UNITED STATES DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
NATIONAL MARINE FISHERIES SERVICE
West Coast Region
1201 NE Lloyd Boulevard, Suite 1100
PORTLAND, OREGON 97232

Refer to NMFS No: WCRO-2019-01915

October 25, 2019

Charles Mark
Forest Supervisor
Salmon-Challis National Forest
1206 South Challis Road
Salmon, Idaho 83467

Re: Reinitiation of Endangered Species Act Section 7 Formal Consultation and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Consultation for the Frank Church-River of No Return Weeds Management Program – Salmon-Challis, Boise, Payette, Nez Perce, and Bitterroot National Forests; Upper Salmon (17060201), Middle Salmon-Panther (17060203), Upper Middle Fork Salmon (17060205), Lower Middle Fork Salmon (17060206), Middle Salmon-Chamberlain (17060207), South Fork Salmon (17060208), and Upper Selway (17060301) Subbasins; Custer, Idaho, Lemhi, and Valley Counties, Idaho

Dear Mr. Mark:

Thank you for your letter of July 12, 2019 requesting initiation of consultation with NOAA's National Marine Fisheries Service (NMFS) pursuant to section 7 of the Endangered Species Act of 1973 (ESA) (16 U.S.C. 1531 et seq.) for the treatment of noxious weeds in the Frank Church-River of No Return Wilderness; Salmon-Challis, Boise, Payette, Nez Perce, and Bitterroot National Forests' (Forests); for the 2019 through 2039 management seasons. The noxious weeds program will be carried out through authority under the Federal Noxious Weed Control Act of 1974, the Sikes Act, the Plant Protection Act of 2000, the Noxious Weed Control and Eradication Act of 2004, U.S. Department of Agriculture Policy 9500-10, and U.S. Forest Service (USFS) Policy (FSM 2080).

Thank you, also, for your request for consultation pursuant to the essential fish habitat (EFH) provisions in section 305(b) of the Magnuson-Stevens Fishery Conservation and Management Act (MSA)(16 U.S.C. 1855(b)) for this action.

In this biological opinion (Opinion), NMFS concludes that the action, as proposed, is not likely to jeopardize the continued existence of Snake River Basin steelhead (*Oncorhynchus mykiss*), Snake River spring/summer Chinook salmon (*O. tshawytscha*), and Snake River sockeye salmon (*O. nerka*). NMFS also determined the action will not destroy or adversely modify designated critical habitat for designated critical habitat for all three species. Rationale for our conclusions is provided in the attached Opinion.

NMFS also reviewed the likely effects of the proposed action on essential fish habitat (EFH), pursuant to section 305(b) of the Magnuson-Stevens Fishery Conservation and Management Act (16 U.S.C. 1855(b)), and concluded that the action would adversely affect the EFH of Pacific Coast Salmon Plan. Therefore, we have included the results of that review in Section 3 of this document.

As required by section 7 of the ESA, NMFS provides an incidental take statement (ITS) with the Opinion. The ITS describes reasonable and prudent measures (RPM) NMFS considers necessary or appropriate to minimize the impact of incidental take associated with this action. The take statement sets forth nondiscretionary terms and conditions, including reporting requirements, that the USFS and any permittee who performs any portion of the action must comply with to carry out the RPM. Incidental take from actions that meet these terms and conditions will be exempt from the ESA take prohibition.

This document also includes the results of our analysis of the action's effects on EFH pursuant to section 305(b) of the MSA, and includes seven Conservation Recommendations to avoid, minimize, or otherwise offset potential adverse effects on EFH. These Conservation Recommendations are a non-identical set of the ESA Terms and Conditions. Section 305(b)(4)(B) of the MSA requires federal agencies provide a detailed written response to NMFS within 30 days after receiving these recommendations.

If the response is inconsistent with the EFH Conservation Recommendations, the Forests must explain why the recommendations will not be followed, including the justification for any disagreements over the effects of the action and the recommendations. In response to increased oversight of overall EFH program effectiveness by the Office of Management and Budget, NMFS established a quarterly reporting requirement to determine how many Conservation Recommendations are provided as part of each EFH consultation and how many are adopted by the action agency. Therefore, in your statutory reply to the EFH portion of this consultation, NMFS asks that you clearly identify the number of Conservation Recommendations accepted.

Please contact Ms. Kimberly Murphy (Salmon Field Office, (208) 756-5180, kimberly.murphy@noaa.gov) if you have any questions concerning this consultation, or if you require additional information.

Sincerely,

Michael P. Tehan

Michael P. Tehan
Assistant Regional Administrator
Interior Columbia Basin Office

Enclosure

cc: T. Ford – SCNF
K. Krieger – SCNF
M. Jakober – BNF
M. Anderson – BNF
C. Seesholtz– BNF
T. Brummett – PNF
C. Probert –NPCNF
S. Fisher – USFWS
C. Colter – SBT
A. Rogerson – NPT

Endangered Species Act Section 7(a)(2) Biological Opinion and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Consultation

Reinitiation for the Effects of the Treatment of Noxious Weeds in the Frank Church-River of No Return Wilderness – Salmon-Challis, Boise, Payette, Nez Perce, and Bitterroot National Forests; Hydrologic Units: Upper Salmon (17060201), Middle Salmon-Panther (17060203), Upper Middle Fork Salmon (17060205), Lower Middle Fork Salmon (17060206), Middle Salmon-Chamberlain (17060207), South Fork Salmon (17060208), and Upper Selway (17060301); Custer, Idaho, Lemhi, and Valley Counties, Idaho

NMFS Consultation Number: WCRO-2019-01915

Lead Action Agency: USDA Forest Service, Salmon-Challis National Forest

Affected Species and NMFS' Determinations:

ESA-Listed Species	Status	Is Action Likely to Adversely Affect Species?	Is Action Likely To Jeopardize the Species?	Is Action Likely to Adversely Affect Critical Habitat?	Is Action Likely To Destroy or Adversely Modify Critical Habitat?
Snake River steelhead (*Oncorhynchus mykiss*)	Threatened	Yes	No	Yes	No
Snake River spring/summer Chinook Salmon (*O. tshawytscha*)	Threatened	Yes	No	Yes	No
Snake River sockeye salmon (*O. nerka*)	Endangered	Yes	No	Yes	No

Fishery Management Plan That Identifies EFH in the Project Area	Does Action Have an Adverse Effect on EFH?	Are EFH Conservation Recommendations Provided?
Pacific Coast Salmon	Yes	Yes

