These models excluded sublethal and indirect effects of the pesticide exposures and focused on the population-level outcomes resulting from an annual 4 day exposure of juveniles to chlorpyrifos, diazinon, or malathion. We did not address mixture toxicity in the model. Death of juveniles was implemented as a change in first year survival rate for each of the salmon life history strategies modeled. We display the model output in Tables 52-55 below.

The percent changes in lambdas increased as concentrations of the three OPs 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 LC50 from the available information to ensure that risk is not underestimated. However, if the actual environmental acute LC50 is lower, then the model will underpredict mortality. If the actual environmental acute LC50 is higher, then the model will over-predict mortality.

These results indicate that salmonid populations exposed to chlorpyrifos, diazinon, or malathion for 4 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 abundance would decline 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. When we compare the concentrations listed below to expected levels in salmonid habitats described in the exposure section, it is highly likely that some portions of, or all of the individuals within a population will be exposed at sometime in their juvenile lifestage. This is even more likely for those individuals that spend longer periods in freshwaters such as steelhead and coho salmon. For those populations with lambdas greater than one, reductions in lambda from death of juveniles can also lead to consequences to abundance and productivity. Attainment of recovery and time-associated goals would likely not be met for populations with reduced lambdas. Many of the populations that are categorized as core populations 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 increased time to meet recovery goals. We discuss in more detail the effects to populations in the *Integration and Synthesis* section.

Table 62. Modeled output for Ocean-type Chinook salmon exposed to 4 d exposures of chlorpyrifos, diazinon, or malathion reporting the impacted factors of survival as percent dead, lambda and standard deviation, and percent change in lambda compared to an unexposed population. 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)

Chlorpyrifos	0 ug/L	1.0 ug/L	1.8 ug/L	3.0 ug/L	3.8 ug/L	5.0 ug/L	10.0 ug/L	100.0 ug/L
% dead	0	1.8	13	50	70	86	98.7	99.9
Lambda (STD)	1.09 (0.1)	1.08 (0.1)	1.04 (0.1)	0.89 (0.08)	0.77 (0.7)	0.62 (0.05)	0.33 (0.03)	0.05 (0.004)
% change in lambda	**NA**	**NS**	**NS (-4)**	**-18**	**-29**	**-43**	**-69**	**-95**

Diazinon	0 ug/L	10.0 ug/L	50 ug/L	75 ug/L	90 ug/L	125 ug/L	200 ug/L	400 ug/L
% dead	0	0.03	11	34	50	76	95	99
Lambda (STD)	1.09 (0.10)	1.09 (0.10)	1.05 (0.10)	0.97 (0.09)	0.89 (0.08)	0.72 (0.06)	0.48 (0.04)	0.26 (0.02)
% change in lambda	**NA**	**NS**	**NS (-3)**	**-12**	**-18**	**-34**	**-56**	**-76**

Malathion	0 ug/L	1.0 ug/L	10.0 ug/L	25.0 ug/L	30.0 ug/L	50 ug/L	75 ug/L	100 ug/L
% dead	0	0.00	1.8	34	50	86	96	99
Lambda (STD)	1.09 (0.10)	1.09 (0.10)	1.08 (0.10)	0.97 (0.08)	0.89 (0.08)	0.62 (0.06)	0.43 (0.04)	0.33 (0.01)
% change in lambda	**NA**	**NS**	**NS**	**-12**	**-18**	**-43**	**-60**	**-69**

Table 63. Modeled output for Stream-type Chinook salmon exposed to 4 d exposures of chlorpyrifos, diazinon, or malathion reporting the impacted factors of survival as percent dead, lambda and standard deviation, and percent change in lambda compared to an unexposed population. 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)

Chlorpyrifos	0 ug/L	1.0 ug/L	1.8 ug/L	3.0 ug/L	3.8 ug/L	5.0 ug/L	10.0 ug/L	100.0 ug/L
% dead	0	1.8	13	50	70	86	98.7	99.9
Lambda (STD)	1.00 (0.03)	0.99 (0.03)	0.96 (0.03)	0.84 (0.02)	0.74 (0.02)	0.61 (0.02)	0.34 (0.01)	0.05 (0.001)
% change in lambda	**NA**	**NS**	**-4**	**-16**	**-26**	**-39**	**-66**	**-95**

Diazinon	0 ug/L	10.0 ug/L	50 ug/L	75 ug/L	90 ug/L	125 ug/L	200 ug/L	400 ug/L
% dead	0	0.03	11	34	50	76	95	99
Lambda (STD)	1.0 (0.03)	1.0 (0.03)	0.97 (0.03)	0.90 0.03)	0.84 (0.03)	0.70 (0.02)	0.48 (0.01)	0.27 (0.01)
% change in lambda	**NA**	**NS**	**-3**	**-10**	**-16**	**-30**	**-51**	**-73**

Malathion	0 ug/L	1.0 ug/L	10.0 ug/L	25 ug/L	30 ug/L	50 ug/L	75 ug/L	100 ug/L
% dead	0	0.00	1.8	34	50	86	96	99
Lambda (STD)	1.00 (0.03)	1.00 (0.03)	0.99 (0.03)	0.9 (0.03)	0.84 (0.03)	0.61 (0.02)	0.44 (0.01)	0.34 (0.03)
% change in lambda	**NA**	**NS**	**NS**	**-10**	**-16**	**-39**	**-56**	**-66**

Table 64. Modeled output for Coho salmon exposed to 4 d exposures of chlorpyrifos, diazinon, or malathion reporting the impacted factors of survival as percent dead, lambda and standard deviation, and percent change in lambda compared to an unexposed population. 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)

Chlorpyrifos	0 ug/L	1.0 ug/L	1.8 ug/L	3.0 ug/L	3.8 ug/L	5.0 ug/L	10.0 ug/L	100.0 ug/L
% dead	0	1.8	13	50	70	86	98.7	99.9
Lambda (STD)	1.03 (0.05)	1.02 (0.05)	0.98 (0.05)	0.82 (0.04)	0.69 (0.04)	0.53 (0.03)	0.24 (0.01)	0.015 (0.001)
% change in lambda	NA	NS	-5	-20	-33	-48	-77	-98

Diazinon	0 ug/L	10.0 ug/L	50 ug/L	75 ug/L	90 ug/L	125 ug/L	200 ug/L	400 ug/L
% dead	0	0.03	11	34	50	76	95	99
Lambda (STD)	1.03 (0.05)	1.03 (0.06)	0.99 (0.05)	0.89 (0.05)	0.82 (0.04)	0.63 (0.03)	0.38 (0.02)	0.16 (0.01)
% change in lambda	NA	NS	NS (-4)	-13	-20	-38	-63	-84

Malathion	0 ug/L	1.0 ug/L	10.0 ug/L	25.0 ug/L	30 ug/L	50 ug/L	75 ug/L	100 ug/L
% dead	0	0.00	1.8	34	50	86	96	99
Lambda (STD)	1.03 (0.05)	1.03 (0.05)	1.02 (0.05)	0.89 (0.05)	0.82 (0.04)	0.53 (0.03)	0.34 (0.02)	0.24 (0.01)
% change in lambda	NA	NS	NS	-13	-20	-48	-67	-76

Table 65. Modeled output for Sockeye salmon exposed to 4 d exposures of chlorpyrifos, diazinon, or malathion reporting the impacted factors of survival as percent dead, lambda and standard deviation, and percent change in lambda compared to an unexposed population. 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)

Chlorpyrifos	**0 ug/L**	**1.0 ug/L**	**1.8 ug/L**	**3.0 ug/L**	**3.8 ug/L**	**5.0 ug/L**	**10.0 ug/L**	**100.0 ug/L**
% dead	0	1.8	13	50	70	86	98.7	99.9
Lambda (STD)	1.01 (0.06)	1.0 (0.06)	98 (0.05)	0.85 (0.05)	0.76 (0.04)	0.63 (0.03)	0.36 (0.02)	0.06 (0.003)
% change in lambda	**NA**	**NS**	**NS (-3)**	**-15**	**-25**	**-38**	**-64**	**-94**

Diazinon	**0 ug/L**	**10.0 ug/L**	**50 ug/L**	**75 ug/L**	**90 ug/L**	**125 ug/L**	**200 ug/L**	**400 ug/L**
% dead	0	0.03	11	34	50	76	95	99
Lambda (STD)	1.01 (0.06)	1.01 (0.06)	98 (0.05)	0.91 (0.05)	0.86 (0.05)	0.72 (0.04)	0.05 (0.03)	0.29 (0.01)
% change in lambda	**NA**	**NS**	**NS (-3)**	**-9**	**-15**	**-29**	**-50**	**-72**

Malathion	**0 ug/L**	**1.0 ug/L**	**10.0 ug/L**	**25.0 ug/L**	**30 ug/L**	**50 ug/L**	**75 ug/L**	**100 ug/L**
% dead	0	0.00	1.8	34	50	86	96	99
Lambda (STD)	1.01 (0.06)	1.01 (0.06)	1 (0.06)	0.92 (0.05)	0.86 (0.05)	0.63 (0.03)	0.46 (0.02)	0.36 (0.02)
% change in lambda	**NA**	**NS**	**NS**	**-9**	**-15**	**-38**	**-54**	**-64**

Effects to salmonid populations from reduced size of juveniles

To assess the potential for adverse effects to juvenile growth resulting from the anticholinesterase insecticides chlorpyrifos, diazinon, and malathion on Pacific salmon 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. We integrated two avenues of effect to juvenile salmonids' growth from exposure to the three OPs. The first avenue is based on the impacts of direct AChE inhibition on feeding success and subsequent juvenile growth. Salmon are often found to be food limited, suggesting that a reduction in prey number or size due to insecticide exposure may further stress salmon. Because anticholinesterase insecticides can reduce benthic densities of aquatic invertebrates and alter the composition of aquatic communities, we also incorporated growth reductions in juveniles due to reductions in available prey as the second avenue.

Growth Model Results

Organismal and population model outputs for all scenarios are summarized in the four figures below and in Tables 5-16 in Appendix 1. As expected, greater reductions in population growth resulted from longer exposures to the insecticides. The factors driving the magnitude of change in lambda were the relative AChE Activity and Prey Abundance parameters determined by the toxicity values for each pesticide (Table 3; Appendix 1). Both factors were equally contributing to the impacts for chlorpyrifos which have similar AChE IC_{50} and Prey Abundance EC_{50} values (Tables 3, 5-8; Appendix 1). The low Prey Abundance EC_{50} values drive the effects for diazinon and malathion models which have much higher AChE IC_{50} values (Tables 3, 9-16; Appendix 1). While strong trends in effects were seen for each pesticide across all four life-history strategies modeled, some slight differences were apparent. 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 produced the next most extreme response; coho salmon output (

Figure 45) 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.

These results show that all four general populations can be severely affected by changes in juvenile growth resulting from AChE inhibition and reduced prey availability. The concentrations that elicit reductions in lambdas are expected to occur in salmonid habitats. The degree to which an actual threatened or endangered population is affected will depend on a host of factors including the number of individuals exposed, the duration of exposure, when they are exposed, and if individuals are exposed more than once. It is also important to realize that these are idealized populations. NMFS did not incorporate other factors that can affect the sensitivity of exposed salmonids such as elevated temperatures, presence of mixtures of OPs and carbamates, and the condition of the fish. We also did not incorporate incidences of death due to acute toxicity in the growth model. We show however, that even without these other stressors taken into account, there is strong evidence that the expected concentrations in salmonid habitats will adversely affect populations if juvenile life stages are exposed. The longer the exposure duration to effect concentrations and the greater number of individuals exposed, the greater the adverse population-level effect.

Figure 43. Percent change in lambda for Ocean-type Chinook salmon following 4 d, 21 d, and 60 d exposures to chlorpyrifos, diazinon, and malathion. 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.

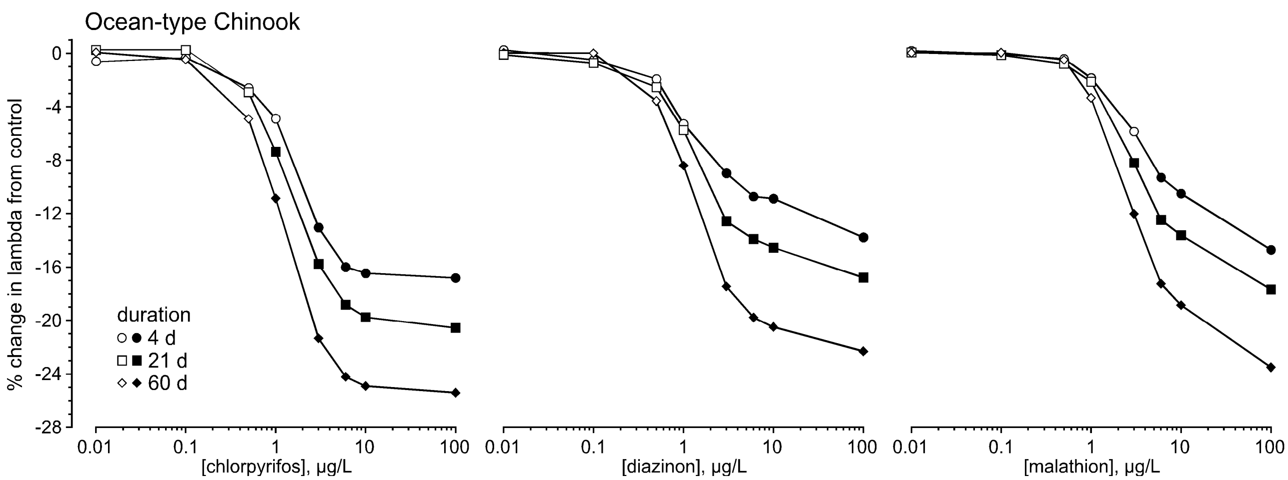


Figure 44. Percent change in lambda for Stream-type Chinook salmon following 4 d, 21 d, and 60 d exposures to chlorpyrifos, diazinon, and malathion. Open symbols denote a percent change in lambda of less than one standard deviation from control population. Closed symbols represent a percent change in lambda of more than one standard deviation from control population.

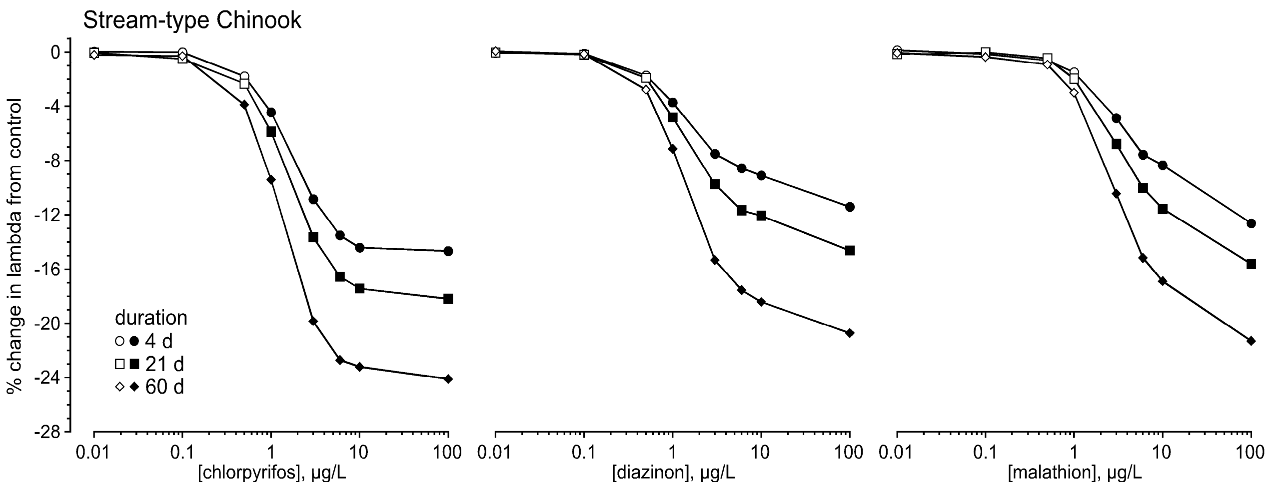


Figure 45. Percent change in lambda for Coho salmon following 4 d, 21 d, and 60 d exposures to chlorpyrifos, diazinon, and malathion. Open symbols denote a percent change in lambda of less than one standard deviation from control population. Closed symbols represent a percent change in lambda of more than one standard deviation from control population.

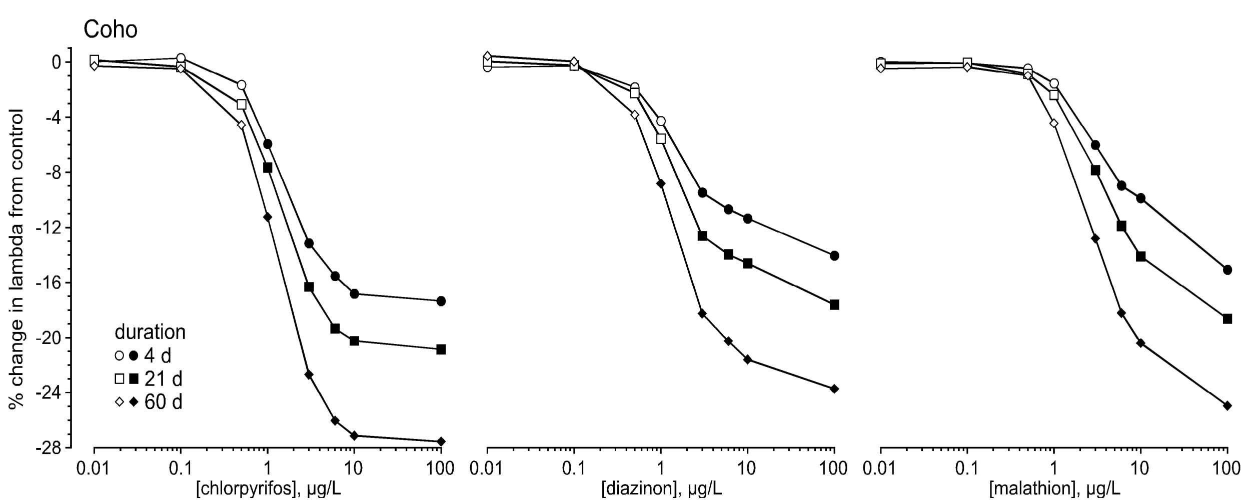
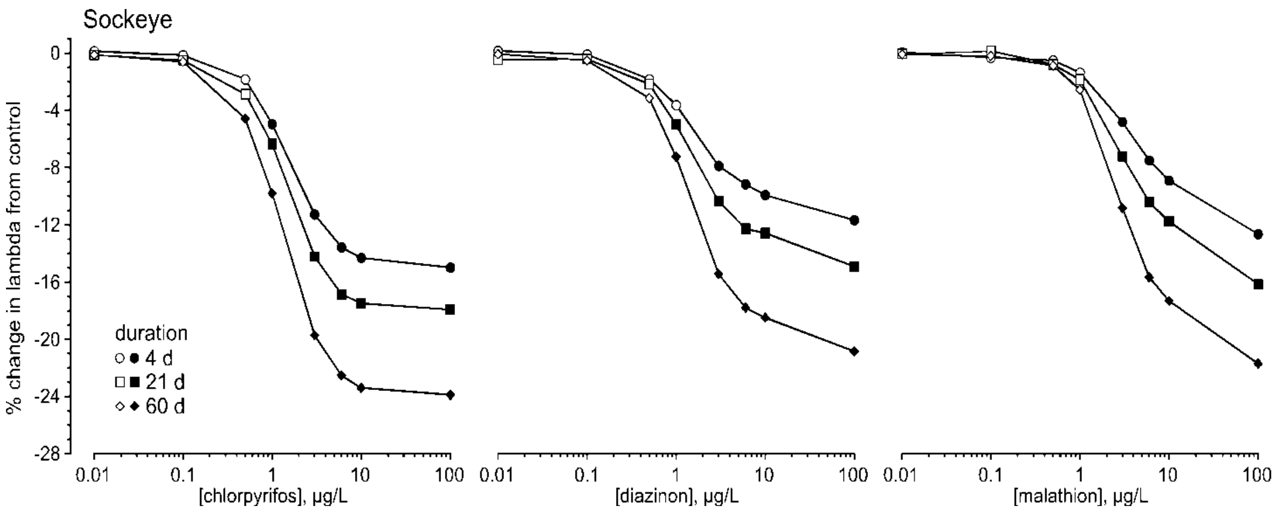


Figure 46. Percent change in lambda for Sockeye salmon following 4 d, 21 d, and 60 d exposures to chlorpyrifos, diazinon, and malathion. Open symbols denote a percent change in lambda of less than one standard deviation from control population. Closed symbols represent a percent change in lambda of more 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 expected. For example, if the Puget Sound Chinook salmon Green River population with a lambda of 0.67 is exposed to chlorpyrifos at 3.0 ug/L for 96 h, an environmentally relevant concentration and certainly not the highest concentration expected, we would expect a reduction in lambda by 0.18 (Table 53) or 0.16 (Table 54) depending whether the individuals exhibit ocean type or stream type life histories. The resulting lambda would be either 0.49 or 0.51 based on acute mortality of juveniles. Taking this example one step further we can also infer what the population's response would be from reductions in juvenile growth. Recall reductions in juvenile growth result from direct effects to juveniles and reduced prey availability. With the same concentration (3 ug/L chlorpyrifos) and Green River population, we would see reduction in lambda from the current 0.67 to either 0.54 or 0.56; both result in steep reductions in viability. Even for those lambdas that are well above one such as Central Valley Chinook salmon Spring Runs' Butte Creek population (lambda = 1.3), reductions of 10- 20% would have major consequences to a population's viability from death of juveniles and reduced growth of juveniles. The repercussions to these populations' viabilities are increased with increasing concentrations, durations, and when mixtures are incorporated.

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 for 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 of the proposed action. 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 affected over multiple generations. If the adverse effect(s) results in reducing a population's survival or recovery potential, than we look at 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 the spatial and temporal uses of the registered formulations containing chlorpyrifos, diazinon, and malathion which is 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 and to swim to successfully navigate through a variety of habitats over their life span and to ultimately spawn successfully in natal waters- thus completing their lifecycle. We have shown that exposure concentrations coupled with effect concentrations are sufficient to affect salmonids. 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.

A suite of ecologically relevant behaviors are likely affected when an individual's olfaction is impaired. Lack of predator avoidance behaviors by juvenile and adult salmonids likely reduces the probability of surviving predation events. Juvenile salmonids with impaired olfaction likely 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 OPs may still 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, thus spawning synchronization could be compromised and recently laid eggs may go unfertilized; resulting in reduced productivity and abundance for a population. 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 and difficulty in conducting field experiments with adult salmonids. Because imprinting, avoiding predators, homing, and spawning are likely affected when exposed to the stressors of the action, we conclude these additional effects cannot be dismissed. Therefore, we expect exposed populations to show reduced reproductive rates, reduced return rates, and reduced intrinsic rates of growth when sufficient numbers of individuals are affected. We further conclude that exposed populations are likely to have reduced abundance and productivity as a result of impaired swimming and olfactory-mediated behaviors.

Starvation during a critical life stage transition

Salmonids emerge from redds (nests) with a yolk-sac, hence they are referred to as yolk-sac fry. Following the complete utilization of the yolk sac, fry must feed frequently to properly develop and grow. If fry are unable to properly swim or detect and capture prey the onset of starvation occurs rapidly. Fry will likely be consumed by predators before they starve to death. The stressors of the action likely affect this critical life-stage transition in several ways leading to increased early lifestage mortality. Impaired swimming and olfaction affects their 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. The complete loss of juvenile recruitment from contaminated stream or river reaches is possible for spawning grounds that occur in areas susceptible to pesticide drift and runoff. These same areas also have off-channel habitats where fry seek shelter and food; however these 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 life histories 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 that is 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 lifecycle. For populations with lambdas well below 1, every adult is crucial to a population's viability. We expect that some of the populations will lose adults before they spawn due to acute lethality of these pesticides. We can not specify the number adults lost to a given population in a given year. However, it is reasonably certain to occur given the expected exposures. Additionally, for those areas with elevated temperatures, we expect an even greater number of returning adults to die before spawning due to temperature's enhanced effect on pesticide-induced lethality in salmonids. Many stream and river miles throughout salmonids ranges are impaired by elevated temperatures. For those populations affected, we expect both productivity and abundances to decline.

Additive toxicity

As discussed in this Opinion, we expect surface waters that contain chlorpyrifos, diazinon, and malathion to affect individuals and prey by additive toxicity as a result of the cumulative impairment of AChE activity and all AChE-associated physiological functions. We expect that changes in lambdas will be more severe due to additive toxicity. Additionally, we also expect to see additive toxicity in the form of AChE inhibition in salmonids and their prey in surface waters containing other OPs and carbamates. Monitoring data confirm that other OPs and carbamates are common in salmonid habitats. Therefore reductions in both abundance and productivity are likely in populations exposed to mixtures containing the three OPs, other OPs, and co-occurring carbamates.

Synergistic toxicity

With certain combinations and specific concentrations of chlorpyrifos, diazinon, and malathion synergism occurs, translating into increased rates of mortality among exposed salmonids. We have no predictive models for this phenomenon. However, where we expect co-occurrence of the three insecticides, we would expect synergism if specific levels are attained. In these areas, even more fish would die from synergism than deaths predicted from additive toxicity. Therefore, population-level effects could be more pronounced as well, 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.

Toxicity from other stressors of the action-

We identified inert ingredients, adjuvants (nonylphenol), tank mixtures (recommended on pesticide product labels), oxons (degradates of chlorpyrifos, diazinon, and malathion), and other pesticide active ingredients (permethrin, methoxychlor, resmethrin, carbaryl, and others) as toxic to salmonids and their prey. There remain substantial data gaps on the concentrations expected of many 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., malaoxon, nonylphenol, carbaryl. The risk posed by these other stressors to salmonid populations is complicated by the same factors we discussed for chlorpyrifos, diazinon, and malathion (i.e., the numbers of individuals exposed, the uncertainty surrounding the temporal and spatial uses of these chemicals, etc.). That said population crashes of Atlantic salmon in Canada were attributed specifically to the use of nonylphenol within a pesticide formulation (Brown and Fairchild 2003; Fairchild et al. 1999). We conclude that given the use and co-application of these chemicals with chlorpyrifos, diazinon, and malathion, that exposed individuals are at increased risk of the suite of toxic effects expected from a particular substance. We also recognize that substantial uncertainty exists on the identity of other ingredients found in applied formulations which further complicates our ability to predict toxicity to salmonids and their prey. 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 all populations of threatened and endangered salmonids covered by this consultation will likely show reductions in viability. The extent or magnitude of these reductions will vary temporally and spatially. However, we expect that all populations 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, Columbia River chum salmon, Hood Canal summer run chum salmon, Central California Coast coho salmon, Southern Oregon and Northern Coastal California coho salmon, Oregon Coast coho salmon, Ozette Lake sockeye salmon, Snake River sockeye salmon, Central California Coast steelhead, California Central Valley steelhead, LCR steelhead, MCR steelhead, Northern California steelhead, Snake River Basin steelhead, South-

Central California coast steelhead, Southern California steelhead, UCR steelhead, and Upper Willamette River steelhead will show reduced viability due to the proposed action.

Effects to Designated Critical Habitat: Evaluation of Risk Hypotheses

Presently, critical habitats have been designated for 26 of the 28 listed salmonids and all fall within the action area. Designated critical habitat within the action area consists of spawning and rearing areas, freshwater migratory corridors, and nearshore and estuarine areas, and includes essential physical and biological features. The effects of the proposed action on prey and water quality PCEs are addressed below by the following risk hypotheses. If the PCEs are impacted, we address the potential for reductions to the associated conservation value of the designated critical habitat.

Risk Hypotheses:

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 abundances in 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 malathion, chlorpyrifos, and diazinon are present together. The concentrations we summarized indicate that alone each of the insecticides can also kill prey at expected environmental concentrations.

Indices of biological integrity 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 predominant salmonid prey items are frequently in low numbers or absent in these areas. We see similar depauperate communities in urban areas as well. We understand that many other limiting factors are also partly responsible for the poor conditions of these aquatic communities. However, the role of these insecticides and their formulations likely bear a portion of the responsibility. In fact, several studies have shown toxicity to salmonid prey items from field collected waters and sediment resulting from chlorpyrifos, diazinon, and malathion (Anderson et al. 2003a; Anderson et al. 2003b; Anderson et al. 2006; Werner et al. 2000; Werner et al. 2002; Werner et al. 2004).

In summary, the available information shows that prey items of ESA-listed salmonids are affected by the stressors of the action to such an extent that warrants an analysis of whether the conservation value of designated critical habitat is reduced.

2. Exposure to the stressors of the action is sufficient to degrade water quality in designated critical habitat.

We evaluated this hypothesis by applying exposure concentrations evaluated in the *Exposure Analysis* and toxicity data from the *Response Analysis*. We also compared expected concentrations in salmonid habitats to U.S. Water Quality Criteria to determine if thresholds are exceeded. Further, we evaluated if any of the state waters within designated critical habitat are listed as impaired by chlorpyrifos, diazinon, or malathion by searching 303(d) lists.

- The expected concentrations from the proposed action trigger adverse effect levels for salmonids and their prey (see *Exposure Analysis* and *Response Analysis*). We expect these concentrations to be present in designated critical habitat and therefore to degrade water quality.

- Chlorpyrifos, diazinon, and malathion are listed as priority pollutants under the Clean Water Act. We expect that concentrations from the proposed action will frequently exceed both acute and chronic levels in designated critical habitats.

- Rivers and stream reaches within designated critical habitats in California have been listed as impaired due to contamination with diazinon and chlorpyrifos.

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. Collectively, this information supports 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.

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 that 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 chlorpyrifos, diazinon, and malathion as described in labeling of all pesticide products containing these active ingredients.

- Exposure to non-agricultural uses. We lacked exposure estimates of stressors of the action associated with non-agricultural uses of these pesticides.

- Exposure and toxicity to 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 distribution. 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 chlorpyrifos, diazinon, and malathion, 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 as defined in 50 CFR 402.2 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 First Search, Google, and other electronic search engines. Those searches produce 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. 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 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 to be from 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, copper, PAHs, and other chemical pollutants and flow into state surface waters. Inputs of these point and nonpoint 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 continue to add towards 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. Chlorpyrifos, diazinon, and malathion 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 and 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 and unanticipated negative effects on water quality. They include increases in sedimentation, loss of riparian shade (increasing temperatures), increased point and nonpoint pollution discharges, decreased infiltration of rainwater (leading to decreases in shallow groundwater recharge, leading to decreases in hyporheic flow, leading to 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 best management practices (e.g., associated with timber harvest, grazing, agricultural activities, urban development, road building, recreational activities, and other non-point source pollution controls).

NMFS expects many of the current anthropogenic effects described in the *Environmental Baseline* to continue. Listed Pacific salmonids are exposed to harvest, hatchery, hydropower, and habitat degradation activities. Regarding 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 of 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 chlorpyrifos, diazinon, and malathion, 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 species and critical habitat as a result of EPA's registration of chlorpyrifos, diazinon, and malathion. 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 (as captured by the likelihood of reductions in the species' viability) and (2) result in the destruction or adverse modification of designated critical habitat. Specifically, we discuss the likelihood of the proposed action to reduce annual population growth rates, or viability, of the salmonid species to such an extent that increased extinction rates are likely. We also address whether the critical habitat will remain functional to serve the intended conservation role for 26 listed Pacific salmonid ESUs or retain its current ability to establish those features and functions essential to the conservation of the species. The *Approach to the Assessment* section described the analysis and tools we have used to complete this evaluation. Conclusions for each ESU/DPS and associated designated critical habits are found in the *Conclusion* section.

Status and environmental baseline information

For each ESU/DPS we summarize the current status information, focusing on the current condition, trends in parameters, and historic run sizes of populations. We also note any population extinctions. Other stressors described in the *Environmental Baseline* section are summarized. Elevated temperature and pesticide detections are noted when information is available. Prior to each ESU level analysis, we highlight two factors which are common to all listed species addressed in this Opinion: the effects of climate change and oceanic conditions, and the prevalence of pesticides in the aquatic environment.

Climate Change and Oceanic Conditions

As described in the *Environmental Baseline* section, climate change may have direct and indirect effects on individuals, populations, species, and the structure and function of marine, coastal and terrestrial ecosystems, including Pacific salmonids, in the foreseeable future. The effects of climate change include increases in atmospheric temperatures, and changes in sea surface temperatures, patterns of precipitation, sea level, distribution and abundance of prey, and the distribution and abundance of competitors or predators.

Oceanic conditions may also influence prey availability and habitat for all Pacific salmonids. The primary effects of the ocean on salmon productivity involve both growth and survival of salmon. Collectively, changes in climate and oceanographic conditions may affect prey availability, temperature and water flow in habitat, and growth for all 28 ESUs. Given the variability in oceanic conditions and increasing effects of climate change, we expect that elevated water temperature will have a significant impact on all listed Pacific salmonids. We further expect negative and positive effects on the species' ability to attain viability. Consequently, we expect the long-term survival and reproductive success for listed salmonids to be greatly affected from oceanic conditions and climatic variability.

Pesticide use and salmonid habitats

A significant risk to threatened and endangered ESUs/DPSs is pesticide drift and runoff to salmonid aquatic habitats. Listed salmonids occupy habitats ranging from shallow, low flow freshwaters, to open reaches of the Pacific Ocean. The temporal and spatial use of habitats by salmonids depend on the species and the individual's life history and life stage. Salmon use freshwater and estuarine wetlands for physiological transition to and from salt water and rearing habitat. Salmon use marine nearshore areas for rearing and migration, with juveniles using shallow shoreline habitats. Given the species' life history, salmonids may be exposed to chlorpyrifos, diazinon, and malathion through direct contact with contaminated surface water or pore water. Of particular concern are small streams and off-channel habitats used by salmonids that have a lower capacity to dilute pesticide contaminants. These habitats are frequently in floodplain areas that overlap with agricultural, residential, and urban land uses. Dietary consumption via salmonid prey is a likely route of exposure and is significant exposure for chlorpyrifos. Chlorpyrifos accumulates in tissues of aquatic organisms and may be consumed by other fish and animals throughout the food chain. Salmonid prey items include dead or dying aquatic terrestrial insects that have been exposed to the three active ingredients.

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). Diverse and abundant communities of invertebrates (many of which are slamonid prey) also population these habitats, and, in part, are responsible for juvenile salmonid reliance on off-channel habitats. Life history attributes of listed salmonids are key determinants of exposure to the stressors of the action.

We also recognize, as indentified in the *Exposure Analysis* section, that significant uncertainty exists with current and future non-agricultural uses of the three active ingredients within the ranges of the ESUs/DPSs. This is further complicated by the lack of available methods to estimate exposure for non-agricultural uses. Monitoring studies indicate that detection of chlorpyrifos, diazinon, and malathion occurs frequently throughout the action area in freshwater and nearshore environments associated with urban, agricultural or mixed land use watersheds (Andersen et al. 2007; Burke et al.

2006; CDPR 1995; CDPR 2008b, Gilliom et al. 2006). However, there is a limited amount of monitoring data available for streams and off-channel habitats. The available monitoring data are not adequate to define exposure at the ESU/DPS level.

We also expect use of many currently registered acetylcholinesterase-inhibiting insecticides within all of the ESU/DPS ranges. As discussed in this Opinion, we expect surface waters that contain chlorpyrifos, diazinon, and malathion to affect individuals and prey by additive toxicity as a result of the cumulative impairment of AChE activity and all AChE-associated physiological functions. Additionally, we also expect to see additive toxicity in the form of AChE inhibition in salmonids and their prey in surface waters containing other OPs and carbamates. Similarly, synergism occurs with certain combinations and specific concentrations of chlorpyrifos, diazinon, and malathion. This interaction translates into increased rates of mortality among exposed salmonids. While we have no predictive models for this phenomenon, we expect to see synergistic effects where these three pesticides co-occur in specific levels.

Due to the registered uses of the three pesticides for a variety of agricultural crops, we highlight for each ESU/DPS the percentage of agricultural land uses to provide a coarse analysis of potential pesticide use. Note, that the NLCD land use categories we used do not directly correspond to EPA-approved use sites especially non-agricultural uses. For example, chlorpyrifos is registered for use on many food crops as well as for Christmas trees even though Christmas trees may be grown in forested, suburban, or typical croplands.

Effects of the proposed action at the species level

California Coastal Chinook salmon

The California Coastal (CC) Chinook salmon ESU includes Chinook salmon in 10 larger and smaller coastal river basins in Humboldt and Mendocino Counties, California. Dams and other impediments to migration such as road culverts have substantially reduced salmon distribution within watersheds. Nevertheless, Chinook salmon runs still exist in all streams that are believed to have supported Chinook salmon historically but in a reduced state. Historic production estimates for the ESU imply that annual escapement may have been around 73,000 fish with most of these being produced in the Eel River. Today, annual adult returns to the Eel River system is estimated at 150 to 2,800 fish. However, both long-term estimates and recent snapshot surveys are lacking for most of the streams within the ESU. Video monitoring at the Maribel Dam in the Russian River has counted between 1,000 and 7,000 adult Chinook passing the dam since 2002. Hatchery stocking has been common in many rivers. However, the CC Chinook salmon seems to have maintained a high interpopulation genetic diversity. For example, the Eel River system may consist of up to five distinct runs. Genetic studies of Russian River Chinook salmon have shown that this run is genetically distinct despite massive stocking of out-of-basin smolts.

The major threats to this ESU identified in the *Status of Listed Resources* and *Environmental Baseline* sections indicate that fisheries, vineyard development, dams and other migration barriers negatively affect this ESU. Other challenges faced by this species include introduced fish species, timber harvest and other agricultural activities. 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, increased predation, and barriers that limit access to tributaries. About 40 km of streams are 303(d) listed for elevated temperature. Pesticides used in forestry, vineyards, and other agricultural activities likely enter streams. Further, the Mediterranean climate in California with dry summers and wet winters may result in high concentration of contaminants in run-off during the onset of the rainy season. The cumulative impacts from these multiple threats continue to affect the CC Chinook salmon.

The three active ingredients are applied for a variety of pest control purposes and span multiple land uses within the CC Chinook salmon ESU (Table 38). According to NLCD, a relatively small portion of the ESU (about 1%) is developed for cultivation of crops (Table 25). However, a high density of vineyards exists in Mendocino County, especially in the Russian River basin. All three active ingredients are expected to be used for several crops and some as part of forestry activities. It is unclear how these vineyards would be categorized. Farming activities are concentrated in low laying areas and floodplains along the estuaries and lower reaches of streams. Consequently, these areas are in close proximity to migration corridors, spawning habitat, and rearing habitat utilized by the CC Chinook ESU. The majority of the ESU (about 52%) is categorized as coniferous forests, and forestry activities occur in watersheds throughout the ESU. Chlorpyrifos is registered for use on pine seedlings, Christmas trees, and woodlands although the extent to which it may be used for these purposes is unclear (EPA 2004a). Approximately 5% of the ESU consists of urban development. The urban environment could potentially contribute to a significant level of exposure given the variety of approved uses, particularly for chlorpyrifos and malathion. Diazinon use in developed areas has been restricted in recent years, although it may be applied to outdoor ornamental plants in nurseries. There is also a special local needs registration for drenching of residential fruit trees in California. Coastal communities in many places are located at the mouth of streams within the CC Chinook salmon ESU. Non-agricultural uses of malathion (particularly for mosquito control) across the various land uses are expected and can pose significant risk to the CC Chinook salmon. The most significant risk posed by chlorpyrifos and diazinon are likely from agricultural uses. However, non-agricultural uses of these compounds across the ESU are expected to contribute additional risk to the CC Chinook salmon. Further, the occurrence of malathion and other cholinesterase-inhibiting insecticides in the environmental baseline are likely to exacerbate the risk.

The CC Chinook salmon are all fall-run or ocean-type. The CC Chinook salmon enter rivers from late August through December. Peak migration in the Russian River usually occurs in October or early November. Once in the stream, the adults move quickly to spawning grounds but may hold in deeper pools in rivers to wait for increased flow if migration is restricted by low water levels. Spawning occurs on gravel beds in the mainstem of rivers and in larger tributaries. Offspring do not remain long in river and streams to rear, although their residence in these systems overlaps significant periods of agricultural use of diazinon and chlorpyrifos. Downstream migration starts as early as February and most juveniles have entered marine waters before mid-summer. Peak downstream migration is usually recorded at the Russian River monitoring station between late April and mid-May. While ocean residency may last from two to five years, the majority of the Chinook salmon returns as three year olds. The timing of this species' migration is of particular import, as the early spawning runs expose them to the first flush run-off of pesticides.

Given the presence of agricultural and other use sites throughout the watersheds used by the species, we expect that the proposed uses of chlorpyrifos and diazinon pesticides products that contaminate aquatic habitat will lead to both individual fitness level consequences and subsequent population level consequences, i.e., reductions in population viability. The widespread uses of these materials, and potential future uses, indicate substantial overlap with the populations that comprise the CC Chinook salmon ESU. Given the reduced abundance of the species in many rivers, presence of dense agricultural development along rivers like the Russian River, degraded channel habitat, and elevated water temperature, the risk to this species' survival and recovery from the stressors of the proposed action is high.

Central Valley Spring-run Chinook salmon

The Central Valley Spring-run Chinook salmon ESU includes populations in the Sacramento River, California. Historically there were runs of spring-run Chinook salmon in both the San Joaquin River and Sacramento River. Today, the run is extirpated from the San Joaquin River basin and from the American River, a large tributary to the lower Sacramento River. The distribution within the Sacramento River basin has been extensively reduced and is now restricted to accessible areas below dams in the mainstem river and in three of its tributaries. Over the last decade, there has also been a shift in spawning distribution. Most spawning now occurs in Deer, Mill, and Butte Creeks, while the number of fish using the Sacramento River mainstem has diminished. Thirty years ago, the Sacramento River held the largest spawner runs. Although the total annual number of returning spawners has increased over the last decade, fish 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. The species has had an exceptional cohort replacement rate over the last few decades. The exception to this trend is the Sacramento River spawners, which has been dwindling steadily.

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. None of the streams within the ESU's habitat are on the state's 303(d) list for elevated temperature. However, high water temperature still constitutes a significant risk to the species due to its preference for shallow off-channel habitats. Also, the 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. Contaminants from urban and agricultural runoff, and ammonia releases from wastewater treatment plants have been identified as sources of mortality to salmon in the Central Valley region. Modification of hydrology has been significant. State and Federal water diversions in the south Sacramento-San Joaquin Delta (Delta) have resulted in increased mortality through stranding, increased predation, prolonged migration, and entrainment at the water diversion facilities.

The three active ingredients are applied for a variety of pest control purposes that span multiple land uses within the Central Valley spring-run ESU (Table 25). About 21% of the land use within the ESU's habitat is developed for crop cultivation and about 10% is developed municipal and industrial. All three active ingredients are expected to be applied to several crops within the ESU. Large areas of urban centers occur along the Sacramento River and San Francisco Bay. Several of these urban centers have mosquito control programs, including spraying, to eliminate West Nile virus. 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. The Mediterranean climate in California, with dry summers and wet winters, may result in high concentration of contaminants during the onset of the rainy season.

Spring-run adults move up the Sacramento River and into tributaries from March through July, corresponding with pesticide applications. They then hold in deeper and cooler waters over summer before spawning starts between late August and early October. The spring-run is categorized as an ocean-type fish; the young salmon starts outmigration within four months following hatching. Fry rearing and migratory periods overlap with dormant spray periods of orchards. Chlorpyrifos and diazinon are commonly applied in dormant spray formulations. During migration, the young salmon migrate down the Sacramento River, through the Delta and San Francisco Bay.

Given fish migration along the Sacramento River and through the Delta and San Francisco Bay, we expect the proposed uses of chlorpyrifos, diazinon, and malathion pesticides products will lead to both individual fitness level consequences and subsequent population level consequences. The widespread uses of these materials indicate substantial overlap with the Central Valley Spring-run Chinook salmon ESU. With the species' oscillations in abundance, loss of habitat from dams, and limited access to cool water, the risk to this species' survival and recovery from the stressors of the proposed action is high.

Lower Columbia River Chinook salmon

The Lower Columbia River (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. Seven populations are considered extinct.

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. Industrial harbor and port developments, urbanization, logging, and agricultural practices further degrade freshwater and marine habitats for this ESU. LCR 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 are also affected by habitat factors in the estuary, such as changes in food availability and the presence of contaminants.

Collectively, about 287 km of this ESU's habitat in the states of Oregon (57 km) and Washington (230 km) have been listed as 303(d) waters for elevated temperature. NAWQA sampling in surface waters within the ESU range detected more than 50 pesticides in streams. Ten pesticides also exceeded EPA's criteria for the protection of aquatic life from chronic toxicity, including chlorpyrifos and malathion. The cumulative impacts from these multiple threats continue to affect this ESU.

Land use data (Table 30) indicate the majority of the ESU's landscape is covered by evergreen forest (49%), shrub/scrub (12%), and mixed forest (7%). About 12% of the land has been developed (1% high intensity) and 2% has been cultivated for crops. We expect application of the three active ingredients for a variety of pest control purposes across multiple land uses within LCR Chinook salmon habitat. Registered uses of chlorpyrifos, diazinon, and malathion include applications to crop agricultural sites, non-crop agricultural sites, residential sites, commercial sites, and animal and structural treatments. Some of these uses allow applications at rates that exceed those allowed in agricultural crops. Non-agricultural uses of malathion (particularly for mosquito control) across the various land uses is also expected and may pose significant risk to LCR Chinook salmon. Thus, ESU exposure to these pesticides is likely.

Mature adults (four to five years) of LCR Fall-run Chinook salmon enter freshwater in August through October to spawn in large river mainstems. Adults migrate and spawn in river reaches extending from above the tidewater to as far as 1,200 miles from the sea. The alevin life stage resides just below the gravel surface until they approach or reach the fry stage. Chinook fry typically select off-channel habitats associated with their natal rivers and streams. Juveniles eventually emigrate from freshwater as subyearling (ocean-type) 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 (stream-type). Stream-type fish migrate to the sea in the spring of their second year. 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.

Given the life history of LCR Chinook salmon, we expect the proposed uses of chlorpyrifos, diazinon, and malathion pesticide products may contaminate aquatic habitats and lead to individual fitness and subsequent population level consequences. The widespread uses of these materials indicate overlap with the 32 historical populations of LCR Chinook salmon. The risk to this species' survival and recovery from the stressors of the action is high.

Upper Columbia River Spring-run Chinook salmon

The Upper Columbia River (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. Should annual population growth rates at the 1980-2004 levels continue, UCR Spring-run Chinook salmon populations are likely to decline in 50 years. 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. About 255 km of this ESU's habitat in the state of Washington are listed as 303(d) waters for elevated temperature. Pesticide use and 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 et al. 1998). Concentrations of six pesticides in one or more surface water samples also exceeded EPA criteria for the protection of aquatic life, including chlorpyrifos, and diazinon. The cumulative impacts from these multiple threats continue to affect UCR Chinook salmon.

Registered uses of chlorpyrifos, diazinon, and malathion include applications to crop agricultural sites, non-crop agricultural sites, residential sites, commercial sites, and animal and structural treatments. Some of these uses allow applications at rates that exceed those allowed in agricultural crops. Land use data (Table 30) indicate that the majority of the ESU's landscape is covered by evergreen forest (45%), shrub/scrub (34%), and herbaceous cover (10%). About 3% of the land has been developed (0.1% high intensity) and 3.5% has been cultivated for crops. The three active ingredients are applied for a variety of pest control purposes that span multiple land uses within UCR Spring-run Chinook habitat. Thus, ESU exposure to these products is likely. Non-agricultural uses of malathion (particularly for mosquito control) across the various land uses is also expected and may pose significant risk to UCR Spring-run Chinook salmon.

UCR Spring-run Chinook salmon begin returning from the ocean in the early spring. The salmon enter the upper Columbia River tributaries from April through July. After migration, they hold in freshwater tributaries until spawning. Peak spawning occurs in mid- to late August. Fish spawn and rear 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 chlorpyrifos, diazinon, and malathion pesticide products that contaminate aquatic habitats will lead to individual fitness and likely lead to subsequent population level consequences, i.e., reductions in population viability. The widespread uses of these materials indicate substantial overlap with the 11 populations that comprise UCR Spring-run Chinook salmon. The risk to this species' survival and recovery from the stressors of the action is high.

Puget Sound Chinook Salmon

The Puget Sound Chinook salmon ESU is composed of 31 historic quasi-independent populations in Washington and Oregon. Of these, 22 are believed to be extant. Most of these populations are in the mid- to southern Puget Sound or Hood Canal and the Strait of Juan de Fuca. The Puget Sound Chinook salmon ESU also includes 26 artificial propagation programs. The estimated total run size for this ESU in the early 1990s was 240,000 fish. That estimate indicates the loss of nearly 450,000 fish from historic numbers. 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. Populations in the ESU have not experienced dramatic increases in abundance over the last two to three years. However, more populations (13) have shown modest increases in escapement in recent years than have declined (9). Recent five-year and long-term productivity trends remain below replacement for the majority of the two extant populations of Puget Sound Chinook salmon. The annual population growth rate known for 22 independent 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 degraded 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. About 705 km of the ESU's habitat is listed as 303(d) waters in the state of Washington for elevated temperature. 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 manmade organic chemicals in streams and rivers. Different mixtures of chemicals were 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).

A unique feature of Puget Sound is its fjord-like features and hydrological isolation from ocean water entry. As a result, toxic chemicals entering Puget Sound have longer residence times within the system and create elevated contaminant levels in the aquatic environment. The pelagic food web in Puget Sound is therefore exposed to increased levels of contaminants. Fall Chinook salmon from Puget Sound has been found to be more contaminated with PCBs (two to six times) and PBDEs (five to 17 times) compared to fish elsewhere. These contaminated conditions may lead to increased susceptibility of juvenile salmon to disease, acute die-offs of salmon returning to spawn in urban watersheds, and egg and larval mortality. The contaminated nature of Puget Sound and the cumulative impacts from the above threats continue to affect Puget Sound Chinook salmon.

Registered uses of chlorpyrifos, diazinon, and malathion include applications to crop agricultural sites, non-crop agricultural sites, residential sites, commercial sites, and animal and structural treatments. Some of these uses allow applications at rates that exceed those allowed in agricultural crops. Land use data (Table 30) indicate that the majority of the ESU landscape is covered by evergreen forest (49%), mixed forest (9%), and shrub/scrub (8.2%). About 14% of the land has been developed. Additionally, 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. We expect application of the three active ingredients for a variety of pest control purposes across land uses within Puget Sound Chinook salmon habitat. Thus, ESU exposure to these products is likely. Non-agricultural uses of malathion (particularly for mosquito control) across the various land uses is also expected and may pose significant risk to Puget Sound Chinook salmon.

Puget Sound Chinook salmon exhibit both spring- and ocean-type life histories. Puget Sound stream-type Chinook salmon adults travel long distances offshore and return to their natal rivers in the spring or summer months prior to spawning. Chinook salmon fry typically select 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. Occasionally males mature precociously without going to sea. The Puget Sound Chinook salmon's extended rearing period in shallow freshwater habitats increases their susceptibility to higher exposures of pesticides, contaminants, and elevated temperature. Ocean-type Chinook salmon adults return to their natal rivers in the fall and migrate to sea during their first year of life as fry or parr (Healey 1991). They spend most of their ocean life in coastal waters and return to freshwater a few days or weeks before spawning.