Consultation Conducted By: National Marine Fisheries Service, West Coast Region

Issued By:

Michael P. Tehan
Michael P. Tehan
Assistant Regional Administrator

Date: October 25, 2019

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ACRONYMS	DEFINITIONS
a.e.	Acid Equivalent
a.i.	Active Ingredient
APHIS	Animal and Plant Health Inspection Service
BA	Biological Assessment
BMPs	Best Management Practices
DPS	Distinct Population Segment
DQA	Data Quality Act
ECA	Equivalent Clearcut Area
EDRR	Early Detection/Rapid Response
EEC	Estimated Environmental Concentration
EFH	Essential Fish Habitat
EPA	Environmental Protection Agency
ESA	Endangered Species Act
ESU	Evolutionarily Significant Unit
EUP	End Use Product
FC-RONRW	Frank Church River of No Return Wilderness
FIFRA	Federal Insecticide, Fungicide, and Rodenticide Act
FSH	Forest Service Handbook
FSM	Forest Service Manual
GLEAMS	Groundwater Loading Effects of Agricultural Management Systems
ICTRT	Interior Columbia Basin Technical Recovery Team
ITS	Incidental Take Statement
lb	Pound
LOEC	Lowest Observed Effects Concentration
MATC	Maximum Allowable Toxicant Concentrations
MFSR	Middle Fork Salmon River
mg/L	Milligrams per Liter
MPGs	Major Population Groups
mph	miles per hour
MSA	Magnuson-Stevens Fishery Conservation and Management Act
MSDS	Material Safety Date Sheet
MSO	Methylated Seed Oil
MUP	Manufacturing Use Product
NMFS	National Marine Fisheries Service
NFS	National Forest System
NOAEL	No Observed Adverse Effect Level
NOEC	No Observed Effects Concentration
NOEL	No Observable Effect Level
NPDES	National Pollutant Discharge Elimination System
OHWM	Ordinary High Water Mark
Opinion	Biological Opinion
PBF	Physical and Biological Feature
PCEs	Primary Constituent Elements
PDC	Project Design Criteria
POEA	Polyoxyethylamine

ACRONYMS	DEFINITIONS
RPMs	Reasonable and Prudent Measures
SCNF	Salmon-Challis National Forest
SERA	Syracuse Environmental Research Associates
TCP	Trichloropyridinol
TEA	Triethylamine Salt
TGAI	Technical Grade Active Ingredient
TMDL	Total Maximum Daily Load
Tribes	Shoshone-Bannock and Nez Pierce Tribes
USFS	U.S. Forest Service
VSP	Viable Salmonid Population
WCR	Water Contamination Rate

1. INTRODUCTION

This Introduction section provides information relevant to the other sections of this document and is incorporated by reference into Sections 2 and 3 below.

1.1 Background

National Marine Fisheries Service (NMFS) prepared the biological opinion (Opinion) and incidental take statement (ITS) portions of this document in accordance with section 7(b) of the Endangered Species Act (ESA) of 1973 (16 U.S.C. 1531 et seq.), and implementing regulations at 50 CFR 402. We also completed an essential fish habitat (EFH) consultation on the proposed action, in accordance with section 305(b)(2) of the Magnuson-Stevens Fishery Conservation and Management Act (MSA) (16 U.S.C. 1801 et seq.) and implementing regulations at 50 CFR 600.

We completed pre-dissemination review of this document using standards for utility, integrity, and objectivity in compliance with applicable guidelines issued under the Data Quality Act (DQA) (section 515 of the Treasury and General Government Appropriations Act for Fiscal Year 2001, Public Law 106-554). A complete record of this consultation is on file at the Snake Basin Office, Boise, Idaho.

The lead action agency for this consultation is the U.S. Forest Service (USFS), including the Salmon-Challis, Boise, Payette, Nez Perce, and Bitterroot National Forests (Forests). The Salmon-Challis National Forest (SCNF) is the lead office for this consultation. The USFS is proposing the action according to its authority under Forest Service Manual (FSM) 2080 and Region 4 Supplement #2000-00-1 (USFS 1995; USFS 2000), the USFS Noxious Weed Strategy (USFS 1996), and the USFS Strategy for Noxious Weed and Non-native Invasive Plants (USFS 1999). Noxious weed management and control has been recognized through national policy, forest plan development, broad scale assessments, and site-specific National Environmental Policy Act decisions. Laws that require management of noxious weeds include: the Federal Noxious Weed Act of 1974, as amended, the Forest and Rangeland Renewable Resource Planning Act of 1974, the Public Rangelands Improvement Act of 1978, and the Carlson-Foley Act of 1968. In addition, Executive Order 13112, signed in February 1999, directs Federal agencies to conduct activities that will reduce noxious weed populations. The Forests also cooperate with the State, but are not bound by most State laws.

Updates to the regulations governing interagency consultation (50 CFR part 402) will become effective on October 28, 2019 [84 FR 44976]. Because this consultation was pending and will be completed prior to that time, we are applying the previous regulations to the consultation. However, as the preamble to the final rule adopting the regulations noted, “[t]his final rule does not lower or raise the bar on section 7 consultations, and it does not alter what is required or analyzed during a consultation. Instead, it improves clarity and, consistency, streamlines consultations, and codifies existing practice.” Thus, the updated regulations would not be expected to alter our analysis.

1.2 Consultation History

NMFS received a request from the SCNF to reinitiate consultation on the proposed action on July 12, 2019. This Opinion is based on information provided in the June 10, 2019 biological assessment (BA), an ongoing action developed through previous consultation efforts and four Opinions issued over the past 15 years (NMFS Tracking #s 2004/01889, 2009/03999, 2012/02094, and 2014/0539).

NMFS received a draft BA on February 7, 2019, and provided the SCNF with comments during a March 5, 2019, Level 1 meeting. A second draft BA was submitted to NMFS on April 2, 2019, and NMFS provided additional comments to the Forests on April 12, 2019. NMFS provided preliminary consensus on the BA's adequacy during that meeting with the agreement that the recommended edits would be adequately incorporated into the final document. The SCNF submitted a revised final BA on July 12, 2019. NMFS shared the draft proposed action and proposed conservation measures with the SCNF on September 26, 2019. The SCNF suggested minor revisions to the draft Opinion on October 18, 2019.