Given the life history of the Puget Sound Chinook salmon, we expect that the proposed uses of chlorpyrifos, diazinon, and malathion pesticide products may lead to individual fitness consequences and subsequent population level consequences, i.e., reductions in population viability. The widespread uses of these materials indicate substantial overlap with the 22 populations of Puget Sound Chinook salmon. The risk to this species' survival and recovery from the stressors of the action is high.

Sacramento River Winter-run Chinook salmon

The Sacramento winter-run Chinook salmon ESU includes only one population in Sacramento River, California. The current spawning distribution of Sacramento River winter-run salmon is severely altered from its historical distribution in cold headwaters. All historic spawning habitats have been blocked by dams in the upper Sacramento River. Historic run estimates for the Sacramento River imply that annual species abundance may have been as large as 200,000 fish (Brown et al. 1994). Estimated escapement dropped to critically low levels during the period from 1986 to 1992 with a low of 186 winter-run Chinook. Natural production between 1992 and 2000 ranged between 588 and 6,727 fish. Estimated natural production between 2001 and 2006 was substantially greater than during the 1992-2000 period, and ranged between 12,627 and 26,860 fish. In 2007, estimated natural production of adult winter-run Chinook salmon from the Sacramento River mainstem fell to 4,461 fish and preliminary analysis indicate a production between 2,600 and 2,950 adults for 2008. 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 Bureau of Reclamation (BOR) to manage cold water through reservoir storage and releases to support adult holding, spawning, incubation, and rearing. Climatic predictions combined with an expected population growth in southern California suggest that it will be increasingly difficult for BOR to continue to manage cold water in the future. 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 malathion, diazinon, chlorpyrifos, and other cholinesterase inhibiting insecticides. Contaminants from urban and agricultural runoff, and ammonia releases from wastewater treatment plants have been identified as sources of salmon mortality. Modification of hydrology has resulted in increased mortality through stranding, increased predation, prolonged migration and entrainment at water diversion facilities.

The three active ingredients are applied for a variety of pest control purposes that span multiple land uses within the Sacramento winter-run ESU. About 10% of the land overlapping with this ESU is developed and large areas of urban centers occur along the Sacramento River and San Francisco Bay (Table 25). Several of these urban centers and other developed areas have mosquito control programs, including spraying malathion products, to eliminate West Nile virus. All three active ingredients are expected to be applied to several crops within the ESU. About 21% of land within the ESU is cultivated for crops. Agriculture activity is prominent in the lower Sacramento River and within the Delta. 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. The Mediterranean climate in California, with dry summers and wet winters, may result in high concentration of these contaminants during the onset of the rainy season.

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 short time before starting outmigration to the sea in November and December. During outmigration, the young salmon migrate down the Sacramento River, through the Delta and San Francisco Bay. The majority of winter-run Chinook return from the sea to migrate upstream along the Sacramento River as three year olds; approximately 30 percent returns as two or four year olds.

Given the migration in the Sacramento River and through the Delta and San Francisco Bay, we expect that the proposed uses of chlorpyrifos, diazinon, and malathion pesticides products that contaminate aquatic habitat will lead to both individual fitness level consequences and subsequent population level consequences. The widespread use of these materials indicate substantial overlap with the only population that comprises the winter-run Chinook salmon ESU. Given that the Sacramento winter-run Chinook salmon consists of only one population, has a low abundance, has limited habitat due to water temperature, and is exposed to multiple contaminants along its migratory route, the risk to this species' survival and recovery from the stressors of the proposed action is high.

Snake River Fall-run Chinook salmon

The Snake River 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. The range for this ESU includes the Snake River basin in the states of Idaho, Oregon, and Washington... Historically, the primary Fall-run Chinook salmon spawning areas were located on the upper mainstem Snake River (Connor et al. 2005). A series of dams block access to the upper Snake River and has significantly reduced spawning and rearing habitat for this ESU. Natural spawning is now limited to the Snake River from the upper end of the Lower Granite Reservoir to Hells Canyon Dam, the lower reaches of the Imnaha, Grande Ronde, Clearwater, Salmon, and Tucannon Rivers, and small areas in the tail races of the lower Snake River hydroelectric dams (Good et al. 2005). Only 10 to 15% of the historical range of this ESU remains. Today, the vast majority of spawning occurs upstream from the Lower Granite Dam. Estimated annual returns from 1938 to 1949 were 72,000 fish (Bjornn and Horner 1980). In 1975, counts of natural-origin adult Snake River Fall-run Chinook salmon at Lower Granite Dam were 1,000 fish. Over the next 25 years, the number of natural-origin fish for this ESU ranged from 78 to 905. The average abundance (1,273) of Snake River 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. Elevated temperature, water diversions, and poor water quality pose significant threats to the status of Snake River Fall-run Chinook salmon. As agricultural activities, urban communities, and industries are concentrated along the Snake River and near the mouths of major tributary valleys, stream water quality and biological communities in the downstream portion of the upper Snake River basin are degraded. Collectively, 1,257 km of the Snake River Fall-run habitat in the states of Idaho (400 km), Oregon (610 km), and Washington (247 km), are listed as 303(d) waters for elevated temperature. The cumulative impacts from these multiple threats continue to affect Snake River Fall-run Chinook salmon.

Registered uses of chlorpyrifos, diazinon, and malathion include applications to crop agricultural sites, non-crop agricultural sites, residential sites, commercial sites, and animal and structural treatments. Some of these uses allow applications at rates that exceed those allowed in agricultural crops. Land use data (Table 30) indicate the majority of the ESU's landscape is covered by evergreen forest (49%), mixed forest (9%), and shrub/scrub (8.2%). About 14% of the land has been developed (1% high intensity) and less than 1% has been cultivated for crops. We expect application of the three active ingredients for a variety of pest control purposes across multiple land uses within Snake River Fall-run Chinook salmon habitat. Thus, ESU exposure to these pesticides is likely. Non-agricultural uses of malathion (particularly for mosquito control) across the various land uses is expected and may pose significant risk to Snake River Fall-run Chinook salmon.

Snake River Fall-run Chinook salmon generally spawn and rear in larger, mainstem rivers, such as the Salmon, Snake, and Clearwater Rivers. The largest concentrations of spawning sites occur in the Clearwater River, downstream from Lolo Creek. As a consequence of losing access to historic spawning and rearing sites in the Upper Snake River, Fall-run Chinook salmon now reside in waters that are generally cooler than the majority of historic spawning areas. Prior to alteration of the Snake River basin by dams, Snake River Fall-run Chinook salmon exhibited a largely ocean-type life history. These fish migrate downstream and rear in the mainstem Snake River during their first year. Today, Snake River Fall-run Chinook salmon in the Snake River basin exhibit ocean- and reservoir- type life history. Fish with a reservoir-type life history overwinter in low velocity pools created by the hydroelectric dams. This particular life history is likely a response to early development in cooler temperatures. This condition prevents juveniles from reaching a suitable size before they migrate out of the Snake River.

Adult Snake River Fall-run Chinook salmon enter the Columbia River in July and August. Spawning occurs above Lower Granite Dam in the mainstem Snake River and in the lower reaches of the larger tributaries. Spawning occurs from October through November and fry emerge from the redds beginning in March or April of the following year. They rear for two months or more in the sandy littoral zone along the river margin. Parr and presmolts move downstream from natal spawning and early rearing areas from June through early fall. Their duration in shallow freshwater habitats increases their chances of higher exposure to pesticides, contaminants, and elevated temperature. Juveniles eventually begin downstream migration o begin extended rearing in the deeper waters of the flowing river and reservoirs. Juveniles migrate along the edges of rivers, where they are at risk of exposure to higher concentrations of pesticides from drift and runoff. Subyearlings that enter the estuary as smolt reside there for a few weeks before moving into the plume and offshore waters (Fresh et al. 2005).

Given the life history of Snake River Fall-run Chinook salmon, we expect the proposed uses of chlorpyrifos, diazinon, and malathion pesticide products may lead to individual fitness level consequences and subsequent population level consequences. The widespread uses of these materials indicate substantial overlap with the only population for Snake River Fall-run Chinook salmon. The risk to this species’ survival and recovery from the stressors of the action is high.

Snake River Spring/Summer-run Chinook salmon

This ESU includes 32 populations in five major population groups. This species occupies 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, buoyed by the last five years, are approaching 1. 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 Snake River Spring/Summer-run Chinook salmon, across different habitats and land use categories, include elevated temperature, water diversions, and poor water quality. As agricultural activities, urban communities, and industries are concentrated along the Snake River and near the mouths of major tributary valleys, stream water quality and biological communities in the downstream portion of the upper Snake River basin are degraded. Collectively, about 1,596 km in Idaho (543 km), Oregon (809 km), and Washington (243 km) are listed as 303(d) waters for elevated temperature. The cumulative impacts from these multiple threats continue to affect Snake River Spring/Summer-run Chinook salmon.

Land use data (Table 30) indicate the majority of the ESU landscape is covered by evergreen forest (48%), shrub/scrub (24%), and herbaceous cover (19%). Although less than 1% of the land has been developed and roughly 7% has been cultivated for crops within the ESU's inland boundary, substantial risk to migrating salmonids exists. As juveniles migrate down the Snake River system to the Pacific Ocean and adults return to spawn, each follows a migratory path that overlaps extensively with uses of the three OPs. Juveniles in particular migrate along river shorelines and take advantage of off-channel rearing opportunities that are proximate to cultivated crops and other land uses where the threes OPs are likely applied. We expect application of the three active ingredients for a variety of pest control purposes across multiple land uses within Snake River Spring/Summer-run Chinook salmon habitat

Snake River 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. Natural-origin juveniles start moving downstream during the following autumn. They typically overwinter in shallow off-channel habitats associated with their natal rivers and streams. The duration of juvenile rearing in these habitat increases their susceptibility to higher exposures of pesticides, contaminants, and elevated temperature. Juveniles become active seaward migrants during the following spring as yearlings (stream-type juvenile life history) (Connor et al. 2005).

Given the life history of Snake River Spring/Summer-run Chinook salmon, we expect the proposed uses of chlorpyrifos, diazinon, and malathion pesticide products that contaminate aquatic habitats will lead to both individual fitness level consequences and subsequent population level consequences. The widespread uses of these materials indicate substantial overlap with the 32 populations that comprise Snake River Spring/Summer-run Chinook salmon. The risk to this species' survival and recovery from the stressors of the action is high.

Upper Willamette River Chinook Salmon

The Upper Willamette River (UWR) Chinook salmon ESU includes all naturally spawned populations residing in the Clackamas River, in the Upper Willamette River above Willamette Falls, and below impassable natural barriers in Oregon. The Willamette River valley is a major agricultural basin in the state of Oregon. A wide array of crops are grown throughout the year and a wide array of pesticides are used on these crops. The ESU is comprised of one major population group with eight historical independent populations. The ESU also includes seven artificial propagation programs. 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. Natural production occurs in only two populations – the Clackamas and McKenzie rivers. The annual population growth rate known for the two independent populations ranged from 0.92 to 0.96.

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. About 2,468 km of the ESU's habitat in the state of Oregon are listed as 303(d) waters for elevated temperature. Fifty pesticides were detected in streams, with 10 pesticides exceeding EPA criteria for the protection of freshwater aquatic life from chronic toxicity(Wentzet al. 1998). Forty-nine pesticides were also detected in streams draining agricultural land. About 25 pesticides were also detected in streams draining mostly urban areas. The cumulative impacts from these threats continue to affect UWR Chinook salmon.

Registered uses of chlorpyrifos, diazinon, and malathion include applications to crops, non-crop agricultural sites, residential sites, commercial sites, and animal and structural treatments. Some of these uses allow applications at rates that exceed those allowed in agricultural crops. Land use data (Table 30) indicate the majority of the ESU's landscape is covered by evergreen forest (43%). However, in 1992 the Willamette Basin accounted for 51% of Oregon's total gross farm sales and 58% of Oregon's crop sales. Roughly 11 % of land has been cultivated for crops and 16% is classified as hay or pasture. About one-third of the agricultural land is irrigated and most of it is adjacent to the mainstem Willamette River. Only 8% of the land has been developed, though urban land is located primarily in the valley along the mainstem Willamette River. Non-agricultural uses of malathion (particularly for mosquito control) across the various land uses is also expected and may pose significant risk to UWR Chinook salmon. We expect application of the three active ingredients for a variety of pest control purposes within UWR Chinook salmon habitat. Given that major urban and agricultural areas are located adjacent to the mainstem Willamette, ESU exposure to these pesticides is likely.

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 salmon ESUs (Meyers 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 subyearlings (Friesen et al. 2004).

Given the life history of UWR Chinook salmon, we expect the proposed uses of chlorpyrifos, diazinon, and malathion pesticide products may lead to both individual fitness level consequences and subsequent population level consequences, i.e., reductions in population viability. The widespread uses of these materials indicate substantial overlap with the eight independent populations that comprise UCR Chinook salmon. The risk to this species' survival and recovery from the stressors of the action is high.

Columbia River Chum Salmon

This ESU includes two remaining populations of 16 historical populations in the lower reaches (the Lower Gorge tributaries and Gray's River) of the Columbia River. Thus, about 88% of the historic populations are extirpated or nearly so.The number of chum salmon in the Columbia River system has been drastically reduced concurrently with the loss of spawning populations. Historically, the Columbia River chum salmon ESU was highly prolific. 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 et al. 2005). Previous estimates of the Gray's River population range from 331 to 812 individuals. However, the population increased in 2002 to as many as 10,000 individuals (Good et al. 2005). Because significant spawning occurs in two locations, the ESU is highly vulnerable to catastrophic events. 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 et al. 1997b). 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 diazinon and malathion (Ebbert and Embry 2001). Additionally, the boundaries of the ESU include 281.6 km of rivers, streams, and estuaries on the 303(d) list for exceeding temperature thresholds – 56.6 km in Oregon and 225.0 km in Washington. Though, the habitat restoration project for the Gray's River will likely provide some benefit to the population.

Land use data indicate that the Columbia River chum salmon may be at risk of pesticide exposure (Table 31). The majority of the ESU is covered by forests (evergreen forest-40.4%, scrub/shrub land-12.7%, and mixed forest-8.4%). Only 2% of the ESU land area is used for cultivated crops and 14% has been developed (1% high intensity). However, 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. The Columbia River itself is on the 303(d) list for over 60 contaminants, including temperature.

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. Juveniles are most likely to be exposure to pesticides when they utilize nearshore habitats. This behavior has been observed in juveniles and is commonly correlated to warming water temperatures and plankton blooms (Burgner 1991). Adults return to spawn in the lower reaches of the Columbia River between the ages of two and five from mid-October through December. A period of milling in front of their stream of origin of approximately ten to twelve days before entering freshwater is common in Fall run chum (Tynan 1997). An average of ten days is spent in the freshwater by the spawning adults. This behavior is likely related t the amount of time required for the chum to complete maturation and acclimate to freshwater, and represents a period where chum may be most susceptible to pesticide exposure.

Given the life history of the Columbia River chum salmon, we expect that the proposed uses of chlorpyrifos, diazinon, and malathion pesticide products may lead to individual fitness level and subsequent population level consequences. The widespread uses of these materials indicate overlap with the two extant populations of Columbia River chum salmon. Given that the ESU consists of two populations at very low numbers, the productivity trend line is flat or negative, it preferentially uses edge habitat and estuaries as fry, and the baseline presence of multiple contaminants in the river system, risk to this species' survival and recovery from the proposed stressors of the action is high.

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 estimated populations, seven are believed to be extirpated. Most of the extirpated populations occur 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 et al. 2005). The Hood Canal 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 degrading floodplains, estuarine, and riparian habitats, along with reduced stream flow and sedimentation, have had a profound negative impact on this ESU. Only 90 km of stream in the ESU are listed on the state of Washington's 303(d) list of impaired waters for elevated temperature.

The land use and environmental data indicate that the Hood Canal chum may be exposed to chlorpyrifos, diazinon, and malathion. The majority (64%) of the Hood Canal ESU land cover is evergreen forest (Table 31). There is no cultivated crop land and less than 6% of the ESU is developed. The land use data, however, may be misleading, as the impacts of urbanization on aquatic ecosystems are severe and long lasting (Spence et al.

1996). Studies on the Chinook salmon populations found in the Puget Sound area show elevated levels of pesticides and other contaminants (Brennan et al. 2004; O'Neill et al. 2006). These data imply that pesticide load on Hood Canal chum salmon may be higher than land use data alone would indicate.

The Hood Canal summer-run chum spawn from mid-September through mid-October (Tynan 1997). Hatching and fry emergence may both be tied to a number of factors, including temperature, salinity, dissolved oxygen, and gravel size. 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. After a period of residence in the nearshore estuary, they migrate up Hood Canal and out into Puget Sound. Adults return to spawn generally between two and five years, though they may not return for up to seven. Mature adults may also be susceptible to pesticide exposure in the estuary during their spawning migration. Upon return to the estuary in August and September, Hood Canal summer-run chum frequently mill in front of their stream of origin for approximately ten to twelve days before entering freshwater (Tynan 1997). Once the adults enter the freshwater (September-October) an average of ten days is spent there. This behavior is likely related to the amount of time required for the chum to complete maturation and acclimate to freshwater, and represents a period where chum may be most susceptible to pesticide exposure.

Given the life history of the Hood Canal chum, we expect the proposed uses of chlorpyrifos, diazinon, and malathion pesticide products may contaminate aquatic habitats and lead to individual fitness level and subsequent population level consequences. The widespread uses of these materials indicate overlap with the nine extant populations of Hood Canal chum salmon. The risk to this species' survival and recovery from the stressors of the action is high.

Central California Coast Coho Salmon

The Central California Coast (CCC) coho salmon ESU includes eight larger watersheds and several smaller streams within Mendocino, Sonoma, Marin, San Mateo, and Santa Cruz Counties 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 from most rivers within the ESU. However, a minimum of 6,570 adult coho salmon are estimated to return to coastal streams within the ESU. Where surveys have been undertaken, the number of returning spawners is low. Long-term trends for the annual population growth rate do not exist for any of the populations in this DPS. 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.

The threats to this ESU as described in the *Status of Listed Resources* and *Environmental Baseline* sections indicate that habitat modification and fishing negatively affect this ESU. Habitat threats such as loss of riparian cover, elevated water temperatures, alteration of channel morphology, loss of winter habitat, and siltation of stream substrate have been identified to influence the status of the CCC coho salmon ESU. About 39 km within the ESU's habitat are included in the state's 303(d) list of impaired waters for elevated temperature. High water temperature excludes coho salmon from inhabiting several streams within the ESU that are not 303(d) listed for temperature. Climate modeling predicts that elevated stream water temperatures will continue to be a limiting factor in the future. Land cover in large portions of some watersheds, such as the Russian River, consists of agriculture and urban development. These land uses may result in high levels of contaminated runoff into the mainstems and their tributaries.

Approximately two percent of the inland boundary of the ESU is in agriculture and 9.3% consists of urban development (Table 26). However, the majority of urban and agricultural development is concentrated in the Russian River watershed and watersheds to the south. Within this southern portion of the range, the three active ingredients are used for a variety of pest control purposes that span multiple land uses. Further, in many of these watersheds both agriculture and urban development are concentrated in the valleys along mainstem river channels. All three active ingredients are expected to be applied to several crops within the ESU. A considerable concern is drift and runoff of the three pesticide products into coho salmon habitat during aerial application. Non-agricultural uses of malathion (particularly for mosquito control) across the various land uses are expected and may pose significant risk to the CCC coho salmon.

Mature adults enter streams in winter, usually peaking in January. Stream entry and movement are influenced by stream flow. Stream movements usually occur during the first large storms. In many streams, fish entry into the stream also depends on breaching of a sandbar at the mouth of the estuary. Once in the stream, the adult coho salmon move quickly to spawning grounds higher up in the watershed. Fry emerge in spring and remain in the stream for up to 18 months. During winter, the juveniles move into side channels, sloughs, backwater, and other areas that protect against high flows and water currents. Following freshwater rearing, coho salmon spend about 18 months in the marine environment, returning to spawn as three year olds. There is very limited variation in this life history and streams usually consist of three partially isolated cohorts.

Given the life cycle of coho salmon juveniles, with more than one year of stream rearing and adults run timing coinciding with the season's first flushes, we expect the proposed use of chlorpyrifos, diazinon, and malathion pesticide products will lead to both individual fitness level consequences and subsequent population level consequences, i.e., reductions in population viability. The widespread uses of these materials within agriculture indicate substantial overlap with fish runs in the Russian River, and with many rivers to the south. These runs are small and influenced by many other stressors, including elevated temperatures. The southern portion of the ESU's geographic range represents an important environmental diversity within the range, and these southern runs may provide genetic diversity as well. Loss of these populations may reduce the ESU's ability to survive changes in climatic conditions. The risk to this species' survival from the proposed action is high, as the related stressors are expected to further decrease survival and fecundity of coho salmon already at risk of extirpation.

Lower Columbia River Coho Salmon

The Lower Columbia River (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 anadromous populations in three major population groups and 25 artificial propagation programs. LCR coho salmon populations have been in decline over the last 70 years. Data on the status of natural-origin LCR coho salmon are very limited. Most populations have low or very low numbers and have been replaced by hatchery production. The annual population growth rate known for two independent populations ranged from 1.028 – 1.102. Over 90% of the historic population for LCR coho salmon appear to be extirpated.

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 affect this ESU. Within the various types of habitat threats, elevated temperature, water diversions, and poor water quality pose significant influences on the status of LCR coho salmon. Collectively, about 525 km of the ESU's habitat in the states of Oregon (292 km) and Washington (234 km) are listed as 303(d) waters for elevated temperature. 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 chlorpyrifos and malathion. The cumulative impacts from these multiple threats continue to affect this ESU.

Registered uses of chlorpyrifos, diazinon, and malathion include applications to crop agricultural sites, non-crop agricultural sites, residential sites, commercial sites, and animal and structural treatments. Some of these uses allow applications at rates that exceed those allowed in agricultural crops. Land use data (Table 31) indicate the majority of the ESU's landscape is covered by evergreen forest (52%), shrub/scrub (12%), and mixed forest (6%). About 11% of the land has been developed and 2% has been cultivated for crops. We expect application of the three active ingredients for a variety of pest control purposes across multiple land uses within LCR coho salmon habitat. Thus, ESU exposure to these pesticides is likely. Non-agricultural uses of malathion (particularly for mosquito control) across the various land uses is also expected and may pose significant risk to LCR coho salmon.

LCR coho salmon enter freshwater from August through December. Coho salmon spawn in November and December, with exceptionally early and late runs occurring along the Washington coast, in the Columbia River, and in Puget Sound. Coho salmon 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. The early returning (Type S) coho salmon generally migrate south of the Columbia River once they reach the ocean. They return to freshwater in mid-August and to the spawning tributaries in early September. Spawning peaks from mid-October to early November. The late returning (Type N) coho salmon have a northern distribution in the ocean. They 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).

Given the life history of LCR coho salmon, we expect the proposed uses of chlorpyrifos, diazinon, and malathion pesticide products may lead to both individual fitness level consequences and subsequent population level consequences, i.e., reductions in population viability. The widespread uses of these materials indicate substantial overlap with the 26 populations that comprise the LCR coho salmon. The risk to this species' survival and recovery from the stressors of the action is high.

Southern Oregon and Northern California Coast Coho Salmon

The Southern Oregon/Northern California Coast (SONCC) coho salmon ESU includes all naturally spawning populations of coho salmon in streams between Punta Gorda, California, and Cape Blanco, Oregon. Three larger rivers (Klamath, Mattole, and Eel Rivers) and many small and medium sized streams exist within this range. Little information on escapement trends exists for most of the streams within the ESU. However, numbers in the largest rivers are believed to have decreased substantially compared to the early 1900s. Estimated escapement in the Klamath River dropped from approximately 15,400 in the mid-1960s to about 3,000 in the mid 1980s, and more recently to 2,000 fish. In the Eel River estimated escapement dropped from 14,000, to 4,000 to about 2,000 during the same period.

The threats to this ESU as described in the *Status of Listed Resources* and *Environmental Baseline* sections indicate that impaired or loss of habitat, road crossings and other migration barriers, timber harvest and agricultural activities negatively affect this ESU. Adverse effects on the SONCC coho salmon consist of high percentage of fines in the streams' bottom substrate, barriers that limit access to tributaries, lack of large instream woody debris, reduced riparian vegetation, and elevated water temperature. About 3,250 river kilometers within the ESU's habitat are listed as 303(d) waters for elevated water temperature. Climate modeling predicts that elevated stream water temperature will continue to affect the species in the future. Pesticides used in forestry and agricultural activities are expected to enter streams during application and as runoff into the mainstems and their tributaries.

About 1% of the inland boundary of the ESU is in agriculture and less than 1% consists of low to high intensity urban development (Table 25). However, in many of these watersheds, both agriculture and urban developments are concentrated in the valleys along mainstem river channels. Approximately 60% of the land use overlapping with this ESU consists of evergreen forests. Active forest management occurs throughout the watersheds within this ESU, and application of pesticide products is anticipated. The three active ingredients are used for a variety of pest control purposes that span multiple land uses and are expected to be applied to areas within the range of this ESU. Aerial application of pesticide products is of considerable concern as it may contribute to run off and drift into coho salmon habitat. Non-agricultural uses of malathion (particularly for mosquito control) across the various land uses are expected and may pose significant risk to the SONCC coho salmon.

In the northern portion of the ESU, mature adults enter streams in September and October while south of Klamath River mature adults usually will not enter until November and December. Stream entry and movement are influenced by stream flow; entry often occurs during the first large storms of the season. In the southern portion of the ESU, adult coho salmon move quickly to spawning grounds higher up in the watershed; in the northern portion of the ESU they may remain for a longer time in the river before spawning. Fry emerge in spring and remain in the stream for up to 18 months. During winter, the juveniles move into side channels, sloughs, backwater, and other stream features that protect against high flows and water currents. Streams are usually inhabited by three partially isolated cohorts. Following freshwater rearing, coho salmon spend about 18 months in the marine environment and, thus, return as three year olds.

Given the life cycle of coho salmon juveniles with more than one year of stream rearing and adults run timing coinciding with the season's first flushes, we expect that the proposed use of chlorpyrifos, diazinon, and malathion pesticide products will lead to both individual fitness level consequences and subsequent population level consequences. The widespread uses of these materials in agriculture and forestry indicate substantial overlap with SONCC coho salmon. SONCC coho runs are small and adversely affected by many other stressors, including elevated temperatures. The risk from the proposed action to the species' survival and recovery is high as many populations are already at risk of extirpation from other stressors.

Oregon Coast Coho Salmon

The Oregon Coast coho salmon ESU includes 11 naturally spawned populations and one hatchery stock in Oregon. While none of the populations have become extinct, it is estimated that current abundance levels are less than 10% of historic populations. In 2001 and 2002, yearly adult returns exceeded 160,000 natural spawners. The five-year geometric mean abundance from 2002-2006 was 152,960 total natural spawners, exceeding 1992-1996 mean abundance of 52,845 individuals. From 2003 to 2006, productivity declined, resulting in the current listing of the Oregon Coast coho ESU. 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 indicate habitat degradation from logging, road construction, urban development, mining, agriculture, recreation, water diversions, and poor water quality negatively affect this ESU. Within the various types of habitat, elevated temperatures, water diversions, and poor water quality pose significant threats to the status of Oregon Coast coho salmon. About 3,716 km of salmonid habitat in Oregon are identified as 303(d) impaired waters for elevated temperature. High water temperature may affect the migration, rearing, and emergence needs of fish and the aquatic organisms upon which they depend. The cumulative impacts from these multiple threats continue to affect Oregon Coast coho salmon.

The three active ingredients are applied for pest control purposes that span multiple land uses within Oregon Coast coho salmon habitat. Agricultural land comprise about 0.23% of the land within the ESU's range, while hay and pastures account for 3.1 % (Table 31). The most dominant land cover is evergreen forest (54.7 %). All three active ingredients are expected to be applied within the ESU habitat for urban and forestry uses, placing all populations at risk of exposure.

Oregon Coastal coho salmon enter rivers in September or October; spawning occurs in December. Emergence occurs within a few weeks of hatching. Following emergence, fry 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, contaminants, and elevated temperature. After approximately 18 months, they return to freshwater to spawn.

Given the life history of Oregon Coast coho salmon, we expect the proposed uses of chlorpyrifos, diazinon, and malathion pesticide products that contaminate aquatic habitats may lead to both individual fitness level consequences and subsequent population level consequences, i.e., reductions in population viability. The widespread uses of these materials indicate substantial overlap with the 11 populations that comprise the Oregon Coast coho salmon. The risk to this species' survival and recovery from the stressors of the action is high.

Ozette Lake Sockeye Salmon

This ESU is made up of only one historic population, with substantial substructuring of individuals into multiple spawning aggregations. Today natural spawning aggregations remain on two beaches of Ozette Lake. It is likely that originally there were multiple spawning aggregations along Ozette Lake's shoreline. However, there is limited evidence to determine the exact number of subpopulations that occurred historically. Hatchery operations and spawner returns occur in two tributaries. The tributary spawning groups were initiated 1992 through hatchery programs, with the first returns of hatchery fish in 1995. It is unclear if tributary spawning ever occurred historically. Peak run size in the 1940s has been estimated to be from about 3,000 to 18,000 fish, and actual production (i.e., including harvest) may have been as high as 50,000. The five year average (geometric mean)estimated abundance from 1994-1998 was 580 (Good et al. 2005). More recent estimates put the population at 3,600 – 4,600 individuals (Haggerty et al. 2007). Given the uncertainty in past population counts coupled with poorly documented historical abundance, it is difficult to determine population growth rates and trends. The supplemental hatchery program began with out-of-basin stocks and make up an average of 10% of the run. The proportion of beach-spawners originating from the hatchery is unknown but it is likely that straying is low.

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 et al. 1996). The extent to which pesticide products are currently used by these companies is unknown. Another more recently identified threat is predation of pre-spawning adults by harbor seals and river otters.

Ozette Lake is in a sparsely populated area, with less than 1% of developed area (0.3% open space, 0.2 % low intensity) and no crop land was identified in NLCD data (Table 32). This ESU has 4.8 km of its habitat listed on the state of Washington's 303(d) list of impaired waters for exceeding temperature thresholds. The land use and environmental data indicate that the Ozette Lake sockeye salmon may be exposed to chlorpyrifos, diazinon, and malathion if applied in the watershed. However, there are few data available on use and no monitoring data are currently available.

Ozette Lake sockeye salmon enter the lake between April and August, and spawning occurs late October through February. Natural spawning occurs on gravely beaches, while hatchery-origin fish spawn in tributaries to the lake. The 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 chlorpyrifos, diazinon, and malathion pesticide products may contaminate aquatic habitats used by sockeye in a way that might lead to individual fitness level and subsequent population level consequences. While the widespread permitted uses of these materials likely leads to some overlap with the Ozette Lake sockeye, the existing and likely future land uses should limit the applications of chlorpyrifos, diazinon, and malathion containing pesticides. Consequently, the risk posed by the proposed action to Ozette Lake sockeye salmon's survival and recovery is low.

Snake River Sockeye Salmon

The Snake River 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 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). Given the low abundances and high proportion of hatchery origin fish, the probability of genetic impacts and loss of genetic diversity is of concern for this ESU.

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 Snake River sockeye occupy a relatively undeveloped area (< 1% developed) with very little cropland Table (26). None of the primary habitat is listed for elevated temperature on Idaho's 303(d) list for impaired waters. However, the Snake River sockeye have the longest migration of any sockeye salmon, traveling 900 miles inland. Much of the migratory path is listed on the 303(d) lists for Oregon and Washington. These waters are contaminated by drift and runoff from both agricultural and urban areas. This exposure during migration likely adds to the low survivorship of smolt. Mortality during migration is also due to traversing the 18 dams located on the mainstem Columbia River and the Snake River. The land use and environmental data indicate that the Snake River sockeye may be exposed to chlorpyrifos, diazinon, and malathion.

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. Chlorpyrifos, diazinon, and malathion applications are permitted on these crops. Drift and runoff occurring in conjunction with sockeye salmon migration is expected to cause adverse effect.

Given the life history, extremely low abundances and high vulnerability of the Snake River sockeye, we expect the proposed uses of chlorpyrifos, diazinon, and malathion pesticide products may lead to individual fitness level and subsequent population level consequences. The widespread uses of these materials indicate overlap with this highly sensitive population of Snake River sockeye. The risk to this species' survival and recovery from the stressors of the action is very high.

Central California Coast Steelhead

The Central California Coast (CCC) steelhead DPS includes all naturally spawned steelhead in streams from the Russian River (inclusive) to Aptos Creek (inclusive), California. This area includes streams entering the San Francisco Bay, San Pablo Bay, and Suisun Bay up to Chipps Island. In total, the DPS consists of nine larger streams, of which the Russian and San Lorenzo Rivers have historically been the most productive. Several smaller streams that support steelhead production on an annual or intermittent basis also exist within the DPS. However, steelhead runs have gone extinct from many of these smaller streams, especially within the San Francisco and San Pablo Bays where steelhead is in danger of being extirpated. Overall production within the DPS has decreased substantially compared to earlier estimates. For example, Waddell Creek averaged about 500 spawners from the 1930s through the 1940s. The same stream supported 150 spawners in 1994. Historically, the Russian River is believed to have had a return of up to 65,000 adults but today it is estimated to have an annual return consisting of a few thousand adults. The return of adult steelhead in San Lorenzo River, once having an annual adult return of up to 20,000 fish, has been reduced to 85% from 30 years ago. Today, the population is on the brink of being extirpated. Hatchery operations have occurred within the DPS and genetic studies indicate that populations in some streams, e.g., Russian River, have been genetically compromised.

The major threats to this DPS identified in the *Status of Listed Resources* and *Environmental Baseline* sections indicate that dams and other migration barriers, urbanization and channel modification, agricultural activities, predators, hatcheries, and water diversions negatively affect this DPS. Throughout the species' range, the activities and disturbances occurring within watersheds have resulted in degraded habitat conditions and water quality. They include a high proportion of fines in stream substrate, 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. An increased abundance of pinnipeds and introduction of fish predators has increased mortality of both young and returning adult steelhead in many watersheds. The cumulative impacts of these threats continue to affect the CCC steelhead.

The three active ingredients are applied for a variety of pest control purposes that span multiple land uses within the CCC steelhead DPS. About 3.2% of the DPS is developed for cultivation of crops. Crop farming is concentrated in low laying areas and floodplains along the estuaries and lower reaches of streams, especially in the Russian and San Lorenzo River basins. These same waters serve as important rearing habitats for CCC steelhead. All three active ingredients are expected to be used for several crops grown in the area. Further, the DPS has the highest density of urban development of all salmonid species with about 16% of the area within the DPS consisting of low to high intensity developed land. About 10% consists of medium to high density urban and industrial developments. Many developed areas are located at the mouth of streams within the CCC steelhead DPS. Non-agricultural uses of malathion (particularly for mosquito control) across the various land uses are expected and can pose significant risk to the CCC steelhead. A considerable concern is drift and runoff of the three pesticide products into steelhead habitat during aerial application.

All of the steelhead populations within this DPS are of the winter-type life history. The winter-type enters rivers as mature adults in fall and winter to spawn. Stream entry is highly dependent on rainfall. Juvenile steelhead remain in freshwater for one or more years before migrating downstream to smolt. They often remain in estuaries for a longer period before fully entering the marine environment. CCC steelhead typically spend one or two years in the ocean. However, in many populations, a small fraction of fish will spend a third year at sea. A fraction of the adults, especially females, survives the spawning season to spawn a second, third, or even a fourth time.

Given the long freshwater residence time by steelhead juveniles, and the relatively high urbanization and presence of agriculture within watersheds used by the species, we expect the proposed uses of chlorpyrifos, diazinon, and malathion pesticides products that contaminate aquatic habitat will lead to both individual fitness level consequences and subsequent population level consequences, i.e., reductions in population viability. The widespread uses of these materials indicate substantial overlap with the populations that comprises the CCC steelhead DPS. Given the low abundance of the DPS, the extensive habitat modification that has occurred, and the prevalence of elevated water temperature, the risk to this species' survival and recovery from the stressors of the proposed action is high.

California Central Valley Steelhead

The California Central Valley (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. The current distribution is severely reduced and fragmented compared to historical distributions. About 6,000 river miles of river access has now been reduced to 300 miles. The majority of this loss was caused by the construction of dams in the upper Sacramento River, San Joaquin River and its tributaries, American River, Yuba River and Stanislaus River. Historical returns within the DPS may have approached two million adults annually but declined to an estimated 40,000 adults by the early 1960s. Current annual run size for the entire Sacramento-San Joaquin system today is estimated to less than 10,000 returning adults. In the Sacramento River, an average of 11,187 adults was counted at the Red Bluff Diversion Dam for the period 1967 to 1977. This number dropped to an average of approximately 2,000 through the early 1990s.

The major threats to this DPS identified in the *Status of Listed Resources* and *Environmental Baseline* sections indicate that dams and other migration barriers, urbanization and channel modification, agricultural activities, 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, DPR, and other assessments found high concentration of contaminants in both the San Joaquin and Sacramento Rivers and their tributaries, including detections of chlorpyrifos, malathion, and diazinon. 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. Multiple river and stream reaches are currently 303(d) listed. These factors affect the species throughout its range. The cumulative impacts from these threats continue to affect the CCV steelhead.

The three active ingredients are applied for a variety of pest control purposes that span multiple land uses within the CCV steelhead DPS. Approximately 27% of the DPS is developed for cultivation of crops (Table 26). High densities of crop farming occur throughout the San Joaquin Basin, in the Sacramento-San Joaquin Delta, and along lower Sacramento River. All three active ingredients are expected to be used for several crops grown in the area. Further, the DPS has a 9.2% of urban development with about 3% of the area within the DPS consisting of medium to high intensity developed land. Non-agricultural uses of malathion (particularly for mosquito control) across the various land uses are expected and may pose significant risk to the CCV steelhead. A considerable concern is drift and runoff of the three pesticide products into steelhead habitat during aerial application.

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. The winter type steelhead enters rivers as mature adults in fall and winter to spawn. Stream entry is highly dependent on rainfall. Juvenile steelhead remain in freshwater for one or more years before migrating downstream to enter the ocean. Steelhead often remain in estuaries for a longer period before fully entering the marine environment and the species may hold in the Delta for some time before entering the ocean.

Given the long freshwater residence time by steelhead juveniles, and the relatively high urbanization and agricultural development within watersheds used by the species, we expect the proposed uses of chlorpyrifos, diazinon, and malathion pesticides products will lead to both individual fitness level consequences and subsequent population level consequences, i.e., reductions in population viability. The widespread uses of these materials indicate substantial overlap with the populations that comprises the CCV steelhead DPS. Given the reduced abundance of the DPS, the extensive habitat modification and loss that has occurred, water volume loss to water diversions, increased water temperatures, and presence of multiple contaminants, the risk to this species' survival and recovery from the stressors of the proposed action is high.

Lower Columbia River Steelhead

The Lower Columbia River (LCR) Steelhead DPS includes 23 historical populations in four major population groups. This DPS includes naturally-spawned 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. 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. A number of the populations have a substantial fraction of hatchery-origin spawners. Many of the populations in this DPS are small. The long- and short-term trends in abundance of all individual populations are negative. For most populations, the trend in natural-origin spawners is less than one. These populations have relatively low recent abundance estimates. The largest is the Kalama River population with 726 spawners. The data series for most stocks is short and downward trends may reflect the general coast-wide decline in steelhead. The annual population growth rate known for nine independent populations ranged from 0.945 to 1.06.

The major threats to this ESU identified in the *Status of Listed Resources* and *Environmental Baseline* sections include water diversions that deplete water levels in rivers and streams; destruction or degradation of riparian habitat; and land use practices (logging, agriculture, and urbanization) that destroy wetland and riparian ecosystems. Within the various types of habitat, elevated temperature, water diversions, and poor water quality pose significant influences on the status of LCR steelhead. About 282 km of the ESU's habitat in the state of Washington are identified as 303(d) impaired waters for elevated temperature. Pesticide use and detections in LCR steelhead habitats are also well documented. NAWQA sampling from 1991-1995 in surface waters within the ESU range detected more than 50 pesticides in streams. Ten pesticides exceeded EPA's criteria for the protection of aquatic life from chronic toxicity. These pesticides include atrazine, chlorpyrifos, and malathion. The cumulative impacts from these multiple threats continue to affect this ESU.

Registered uses of chlorpyrifos, diazinon, and malathion include applications to crop agricultural sites, non-crop agricultural sites, residential sites, commercial sites, and animal and structural treatments. Some of these uses allow applications at rates that exceed those allowed in agricultural crops. Land use data (Table 32) indicate the majority of the ESU's landscape is covered by evergreen forest (55%), shrub/scrub (12%), mixed forest (5%), and hay/pasture (5%). About 11% of the land has been developed and 3% has been cultivated for crops. We expect application of the three active ingredients for pest control purposes across multiple land uses within LCR steelhead habitat. Thus, ESU exposure to pesticides is likely. Non-agricultural uses of malathion (particular for mosquito control) across the various land uses is also expected and may pose significant risk to LCR steelhead.

This DPS includes winter- and summer-run types. Summer-run steelhead return to freshwater from May to 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 select off-channel habitats associated with their natal rivers and streams. Juveniles rear in the freshwater habitats for more than a year. Given this duration, juveniles are most likely to experience higher pesticide exposure during their first year of rearing. Where both runs spawn in the same stream, summer-run steelhead tend to spawn at higher elevations than the winter-run forms.

Given the life history of LCR steelhead, we expect the proposed uses of chlorpyrifos, diazinon, and malathion pesticide products that contaminate aquatic habitats may lead to both individual fitness level consequences and subsequent population level consequences, i.e., reductions in population viability. The widespread uses of these materials indicate substantial overlap with the 23 populations that comprise the LCR steelhead. The risk to this species' survival and recovery from the stressors of the action is high.

Middle Columbia River Steelhead

The Middle Columbia River (MCR) steelhead DPS includes four major population groups with 17 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 et al. 1996) where as 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. The annual population growth rate known for 11 independent populations ranged from 0.97 to 1.02. Two historical populations are considered extinct.

The major threats to this ESU identified in the *Status of Listed Resources* and *Environmental Baseline* sections include barriers preventing steelhead migration above dams and fish mortalities from the Columbia River hydroelectric system. Additionally, agricultural practices, especially grazing, water diversions, and withdrawals negatively affect this DPS. Within the various types of habitat threats, elevated temperature, water diversion, and poor water quality from contaminants are significant influences on the status of MCR steelhead. Roughly 3,905 km of the ESU's habitat in the states of Oregon (3,519 km) and Washington (386 km) are listed as 303(d) waters for elevated temperature. In the Yakima River, 72 streams and river segments are listed as impaired waters and 83% exceed temperature standards. Within the Yakima River Basin, 76 pesticide compounds were detected. They include 38 herbicides, 17 insecticides (such as carbaryl, diazinon, chlorpyrifos, and malathion), 15 breakdown products, and 6 others. The median and maximum numbers of chemicals in a mixture were six and eight, respectively (Fuhrer et al. 2004). Atrazine was the most frequently detected pesticide in agricultural streams. Acetylcholinesterase-inhibiting pesticides were also detected. They include azinphos-methyl, dimethoate, ethoprop, disulfuton, aldicarb, aldicar sulfone, and carbaryl. In the Granger drainage of the lower Yakima, atrazine, simazine, chlorpyrifos, diazinon, and malathion were also detected. The co-occurrence of atrazine with chlorpyrifos, diazinon, malathion, and other OPs in aquatic habitats increases the likelihood of adverse responses in salmonids and their aquatic prey. The cumulative impacts from these multiple threats continue to affect MCR steelhead.

Registered uses of chlorpyrifos, diazinon, and malathion include applications to crop agricultural sites, non-crop agricultural sites, residential sites, commercial sites, and animal and structural treatments. Some of these uses allow applications at rates that exceed those allowed in agricultural crops. Land use data (Table 32) indicate the majority of the ESU's landscape is covered by evergreen forest (55%), shrub/scrub (12%), mixed forest (5%), and hay/pasture (5%). About 11% of the land has been developed and 3% has been cultivated for crops. We expect application of the three active ingredients for a variety of pest control purposes across multiple land uses within MCR steelhead habitat. Thus, ESU exposure to pesticides is likely. Non-agricultural uses of malathion (particularly for mosquito control) across the various land uses is also expected and may pose significant risk to MCR steelhead.

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.

Given the life history of MCR steelhead, we expect the proposed uses of chlorpyrifos, diazinon, and malathion pesticide products may lead to both individual fitness consequences to juveniles and adults and subsequent population level consequences, i.e., reductions in population viability. The widespread uses of these materials indicate substantial overlap with the 17 populations that comprise the MCR steelhead. The risk to this species' survival and recovery from the stressors of the action is high.

Northern California Steelhead

The Northern California (NC) steelhead DPS includes steelhead in 10 larger and several small coastal river basins in Humboldt and Mendocino Counties, California. Historical production estimates for the DPS imply that annual production may have been up towards 200,000 fish. Both long-term estimates and recent snapshot surveys are lacking for most of the streams within the DPS. However, information from the Eel River indicates that the species has declined drastically since the 1960s. Estimates of population growth rate calculated for a sub-population of summer-run steelhead in the Middle Fork Eel River indicated that the population generally had a negative annual escapement during the period from 1966 through 2002. Though hatchery operations have occurred within the DPS, little is known of its contribution to the natural spawning population.

The major threats to this DPS identified in the *Status of Listed Resources* and *Environmental Baseline* sections indicate that impaired or loss of habitat, road crossings and other migration barriers, introduced fish species, timber harvest and agricultural activities, especially vineyard development, negatively affect this DPS. Stressors to the NC steelhead consist of a high percentage of fines in the streams' bottom substrate, lack of large instream woody debris, reduced riparian vegetation, elevated water temperature, increased predation, and barriers that limit access to tributaries. Pesticides used in forestry, vineyards and other agricultural activities likely enter streams where they impair water quality in the aquatic environment for salmonids. The Mediterranean climate in California, with dry summers and wet winters, may result in high concentration of contaminants in run-off during the onset of the rainy season. The cumulative impacts from these multiple threats continue to affect the NC steelhead.

The three active ingredients are applied for a variety of pest control purposes that span multiple land uses within the NC steelhead DPS. Less than one percent of the DPS is developed for cultivation of crops. However, high density of vineyards exists in Mendocino County. Crop farming is concentrated in low laying areas and floodplains along the estuaries and lower reaches of streams. About 60% of the DPS consists of coniferous forests and forestry activities occur throughout the DPS (Table 26). All three active ingredients are expected to be used for several crops and some as part of forestry activities. Several coastal communities are located at the mouth of streams within the NC steelhead DPS. Non-agricultural uses of malathion (particularly for mosquito control) across the various land uses are expected and can pose significant risk to the NC steelhead.

Most of the steelhead populations within this DPS are of the winter-type life history. However, four rivers are known to support small runs of summer type steelhead: Redwood Creek, Mad River, the Middle Fork Eel River, and Matole River. The winter-type enters rivers as mature adults from November through April to spawn while the summer-type enters the stream in immature condition in spring and summer. The summer-type then holds in deep pools at higher altitudes of the rivers throughout the summer. They can hold in the pools for as long as 6-8 months before moving into natal reaches to spawn. Juvenile steelhead remain in freshwater for one or more years before migrating downstream to smolt. In many streams, steelhead often rear for a shorter or longer period in the estuary before fully entering the marine environment. California steelhead typically spends one or two years in the ocean. However, in many populations, a small fraction of fish will spend a third year at sea. A fraction of the adults, especially females, survives the spawning season to spawn a second, third, or even a fourth time.

Given the long freshwater residence time by steelhead juveniles, more than a year, and the presence of agricultural and forestry activities within watersheds used by the species, we expect the proposed uses of chlorpyrifos, diazinon, and malathion pesticides products that contaminate aquatic habitat will lead to both individual fitness level consequences and subsequent population level consequences, i.e., reductions in population viability. The widespread uses of these materials indicate substantial overlap with the populations that comprises the NC steelhead DPS. The risk to this species' survival and recovery from the stressors of the action is high.

Puget Sound Steelhead

The Puget Sound steelhead is comprised of 21 populations. Of these, 17 had declining trends and four had increasing trends for the late-run naturally produced component of the winter-run steelhead populations. No adult trend data were available for summer-run steelhead. No estimates of historical (pre-1960s) abundance specific to the Puget Sound steelhead ESU are available. Total run size for Puget Sound steelhead in the early 1980s is approximately 100,000 winter-run steelhead and 20,000 summer-run steelhead.

This DPS has two life history types: summer and winter steelhead. Steelhead that enter freshwater between May and October are considered summer steelhead. Meanwhile, steelhead that enter freshwater between November and April are considered winter steelhead. Mature adults (three to five years old) may enter rivers any month of the year and spawn in late winter or spring. Adults 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 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 as well. Afterwards, they smolt and migrate to sea in the spring.

The major threats to this ESU identified in the *Status of Listed Resources* and *Environmental Baseline* sections indicate habitat degradation from logging, road construction, urban development, mining, agriculture, and recreation; water diversions; and poor water quality negatively affect this ESU. Within the various types of habitat threats, elevated temperature, water diversions, and poor water quality pose significant influences on the status of Puget Sound steelhead. About 705 km of this ESU's habitat in the state of Washington are identified as 303(d) impaired waters due to elevated temperature. Pesticide use and detections in the ESU's watershed are well documented. 2006 NAWQA sampling in the Puget Sound basin detected 26 pesticides and 74 manmade 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).

A unique feature of Puget Sound is its fjord-like features and hydrological isolation from ocean water entry. As a result, toxic chemicals entering Puget Sound have longer residence times within the system and create elevated contaminant levels in the aquatic environment. Because Puget Sound is a deep, almost oceanic habitat, the tendency of a number of species to migrate outside Puget Sound is limited, relative to similar species in other urban estuaries. The 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 estuarine ecosystems (Collier et al. 2006). The pelagic food web and fish in Puget Sound are therefore exposed to increased levels of contaminants. Fall Chinook salmon from Puget Sound have been found to be more contaminated with PCBs (two to six times) and PBDEs (five to 17 times) compared to fish elsewhere. These contaminated conditions may lead to increased susceptibility of juvenile salmon to disease, acute die-offs of salmon returning to spawn in urban watersheds, and egg and larval mortality in a variety of fish. The contaminated nature of Puget Sound and the cumulative impacts from the above threats continue to affect Puget Sound steelhead. The cumulative impacts from the above multiple threats continue to affect Puget Sound steelhead.

Registered uses of chlorpyrifos, diazinon, and malathion include applications to crop agricultural sites, non-crop agricultural sites, residential sites, commercial sites, and animal and structural treatments. Some of these uses allow applications at rates that exceed those allowed in agricultural crops. Land use data (Table 32) indicate that Puget Sound steelhead may be exposed to chlorpyrifos, diazinon, and malathion. The majority of the ESU landscape is covered by evergreen forest (49%), mixed forest (9%), shrub/scrub (8%), and hay/pasture (4%). About 14% of the land has been developed and less than 1% has been cultivated for crops. Additionally, 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. We expect application of the three active ingredients for pest control purposes across multiple land uses within Puget Sound steelhead habitat. Thus, ESU exposure to pesticides is likely. Non-agricultural uses of malathion (particular for mosquito control) across the various land uses is also expected and may pose significant risk to Puget Sound steelhead.