NMFS provided copies of the draft proposed action and terms and conditions to the Nez Perce and Shoshone-Bannock Tribes (Tribes) and requested comments on September 30, 2019, because the proposed action would likely affect tribal trust resources. The Tribes did not respond.

1.3 Proposed Federal Action

"Action" means all activities or programs of any kind authorized, funded, or carried out, in whole or in part, by federal agencies (50 CFR 402.02). "Interrelated actions" are those that are part of a larger action and depend on the larger action for their justification. "Interdependent actions" are those that have no independent utility apart from the action under consideration (50 CFR 402.02). No interdependent or interrelated actions have been identified for this action.

The federal action is to control and manage noxious and invasive weeds within the administrative boundaries of the Frank Church River of No Return Wilderness (FC-RONRW) during the 2019 to 2039 field seasons, using an adaptive, integrated invasive species management strategy.

The differences between this proposed action and previous FC-RONRW weed consultations are:

1. The addition of five new herbicide active ingredients and several End Use Products (EUP) that have not been included in previous consultations (i.e., imazapyr, imazamox, sulfometuron-methyl, triclopyr trimethylamine salt [TEA], and fluroxypyr) (Table 1).

2. All herbicide active ingredients will be applied according to current label direction using the aquatic buffers described in Table 3. Only aquatic approved adjuvants will be used within riparian areas.

3. Jet boats will not be used to transport or apply chemicals. Jet boats were allowed in previous consultations but were mostly not used due to impracticality. None of the units anticipate using jet boats during the lifespan of this project. If that changes, the units would inform the regulatory agencies and reinitiate consultation.

4. The Prevention, Early Detection/Rapid Response (EDRR) and Rehabilitation/Restoration elements of the program are specifically described in this consultation, particularly in light of the increased threat posed by aquatic invasive plants. In previous consultations, these elements were implied but not directly described.

General program elements of the Weed Management Strategy are summarized below.

1.3.1 Prevention

Prevention of new infestations and the spread of noxious and invasive weeds will be enhanced through education/outreach and a noxious and invasive weeds prevention plan. The plan includes a variety of educational materials such as signage, exhibits, websites, and workshops; all of which will be used by the USFS and cooperative partners to raise public awareness of invasive plants and the ecological and economic damage created by their establishment and spread. Legal instruments such as weed seed free hay and forage orders assist in the implementation of preventative measures. Internal training will be used to educate personnel to recognize invasive plant species, understand vectors and preventative measures, incorporate preventative measures into the project design of all projects and activities, follow procedures for reporting and mapping invasive plant infestations, and communicate with other programs, agencies, and partners. This is a non-treatment aspect of the weed management program approach.

1.3.2 Early Detection/Rapid Response

The EDRR is a critical component of the weed management program. As new invasive plant infestations are detected, a quick and coordinated inventory and eradication response will reduce negative environmental and economic impacts.

The EDRR allows for detection and eradication of new invasive plant infestations located outside of currently identified infested areas. Infestations outside of currently identified areas may include: (1) New infestations of invasive plants currently known to exist in the FC-RONRW; (2) sites that currently exist, but have not been identified in inventories to date; or, (3) invasive plant species that have not previously occurred in the FC-RONRW (i.e., Watch List species). The EDRR is based on the premise that the impacts of similar treatment methods are predictable, even though the exact location or timing of the treatment may be unknown. Invasive plant infestations that are discovered subsequent to the current FC-RONRW invasive plant inventory would be evaluated to determine that treatment methods and environmental impacts are consistent with those already analyzed.