Given the life history of Puget Sound steelhead, we expect the proposed uses of chlorpyrifos, diazinon, and malathion pesticide products that contaminate aquatic habitats may lead to both individual fitness level consequences and subsequent population level consequences, i.e., reductions in population viability. The widespread uses of these materials indicate substantial overlap with the 21 populations that comprise the Puget Sound steelhead. The risk to this species' survival and recovery from the stressors of the action is high.

Snake River Basin Steelhead

The Snake River Basin (SRB) steelhead includes all naturally spawned populations of steelhead (and their progeny) in streams in the Snake River Basin of Idaho, northeast Oregon, and southeast Washington. This DPS is comprised of 23 populations in six major population groups; it excludes resident forms of *O. mykiss* (rainbow trout) co-occurring with these steelhead. SRB steelhead remain spatially well distributed in each of the six major geographic areas in the Snake River basin (Good et al. 2005). 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. Regarding annual population growth rate, there are mixed long- and short-term trends in abundance and productivity. The annual population growth rate known for eight independent populations ranged from 0.89 to 1.08. One historical population is likely extirpated.

The major threats to this ESU identified in the *Status of Listed Resources* and *Environmental Baseline* sections include hydrosystem mortality, reduced stream flow, excessive sediment, and degraded water quality. Within the various types of habitat threats, elevated temperature, water diversions, and poor water quality pose significant influences on the status of SRB steelhead. Collectively, 1,975 km of the ESU's habitat in the states of Idaho (738 km), Oregon (991 km), and Washington (247km) are listed as 303(d) waters for elevated temperature. Pesticide use and detections in SRB steelhead freshwater habitats are well documented. NAWQA sampling in 1992-1995 in the ESU's watershed detected Eptam, atrazine, desethylatrazine, metolachlor, and alachlor. The cumulative impacts from these multiple threats continue to affect SNB steelhead.

Registered uses of chlorpyrifos, diazinon, and malathion include applications to crop agricultural sites, non-crop agricultural sites, residential sites, commercial sites, and animal and structural treatments. Some of these uses allow applications at rates that exceed those allowed in agricultural crops. Land use data (Table 32) indicate the majority of the ESU's landscape is covered by evergreen forest (52%), shrub/scrub (21%), and herbaceous cover (16%). About 1% of the land has been developed and 8% has been cultivated for crops. We expect application of the three active ingredients for a variety of pest control purposes across multiple land uses within SRB steelhead habitat. Thus, ESU exposure to pesticides is likely. Non-agricultural uses of malathion (particularly for mosquito control) across the various land uses is also expected and may pose significant risk to SRB steelhead.

Sexually immature adult Snake River summer steelheads enter the Columbia River from late June to October. Snake River 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 migrate upriver until they reach tributaries from 1,000 to 2,000 meters above sea level where they spawn between March and May of the following year. Emergence occurs by early June from low elevation streams and as late mid-July at higher elevations. After hatching, juvenile Snake River steelhead typically select off-channel habitats associated with their natal rivers and streams. They spend two to three years in freshwater before they smolt and migrate to the ocean. Given their residency time in shallow freshwater habitats, juveniles likely experience higher exposure to pesticides, other contaminants, and elevated temperature.

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 extends up to 900 miles from the ocean in the Snake River. Spawning usually occurs 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 stream margins. Steelhead rear in a wide variety of freshwater habitats, generally for two to three years, and up to six or seven years is possible. They smolt and migrate to sea in the spring. Some adult steelhead survive spawning, return to the sea, and later return to spawn a second time in river tributaries.

Given the life history of SRB Steelhead, we expect the proposed uses of chlorpyrifos, diazinon, and malathion pesticide products that contaminate aquatic habitats may lead to both individual fitness level consequences and subsequent population level consequences i.e., reductions in population viability. The widespread uses of these materials indicate substantial overlap with the populations that comprise SRB steelhead. The risk to this species' survival and recovery from the stressors of the action is high.

South-Central California Coast Steelhead

The South-Central California Coast (S-CCC) steelhead DPS includes all naturally spawned steelhead in streams from the Pajaro River to, but not including, the Santa Maria River, California. Runs have been lost in many streams within the DPS' range. Historic adult abundance estimates for the DPS imply an annual return may have been up towards 20,000 fish. Current estimates have not been made for the DPS but estimated production in five of the major rivers indicates a return of less than 500 adults. During the years from 1963 to 1993, annual return in Carmel River decreased with an average of 22% per year. In the early 1990s returns increased from one in 1991 to several hundred. This may indicate improved conditions in this river. Though hatchery operations have occurred within the DPS, little is known of its contribution to the natural spawning population.

The major threats to this DPS identified in the *Status of Listed Resources* and *Environmental Baseline* sections indicate that dams and other migration barriers, urbanization and channel modification, agricultural activities, and wildfires negatively affect this DPS. Because of the activities and disturbances occurring within watersheds, the stream substrate contains a high proportion of fines, stream channels lack complexity, banks are eroding, the water is turbid and contains contaminants, and migration barriers restrict fish access to cooler head waters. The southern distribution of the S-CCC steelhead along the Pacific coast naturally exposes the species to stressors such as high water temperature and low flows during summer. The cumulative impacts from these multiple threats continue to affect the S-CCC steelhead.

The three active ingredients are applied for a variety of pest control purposes that span multiple land uses within the S-CCC steelhead DPS. About 7% of the DPS is developed for cultivation of crops, and about 8% of the area within the DPS consists of developed land (Table 26). Crops are concentrated in low laying areas and floodplains along the estuaries and lower reaches of streams. Developed areas occur in many places located at the mouth of streams within the S-CCC steelhead DPS. All three active ingredients are expected to be used for several crops. Non-agricultural uses of malathion (particularly for mosquito control) across the various land uses are expected and may pose significant risk to the S-CCC steelhead. A considerable concern is drift and runoff of the three pesticide products into steelhead habitat during aerial application.

All of the steelhead populations within this DPS are of the winter-type life history. The winter-type enters rivers as mature adults in fall and winter to spawn. Stream entry is highly dependent on rainfall. However, detailed information about the S-CCC steelhead life history is not available. Juvenile steelhead remain in freshwater for one or more years before migrating downstream to smolt. They often remain in estuaries for a longer period before entering the marine environment. The S-CCC steelhead has adapted to the warmer climate and can withstand higher temperatures than northern populations.

Given the long freshwater residence time by steelhead juveniles, and the relatively high urbanization and presence of agriculture within watersheds used by the species, we expect the proposed uses of chlorpyrifos, diazinon, and malathion pesticides products that contaminate aquatic habitat will lead to both individual fitness level consequences and subsequent population level consequences, i.e., reductions in population viability. The widespread uses of these materials indicate substantial overlap with the populations that comprises the S-CCC steelhead DPS. Given the low abundance of the DPS, the extensive habitat modification and loss that has occurred, and the high water temperatures, the risk to this species' survival and recovery from the stressors of the proposed action is high.

Southern California Steelhead

The Southern California (SC) steelhead DPS includes steelhead 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. Historical production estimates for the DPS imply that annual production may have been up toward 50,000 fish. 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. Of concern is the lack of information to assess population structure within the DPS. Although hatchery operations have occurred within the DPS, little is known of its contribution to the natural spawning population.

The major threats to this DPS identified in the *Status of Listed Resources* and *Environmental Baseline* sections indicate that dams and other migration barriers, urbanization and channel modification, agricultural activities and wildfires negatively affect this DPS. 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. The NAWQA analysis detected more than 5 pesticides in ground and surface waters within the heavily populated Santa Ana basin, including multiple acetylcholinesterase inhibitors. The SC steelhead represents the southern range of steelhead along the Pacific coast. As such, it is naturally exposed to stressors such as high water temperatures and low flows during summer.

The three active ingredients are applied for a variety of pest control purposes that span multiple land uses within the SC steelhead DPS. About 7% of the DPS is developed for cultivation of crops and about 13% is developed land of which 3% of the area within the DPS consists of medium to high intensity developed land (Table 26). Farming is concentrated in low laying areas and floodplains along the estuaries and lower reaches of streams. Developed areas in many places are located at the mouth of streams within the SC steelhead DPS. All three active ingredients are expected to be used for several crops. Non-agricultural uses of malathion (particularly for mosquito control) across the various land uses are expected and may pose significant risk to the SC steelhead. A considerable concern is drift and runoff of the three pesticide products into steelhead habitat during aerial application.

All of the steelhead populations within this DPS are of the winter-type life history. The winter-type enters rivers as mature adults in fall and winter to spawn. Stream entry is highly dependent on rainfall. However, detailed information about the SC steelhead life history is not available. Juvenile steelhead remain in freshwater for one or more years before migrating downstream to smolt. They often remain in estuaries for a longer period before fully entering the marine environment. The SC steelhead has adapted to the warmer climate and can withstand higher temperatures than northern populations. Studies of steelhead in Ventura River found that the population has a relative high growth rate during freshwater rearing consistent with rearing in warmer water.

Given the long freshwater residence time by steelhead juveniles, and the relatively high urbanization and presence of agriculture within watersheds used by the species, we expect the proposed uses of chlorpyrifos, diazinon, and malathion pesticides products that contaminate aquatic habitat will lead to both individual fitness level consequences and subsequent population level consequences, i.e., reductions in population viability. The widespread uses of these materials indicate substantial overlap with the populations that comprises the SC steelhead DPS. Given the low abundance of the DPS, the extensive habitat modification and loss that has occurred, presence of contaminants in surface waters, and the high water temperature, the risk to this species' survival and recovery from the stressors of the proposed action is high.

Upper Columbia River Steelhead

The Upper Columbia River (UCR) steelhead DPS includes all populations that spawn and rear in the middle reaches of the rivers and tributaries draining the eastern slope of the Cascade Mountains upstream of Rock Island Dam. The DPS is comprised of a single major population group with five naturally-spawned populations, and six artificial programs in the state of Washington. For all naturally spawned populations, abundance over the most recent 10-year period is below identified thresholds as a minimum for recovery. Returns of both hatchery and naturally produced steelhead in the upper Columbia River have increased in recent years. The average 1997 to 2001 return was about 12,900 fish. On average, from 1980 through 2000, including adult returns through 2004 – 2005, UCR steelhead populations have not replaced themselves. Regarding the population growth rate of natural production, on average, UCR steelhead populations have not replaced themselves over the past 25 years. Overall adult returns are dominated by hatchery fish. Detailed information is lacking on the productivity of the natural populations. The annual population growth rate known for two independent populations ranged from 1.067 to 1.086. All UCR steelhead populations have reduced genetic diversity from homogenization of populations during the Grand Coulee Fish Maintenance project from 1939-1943, from 1960, and 1981 (Chapman et al. 1994).

The major threats to this ESU identified in the *Status of Listed Resources* and *Environmental Baseline* sections include dams that block fish migration and alter river hydrology. Additionally, water diversions that deplete water levels in rivers and streams; destruction or degradation of riparian habitat; and land use practices that destroy wetland and riparian ecosystems (logging, agriculture, urbanization) negatively affect this ESU. Elevated water temperature, water diversion, and poor water quality pose significant threats to the status of UCR steelhead. About 282 km of the ESU's habitat in the state of Washington are listed as 303(d) waters for elevated temperature. Pesticide use and detections in UCR steelhead freshwater habitats are well documented. Within the Yakima River Basin, 76 pesticide compounds were detected. They include 38 herbicides, 17 insecticides (such as carbaryl, diazinon, and malathion), 15 breakdown products, and 6 others. The median and maximum numbers of chemicals in a mixture were eight and six, respectively (Fuhrer et al. 2004). Atrazine was the most frequently detected pesticide in agricultural streams. Cholinesterase-inhibiting pesticides were also detected. They include azinphos-methyl, dimethoate, ethoprop, disulfuton, aldicarb, aldicar sulfone, and carbaryl. In the Granger drainage of the lower Yakima, atrazine, simazine, chlorpyrifos, diazinon, and malathion were also detected. The co-occurrence of atrazine with chlorpyrifos, diazinon, malathion, and other OPs in aquatic habitats increases the likelihood of adverse responses in salmonids and their aquatic prey. The cumulative impacts from these multiple threats continue to affect UCR steelhead.

Registered uses of chlorpyrifos, diazinon, and malathion include applications to crop agricultural sites, non-crop agricultural sites, residential sites, commercial sites, and animal and structural treatments. Some of these uses allow applications at rates that exceed those allowed in agricultural crops. Land use data (Table 32) indicate that UCR steelhead may be exposed to chlorpyrifos, diazinon, and malathion. The majority of the ESU landscape is covered by shrub/scrub (38%), evergreen forest (34%), and herbaceous cover (8%). About 3% of the land has been developed and 13% has been cultivated for crops. We expect application of the three active ingredients for a variety of pest control purposes across multiple land uses within UCR steelhead habitat. Thus, ESU exposure to pesticides is likely. Non-agricultural uses of malathion (particularly for mosquito control) across the various land uses is also expected and may pose significant risk to UCR steelhead.

UCR 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. 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. Fry move downstream in the fall in search of suitable overwintering habitat (Chapman et al. 1994). Larger juvenile life stages use progressively deeper and faster water, sheltering behind boulders in the highest gradient riffles and cascades. 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. Smolt outmigrations are predominantly age-two and age-three juveniles. Given the long duration in shallow freshwater habitats, juveniles are more likely to experience higher pesticide exposure, contaminants, and elevated temperature. Most adult steelhead return after one or two years at sea to start the cycle again.

Given the life history of UCR steelhead, we expect that the proposed uses of chlorpyrifos, diazinon, and malathion pesticide products that contaminate aquatic habitats will lead to both individual fitness consequences and subsequent population level consequences, i.e., reductions in population viability. The widespread uses of these materials indicate substantial overlap with the four populations that comprise the UCR steelhead. The risk to this species' survival and recovery from the stressors of the action is high.

Upper Willamette River Steelhead

The Upper Willamette River (UWR) steelhead DPS includes all naturally spawned steelhead populations below natural and manmade impassable barriers in the Willamette River, Oregon, and its tributaries upstream from Willamette Falls to the Calapooia River. The DPS is comprised of a single major population group with four historical populations. These populations remain extant and produce moderate numbers of natural-origin steelhead each year. Steelhead in this DPS are depressed from historical levels. Native winter-run steelhead 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 1. Conversely, short-term trends are 1 or higher. The annual population growth rate of the four independent populations ranged from 0.97 to 1.023. The long-term risk of extinction is considered moderate for all four populations.

The major threats to this ESU identified in the *Status of Listed Resources* and *Environmental Baseline* sections include hydroelectric dams that block fish migration and alter river hydrology. Additionally, water diversions that deplete water levels in rivers and streams; destruction or degradation of riparian habitat; and land use practices (logging, agriculture, urbanization) that degrade water quality and destroy wetland and riparian ecosystems negatively affect this ESU. Within the various types of habitat threats, elevated temperature, water diversion, and poor water quality pose significant influences on the status of UWR steelhead. About 1,668 km of the ESU's habitat in the state of Oregon are listed as 303(d) waters for elevated temperature. Additionally, pesticide use and detections in UWR steelhead freshwater habitats are well documented. Wentz et al. (1998) reported that 50 pesticides were detected in streams and 10 pesticides exceeded EPA criteria for the protection of freshwater aquatic life for chronic toxicity. Forty-nine pesticides were detected in streams draining predominantly agricultural land. About 25 pesticides were also detected in streams draining mostly urban areas. The highest pesticide concentrations generally occurred in streams draining agricultural land. The cumulative impacts from these multiple threats continue to affect UWR steelhead.

Registered uses of chlorpyrifos, diazinon, and malathion include applications to crop agricultural sites, non-crop agricultural sites, residential sites, commercial sites, and animal and structural treatments. Some of these uses allow applications at rates that exceed those allowed in agricultural crops. Land use data (Table 32) indicate the majority of the ESU's landscape is covered by evergreen forest (33%), hay/pasture (20%), and shrub/scrub (8%). About 10% of the land has been developed and 15% has been cultivated for crops. We expect application of the three active ingredients across span multiple land uses within UWR steelhead habitat. Thus, ESU exposure to pesticides is likely. Non-agricultural uses of malathion (mosquito control) across the various land uses is also expected and may pose significant risk to UWR steelhead.

Upper Willamette winter-run Steelhead enter the Willamette River in January and February. They ascend to their spawning areas in late March or April. Spawning occurs from April to June. Steelhead fry typically select off-channel habitats associated with their natal rivers and stream. Smolt migration past Willamette Falls begins in early April and extends through early June, with peak migration in early to mid-May. Steelhead smolts migrate away from the shoreline and enter the Columbia via Multnomah Channel. Most spend two years in the ocean before re-entering freshwater to spawn. Steelhead in this DPS generally spawn once or twice. Repeat spawners are predominantly female and account for less than 10% of the total run size.

Given the life history of UWR steelhead, we expect the proposed uses of chlorpyrifos, diazinon, and malathion pesticide products that contaminate aquatic habitats will lead to both individual fitness level consequences and subsequent population level consequences, i.e., reductions in population viability. The widespread uses of these materials indicate substantial overlap with the independent populations that comprise the UWR steelhead. The risk to this species' survival and recovery from the stressors of the action is high.

Summary of Species-Level Effects

In the preceding section NMFS described expected population level effects in terms of reductions in annual growth rate as well as reductions in productivity (reproduction) and abundance (numbers of salmonids). We concluded that all but Ozette sockeye populations will likely show reductions in viability. The effects of EPA's proposed action are first manifested at the individual level where reductions in individual fitness is expected. We showed that an individual's survival, reproduction, 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 OPs and carbamate insecticides, and elevated temperatures in the action area. The latter is expected to increase range–wide if global climate change intensifies as predicted.

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, Columbia River chum salmon, Hood Canal summer run chum salmon, Central California Coast coho salmon, Southern Oregon and Northern Coastal California coho salmon, Oregon Coast coho salmon, Snake River sockeye salmon, Central California Coast steelhead, California Central Valley steelhead, LCR steelhead, MCR steelhead, Northern California 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 ESU will not likely experience reduction in viability.

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 and effects to these PCEs are analyzed under *Effects to Designated Critical Habitat* Section. 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. Based on our *Effects Analysis*, the proposed action will affect freshwater rearing, spawning, migration, and foraging areas, and the PCEs that these habitat types provide listed salmon and steelhead. Of particular concern is the effects of EPA's proposed registration of chlorpyrifos, malathion, and diazinon on salmonid prey and water quality in these areas.

Direct exposure to chlorpyrifos, malathion, diazinon and the other chemical stressors of the action within freshwater or the riparian zone within will have an effect on Pacific salmon or steelhead critical habitat. As noted in the *Effects Analysis*, pesticides most often occur in the aquatic environment as mixtures. Chlorpyrifos, diazinon, and malathion are among the most common insecticides found in 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. Consequently, salmonid growth may be affected by the reduced ration available in addition to being directly affected due to AChE inhibition. Smaller fish are further susceptible to larger predators, dietary and energetic stress, which may ultimately affect individual reproductive success and survival.

Additionally, in areas of intensive urban and agricultural land uses, runoff will likely contain other pesticides, chemical pollutants, and sediments that also degrade water quality. Depending on the available water flow, amount of shade from large woody debris, and water temperature in aquatic habitats, the toxicity of chlorpyrifos, diazinon, and malathion in tributary and stream waters may become more pronounced. These overall reductions in water quality will reduce areas available for spawning, rearing, migrating and foraging 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, Hood Canal summer run chum salmon, Central California Coast coho salmon, Southern Oregon and Northern Coastal California coho salmon, Oregon Coast coho salmon, Snake River sockeye salmon, Central California Coast steelhead, California Central Valley steelhead, LCR steelhead, MCR steelhead, Northern California steelhead, Snake River Basin steelhead, South-Central California coast steelhead, Southern California steelhead, UCR steelhead, and Upper Willamette River steelhead. 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 the critical habitat designated for these species, exceptfor that of Ozette Lake sockeye ESU. The duration, frequency, and severity of these reductions will vary according to overall numbers and volume of applications of chlorpyrifos, diazinon, and malathion in areas of designated critical habitat, among other variables.

Conclusion

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, Columbia River chum salmon, Hood Canal summer run chum salmon, Central California Coast coho salmon, LCR coho salmon, Southern Oregon and Northern Coastal California coho salmon, Oregon Coast coho salmon, , Snake River sockeye salmon, California Central Valley steelhead, Central California Coast steelhead, LCR steelhead, MCR steelhead, Northern California 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 project, as proposed, is likely to jeopardize the continued existence of these endangered or threatened species.

It is NMFS' Opinion that the project, as proposed, is not likely to jeopardize the continued existence of Ozette Lake sockeye salmon.

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, Hood Canal summer run chum salmon, Central California Coast coho salmon, Southern Oregon/Northern Coastal California coho salmon, Oregon Coast coho, Snake River sockeye salmon, California Central Valley steelhead, Central California Coast steelhead, LCR steelhead, MCR steelhead, Northern California 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 project, as proposed, is likely to result in the destruction or adverse modification of critical habitat of these endangered and threatened species.

It is NMFS' Opinion that the project, as proposed, is not likely to result in the destruction or adverse modification of critical habitat of Ozette Lake sockeye salmon.

Reasonable and Prudent Alternative

This Opinion has concluded that EPA's proposed registration of pesticides containing chlorpyrifos, diazinon, and malathion is likely to jeopardize the continued existence of 27 endangered and threatened Pacific salmonids and is likely to destroy or adversely modify designated critical habitat for 25 threatened and endangered salmonids. The clause "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 result in the destruction or adverse modification of critical habitat.

NMFS reached this conclusion because measured and predicted concentrations of the three active ingredients in salmonid habitats, particularly in off-channel habitats, are likely to cause adverse effects to listed species including significant reductions in survival, reproduction, migration, and growth. Further, all but one population of listed Pacific salmonids are likely to suffer reductions in viability given the severity of expected changes in abundance and productivity associated with the proposed action. These adverse effects are expected to appreciably reduce the likelihood of both the survival and recovery of the listed Pacific salmonids. EPA's proposed registration of chlorpyrifos, diazinon, and malathion is likely to result in the destruction or adverse modification of critical habitat of these endangered and threatened species because of adverse effects on salmonid prey and water quality in freshwater rearing, spawning, migration, 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 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 chlorpyrifos, diazinon, and malathion and other cholinesterase-inhibiting compounds result in additive and synergistic responses; (3) exposure to other chemicals and physical stressors (e.g., temperature) in the baseline habitat will likely intensify response to chlorpyrifos, diazinon, and malathion.

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 is not attempting to ensure that 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, as long as the action is conducted according to the RPA and reasonable and prudent measures (RPM). Avoiding take would most likely entail cancelling registration, or prohibiting use in watersheds inhabitated by salmonids. 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 the Opinion to ensure that the proposed registration of these pesticides is not likely to jeopardize endangered or threatened species under the jurisdiction of NMFS or destroy or adversely modify critical habitat that has been designated for these species. 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 1-5 shall be specified on FIFRA labels of all pesticide products containing chlorpyrifos, diazinon, and malathion used in California, Idaho, Oregon and Washington. Alternatively, the label could direct pesticide users to the EPA Endangered Species Protection Program's bulletins that specify elements 1-5.

Element 1. Apply the following no-application buffers/setbacks (buffers):
<u>A. Where ground applications are permitted.</u> Do not apply pesticide products[13] within 500 ft (152. 4 m) of salmonid habitats[14].
<u>B. Where aerial applications are permitted.</u> Do not apply pesticide products within 1,000 ft (304.8 m) of salmonid habitats.

<u>Rationale:</u>

1). Use of buffers in other programs.
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 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 chlorpyrifos containing end-use products have mandated buffers of 25, 50, and 100 ft for ground, airblast, and aerial applications, respectively. Malathion containing pesticides have mandated buffers for aerial applications of 25 and

[13] Use of the term "pesticide products" in the Reasonable and Prudent Alternative section of the Opinion refers to pesticide products containing chlorpyrifos, diazinon, or malathion.

[14] 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.

50 ft, for non-ULV and ULV, respectively (RED). CDPR has pesticide use limitations of 120 and 600 ft buffers for chlorpyrifos, diazinon, and malathion-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 active ingredients. 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.

2). AgDrift modeling results for ground and aerial applications into off-channel habitats. NMFS generated estimated environmental concentrations for the three OPs 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. The estimated concentrations decline as buffer size increases (Table 66). We note the disparity between the concentrations predicted at the 500 ft ground application buffer versus the 1,000 ft aerial buffer. The two results are not directly comparable because the models use different methods to predict amount of drift. Additionally, the buffer for ground applications addresses both drift and potential runoff, where as the aerial buffer applies primarily to drift as runoff is expected to be minimal relative to drift at 1,000 ft.

Table 66. Estimated environmental concentrations of chlorpyrifos, diazinon, and malathion applied at the rate or 1lb per acre for ground and aerial applications.

Ground application, low boom, ASAE very fine-fine droplet distribution, 50th percentile estimates. EPA Tier 1 simulation	
Buffer	Off-Channel (10 m * 0.1 m)
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	Off-Channel (10 m * 0.1 m)
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

3). Comparisons of estimated concentrations from AgDrift model runs with biological effects information presented in the Opinion.

With a 1,000 ft buffer, an aerial application of 1 lb/acre resulted in a pesticide concentration of approximately 13 ug/L in an off-channel habitat 10 m wide, 0.1 m deep. If juvenile salmonids were present, we would expect mortalities for each of the OPs, with the greatest number of mortalities for chlorpyrifos-exposed fish. We would also expect other non-lethal fish endpoints to be affected. Salmonid prey items would be severely affected by these concentrations. With a 500 ft buffer, a ground application of 1 lb/acre resulted in a predicted pesticide concentration of approximately 1.12 ug/L in off-channel habitats. Some juvenile salmonids would die from this exposure and other sub-lethal effects would also be expected. Sensitive salmonid prey items would also be adversely affected at 1.12 ug/L.

The majority of buffers described earlier are smaller than the 500 ft (ground applications) and 1,000 ft (aerial applications) buffers and for this action would result in substantially greater risk to salmonids and salmonid prey items. For example, a 10 ft buffer for a common application rate of 1 lb/acre would result in an estimated concentration of 20 ug/L for a ground application; a value that is 20 times higher than the concentration predicted at 500 ft. For an aerial application, a 300 ft buffer would result in a pesticide concentration of 33 ug/L (approximately three times higher than a concentration at 1,000 ft).

While the concentrations predicted by the modeling could result in unknown numbers of lethal and non-lethal takes of salmonids as well as reduction in prey, NMFS believes that even with the selected buffers most pesticide applications will not result in these estimated concentrations. Several factors must be weighed when using these model estimates to describe the relative risk to salmonid habitats. First, these estimates are generated for a level field with wind blowing directly toward aquatic habitats and with no interception of pesticide drift by riparian or other vegetation. Many agricultural fields are not flat and wind may change directions quickly or may not be blowing directly into salmonid habitats. Second, many aquatic habitats are flowing and are much larger than the off-channel habitat modeled in Table 66. Third, the model's predictive capabilities become less certain as buffer size increases (Bird et al. 2002).

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, however the frequency of this exact scenario occurring remains unknown. We selected this scenario to represent off-channel habitats utilized by a sensitive salmonid lifestage 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. When applying pesticide products, commence applications on the side nearest the aquatic habitat and proceed away from the aquatic habitat.

Element 3. For agricultural uses, provide a 20 ft (6.1 m) minimum strip of non-crop vegetation (on which no pesticides shall be applied) on the downhill side of the application site immediately adjacent to any surface waters that have a connection to salmonid-bearing waters. This includes drainage systems that have salmonid exclusion devices, but drain to salmonid-bearing waters.

Element 4. 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 hours following application.

Element 5. Report all incidents of fish mortality that occur within four days of application and within the vicinity of the treatment area to EPA Office of Pesticide Programs (703-305-7695).

Element 6. 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, DPS) i.e., coho salmon, chum salmon, steelhead, sockeye salmon, and ocean-type Chinook and stream-type Chinook salmon. Additionally, each state shall have at least three sites within their borders. One site in each state shall target where juvenile ESA-listed salmonids migrate to the Pacific Ocean. 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.

Although NMFS has concluded that EPA's action is likely to jeopardize 27 listed ESUs and destroy or adversely modify 25 designated critical habitats, NMFS does not believe that these effects will occur in the year between issuance of this Opinion and EPA's implementation of the RPA. Products containing these three active ingredients have been in use for some time. NMFS believes that these products have contributed to ESU declines, but not to the extent that one year of additional use as now authorized would lead to likely jeopardy or adverse modification.

Because this Opinion has concluded that the EPA's proposed registration of pesticides containing chlorpyrifos, diazinon, and malathion is likely to jeopardize the continued existence of 27 endangered and threatened Pacific salmonids under the jurisdiction of the NMFS and is likely to result in the destruction or adverse modification of designated critical habitat for 25 threatened and endangered salmonids,the EPA is required to notify NMFS OPR of its final decision on implementation of the reasonable and prudent alternative.

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 chlorpyrifos, diazinon, and malathion, and their formulations as they are used in the Pacific Northwest and the impacts of these applications on listed ESUs 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, 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 the EPA's registration of pesticide products across the states of California, Idaho, Oregon, and Washington to 28 listed Pacific salmonids under NMFS' jurisdiction. Recent and historical surveys indicate that listed salmonids occur in the action area, in places where they will be exposed to all stressors of the action. The RPAs are designed to reduce this exposure but not eliminate it. Pesticide runoff and drift of chlorpyrifos, diazinon, and malathion 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 water quality conditions of receiving waters used by salmon. Urban runoff is generally warmer in temperature and elevated water temperature pose negative effects on certain life history phases for salmon. The range of effects of the three active ingredients 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 Action* and the *Effects of the Proposed Actions* 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 chlorpyrifos, diazinon, and malathion);
2. Limited 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 active ingredients;
7. Annual variable conditions regarding land use, crop cover, and pest pressure;
9. Variable temporal and spatial conditions within each ESU, especially at the population level; and
10. Variable conditions of water bodies in which salmonids live.

NMFS therefore identifies as a surrogate for the allowable extent of take the ability of this action to proceed without any fish kills attributed to the use of malathion, diazinon or chlorpyrifos, or any compounds, degradates, or mixtures thereof in any stream containing individuals from any ESU. Because of the difficulty of detecting salmonid deaths, the fishes killed do not have to be listed salmonids. Salmonids appear to be more sensitive to these compounds, so that if there are kills of other freshwater fishes that can be 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 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 chlorpyrifos, diazinon, and malathion; (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's 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 chlorpyrifos, diazinon, and malathion by reducing the risk of chemicals reaching the water.
2. Monitor any incidental take or surrogate measure of take that occurs from the action.
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, the EPA must comply with the following terms and conditions, which implement the reasonable and prudent measure described above. These terms and conditions are non-discretionary.

1. 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 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, DPS) i.e., coho salmon, chum salmon, steelhead, sockeye salmon, and ocean-type Chinook and stream-type Chinook salmon. Additionally, each state shall have at least three sites within their borders. One site in each state shall target where juvenile ESA-listed salmonids migrate to the Pacific Ocean. The plan shall collect daily surface water samples for seven consecutive days for at least three seven-day periods during the application season. Collected water samples will be analyzed for current-use OPs and carbamates following USGS schedules for analytical chemistry. The report shall be submitted to NMFS OPR and will summarize annual monitoring data and provide all raw data.

2. For Ozette Lake Sockeye, require the following no-application buffers/setbacks on labels for all malathion, diazinon and chlorpyrifos containing products: Where ground applications are permitted. Do no apply pesticide products within 500 ft (152.4 m) of Ozette Lake sockeye salmon habitat. Where aerial applications are permitted. Do not apply pesticide products within 1,000 ft (304.8 m) of Ozette Lake sockeye salmon habitat.

3. EPA shall include the following instructions requiring reporting of fish kills either on the labels for all products containing malathion, diazinon or chlorpyrifos, or in ESPP Bulletins:

   NOTICE: If landowners and applicators find that salmon appear injured or killed as a result of pesticide exposure or other project-related activities, 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 must contact NMFS Office of Protected Resources at 301-713-1401. The finder may be asked to carry out instructions provided by Protected Resources to collect specimens or take other measures to ensure that evidence intrinsic to the specimen is preserved.

4. EPA shall report to NMFS any incidences from its incident database that it has classified as probable or highly probable.

Conservation Recommendations

Section 7(a) (1) of the ESA directs Federal agencies to utilize 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 active ingredients 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;
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 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 chlorpyrifos, diazinon, and malathion 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 ITS 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 no 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 chlorpyrifos, diazinon, and malathion 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.

In the freshwater portion of their life, Pacific salmon may be exposed to insecticides that act by inhibiting acetylcholinesterase (AChE). Acetylcholinesterase is a crucial enzyme in the proper functioning of cholinergic synapses in the central and peripheral nervous systems of vertebrates and invertebrates. Of consequence to salmon, anticholinesterase insecticides have been shown to interfere with salmon swimming behavior (Beauvais et al 2000, Brewer et al. 2001, Sandahl et al. 2005), feeding behavior (Sandahl et al. 2005), foraging behavior (Morgan and Kiceniuk 1990), homing behavior (Scholz et al. 2000), antipredator behaviors (Scholz et al. 2000) and reproductive physiology (Moore and Waring 1996, Scholz et al. 2000, Waring and Moore 1997).

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 (Liess and Schulz 1999, Anderson et al. 2003, Stark et al. 2004). For example, when an aquatic macroinvertebrate community in a German stream was exposed to runoff containing parathion (an acetylcholinesterase inhibitor) and fenvalerate (another commonly used insecticide), eight of eleven abundant species disappeared and the remaining three were reduced in abundance (Liess and Schulz 1999). Long-term changes in invertebrate densities and community composition likely result in reductions in salmon prey availability. Therefore, in addition to the direct impacts that acetylcholinesterase inhibitors have on salmon, there may also be, independently, significant indirect effects to salmon via their prey (Peterson et al. 2001). Wild juvenile salmon feed primarily on invertebrates in the water column and those trapped on the water's surface, actively selecting the largest items available (Healey 1991, Quinn 2005). Salmon are often found to be food limited (Quinn 2005), suggesting that a reduction in prey number or size due to insecticide exposure may further stress salmon. For example, Davies and Cook (1993) found that several months following a spray drift event, benthic and drift densities were still reduced in exposed stream reaches. Consequently, brown trout in the exposed reaches fed less and grew at a slower rate compared to those in unexposed stream reaches (Davies and Cook 1993). Although the insecticide in their study was cypermethrin (a pyrethroid), similar reductions in macroinvertebrate density and recovery times have been found in studies with acetylcholinesterase 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 (West and Larkin 1987, Healey 1982, 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 chlorpyrifos, diazinon, or malathion 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 mature and reproduce at age 3. Sockeye females reach maturity at age 4 or 5, but the majority of reproductive contributions are provided by age 4 females. Chinook females can mature at age 3, 4 or 5, with the majority of the reproductive contribution from ages 4 and 5. The primary difference between the ocean-type and stream-type chinook is the juvenile freshwater residence with ocean-type juveniles migrating to the ocean as subyearlings and stream-type overwintering in freshwater and migrating to the ocean as yearlings. The models depicted general populations representing each life-history strategy and were constructed based upon literature data described below. Specific populations were not modeled due to the difficulty in finding sufficient demographic and reproductive data for a single population.

A separate acute toxicity model was constructed that estimated the population-level impacts of juvenile mortality resulting from exposure to lethal concentrations of chlorpyrifos, diazinon, and malathion. 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 (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 model was developed to serve as a means to assess the potential effects on ESA-listed salmon populations from exposure to chlorpyrifos, diazinon and malathion. The growth model focuses on the impacts to prey abundance and a salmon's ability to feed as represented by changes in growth. Assessing the results from different pesticide exposure scenarios relative to a control (i.e., unexposed) scenario provides an insight into the extent to which sublethal pesticide exposures may lead to changes in the somatic growth and survival of individual salmon. Consequently, subsequent changes in salmon population dynamics as indicated by per cent change in a population's lambda assists us in forecasting the potential population level impacts to listed populations. Also, the model helps us understand the potential influence of life-history strategies that might explain differential results within the species modeled.

Methods

The model consists of two parts, an organismal portion and a population portion. The organismal portion of the model links AChE inhibition and reduced prey abundance due to insecticide exposure to changes in the growth of individual fish. The population portion of the model links the sizes of individual subyearling salmon to their survival and the subsequent growth of the population. Models were constructed and run using MATLAB 6.5 (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 IC50 values and their slopes were used to determine the level of AChE activity (Figure 2A, 2B, 2C) from the exposure concentration of each pesticide.

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 expose 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 (i.e., under the assumption that exposures would not increase potential ration beyond control). 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.

Sigmoidal dose-response relationships, at steady-state, between a single pesticide exposure and 1) AChE activity and 2) relative prey abundance are modeled using specific IC50s and EC50s and slopes (Table 3, Figure 2B and 3B). The timecourse for the 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). Likewise, the timecourse for relative prey abundance was modeled using two single-order exponential functions, one for the time required for the exposure to reduce prey abundance (i.e., kill prey) and the other for time required for complete recovery of prey abundance (time-to-effect$_{\text{prey}}$ and time-to-recovery$_{\text{prey}}$, respectively; Figure 3C). This 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) produced a relationship representing somatic growth rate over time (Figure 4D), which was then used to model individual growth rate and size over time. More details about the equaltions used in the models can be found in Box 1. The model was run for 1000 subyearling salmon exhibiting a normal distribution of starting weights with a mean of 1.0 g and standard deviation of 0.1 g. The size of 1.0 g was chosen to represent juvenile 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 was 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 was 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. The outputs of the organismal model consisted of mean weights (with standard deviations) after the specified growth period (Table 2). A sensitivity analysis was run to determine the influence of the parameter values on the output of the model. Model output was most sensitive to changes in the control growth rate (Gc, mean sensitivity 2.53). Control prey density and control AChE activity produced the next greatest sensitivity values (Table 1).

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. Exposure scenarios for all three compounds individually consisted of the following concentrations for 4, 21, or 60 days. The concentrations modeled were 0.01ug/L, 0.1ug/L, 0.5ug/L, 1ug/L, 3ug/L, 6ug/L, 10ug/L, and 100ug/L. All combinations of compound, length of exposure and concentration were modeled for each species. For the exposure scenarios presented in this project, the duration of time until full effect for the pesticides was assumed to be within a few days (Ferrari et al., 2004). Therefore we chose a time-to-effect half life for use in the calculations of 0.5 day. Time-to-recovery for salmon and other fish exposed to organophosphate insecticides require weeks to recover AChE activity (Eder et al., 2007; Ferrari et al., 2004; Chambers et al., 2002). This was reflected by assigning the recovery half life a value of 30 days.

The EC50 values and slopes for invertebrate prey were estimated using empirical data from an experiment examining the effects of chlorpyrifos on aquatic invertebrate communities (Van den Brink 1996). Using original data from the authors (Paul van den Brink, personal communication), the relative abundances of taxa known to be salmonid prey (or functionally similar to salmonid prey) were calculated (i.e., sum of the abundances of 14 taxa at 7 days post-exposure divided by pre-treatment abundances). This data set was used because it allowed us to calculate an EC50 and slope for an assemblage of representative prey exposed to a relevant range of chlorpyrifos concentrations in replicated outdoor mesocosms (Table 3). This calculated EC50 (2.3 µg/L, Table 3) is similar to other published values (laboratory 96-hr EC50 for invertebrates range from 0.2 – 2.7 µg/L,Van Wijngaarden et al. 1996), and is consistent with its use as an insecticide. The median chlorpyrifos EC50 from the literature was indeed similar at 1.7 µg/L, and using that value and other EC50 values from the literature for malathion and diazinon, we were able to estimate a relative toxicity of those compared to chlorpyrifos (Table 3).

The Van den Brink et al. (1996) dataset was also used to examine invertebrate community recovery rates following pesticide exposure. The 30-day half-life for recovery that was estimated from their data and used as a constant for these scenarios is consistent with other studies of invertebrate community recovery rates (Davies and Cook 1993, Liess and Schulz 1999). It was also assumed that regardless of the exposure scenario, relative prey abundance would not drop below a 20% 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 available as prey, regardless of pesticide exposure.

Box 1. Below are the relationships and 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/IC50)^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.

Figures 2C and 3C use a series of exponential functions (4A and 4B are repeats of 2F and 3C):

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 3C, c = Pc, i = Pi, ke = ln(2)/prey effect half-life, kr = ln(2)/prey recovery half-life.
For both Figures 2C and 3C, 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.

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.
Figure 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 were multiplied to produce a final ration for a given day. The final ration was used as input for 4C to generate 4D.

Population Model

The weight distributions from the organismal growth portion of the model were 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 incorporated a size-dependent survival rate for a three- or four-month interval (depending upon the species) which takes the juveniles up to 12 months of age. This time represents the 4-month early winter survival in freshwater for stream-type chinook, coho, and sockeye models. For ocean-type chinook, it is the 3-month period the subyearling smolt spend in the estuary and nearshore habitats (i.e., estuary survival). The weight distributions from the organismal model were converted to length distributions by applying condition factors from data for each modeled species. 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 (Kostow 1995, Myers et al. 2006, Howell et al. 1985). The relationship is based on the length of a subyearling salmon relative to the mean length of other competing subyearling salmon of the same species in the system, Equation 1, and relates that relative difference to size-dependent survival based upon Equation 2. The values for α and resulting size-dependent survival (survival ϕ) for control runs for each species are listed in Table 3. The constant α is a species-specific parameter defined such that it produces the correct control survival ϕ value when Δlength equals zero.

Equation 1: $\Delta\text{length} = \text{fish length(mm)} - \text{mean length(mm)}$

Equation 2: $\text{Survival } \phi = (e^{(\alpha+(0.0329*\Delta\text{length}))}) / (1 + e^{(\alpha+(0.0329*\Delta\text{length}))})$

Randomly selecting length values from the normal distribution calculated from the organismal model output size and applying equations 1 and 2 generated a size-dependent survival probability for each fish. This process was replicated 1000 times for each exposure scenario and simultaneously 1000 times for the paired control scenario and resulted in a mean size-dependent survival rate for each population. The resulting size-dependent survival rates were 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 used 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 were 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 were included beyond natural variability as represented by using parameter values selected from a normal distribution about a mean. Ocean conditions, freshwater habitat, fishing pressure, and resource availability were assumed constant and density independent.

In the model an individual fish experiences a 4-day exposure once as a subyearling (during its first spring) and never again. The pesticide exposure is assumed to occur annually. All subyearlings within a given population are assumed to be exposed to the pesticide. No other age classes experience the exposure. The model integrates this as every brood class being exposed as subyearlings and thus the vital demographic rates of the transition matrix are continually impacted in the same manner.

The model recalculated first-year survival each run using a size-dependent survival value selected from a normal distribution with the mean and standard deviation produced by Equation 2. Population model output consisted of the percent change in lambda from unexposed control populations derived from the mean of one 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 6.5.0 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 1000 times selecting the values in the transition matrix from their normal distribution defined by the mean standard deviation liste in Table 4 of the appendix. 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 the population-level responses. Separate models were constructed for coho, sockeye, ocean-type and stream-type chinook. In all cases transition values were determined from literature data on survival and reproductive characteristics of each species.

A life-history model was constructed for coho salmon (*Oncorhynchus kisutch*) with a maximum age of 3. Spawning occurs in late fall and early winter with emergence from March to May. Fry spend 14-18 months in freshwater, smolt and spend 16-20 months in the saltwater before returning to spawn (Pess et al. 2002). Survival numbers were summarized in Knudsen et al. (2002) as follows. The average fecundity of each female is 4500 with a standard deviation of 500. The observed number of males:females was 1:1. Survival from spawning to emergence is 0.3 (0.07). Survival from emergence to smolt is 0.0296 (0.00029) and marine survival is 0.05 (0.01). All parameters followed a normal distribution (Knudson et al. 2002). The calculated values used in the matrix are listed in Table 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 3374 (473) and for age 5 females were 4058 (557) (McGurk 2000). First year survival rates were 0.737/month (Gustafson et al. 1997). Ocean survival rates were calculated based upon brood data and the findings that 90% of ocean mortality occurs during the first 4 months of ocean residence (Pauley et al. 1989). Matrix values used in the sockeye baseline model are listed in Table 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 migrate from their natal stream within a couple months of hatching and spend several months rearing in estuary and nearshore habitats before continuing on to the open ocean. Transition values were determined from literature data on survival and reproductive characteristics from several ocean-type chinook populations in the Columbia River system (Green and Beechie 2004, PSCCTC 2002, Ratner et al. 1997, Healey and Heard 1984, Roni and Quinn 1995, Howell et al.

1985). The sex ratio of spawners was approximately 1:1. Estimated size-based fecundity of 4511(65), 5184(89), and 5812(102) was calculated based on data from Howell et al., 1985, using length-fecundity relationships from Healy and Heard (1984). Control matrix values for the chinook model are listed in Table 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 subyearling chinook condition indices came from data collected in the lower Columbia River and estuary (Johnson et al. 2007).

An age-structured life-history matrix model for stream-type chinook salmon with a maximum age of 5 was defined based upon literature data on Yakima River spring chinook from Knudsen et al 2006 and Fast et al 1988, with sex ratios of 0.035, 0.62 and 0.62 for females spawning at ages 3, 4, and 5, respectively. Length data from Fast et al., 1988 was used to calculate fecundity from Healy and Heard 1984, length-fecundity relationships. The 184 day growth period produces control fish with a mean size of 96mm, within the observed range documented in the fall prior to the first winter (Beckman et al. 2000). The size-dependent survival encompasses the 4 early winter months, up until the fish are 12 months old.

Acute Toxicity Models

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 were modeled as direct reduction in the first year survival rate (S1). 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 LC_{50} and slope (Table 3) for each pesticide. A sigmoid dose-response relationship was used to accurately handle responses well away from LC_{50} and to be consistent with other dose-response relationships. The sigmoidal dose-response slope (3.6; Table 3) was chosen because it produces responses that, once converted to probit values, have a probit slope of 4.5 for the responses within one log unit of the LC_{50}. This model utilized a probit slope of 4.5 for the mortality. Empirical slopes derived from standard acute toxicity studies with aquatic organisms show a range of probit slopes that bracket 4.5 for each of the three insecticides (http://www.ipmcenters.org/Ecotox/index.cfm).

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 was incorporated as described above using mean and standard deviation of normally distributed survival and reproductive rates and model output consisted of the percent change in lambda from unexposed control populations derived from the mean of 1000 calculations of both the unexposed control population and the pesticide exposed population. The percent change in lambda was considered different from control when the difference was 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 is within the variability reported in the literature for other salmonids (reviewed in Weatherley and Gill 1995). Control prey density and control AChE activity produced the next greatest sensitivity values (Table 1). Large changes (0.5 to 2X) in the other key parameters produced proportionate changes in final weight.

The sensitivity analysis of all four of the control population matrices predicted the greatest changes in population growth rate (λ) result from changes in first-year survival. Parameter values and their corresponding sensitivity values are listed in Table 4. The elasticity values for the transition matrices also corresponded to the driving influence of first-year survival, with contributions to lambda of 0.33 for coho, 0.29 for ocean-type chinook, 0.25 for stream-type chinook, and 0.24 for sockeye.

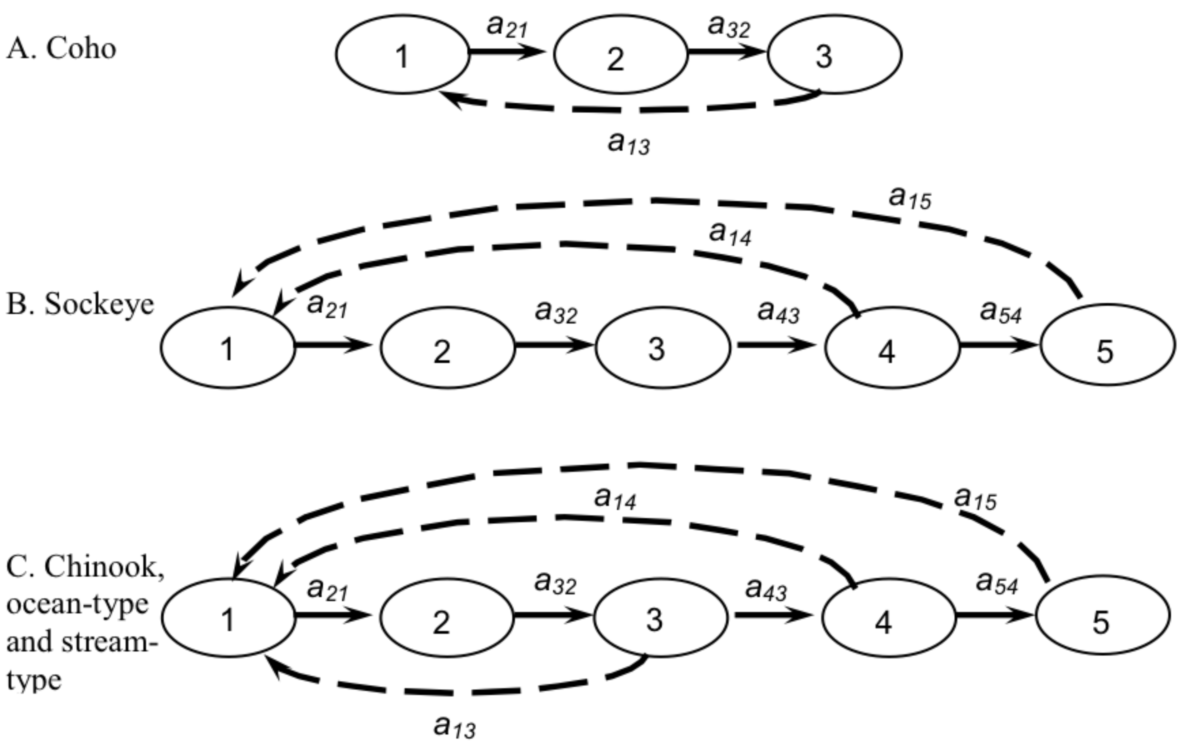
Model Output

Organismal and population model outputs for all scenarios are shown in Tables 5-16 and were summarized in as graphs in the main text. 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 relative AChE Activity and Prey Drift parameters determined by the toxicity values for each pesticide (Table 3). Both factors were equally contributing to the impacts for chlorpyrifos which have similar AChE IC_{50} and Prey Abundance EC_{50} values (Tables 3 & 5-8). The low Prey Abundance EC_{50} values drive the effects for diazinon and malathion models which have much higher AChE IC_{50} values (Tables 3 & 9-16).