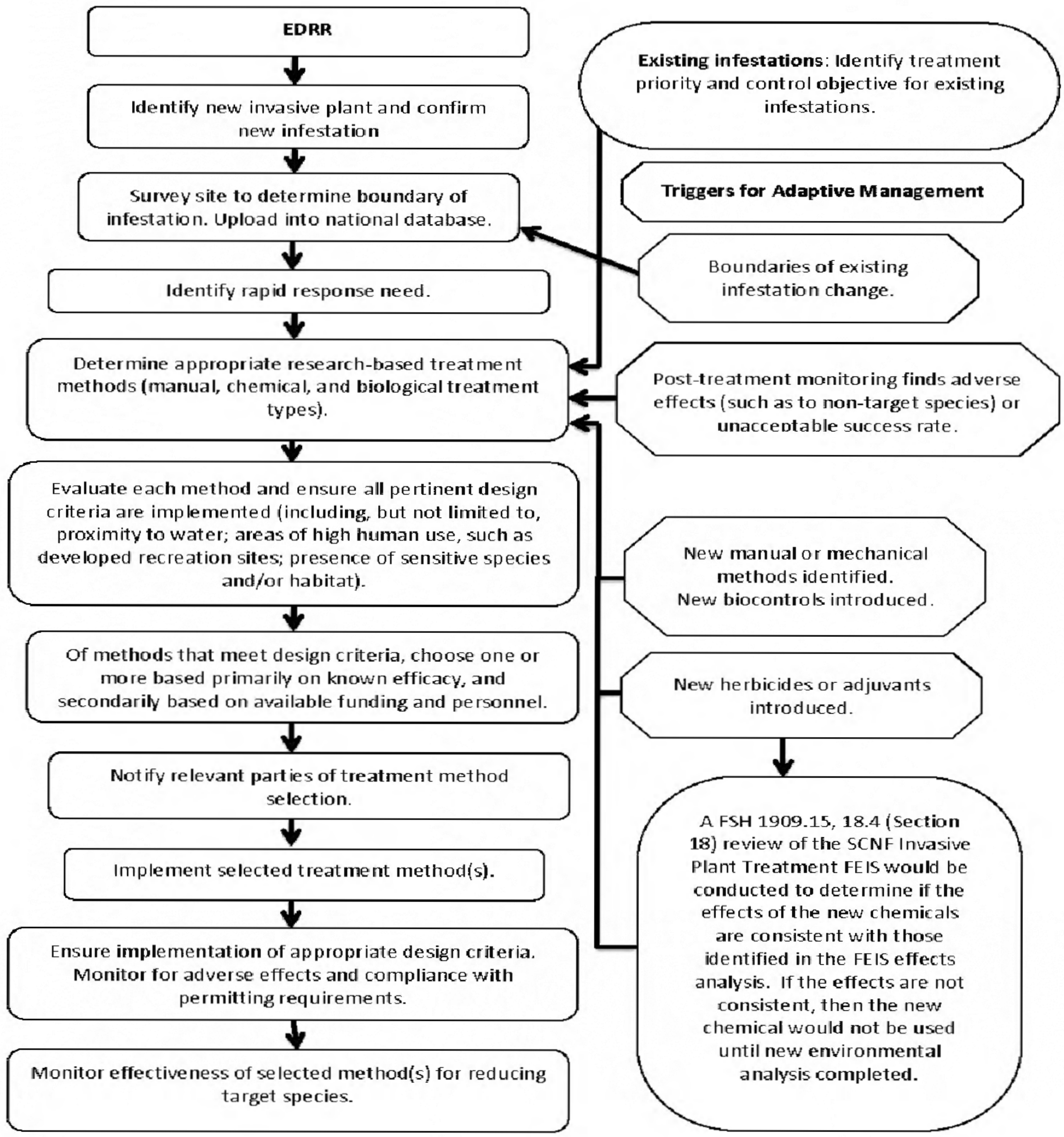

Figure 1. The EDRR and Adaptive Management Decision Tree.

1.3.3 Control

The proposed weed management program will facilitate the use of a variety of treatment options and combinations intended to minimize the effect of invasive plants and limit their spread. This proposed action includes the use of manual, chemical, and biological methods or a combination of any of these methods. A variety of treatment options and combinations that could be applied to a wide range of site conditions and invasive plant species are necessary to provide flexibility to increase effectiveness, reduce cost, and minimize potential for adverse effects from treatments.

Areas infested by invasive plants in the FC-RONRW may exhibit a wide range of site conditions. Effective control relies on a clear understanding of the site conditions and of the target species: its biology, the ecosystem it has infested, associated introduction pathways, and effective control methods. Therefore, multiple biological control agents or herbicides may be used to control invasive plant infestations. Control often requires repeat treatments and follow-up monitoring of control efficacy. As monitoring identifies the effectiveness of treatments, specific control measures can be adjusted.

1.3.4. Monitoring

Monitoring is an integral part of implementing an adaptive weed management program. Monitoring provides the data for adaptive management. Information collected from monitoring may be used by managers to evaluate the efficacy of prevention, EDRR, treatment and rehabilitation actions, and to inform future decision making and strategy.

There are two basic types of monitoring essential to an adaptive weed management plan: implementation monitoring and effectiveness monitoring. Implementation monitoring answers the question, “Did we do what we said we would do?” Effectiveness monitoring answers the questions, “Were treatment and rehabilitation/restoration actions effective?” and “Were intended goals accomplished?” Effectiveness monitoring generates data that aids managers in assessing trends in infestation number, size, and density, the effects of noxious and invasive plant infestations on native vegetation, the effect of treatments on target and non-target species, and the effectiveness of treatments as implemented.

Retreatment and rehabilitation prescriptions will be developed as needed based on post-treatment monitoring results. Changes in treatment methods will occur based on effectiveness of treating the invasive plant infestations. For example, an invasive plant population treated with a broadcast herbicide application may be retreated with spot spraying or hand pulling, once the size of the infestation and density of the seed bank are reduced.

Managers may use monitoring data from one site or set of sites to predict the effects of similar actions on other parts of the project area. This information can be used to promote the use of the most effective techniques for prevention, detection, treatment, and restoration, and avoid the use of ineffective methods.

1.3.4.1 Implementation Monitoring

Program elements and site-specific projects should include the following to accomplish implementation monitoring:

1. Develop a project work plan for herbicide use as described in Forest Service Handbook (FSH) 2109.14.3. This plan will present organizational and operational details including treatment objectives, equipment, materials, and supplies needed;

herbicide application method and rate; field crew organization and lines of responsibility; and a description of any interagency coordination. The plan will also include a job hazard analysis to ensure applicator safety.

2. Conduct site visits during work periods to monitor compliance.

3. Initiate monitoring during implementation to ensure project design criteria (PDC) are implemented as planned. Document daily field conditions, activities, accomplishments and/or difficulties. Use contract administration mechanisms to correct contractor performance deficiencies.

4. Document and report herbicide use and certified applicator information in the National Pesticide Use Database, via the database of record to determine the amount, type and location of herbicide use annually.

5. For biological control releases, monitor a selection of biological control release sites annually, tracking agent establishment and target species' response, to determine the efficacy of the release.

6. For mechanical treatments, monitor rehabilitative and restoration measures throughout the recovery process to quickly identify and correct any problems that may impede successful revegetation.

1.3.4.2 Effectiveness Monitoring

Effectiveness monitoring generates data that aids managers in assessing trends in infestation number, size, and density, the effect of noxious and invasive plant infestations on native vegetation, the effect of treatments on target and non-target species, and the effectiveness of treatments as implemented. Effectiveness monitoring must be done at multiple scales in order to provide the best insight into the effects of treatment actions. All treatment methods (manual, biological, and chemical) are subject to effectiveness monitoring.

1. Monitor size, density, and other biological characteristics of invasive plant infestations.

2. Maintain noxious and invasive plant inventory in the appropriate database of record.

3. Evaluate immediate and short-term impacts of treatment on target invasive plants and non-target vegetation.

4. Monitor and document observations of treated sites as practicable in accordance with established guidelines.

5. Evaluate long-term effects of treatment on target invasive plants and non-target vegetation.

6. Establish permanent monitoring plots for long-term site assessment.

7. Monitor survival, distribution, and effectiveness of biological control agents.

1.3.5 Treatment Methods

The proposed adaptive weed management program will utilize a variety of tools, used alone or in combination, to treat invasive plants on the FC-RONRW. Proposed treatment methods include the following:

- Manual methods, such as hand pulling, mowing, cutting, or torching.

- Biological control through the use of predators, parasites, and pathogens.

- Rehabilitation and restoration methods such as seeding sites to improve competition or prevent establishment of non-native invasive plant species.

- Herbicide control using ground-based application methods.

1.3.5.1 Manual Treatment

Manual treatments are typically used to remove seed heads, individual plants or small infestations. They may be used in sensitive areas to avoid impacts to non-target species or water quality, or to prevent seed production, etc. Manual approaches are slow and very labor intensive; they are effective only for small infestations.