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. In addition to this, the stream-type chinook and sockeye models produced very similar results as measured as the final output of percent change in population growth rate. The ocean-type chinook model output produced the next most extreme response, with coho output showing the greatest changes in lambda resulting from the pesticide exposures. When looking for similarities in parameters to explain the ranking, no single life history parameter or characteristic, such as lifespan, reproductive ages, age distribution, lambda and standard deviation, or first-year survival show a pattern that matches this consistent output. Combining these factors into the transition matrix for each life-history and conducting the sensitivity and elasticity analyses revealed that changes in first-year survival produced the greatest changes in lambda. In addition, the elasticity analysis can be used to predict relative contribution to lambda from changes in first-year survival on a per unit basis. As detailed by the elasticity values reported above, the same change in first-year survival will produce a slightly greater change in the population growth rate for coho and ocean-type chinook than for stream-type chinook and sockeye. While some life-history characteristics may lead a population to be more vulnerable to an impact, the culmination of age structure, survival and reproductive rates as a whole strongly influences the population-level response.

Figure 1: Life-History Graphs and Transition Matrix for coho (A), sockeye (B), and chinook (C) salmon. The life-history graph for a population labeled by age, with each transition element labeled according to the matrix position, a_{ij}, i row, 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.



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. Relationships used to link anticholinesterase exposure to the organism's ability to acquire food (potential ration). See text for details. Relationships in B, C, and D utilize empirical data. Closed circles represent control conditions. Open circles represent the exposed (inhibited) condition. A) Representation of a constant level of anticholinesterase pesticide exposure (either a single compound or mixtures). B) Sigmoidal relationship between exposure concentration and steady-state acetylcholinesterase (AChE) activity showing a dose-dependent reduction defined by control activity (horizontal line, Ac), sigmoid (i.e., hille) slope (AChE slope), and the concentration producing 50% inhibition (vertical line, IC_{50}). C) Timecourse of acetylcholinesterase inhibition based on modeling the time-to-effect and time-to-recovery as single exponential curves with different time-constants. At the start of the exposure AChE activity will be at control and then decline toward the inhibited activity (Ai) based on Panel B. D) Linear model relating acetylcholinesterase activity to feeding behavior using a line that passes through the feeding (Fc) and activity (Ac) control conditions with a slope of Mfa. E) The relationship between feeding behavior and the potential ration an organism could acquire (if not food limited) used a line passing through the control conditions (Fc as in Panel D and the maximum ration possible, Rc) and through the origin producing a slope (Mrf) equal to Rc/Fc. F) Timecourse for effect of exposure to anticholinesterase on potential ration produced by combining C & E. Further details regarding the equations used in the organismal model can be found in Box1.

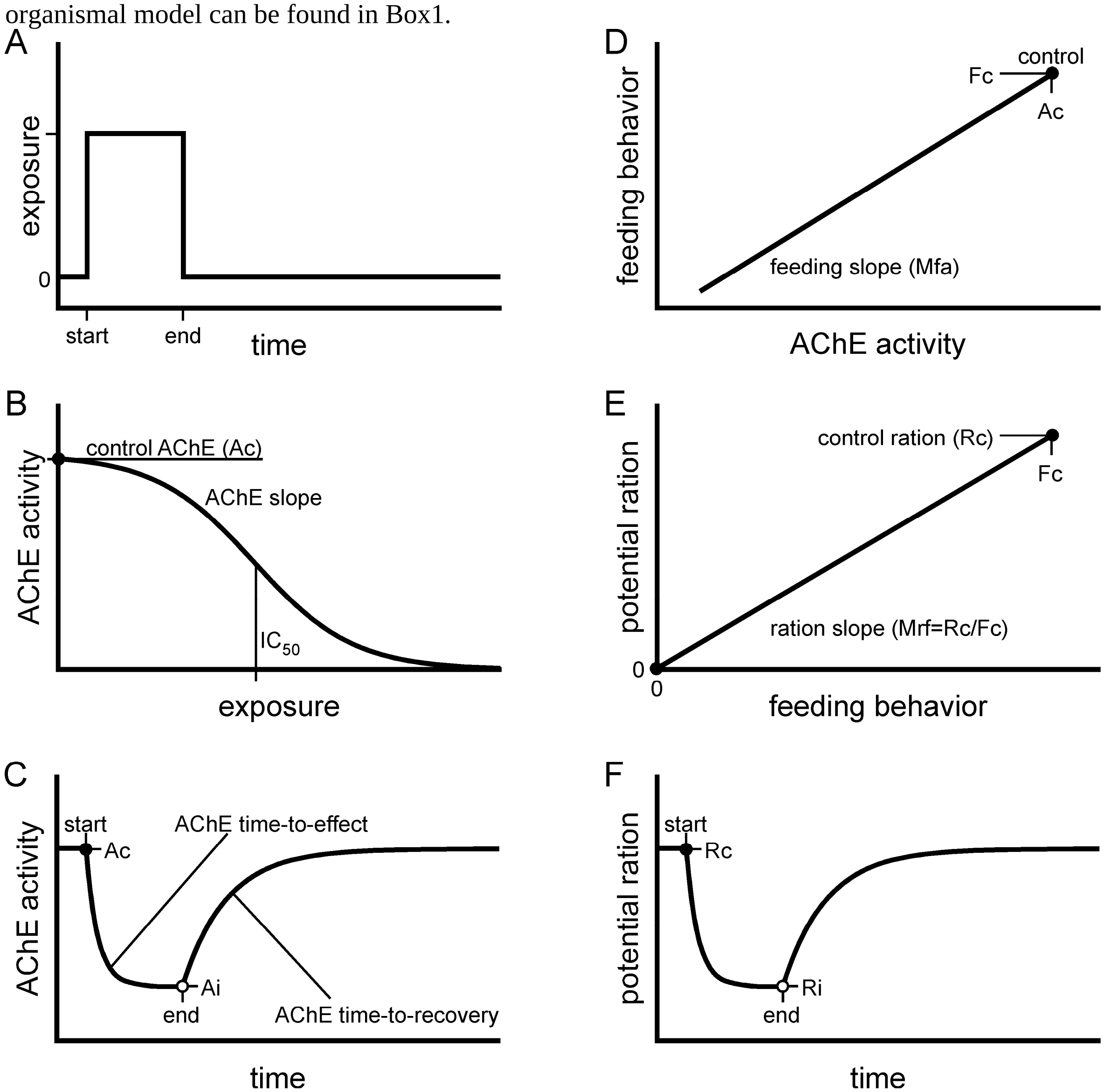


Figure 3. Relationships used to link anticholinesterase exposure to the availability of prey. See text for details. Relationships in B and C utilize empirical data. Closed circles represent control conditions. Open circles represent the exposed (inhibited) condition. A) Representation of a constant level of anticholinesterase pesticide exposure (either a 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., hill) 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 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, relative prey abundance will be at control (defined as 1) and then decline toward the inhibited abundance (Pi) based on Panel B. Further details regarding the equations used in the organismal model can be found in Box 1.

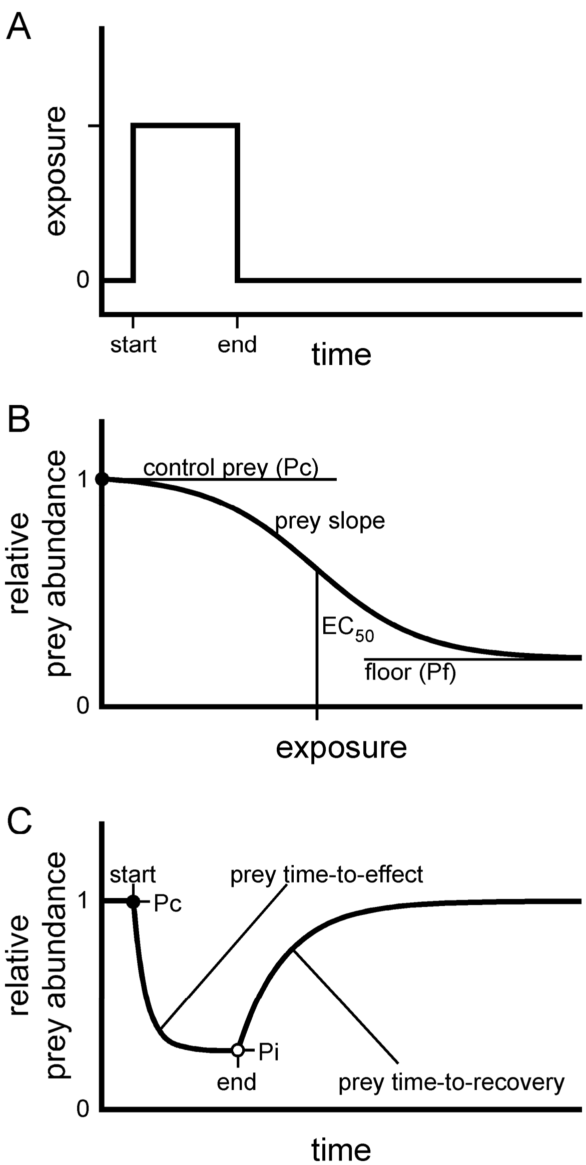


Figure 4. Relationships used to link anticholinesterase exposure to growth rate. See text for details. Relationships in A, B, and C utilize empirical data. Closed circles represent control conditions. Open circles (e.g., Ai) represent an example of an exposed (inhibited) condition. A&B) Relationships describing the timecourse of the effects of anticholinesterase exposure on the organism's 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. This temporal profile of growth rate was then applied to model the consequences of exposure on the long-term weight gain of the animal. Further details regarding the equations used in the organismal model can be found in Box1.

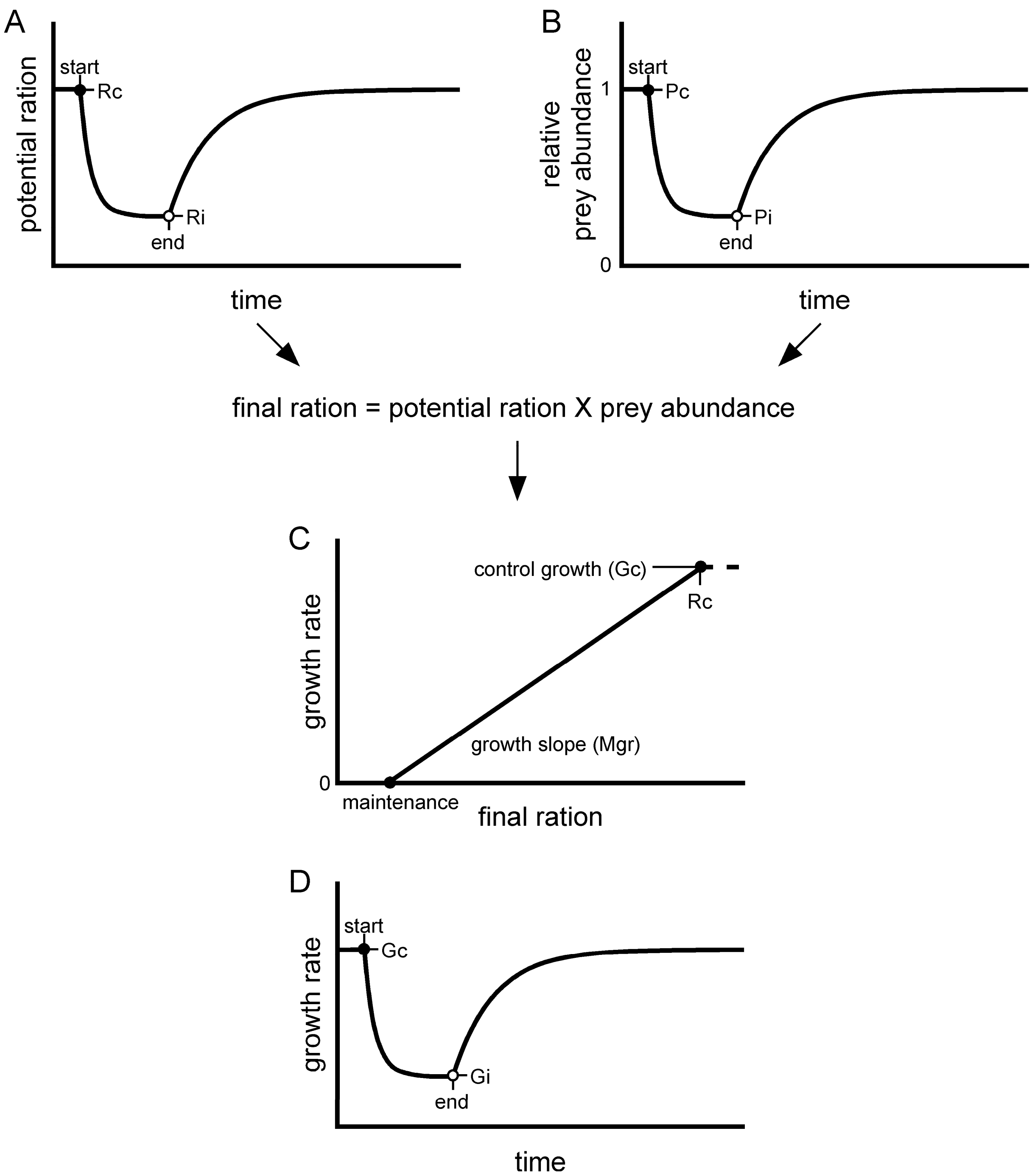


Table 1. List of values used for control parameters to model organismal growth and the model sensitivity to changes in the parameter.

Parameter	Value[1]	Error[2]	Sensitivity[3]
acetylcholinesterase activity (Ac)	1.0[4,5]	0.06[5]	-1.59
feeding (Fc)	1.0[4,5]	0.05[5]	0.304
ration (Rc)	5% weight/day[6]	0.05[7]	-0.530
feeding vs. activity slope (Mfa)	1.0[5]	0.1[5]	-0.294
ration vs. feeding slope (Mrf)	5 (Rc/Fc)	-	-
growth vs. ration slope (Mgr)	0.35[6]	0.02[6]	-0.529
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]	1.64
AChE impact time-to-effect ($t_{1/2}$)	0.5 day[9]	n/a	0.005
AChE time-to-recovery ($t_{1/2}$)	30 days[10]	n/a	-0.22
prey abundance time-to-effect ($t_{1/2}$)	0.5 day[11]	n/a	0.006
prey abundance time-to-recovery ($t_{1/2}$)	30 days[12]	n/a	-0.144
prey floor	0.20[11]	n/a	0.091

[1] mean value of a normal distribution used in the model or constant value when no corresponding error is listed
[2] standard deviation of the normal distribution used in the model
[3] mean sensitivity when baseline parameter is changed over range of 0.5 to 2-fold
[4] other values relative to control
[5] derived from Sandahl et al. (2005)
[6] derived from Brett et al. (1969)
[7] data from Brett et al. (1969) has no variability (ration was the independent variable) so a variability of 1% was selected to introduce some variability
[8] consistent with field-collected data for juvenile Chinook (Nelson et al., 2004)
[9] estimated from Ferrari et al., 2004
[10] consistent with Eder et al., 2007; Ferrari et al., 2004; Chambers et al., 2002
[11] estimated from Van den Brink 1996
[12] derived from Van den Brink 1996, Davies and Cook 1993, Liess and Schulz 1999

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 2	-0.33	-1.99	-0.802	-0.871
Control Survival ϕ	0.418	0.169	0.310	0.295

[1] Values from data in Healy and Heard 1984, Fast et al 1988, Beckman et al., 2000, Knudsen et al 2006
[2] Values from data in Healey and Heard 1984, Howell et al, 1985, Roni and Quinn 1995, Ratner et al 1997, PSCCTC 2002, Green and Beechie, 2004, Johnson et al., 2007
[3] Values from data in Pess et al., 2002, Knudsen et al., 2002
[4] Values from data in Pauley et al., 1989, Gustafson et al., 1997, McGurk 2000

Table 3. Effects values (ug/L) and slopes for AChE activity, acute fish lethality, and prey abundance dose-response curves.

Compound	AChE Activity IC_{50}[1] ug/L	AChE Activity slope[1]	Fish lethality LC_{50}[2] ug/L	Fish lethality slope[3]	Prey Abundance EC_{50}[4] ug/L	Prey Abundance Slope
chlorpyrifos	2.0	1.5	3.0	3.6	2.3	1.8
Malathion	74.5	1.32	30	3.6	2.76[3]	1.8
Diazinon	145	0.79	90	3.6	1.38[3]	1.8

[1]Values from Laetz et al., submitted for malathion and diazinon; Sandahl et al., 2005 for chlorpyrifos.
[2] Values from EPA BEs
[3] sigmoidal slope that produces responses with a probit slope of 4.5, see text.
[4] Chlorpyrifos value from median EC50s from data in EPA BE calculated by multiplying the chlorpyrifos EC_{50} by 1.2 for malathion and 0.6 for diazinon.

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. add EThe influence of each matrix element 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 the transition element, defined by $\delta\lambda / \delta a$. The elasticity of transition element a_{ij} is defined as the proportional change in λ relative to the proportional change in a_{ij}, and equals (a_{ij}/λ) times the sensitivity of a_{ij}. Elasticity values allow comparison of the influence of individual transition elements and comparison across matrices.

Transition Element	Chinook Stream-type			Chinook Ocean-type			Coho			Sockeye		
	Value[1]	S	E	Value[2]	S	E	Value[3]	S	E	Value[4]	S	E
S1	0.0643	3.844	0.247	0.0056	57.13	0.292	0.0296	11.59	0.333	0.0257	9.441	0.239
S2	0.1160	2.132	0.247	0.48	0.670	0.292	0.0505	6.809	0.333	0.183	1.326	0.239
S3	0.17005	1.448	0.246	0.246	0.476	0.106				0.499	0.486	0.239
S4	0.04	0.319	0.0127	0.136	0.136	0.0168				0.1377	0.322	0.0437
R3	0.5807	0.00184	0.0011	313.8	0.0006	0.186	732.8	0.000469	0.333			
R4	746.73	0.000313	0.233	677.1	0.000146	0.0896				379.57	0.000537	0.195
R5	1020.36	1.25E-05	0.0127	1028	1.80E-05	0.0168				608.7	7.28E-05	0.0437

[1] Value calculated from data in Healy and Heard 1984, Fast et al 1988, Beckman et al., 2000, Knudsen et al 2006
[2] Value calculated from data in Healey and Heard 1984, Howell et al, 1985, Roni and Quinn 1995, Ratner et al 1997, PSCCTC 2002, Green and Beechie, 2004, Johnson et al., 2007
[3] Value calculated from data in Pess et al., 2002, Knudsen et al., 2002
[4] Value calculated from data in Pauley et al., 1989, Gustafson et al., 1997, McGurk 2000

Table 5. Model output for ocean-type Chinook growth model exposed to chlorpyrifos reporting the impacted model parameter values for AChE Activity and Prey Abundance corresponding to the pesticide concentration and the resulting first-year survival (S1), population growth rate (λ) and standard deviation, and the percent change in λ from control. NS for percent change in lambda less than 7.

Impacted Model Parameters		**0 ug/L**	**0.01 ug/L**	**0.1 ug/L**	**0.5 ug/L**	**1.0 ug/L**	**3.0 ug/L**	**6.0 ug/L**	**10 ug/L**	**100 ug/L**
	AChE Activity	1.00	1.00	0.99	0.89	0.74	0.35	0.16	0.08	0.003
	Prey Abundance	1.00	1.00	1.00	0.95	0.85	0.51	0.32	0.25	0.20
4 d	S1	0.0056	0.0056	0.0056	0.0052	0.0046	0.0035	0.0031	0.0030	0.0029
	λ	1.09	1.10	1.09	1.07	1.04	0.95	0.92	0.91	0.91
	std of λ	0.08	0.08	0.08	0.08	0.08	0.07	0.06	0.06	0.06
	%△λ	**NA**	NS (1)	NS (0)	NS (-3)	NS (-5)	-13	-16	-16	-17
21 d	S1	0.0056	0.0056	0.0056	0.0050	0.0043	0.0030	0.0027	0.0026	0.0025
	λ	1.09	1.09	1.09	1.06	1.01	0.92	0.89	0.88	0.87
	std of λ	0.08	0.08	0.08	0.08	0.07	0.06	0.06	0.06	0.06
	%△λ	**NA**	NS (0)	NS (0)	NS (-3)	-7	-16	-19	-20	-21
60 d	S1	0.0056	0.0056	0.0055	0.0047	0.0038	0.0024	0.0022	0.0021	0.0020
	λ	1.09	1.09	1.08	1.04	0.97	0.86	0.83	0.83	0.82
	std of λ	0.08	0.08	0.08	0.08	0.07	0.06	0.06	0.06	0.06
	%△λ	**NA**	NS (0)	NS (0)	NS (-5)	-11	-21	-24	-25	-25

Table 6. Model output for stream-type Chinook growth model exposed to chlorpyrifos reporting the impacted model parameter values for AChE Activity and Prey Abundance corresponding to the pesticide concentration and the resulting first-year survival (S1), population growth rate (λ) and standard deviation, and the percent change in λ from control. NS for percent change in lambda less than 3.

Impacted Model Parameters		**0 ug/L**	**0.01 ug/L**	**0.1 ug/L**	**0.5 ug/L**	**1.0 ug/L**	**3.0 ug/L**	**6.0 ug/L**	**10 ug/L**	**100 ug/L**
	AChE Activity	1.00	1.00	0.99	0.89	0.74	0.35	0.16	0.08	0.003
	Prey Abundance	1.00	1.00	1.00	0.95	0.85	0.51	0.32	0.25	0.20
4 d	S1	0.0645	0.064	0.064	0.060	0.054	0.041	0.036	0.035	0.034
	λ	1.00	1.00	1.00	0.98	0.96	0.89	0.87	0.86	0.85
	std of λ	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03
	%△λ	**NA**	**NS (0)**	**NS (0)**	**NS (-2)**	**-4**	**-11**	**-14**	**-14**	**-15**
21 d	S1	0.0645	0.064	0.064	0.058	0.05	0.035	0.031	0.03	0.029
	λ	1.00	1.00	1.00	0.97	0.94	0.86	0.83	0.83	0.82
	std of λ	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03
	%△λ	**NA**	**NS (0)**	**NS (0)**	**NS (-2)**	**-6**	**-14**	**-16**	**-17**	**-18**
60 d	S1	0.0645	0.064	0.063	0.055	0.043	0.027	0.023	0.022	0.021
	λ	1.00	1.00	1.00	0.96	0.90	0.80	0.77	0.77	0.76
	std of λ	0.03	0.03	0.03	0.03	0.03	0.03	0.02	0.02	0.02
	%△λ	**NA**	**NS (0)**	**NS (0)**	**-4**	**-9**	**-20**	**-23**	**-23**	**-24**

Table 7. Model output for Coho growth model exposed to chlorpyrifos reporting the impacted model parameter values for AChE Activity and Prey Abundance corresponding to the pesticide concentration and the resulting first-year survival (S1), population growth rate (λ) and standard deviation, and the percent change in λ from control. NS for percent change in lambda less than 6.

		0 ug/L	**0.01 ug/L**	**0.1 ug/L**	**0.5 ug/L**	**1.0 ug/L**	**3.0 ug/L**	**6.0 ug/L**	**10 ug/L**	**100 ug/L**
Impacted Model Parameters	AChE Activity	1.00	1.00	0.99	0.89	0.74	0.35	0.16	0.08	0.003
	Prey Abundance	1.00	1.00	1.00	0.95	0.85	0.51	0.32	0.25	0.20
4 d	S1	0.030	0.030	0.029	0.028	0.025	0.20	0.018	0.017	0.017
	λ	1.03	1.03	1.03	1.01	0.97	0.90	0.87	0.86	0.85
	std of λ	0.06	0.06	0.06	0.06	0.05	0.06	0.05	0.05	0.05
	%△λ	**NA**	**NS (0)**	**NS (0)**	**NS (-2)**	**-6**	**-13**	**-16**	**-17**	**-17**
21 d	S1	0.030	0.030	0.029	0.027	0.024	0.018	0.016	0.015	0.015
	λ	1.03	1.03	1.02	1.00	0.95	0.86	0.83	0.82	0.81
	std of λ	0.06	0.06	0.06	0.06	0.06	0.06	0.06	0.05	0.06
	%△λ	**NA**	**NS (0)**	**NS (0)**	**NS (-3)**	**-8**	**-16**	**-19**	**-20**	**-21**
60 d	S1	0.030	0.029	0.029	0.026	0.021	0.014	0.012	0.12	0.1100
	λ	1.03	1.02	1.02	0.98	0.91	0.79	0.76	0.75	0.75
	std of λ	0.06	0.06	0.06	0.06	0.06	0.06	0.06	0.06	0.06
	%△λ	**NA**	**NS (0)**	**NS (0)**	**NS (-4)**	**-11**	**-23**	**-26**	**-27**	**-28**

Table 8. Model output for Sockeye growth model exposed to chlorpyrifos reporting the impacted model parameter values for AChE Activity and Prey Abundance corresponding to the pesticide concentration and the resulting first-year survival (S1), population growth rate (λ) and standard deviation, and the percent change in λ from control. NS for percent change in lambda less than 4.

		0 ug/L	**0.01 ug/L**	**0.1 ug/L**	**0.5 ug/L**	**1.0 ug/L**	**3.0 ug/L**	**6.0 ug/L**	**10 ug/L**	**100 ug/L**
Impacted Model Parameters	AChE Activity	1.00	1.00	0.99	0.89	0.74	0.35	0.16	0.08	0.003
	Prey Abundance	1.00	1.00	1.00	0.95	0.85	0.51	0.32	0.25	0.20
4 d	S1	0.026	0.026	0.026	0.024	0.021	0.016	0.014	0.014	0.013
	λ	1.01	1.01	1.01	0.99	0.96	0.90	0.88	0.87	0.86
	std of λ	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04
	%△λ	**NA**	**NS (0)**	**NS (0)**	**NS (-2)**	**-5**	**-11**	**-14**	**-14**	**-15**
21 d	S1	0.026	0.026	0.026	0.023	0.020	0.014	0.012	0.011	0.011
	λ	1.01	1.01	1.01	0.98	0.95	0.87	0.84	0.83	0.83
	std of λ	0.04	0.04	0.04	0.04	0.04	0.03	0.04	0.03	0.03
	%△λ	**NA**	**NS (0)**	**NS (0)**	**NS (-3)**	**-6**	**-14**	**-17**	**-18**	**-18**
60 d	S1	0.026	0.026	0.025	0.022	0.017	0.010	0.009	0.008	0.0080
	λ	1.01	1.01	1.00	0.97	0.91	0.81	0.78	0.78	0.77
	std of λ	0.04	0.04	0.04	0.04	0.04	0.03	0.03	0.03	0.03
	%△λ	**NA**	**NS (0)**	**NS (-1)**	**-5**	**-10**	**-20**	**-23**	**-23**	**-24**

Table 9. Model output for ocean-type Chinook growth model exposed to diazinon reporting the impacted model parameter values for AChE Activity and Prey Abundance corresponding to the pesticide concentration and the resulting first-year survival (S1), population growth rate (λ) and standard deviation, and the percent change in λ from control. NS for percent change in lambda less than 7.

Impacted		**0 ug/L**	**0.01 ug/L**	**0.1 ug/L**	**0.5 ug/L**	**1.0 ug/L**	**3.0 ug/L**	**6.0 ug/L**	**10 ug/L**	**100 ug/L**
Model Parameters	AChE Activity	1.00	1.00	1.00	0.99	0.98	0.96	0.92	0.89	0.57
	Prey Abundance	1.00	1.00	0.99	0.89	0.71	0.36	0.25	0.22	0.20
4 d	S1	0.0056	0.0056	0.0056	0.0053	0.0048	0.0040	0.0038	0.0037	0.0034
	λ	1.09	1.09	1.09	1.07	1.04	0.99	0.98	0.97	0.94
	std of λ	0.08	0.08	0.08	0.08	0.07	0.07	0.07	0.07	0.07
	%△λ	**NA**	**NS (0)**	**NS (0)**	**NS (-2)**	**NS (-5)**	**-9**	**-11**	**-11**	**-14**
21 d	S1	0.0056	0.0056	0.0056	0.0052	0.0046	0.0036	0.0033	0.0032	0.0030
	λ	1.09	1.09	1.09	1.06	1.03	0.96	0.094	0.93	0.91
	std of λ	0.08	0.08	0.08	0.08	0.07	0.07	0.06	0.06	0.06
	%△λ	**NA**	**NS (0)**	**NS (-1)**	**NS (-2)**	**NS (-6)**	**-13**	**-14**	**-15**	**-17**
60 d	S1	0.0056	0.0056	0.0056	0.0049	0.0041	0.0029	0.0026	0.0026	0.0023
	λ	1.09	1.10	1.09	1.05	1.00	0.90	0.88	0.87	0.85
	std of λ	0.08	0.08	0.08	0.08	0.07	0.06	0.06	0.06	0.06
	%△λ	**NA**	**NS (0)**	**NS (0)**	**NS (-4)**	**-8**	**-17**	**-20**	**-21**	**-22**

Table 10. Model output for stream-type Chinook growth model exposed to diazinon reporting the impacted model parameter values for AChE Activity and Prey Abundance corresponding to the pesticide concentration and the resulting first-year survival (S1), population growth rate (λ) and standard deviation, and the percent change in λ from control. NS for percent change in lambda less than 3.

Impacted Model Parameters		**0 ug/L**	**0.01 ug/L**	**0.1 ug/L**	**0.5 ug/L**	**1.0 ug/L**	**3.0 ug/L**	**6.0 ug/L**	**10 ug/L**	**100 ug/L**
	AChE Activity	1.00	1.00	1.00	0.99	0.98	0.96	0.92	0.89	0.57
	Prey Abundance	1.00	1.00	0.99	0.89	0.71	0.36	0.25	0.22	0.20
4 d	S1	0.0645	0.064	0.064	0.061	0.056	0.047	0.044	0.043	0.039
	λ	1.00	1.00	1.00	0.98	0.96	0.92	0.91	0.91	0.88
	std of λ	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03
	%△λ	**NA**	**NS (0)**	**NS (0)**	**NS(-2)**	**-4**	**-8**	**-9**	**-9**	**-11**
21 d	S1	0.0645	0.064	0.064	0.059	0.053	0.042	0.039	0.038	0.034
	λ	1.00	1.00	1.00	0.98	0.95	0.90	0.88	0.88	0.85
	std of λ	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03
	%△λ	**NA**	**NS (0)**	**NS (0)**	**NS (-2)**	**-5**	**-10**	**-12**	**-12**	**-15**
60 d	S1	0.0645	0.064	0.064	0.057	0.048	0.033	0.30	0.028	0.025
	λ	1.00	1.00	1.00	0.97	0.93	0.84	0.82	0.82	0.79
	std of λ	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03
	%△λ	**NA**	**NS (0)**	**NS (0)**	**NS (-3)**	**-7**	**-15**	**-18**	**-18**	**-21**

Table 11. Model output for Coho growth model exposed to diazinon reporting the impacted model parameter values for AChE Activity and Prey Abundance corresponding to the pesticide concentration and the resulting first-year survival (S1), population growth rate (λ) and standard deviation, and the percent change in λ from control. NS for percent change in lambda less than 6.

Impacted		**0 ug/L**	**0.01 ug/L**	**0.1 ug/L**	**0.5 ug/L**	**1.0 ug/L**	**3.0 ug/L**	**6.0 ug/L**	**10 ug/L**	**100 ug/L**
Model Parameters	AChE Activity	1.00	1.00	1.00	0.99	0.98	0.96	0.92	0.89	0.57
	Prey Abundance	1.00	1.00	0.99	0.89	0.71	0.36	0.25	0.22	0.20
4 d	S1	0.030	0.030	0.029	0.028	0.026	0.022	0.021	0.021	0.019
	λ	1.03	1.03	1.03	1.01	0.98	0.93	0.92	0.91	0.88
	std of λ	0.06	0.06	0.06	0.06	0.06	0.06	0.06	0.06	0.05
	%△λ	**NA**	**NS (0)**	**NS (0)**	**NS (-2)**	**NS (-4)**	**-10**	**-11**	**-11**	**-14**
21 d	S1	0.030	0.030	0.029	0.028	0.025	0.020	0.019	0.018	0.017
	λ	1.03	1.03	1.02	1.00	0.97	0.90	0.88	0.88	0.85
	std of λ	0.06	0.06	0.06	0.06	0.06	0.06	0.06	0.06	0.06
	%△λ	**NA**	**NS (0)**	**NS (0)**	**NS (-2)**	**NS (-6)**	**-13**	**-14**	**-15**	**-18**
60 d	S1	0.030	0.030	0.030	0.026	0.022	0.016	0.015	0.014	0.013
	λ	1.03	1.03	1.03	0.99	0.94	0.84	0.82	0.81	0.78
	std of λ	0.06	0.06	0.06	0.06	0.06	0.05	0.06	0.05	0.06
	%△λ	**NA**	**NS (0)**	**NS (0)**	**NS (-4)**	**-9**	**-18**	**-20**	**-22**	**-24**

Table 12. Model output for Sockeye growth model exposed to diazinon reporting the impacted model parameter values for AChE Activity and Prey Abundance corresponding to the pesticide concentration and the resulting first-year survival (S1), population growth rate (λ) and standard deviation, and the percent change in λ from control. NS for percent change in lambda less than 4.

		0 ug/L	**0.01 ug/L**	**0.1 ug/L**	**0.5 ug/L**	**1.0 ug/L**	**3.0 ug/L**	**6.0 ug/L**	**10 ug/L**	**100 ug/L**
Impacted Model Parameters	AChE Activity	1.00	1.00	1.00	0.99	0.98	0.96	0.92	0.89	0.57
	Prey Abundance	1.00	1.00	0.99	0.89	0.71	0.36	0.25	0.22	0.20
4 d	S1	0.026	0.026	0.026	0.024	0.022	0.018	0.017	0.017	0.015
	λ	1.01	1.01	1.01	1.00	0.98	0.93	0.92	0.91	0.89
	std of λ	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04
	%△λ	**NA**	**NS (0)**	**NS (0)**	**NS (-2)**	**NS (-4)**	**-8**	**-9**	**-10**	**-12**
21 d	S1	0.026	0.026	0.026	0.023	0.020	0.016	0.015	0.014	0.013
	λ	1.01	1.01	1.01	0.99	0.96	0.91	0.89	0.88	0.86
	std of λ	0.04	0.05	0.04	0.04	0.04	0.04	0.04	0.04	0.04
	%△λ	**NA**	**NS (0)**	**NS (0)**	**NS (-2)**	**-5**	**-10**	**-12**	**-13**	**-15**
60 d	S1	0.026	0.026	0.025	0.022	0.018	0.013	0.011	0.011	0.010
	λ	1.01	1.01	1.01	0.98	0.94	0.86	0.83	0.82	0.80
	std of λ	0.04	0.04	0.04	0.04	0.04	0.04	0.03	0.03	0.03
	%△λ	**NA**	**NS (0)**	**NS (0)**	**NS (-3)**	**-7**	**-15**	**-18**	**-19**	**-21**

Table 13. Model output for ocean-type Chinook growth model exposed to malathion reporting the impacted model parameter values for AChE Activity and Prey Abundance corresponding to the pesticide concentration and the resulting first-year survival (S1), population growth rate (λ) and standard deviation, and the percent change in λ from control. NS for percent change in lambda less than 7.

Impacted Model Parameters		**0 ug/L**	**0.01 ug/L**	**0.1 ug/L**	**0.5 ug/L**	**1.0 ug/L**	**3.0 ug/L**	**6.0 ug/L**	**10 ug/L**	**100 ug/L**
	AChE Activity	1.00	1.00	1.00	1.00	1.00	0.98	0.96	0.93	0.40
	Prey Abundance	1.00	1.00	1.00	0.96	0.89	0.57	0.36	0.27	0.20
4 d	S1	0.0056	0.0057	0.0056	0.0055	0.0053	0.0046	0.0041	0.0038	0.0032
	λ	1.09	1.09	1.09	1.09	1.07	1.03	0.99	0.98	0.93
	std of λ	0.08	0.08	0.08	0.08	0.08	0.08	0.07	0.07	0.06
	%△λ	**NA**	**NS (0)**	**NS (0)**	**NS (0)**	**NS (-2)**	**NS (-6)**	**-9**	**-11**	**-15**
21 d	S1	0.0056	0.0057	0.0056	0.0054	0.0052	0.0042	0.0036	0.0034	0.0028
	λ	1.09	1.09	1.09	1.08	1.07	1.00	0.96	0.94	0.90
	std of λ	0.08	0.08	0.08	0.08	0.08	0.08	0.07	0.07	0.06
	%△λ	**NA**	**NS (0)**	**NS (0)**	**NS (-1)**	**NS (-2)**	**-8**	**-13**	**-14**	**-18**
60 d	S1	0.0056	0.0056	0.0056	0.0054	0.0050	0.0036	0.0029	0.0027	0.0022
	λ	1.09	1.10	1.10	1.08	1.06	0.96	0.91	0.88	0.84
	std of λ	0.08	0.08	0.08	0.08	0.08	0.07	0.06	0.06	0.06
	%△λ	**NA**	**NS (0)**	**NS (0)**	**NS (0)**	**NS (-3)**	**-12**	**-17**	**-19**	**-24**

Table 14. Model output for stream-type Chinook growth model exposed to malathion reporting the impacted model parameter values for AChE Activity and Prey Abundance corresponding to the pesticide concentration and the resulting first-year survival (S1), population growth rate (λ) and standard deviation, and the percent change in λ from control. NS for percent change in lambda less than 3.

Impacted Model Parameters		**0 ug/L**	**0.01 ug/L**	**0.1 ug/L**	**0.5 ug/L**	**1.0 ug/L**	**3.0 ug/L**	**6.0 ug/L**	**10 ug/L**	**100 ug/L**
	AChE Activity	1.00	1.00	1.00	1.00	1.00	0.98	0.96	0.93	0.40
	Prey Abundance	1.00	1.00	1.00	0.96	0.89	0.57	0.36	0.27	0.20
4 d	S1	0.0645	0.064	0.064	0.063	0.061	0.052	0.047	0.045	0.037
	λ	1.00	1.00	1.00	1.00	0.99	0.95	0.92	0.92	0.87
	std of λ	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03
	%△λ	**NA**	**NS (0)**	**NS (0)**	**NS (-1)**	**NS (-1)**	**-5**	**-8**	**-8**	**-13**
21 d	S1	0.0645	0.064	0.065	0.063	0.06	0.048	0.042	0.039	0.032
	λ	1.00	1.00	1.00	0.99	0.98	0.93	0.9	0.88	0.84
	std of λ	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03
	%△λ	**NA**	**NS (0)**	**NS (0)**	**NS (0)**	**NS (-2)**	**-7**	**-10**	**-12**	**-16**
60 d	S1	0.0645	0.064	0.064	0.062	0.057	0.049	0.033	0.030	0.024
	λ	1.00	1.00	1.00	0.99	0.97	0.89	0.85	0.83	0.78
	std of λ	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.02
	%△λ	**NA**	**NS (0)**	**NS (0)**	**NS (-1)**	**NS (-3)**	**-10**	**-15**	**-17**	**-21**

Table 15. Model output for Coho growth model exposed to malathion reporting the impacted model parameter values for AChE Activity and Prey Abundance corresponding to the pesticide concentration and the resulting first-year survival (S1), population growth rate (λ) and standard deviation, and the percent change in λ from control. NS for percent change in lambda less than 6.

		0 ug/L	**0.01 ug/L**	**0.1 ug/L**	**0.5 ug/L**	**1.0 ug/L**	**3.0 ug/L**	**6.0 ug/L**	**10 ug/L**	**100 ug/L**
Impacted Model Parameters	AChE Activity	1.00	1.00	1.00	1.00	1.00	0.98	0.96	0.93	0.40
	Prey Abundance	1.00	1.00	1.00	0.96	0.89	0.57	0.36	0.27	0.20
4 d	S1	0.030	0.030	0.030	0.029	0.028	0.025	0.022	0.022	0.018
	λ	1.03	1.03	1.03	1.02	1.01	0.97	0.94	0.92	0.87
	std of λ	0.06	0.06	0.06	0.06	0.06	0.06	0.05	0.06	0.06
	%△λ	**NA**	**NS (0)**	**NS (0)**	**NS (0)**	**NS (-2)**	**-6**	**-9**	**-10**	**-15**
21 d	S1	0.030	0.030	0.030	0.029	0.028	0.023	0.020	0.019	0.016
	λ	1.03	1.03	1.03	1.02	1.00	0.95	0.91	0.88	0.84
	std of λ	0.06	0.06	0.06	0.06	0.06	0.05	0.05	0.05	0.05
	%△λ	**NA**	**NS (0)**	**NS (0)**	**NS (-1)**	**NS (-2)**	**-8**	**-12**	**-14**	**-19**
60 d	S1	0.030	0.029	0.030	0.029	0.0260	0.020	0.016	0.015	0.012
	λ	1.03	1.02	1.03	1.02	0.99	0.89	0.84	0.82	0.77
	std of λ	0.06	0.06	0.06	0.06	0.06	0.05	0.06	0.05	0.06
	%△λ	**NA**	**NS (0)**	**NS (0)**	**NS (-1)**	**NS (-4)**	**-13**	**-18**	**20**	**-25**

Table 16. Model output for Sockeye growth model exposed to malathion reporting the impacted model parameter values for AChE Activity and Prey Abundance corresponding to the pesticide concentration and the resulting first-year survival (S1), population growth rate (λ) and standard deviation, and the percent change in λ from control. NS for percent change in lambda less than 4.

Impacted Model Parameters		**0 ug/L**	**0.01 ug/L**	**0.1 ug/L**	**0.5 ug/L**	**1.0 ug/L**	**3.0 ug/L**	**6.0 ug/L**	**10 ug/L**	**100 ug/L**
	AChE Activity	1.00	1.00	1.00	1.00	1.00	0.98	0.96	0.93	0.40
	Prey Abundance	1.00	1.00	1.00	0.96	0.89	0.57	0.36	0.27	0.20
4 d	S1	0.026	0.026	0.026	0.025	0.024	0.021	0.018	0.017	0.014
	λ	1.01	1.01	1.01	1.00	1.00	0.96	0.93	0.92	0.88
	std of λ	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04
	%△λ	**NA**	**NS (0)**	**NS (0)**	**NS (-1)**	**NS (-1)**	**-5**	**-8**	**-9**	**-13**
21 d	S1	0.026	0.026	0.026	0.025	0.024	0.019	0.016	0.015	0.012
	λ	1.01	1.01	1.01	1.00	0.99	0.94	0.91	0.89	0.85
	std of λ	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.03
	%△λ	**NA**	**NS (0)**	**NS (0)**	**NS (-1)**	**NS (-2)**	**-7**	**-10**	**-12**	**-16**
60 d	S1	0.026	0.026	0.026	0.025	0.023	0.016	0.013	0.012	0.009
	λ	1.01	1.01	1.01	1.00	0.98	0.90	0.85	0.84	0.79
	std of λ	0.04	0.04	0.04	0.04	0.04	0.04	0.03	0.04	0.03
	%△λ	**NA**	**NS (0)**	**NS (0)**	**NS (-1)**	**NS (-2)**	**-11**	**-16**	**-17**	**-22**

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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
	Gray's 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
AChE	acetylcholinesterase
ai	active ingredient
APEs	alkylphenol ethoxylates. A group of non-ionic surfactant.
BE	Biological Evaluation
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)
BWEP	Boll Weevil Eradication Program
CALFED	CALFED Bay-Delta Program (California Resource Agency)
CBFWA	Columbia Basin Fish and Wildlife Authority
CBI	Confidential Business Information
CDPR	California Department of Pesticide Regulation
CIDMP	Comprehensive Irrigation District Management Plan
CFR	Code of Federal Regulations
cfs	cubic feet per second
CDFG	California Department of Fish and Game
CSOs	combined sewer/stormwater overflows
CSWP	California State Water Project
CURES	Coalition for Urban/Rural Environmental Stewardship
CVP	Federal Central Valley Projects
CVRWQCB	Central Valley Regional Water Quality Control Board
CWA	Clean Water Act
DDD	Dichloro Diphenyl Dichloroethane
DDE	Diphenyl Dichlorethylene
DDT	Dichloro Diphenyl Trichloroethane
DEQ	Oregon Department of Environmental Quality
DOE	Washington State Department of Ecology
DPS	Distinct Population Segment
EC	Emulsifiable Concentrate Pesticide Formulation
EC_{50}	Median Effect Concentration
EU	European Union
EEC	Estimated Environmental Concentration
EDC	endocrine disruptors
ENSO	El Nino Southern Oscillation
EPA	U.S. Environmental Protection Agency
ESA	Endangered Species Act
ESU	evolutionarily significant unit
EXAMS	Tier II Surface Water Computer Model
FERC	Federal Energy Regulatory Commission
FCRPS	Federal Columbia River Power System
FIFRA	Federal Insecticide, Fungicide, and Rodenticide Act
FQPA	Food Quality Protection Act
ft	feet
HCP	Habitat Conservation Plan

HSRG	Hatchery Scientific Review Group
HUC	Hydrological Unit Code
ICBTRT	Interior Columbia Basin 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
IPM	Integrated Pest Management
ISG	Independent Science Group
Lbs	Pounds
LC_{50}	Median Lethal Concentration. Statistically derived concentration of a substance expected to cause death in 50% of test animals. Usually expressed as the weight of substance per weight or volume of water, air, feed, e.g., mg/l, mg/kg, or ppm.
LOEC	Lowest Observed Adverse Effect Concentration. The lowest concentration with a significant difference from the control.
LOEL	Lowest Observed Adverse Effect level
LOC	Level of Concern
LOEC	Lowest Observed Effect Concentration
mg/L	milligrams per liter
MOA	Memorandum of Agreement
MPG	Major population group
MRID	Master Record Identification Number
MSA	Magnuson Stevens Fishery Conservation and Management Act
MTBE	Methyl tert-butyl ether
NASA	National Aeronautics and Space Administration
NPS	National Parks Services
NRCS	Natural Resources Conservation Service
NAWQA	U.S. Geological Survey National Water-Quality Assessment
NWS	National Weather Service
NEPA	National Environmental Policy Act
NMA	National Mining Association
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
NOEC	No Effect Concentration. The highest concentration with no significant difference from the control
NPDES	National Pollution Discharge Eliminating System
NRC	National Research Council
ODFW	Oregon Division of Fish and Wildlife
OP	Organophosphorus
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
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
RED	Reregistration Eligibility Decision
REI	Restricted Entry Level
RPA	Reasonable and Prudent Alternatives
RPM	reasonable and prudent measures
RQ	Risk Quotient
RTU	Ready to Use
RUP	Restricted Use Pesticide
SAR	smolt-to-adult return rate
SASSI	Salmon and Steelhead Stock Inventory
SLN	Special Local Need (Registrations under Section 24(c) of FIFRA)
T&C	terms and conditions
TCE	Trichloroethylene
TCP	3,5,6-trichloro-2-pyridinal
TDG	total dissolved gas
TGAI	Technical Grade Active Ingredient
TMDL	Total Maximum Daily Load
TRT	Technical Recovery Team
ULV	Ultra-Low Volume
USFS	United States Forest Service
USC	United States Code
USFWS	United States Fish and Wildlife Service
USGS	United States Geological Survey
VOC	Volatile Organic Compounds
VSP	viable salmonid population
WDFW	Washington Department of Fish and Wildlife
WLCRTRT	Willamette/Lower Columbia River Technical Review Team
WP	wettable powder
WQS	water quality standards
WWTIT	Western Washington Treaty Indian Tribes

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. Active ingredients are always listed on the label (FIFRA 2(a)).
Adulticide	A compound that kills the adult lifestage 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 an 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 (ESU)	A group of Pacific salmon or steelhead trout that is (1) substantially reproductively isolated from other conspecific units and (2) represent an important component of the evolutionary legacy of the species.
Fall Chinook Salmon	This salmon stock returns from the ocean in late summer and early fall to head upriver to its spawning grounds, distinguishing it from other stocks which migrate in different seasons.
Fate	Dispersal of a material in various environmental compartments (sediment, water air, biota) as a result of transport, transformation, and degradation.
Flowable	A pesticide formulation that can be mixed with water to form a suspension in a spray tank.
Fry	Stage in salmonid life history when the juvenile has absorbed its yolk sac and leaves the gravel of the redd to swim up into the water column. The fry stage follows the alevin stage and in most salmonid species is followed by the parr, fingerling, and smolt stages. However, chum salmon juveniles share characteristics of both the fry and smolt stages and can enter sea water almost immediately after becoming fry.
Half-pounder	A life history trait of steelhead exhibited in the Rogue, Klamath, Mad, and Eel Rivers of southern Oregon and northern California. Following smoltification, half-pounders spend only 2-4 months in the ocean, then return to fresh water. They overwinter in fresh water and emigrate to salt water again the following spring. This is often termed a false spawning migration, as few half-pounders are sexually mature.
Hatchery	Salmon hatcheries use artificial procedures to spawn adults and raise the resulting progeny in fresh water for release into the natural environment, either directly from the hatchery or by transfer into another area. In some cases, fertilized eggs are outplanted (usually in "hatch-boxes"), but it is more common to release fry or smolts.
Hyporheic Zone	Area of saturated sediment and gravel beneath and beside streams and rivers where groundwater and surface water mix.

Inert ingredients	"an ingredient which is not active" (FIFRA 2(m)). It may be toxic or enhance the toxicity of the active ingredient.
Introgression	Introduction by interbreeding or hybridization of genes from one population or species into another.
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.
Kelts	Steelhead that have spawned but may survive to spawn again, unlike most other anadromous fish.
Kokanee	The self-perpetuating, nonanadromous 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.
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.
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.

Oxon	Oxygen analog transformation products of parent organophosphates.
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 an those portions of flood plains an valley bottoms that support riparian vegetation.
Risk	The probability of harm from actual or predicted concentrations of a chemical in the aquatic environment – a scientific judgement.
Salmonid	Fish of the family *Salmonidae*, including salmon, trout, chars, grayling, and whitefish. In general usage, the term usually refers to salmon, trout, and chars.
SASSI	A cooperative program by WDFW and WWTIT to inventory and evaluate the status of Pacific salmonids in Washington State. The SASSI report is a series of publications from this program.
Semelparous	The condition in an individual organism of reproducing only once in a lifetime.
Smolt	A juvenile salmon or steelhead migrating to the ocean and undergoing physiological changes to adapt from freshwater to a saltwater environment.
Sublethal	Below the concentration that directly causes death. Exposure to sublethal concentrations of a material may produce less obvious

	effect on behavior, biochemical, and/or physiological function of the organism often leading to indirect death.
Surfactant	A substance that reduces the interfacial or surface tension of a system or a surface-active substance.
Synergism	A phenomenon in which the toxicity of a mixture of chemicals is greater than that which would be expected from a simple summation of the toxicities of the individual chemicals present in the mixture.
Technical Grade Active Ingredient (TGAI)	Pure or almost pure active ingredient. Available to formulators. Most toxicology data are developed with the TGAI. The percent AI is listed on all labels.
Technical Recovery Teams (TRT)	Teams convened by NOAA Fisheries to develop technical products related to recovery planning. TRTs are complemented by planning forums unique to specific states, tribes, or reigns, which use TRT and other technical products to identify recovery actions.
Teratogenic	Effects produced during gestation that evidence themselves as altered structural or functional processes in offspring.
Total Maximum Daily Load	defines how much of a pollutant a water body can tolerate (absorb) daily and remain compliant with applicable water quality standards. All pollutant sources in the watershed combined, including nonpoint 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 (VSP)	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.
Wettable powder	Pesticide formulations made by combining the active ingredient with a fine powder. They are made to mix with water.

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 designting 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.