The term "manual" defines treatments such as hand pulling or using hand tools, such as hand clippers, hoes, rakes, shovels, etc., to remove plants or cut off seed heads. Manual treatments can be effective for annual and tap-rooted invasive plant, but are ineffective against perennial invasive plants with deep underground stems or roots, or fine rhizomes that can be easily broken and left behind to re-sprout. Use of manual methods might need to be repeated several times throughout the growing season depending on the species. Manual treatments may require digging below the soil surface to remove the main root of plants.

Manual control methods proposed for use in the FC-RONRW include hand pulling, grubbing, hoeing, and cutting with non-mechanized tools. None of the proposed manual treatment methods would till the ground or otherwise cause significant soil disturbance.

PDC for Manual Treatment Methods

- Minimize soil disturbance as much as possible to prevent deeply buried invasive plant seeds being brought to the soil surface, promoting a sprouting event that could increase the density of the invasive plant species or create areas of bare soil that would provide an optimal seed bed in which new invasive species can sprout.

- Avoid non-target species damage to the extent practicable. Select mechanical methods to effectively control the target species (e.g., grubbing/hoeing is inappropriate for rhizomatous species and may increase the density of the invasive plant population as root fragments sprout and become new plants).

- Apply manual treatments at the proper stage of plant growth when treatment will be most effective at controlling the target invasive plant.

- Thoroughly inspect and clean all equipment and clothing to remove invasive plant seeds or vegetative propagules to prevent the movement of the invasive plant to another site.

- To the extent practicable, conduct clipping and removal of seed stalks prior to seed maturity to reduce inputs to the seed bank or when seeds are easily picked up and transported by vectors such wind, humans or animals(e.g., wind-dispersed seeds such as rush skeletonweed or bur-like seeds such as houndstongue that cling to fur and clothing).

- Specific to aquatic invasive species, hand-pulling, grubbing, raking, smothering and other similar manual methods may be used when an infestation is very limited in extent and occurs close to the shoreline of a water body, but has not yet infested deeper waters.

1.3.5.2 Biological Control

Biological weed control consists of using biological agents such as insects and plant pathogens to attack, weaken, and kill a targeted weeds species and reduce its competitive or reproductive capacity. The USFS also uses biological control agents to supplement herbicide control in larger infestations where treatment cannot be accomplished regularly due to the cost of treatment. Biological controls will be used when the target species occupies extensive portions of the landscape, other methods of control are prohibitive based on cost and location, and an effective biological control regime exists. Biological control activities typically include the release of parasitic and “host specific" insects, mites, nematodes, and pathogens. Biological treatments do not eradicate the target species, but rather reduce target plant densities and competition with desired plant species for space, water, and nutrients. Biological control treatments are not consistent with an eradication objective, but are an integral part of a weed management program approach.

The Animal and Plant Health Inspection Service (APHIS) and the State of Idaho have approved invertebrate plant feeders and plant pathogens that are proven natural control agents that suppress, inhibit, or control specific target invasive plant species. Biological control activities include collection of invertebrate plant feeders and pathogens, development of insectaries for collection, transportation and transplantation of parasitic invertebrate plant feeders and pathogens, and supplemental stocking of populations. Biological control agents are transported in containers that safely enclose the agent until release. Releases will be ground-based. Each release is equivalent to treating approximately 5 acres. The treated areas will continue to be inventoried and monitored to determine the success of the treatments and when the released bio-control agents have reached equilibrium with the target species. Repeat visits may need to be made several times a season and over a series of years to determine if additional releases are needed or if a different agent needs to be released.

The use of biological control treatment usually results in delayed effectiveness, often requiring 5 to 10 years for successful reduction of target invasive plant infestations. However, simultaneous increase of native vegetation often eliminates the need for restoration. Biological control is the preferred method in remote areas where access is limited, on high density extensive populations where other control methods may not be appropriate, on species where biological control agents are available and proven effective, and in conjunction with other control methods to reduce density of the target species.

PDC for Biological Control Methods

- Obtain APHIS permits to move live plant pests, noxious weeds, or soil for those agents when transportation across state lines is involved.

- There will be no intentional releases of unapproved biological control agents.

- There will be no intentional releases of agents known to feed on species other than those that were introduced to control specified invasive plant species.

- Appropriate Forest Service protocols will be used for documentation of releases and monitoring and share release information with the Idaho State Department of Agriculture.

- Where possible, biological control agents will be collected locally or from areas with similar climatic and weather conditions, land and soil types, and cover types to maximize successful establishment to the extent practicable.

- Biological control agents will be distributed at the optimal season and life cycle stage to optimize the likelihood of successful establishment. Quantities sufficient to optimize successful short-term establishment will be distributed.

  For those agents that self-disperse poorly, the distribution throughout target infestations will be actively assisted by redistribution (collecting and moving the agent to new locations). Survival, distribution, and other population characteristics of biological control agents will be monitored.

1.3.5.3 Rehabilitation and Restoration

The goal for invasive plant management efforts is to restore and maintain healthy native or desired plant communities that are resistant to invasive plant establishment, which recover quickly from disturbances, and provide ecosystem functionality. Many invasive plant-infested communities are able to successfully reestablish without intervention after weed control efforts. However, sites that have been severely impacted by weeds can be devoid of desirable plant species or consist of only scattered individual relict plants. Soil erosion may have taken place. Ecosystem structure and function may no longer be in place (e.g., mycorrhizal relationships between plants and soil fungi). Natural revegetation can often be slow, especially in these cases where there are few or no desirable plant species to take the place of invasive plants.

Rehabilitation and restoration are vital components of a successful adaptive, integrated invasive species management program. Rehabilitation is defined as short-term intervention to ensure minimum site stability and functionality. A successful intervention may be as simple as removing sources of soil and vegetative disturbance, such as restricting animal or human access until recovery has been achieved. Restoration is a long-term objective and involves returning sites to natural functions and native or desirable plant species. The long-term objective of rehabilitation/restoration is to reestablish a desired plant community and a return to conditions that foster the recovery of natural ecosystem processes. Equipment that could be used during reseeding activities will be limited to hand tools. Natural revegetation will be the preferred option whenever possible.

The following PDC are a subset of those described for the proposed action. For a complete list please refer to the BA and its appendices. The following list includes only those measures specific to avoiding or minimizing potential effects of the proposed action on ESA-listed fish and their habitat, and pertinent to the analysis of this Opinion:

- When seeding, determine the need for site preparation and protective measures that may need to be taken to allow the seeding to establish successfully.

- Revegetation activities will be planned for the optimal season and site conditions for successful establishment.

- Sites where restoration and rehabilitation treatments have been applied may need to be protected from grazing use through temporary fencing, livestock exclusion, or other method appropriate to the sites to allow seeded plant establishment.

- Following establishment, proper vegetation management will continue to be practiced to maintain a healthy, functioning plant community that is resilient to disturbance and resistant to invasive plant re-invasion.

- Ensure that treatment tools and other equipment are free of invasive plant seed before moving to or using on the project site.

- Ground disturbing activities will be minimized to the extent possible during reseeding efforts.

- Rehabilitation and restoration activities will be conducted only in areas with slope gradients less than 45 percent.

- Rehabilitation and restoration activities will be conducted only in areas with low or moderate land type erosion hazard ratings.

1.3.5.4 Herbicide Application

This method involves the use of herbicides and associated herbicide additives, including surfactants and dyes. This is the primary treatment method that will be used for controlling or eradicating weeds in the FC-RONRW.

Active Ingredients. The BA proposes use of 14 different active ingredients (Table 1). Nine of the 14 active ingredients were previously authorized for use in past FC-RONRW consultations (2,4-D amine, aminopyralid, chlorsulfuron, clopyralid, dicamba, glyphosate, imazapic, metsulfuron-methyl, and picloram). Five of the active ingredients are new herbicides being proposed for use in this BA (imazapyr, imazamox, sulfometuron-methyl, triclopyr trimethylamine salt (TEA), and fluroxypyr).

Table 1. Active Ingredients and End-Use Products that May be Used by the USFS for Noxious Weed Control Activities in the FC-RONRW.

Active Ingredient	End Use Product[1]	Typical Application Rate[2] (lb a.e./acre)	Maximum Label Application Rate[2] (lb a.e./acre)
2-4-D amine	Weedar 64 Amine 4 Riverdale Weedestory AM-40 Weedestroy Clean Amine DMA 4	0.5–1.5	4.0
Aminopyralid	Milestone	0.078–0.11	0.11
Clopyralid	Transline	0.28–0.5	0.5
Chlorsulfuron	Telar XP	0.02 0.09	0.12
Dicamba[3]	Banvel Vanquish	0.25–2.0	2.0
Aquatic Glyphosate	Rodeo GlyPro Accord Concentrate AquaMaster AquaNeat Aquatic Herbicide Foresters Aquaneet	0.26–3.7	8.0
Imazapic	Plateau	0.1–0.19	0.19
Metsulfuron-methyl	Escort XP	0.04–0.11	0.15
Picloram[4]	Tordon 22K Tordon K	0.5–0.75	1.0
Imazapyr[5]	Habitat	0.5–1.0	1.5
Imazamox[5]	Clearcast	0.25–0.5	0.5
Sulfometuron-methyl[5]	Oust XP	0.09 0.23	0.37
Triclopyr trimethylamine salt (TEA) [5]	Garlon 3A	4.5–6.0	9.0
Fluroxypyr[5]	Vista XRT (Ultra)	0.35–0.4	0.48

a.e. = acid equivalent

[1] Other brands not listed in Table 12, but that contain identical U.S. Environmental Protection Agency (EPA) registration numbers may be added or substituted in the future after Level 1 review (reference to EPA Pesticide Registration Manual and 40 CFR 152.113).

[2] Typical application rates are those used; maximum application rates are those on the product labels unless otherwise noted. The range of application rates for each herbicide is derived from Human Health and Ecological Risk Assessments and the herbicide label.
[3] Dicamba application rates on the FC-RONRW are 1 lb per acre for broadcast applications and no more than 2 lb per acre per year on a treatment area, per label requirements for rangeland uses.
[4] Maximum application rate for picloram is 1-pound acid equivalent per acre (1 lb a.e./acre); rates may be higher for smaller portions of the acre, but the total use on the acre cannot exceed 1 lb a.e. per acre per year.
[5] New active ingredient proposed for use as part of this action.

End Use Products. The herbicide active ingredients are blended in with other inert ingredients and sold commercially as patented EUPs (e.g., Milestone, Transline, Rodeo, etc). Numerous EUPs may contain the same active ingredient; however, they may contain different inert ingredients or different proportions of similar inert ingredients. Inert ingredients are any substances, other than the active ingredient, that are intentionally added to an herbicide formulation. Inert ingredients serve to enhance the action of the active ingredient. Inert ingredients may include carriers, surfactants, preservatives, dyes, and anti-foaming agents among other chemicals. Because many manufacturers consider inert ingredients in their herbicide formulations to be proprietary, they do not list specific chemicals. Many of these chemicals are categorized by the EPA as List 3 (inert ingredients of unknown toxicity) or List 4B (other ingredients for which EPA has sufficient information to reasonably conclude that the current use pattern in pesticide or herbicide products will not adversely affect public health or the environment). Listed inert ingredients for the herbicide EUPs being considered for use in the FC-RONRW include water, isopropanol (aka isopropyl alcohol), isopropanolamine, and polyglycol. Water is the inert ingredient that will be used most often in the FC-RONRW because it is free, readily available, and the active ingredients proposed for use are formulated to be effectively applied with water.

Adjuvants. Adjuvants are generally defined as any substance added separately to an EUP (typically as part of a spray tank mixture) to improve its performance. They can either enhance the activity of an active ingredient (activator adjuvant) or offset any problems associated with its application (special purpose or utility modifiers). For example, some adjuvants increase herbicide effectiveness by reducing the surface tension of water, increasing the area of the plant covered by the solution and increasing the plant's uptake of the herbicide itself. Adjuvants can be added during the manufacturing process or by the applicator as needed based on site conditions.

Adjuvants include surfactants, anti-foaming agents, crop oil or crop oil concentrates, drift retardants, compatibility agents, dyes, and pH buffers. Spray adjuvants proposed for use in the FC-RONRW include Activator 90, Spreader 90, LI-700, Syl-Tac-EA, Rl1, and methylated seed oil (MSO). Activator 90, Spreader 90, and LI-700 are non-ionic surfactants, meaning they have no ionic charge and are hydrophilic (water-loving). They are generally biodegradable and are compatible with many fertilizer solutions. The adjuvant Rl1 is a spreading agent that lowers the surface tension on the droplet so it covers the target plant more efficiently. The adjuvant MSO increases the penetration of oil-soluble herbicides into a plant. It is particularly effective during drought, when leaf cuticles are thick (Tu et al. 2001). Both the herbicide and the adjuvant labels include instructions on the use of agents such as these for proper herbicide application.

Dyes proposed for use in the FC-RONRW include Bullseye, Insight, and Hilight. These dyes provide a bright blue color and are non-hazardous. The use of the dye will be used to make it easier to see where the herbicide has been applied, and where or whether it has dripped, spilled, or leaked. Dyes also make it easier to detect missed spots and help the applicator avoid spraying a plant or area twice.

Currently, the state of Idaho does not have a registration system for adjuvants. In order to address toxicity concerns related to the use of adjuvants, only adjuvants that have been certified for use in riparian areas by the state of Washington will be used in FC-RONRW riparian areas. The state of Washington certifies adjuvants for aquatic use with its own state registration numbers. To be certified in Washington, adjuvant manufacturers have to provide a complete ingredient list, display all ingredients on the label and submit efficacy data to the State Department of Water Quality to prove that the product does what they claim it does. Once all requirements are met, the state assigns a unique state registration number that will be listed on the product label. Appendix B, Table B-5 in the BA lists the adjuvants currently certified by the state of Washington for use in riparian areas. All adjuvants proposed for use in the proposed action are on the list currently approved for riparian area use within the State of Washington.

Application Methods. Herbicides will be applied by several different ground-based methods, including backpack spraying, hand pumps, wicking, and wiping. Aerial application is not proposed in the FC-RONW and is not included in the proposed action. Broadcast spraying is also not a spray technique that is used in the FC-RONRW. The selected application technique depends on a number of variables, including the treatment objective; accessibility, topography, and size of the treatment area; characteristics of the target plants and the desired vegetation; the location of sensitive areas in the immediate vicinity; the anticipated costs and equipment limitations; and the meteorological and vegetative conditions at the time of treatment.

The type of herbicide used and its application rate will vary depending on factors such as the target species, phenological stage, abundance and distribution of the target and non-target species, the type of herbicide used, site conditions, soil type, depth to the water table, distance to open water sources, riparian areas, the presence of sensitive plant species, and herbicide label requirements.

Table 2 summarizes the physical properties of the herbicides proposed for use in the FC-RONRW, including soil persistence and mobility, water contamination rates, and expected environmental concentrations at maximum label application rates.

Table 2. Physical Properties of Herbicides Proposed for Use in the FC-RONRW.

Active Ingredient	Persistence in Soil (half-life in days)[1]	Mobile in Soil	Volatilization Potential
2,4-D amine	10 Low	Yes, but degrades quickly	Much less volatile than ester formulations[2]
Aminopyralid	103 High	Yes	Extremely Low[3]
Chlorsulfuron	40 (28–42) Moderate	No	Volatilization plays a minor role in disappearance[4]
Clopyralid	40 Moderate	No	Does not volatize readily in the field[2]
Dicamba5/	7–42 Low-Mod	Yes	Volatilization not significant from soil surfaces, some may occur

Active Ingredient	Persistence in Soil (half-life in days)[1]	Mobile in Soil	Volatilization Potential
			from plant surfaces[5] can be volatile[8]
Aquatic Glyphosate	47 Moderate	No	Does not volatize readily in the field[2]
Imazapic	7–150 Low-High	No	Not volatile[2]
Imazapyr	2,150 (313–2,972) High	Yes	Does not volatize readily in the field[2]
Imazamox	81 Moderate	Yes	Volatilization not significant[6]
Metsulfuron- methyl	30 (7–28) Low	No	Does not volatize readily in the field[7]
Picloram6/	90 (20–300) Mod-High	Yes	Does not volatize readily in the field[2]
Sulfometuron-methyl	20–28 @ pH 6–7 Low	Yes; generally more mobile at higher soil pH	Not volatile[8]
Triclopyr (TEA)	30 Low	Yes	Ester formulations highly volatile[2] (USFS uses only TEA amine formulation)
Fluroxypyr	7–23[9] Low 3.2–185[10] Low-High	Yes	Volatilization is considered negligble[9]

[1] Soil half-life values for herbicides are from Herbicide Handbook (Ahrens 1994) except Imazapyr and Imazamox from respective SERA Risk Assessments. Fluroxypyr soil half-life values are from the SERA Risk Assessment (2009) and the Bureau of Land Management Ecological Risk Assessment (2014). Pesticides that are considered non-persistent are those with a half-life of less than 30 days; moderately persistent herbicides are those with a half-life of 30 to 100 days; pesticides with a half-life of more than 100 days are considered persistent.
[2] Tu et al. 2001; [3] Dow Agro 2010; [4] PMEP 1985; [5] EXTONET 1993; [6] EPA 1997; [7] Information Ventures 2003; [8] BPA 2000; [9] SERA 2009; [10] Bureau of Land Management 2014.

The following PDC for herbicide treatments are a subset of those described for the proposed action. For a complete list please refer to Section 5.4.4.4 and Appendix C of the BA. The following list includes only those measures specific to avoiding or minimizing potential effects of the proposed action on ESA-listed fish and their habitat, and pertinent to the analysis of this Opinion:

- Herbicide application shall comply with applicable laws (Idaho Statute Title 22, Chapter 34 and Idaho Administrative Code Rule 02.03.03), USFS policy and guidelines (FSH 2109 and FSM 2150), ESA section 7 consultation requirements, National Pollutant Discharge Elimination System (NPDES) permit requirements, and product label directions for the herbicide being used to assure worker safety and to manage potential impacts of herbicide application.

- Always read and follow label directions, including instructions for herbicide use, application rates, equipment and techniques, and personal protective equipment for applicators and mixers, and container disposal.

- Make sure Material Safety and Data Sheets (MSDS), safety plan, spill prevention plan and cleanup kits are available to applicators, per the requirements of FSH 2109.

- Keep accurate and detailed application records, per Idaho Department of Agriculture Rules Governing Pesticide and Chemigation Use and Application (https://agri.idaho.gov/main/chemigation/chemigation-resources-and-information/) and NPDES.

- Perform herbicide applications by or under the direct supervision of licensed Idaho professional herbicide applicators for forest and contract crews, per Idaho Department of Agriculture Rules Governing Pesticide and Chemigation Use and Application.

- No carriers other than water shall be used in the application of herbicides.

- Monitor wind speed/direction and equipment and spray parameters throughout an herbicide application. No herbicide shall be applied in sustained wind conditions exceeding 5 miles per hour (mph) in riparian areas or in any wind conditions exceeding product label directions.

- Herbicide applicators will obtain a weather forecast for the area prior to initiating a spraying project to ensure no extreme precipitation or wind events are predicted to occur during or immediately after spraying that could allow runoff or drift into waterbodies.

- Conduct equipment and personnel inspections and calibration as needed to ensure proper herbicide application and to meet regulatory requirements. Regularly check equipment and components for wear. Attend to repairs and parts replacement promptly.

- Transport only the quantity of herbicide and adjuvants needed for a project. Secure containers being transported in such a way to prevent the likelihood of spills. Make periodic checks en route to help avoid spillage. Carry herbicides and adjuvants in watertight, floatable containers when supplies need to be carried over water by boat, raft or other watercraft.

- When out in the field, use practical measures to restrict access to herbicides and adjuvants and spray equipment by unauthorized personnel.

- Follow the procedures in the FC-RONRW Spill Plan in the event of a spill. Keep the FC-RONRW Spill Plan compliant with the NPDES permit.

- Use indicator dye in the herbicide mix to visually ensure uniform coverage and minimize overlapped or skipped areas and treatment of non-target areas. All colorants used will be non-toxic, water-soluble, liquid formulations.

- Within designated recreation sites, boat ramps, trailheads, campsites and other high use areas, utilize treatments methods that minimize potential exposure to the public.

- To minimize herbicide drift during broadcast operations, use low pressure and larger droplet size to the extent possible with the equipment being used. Use nozzles designed for herbicide application.

- Equip water drafting equipment with back siphoning prevention devices.

- Wherever possible, mix and load at a distance greater than 100 feet from water and where spilled materials will not flow into groundwater, wetlands, or streams.

- No broadcast application methods are used in the FC-RONRW.

- Within 15 feet of the ordinary high water mark (OHWM) of all water bodies, clopyralid, chlorosulfuron, and fluroxypyr may be applied using focused spot spraying or hand selective methods. Hand selective methods consist of wicking, wiping, basal bark, hack and squirt, and/or stem injection applications to individual plants.

- To the extent practicable, apply herbicides to infestations containing biological control agents at times when the effects of herbicides to the host plants would not interfere with the agent's life cycle.

- Use a spray pattern that avoids application of herbicides to non-target species.

- The USFS proposes to use only Washington State Department of Ecology and Agriculture (Washington State) aquatic-certified adjuvants in riparian areas within the FC-RONRW(Washington State 2016).

- Ensure that contracts and agreements include all of these PDC as a minimum.

Table 3 identifies current label-specified direction for ground application of herbicides near live waters, and buffer widths from water. These buffer widths would apply to all perennial streams, intermittent streams, and wetlands.

Table 3. Buffer Widths for Near Water Herbicide Applications in the FC-RONRW.

Active Ingredient	End Use Product[1]	General Application	Buffer Width from Water Using Focused Spot Spraying or Hand Selective Applications[2]
Labeled for Aquatic Use			
Aquatic 2,4-D amine[1]	Weedar 64, Amine 4, Weedestroy AM-40, Weedestroy, Clean Amine, DMA 4	Upland & Riparian	Water Edge[3]
Aquatic Glyphosate[1]	Rodeo, Glypro, Accord Concentrate, AquaMaster, Aquaneat Aquatic, Foresters,	Upland & Riparian	Water Edge[3]

Active Ingredient	End Use Product[1]	General Application	Buffer Width from Water Using Focused Spot Spraying or Hand Selective Applications[2]
	Aquaneet		
Aquatic Imazamox[1]	Clearcast	Riparian	Water Edge[3]
Aquatic Imazapyr[1]	Habitat	Upland & Riparian	Water Edge[3]
Aquatic Triclopyr (TEA)[1]	Garlon 3A	Upland & Riparian	Water Edge[3]
Low Risk to Aquatic Organisms			
Aminopyralid	Milestone	Upland & Riparian	Water Edge[3]
Fluroxypyr	Vista XRT (Ultra)	Upland & Riparian	Water Edge [4/]
Imazapic	Plateau	Upland & Riparian	A level, well-maintained vegetative buffer strip between areas to which this product is applied and surface water[3]
Clopyralid	Transline	Upland & Riparian	Water Edge[4]
Metsulfuron-methyl	Escort XP	Upland & Riparian	Water Edge[3]
Moderate Risk to Aquatic Organisms			
Chlorsulfuron	Telar XP	Upland & Riparian	Water Edge [4/]
Dicamba	Vanquish, Banvel	Upland	Outside riparian vegetation OR 50 feet from water's edge, whichever is greater.[5]
Sulfometuron-methyl	Oust XP	Upland & Riparian	15 feet[6]
High Risk to Aquatic Organisms			
Picloram	Tordon 22K	Upland	Outside riparian vegetation OR 50 feet from water's edge, whichever is greater.[5]

[1] Aquatic formulations of 2,4-D amine, glyphosate, imazamox, imazapyr and triclopyr TEA will be used. Only adjuvants approved for aquatic use by the state of Washington will be used within riparian areas (see Appendix B, Table B-5 for a list of approved adjuvants).
[2] No broadcast application of herbicides will occur in the FC-RONRW. Herbicide application will occur by focused spot spraying or hand selective methods. Focused spot spraying – hand held nozzles attached to back pack tanks and/or hand pumped sprayers to apply herbicide directly onto small patches or individual plants. Hand selective methods – wicking, wiping, basal bark, hack & squirt, stem injections – all on individual plants.
[3] Label direction specifies do not apply chemical directly to water or areas where surface water is present, or intertidal areas below the mean high water mark.
[4] Within 15 feet of the OHWM of all water bodies, clopyralid, chlorosulfuron, and fluroxypyr may be applied using focused spot spraying or hand selective methods. Hand selective methods consist of wicking, wiping, basal bark, hack and squirt, and/or stem injection applications to individual plants.
[5] No applications of dicamba or picloram will be made within riparian areas.
[6] Sulfometuron-methyl will not be applied within 15 feet of any water bodies.

2. ENDANGERED SPECIES ACT: BIOLOGICAL OPINION AND INCIDENTAL TAKE STATEMENT

The ESA establishes a national program for conserving threatened and endangered species of fish, wildlife, plants, and the habitat upon which they depend. As required by section 7(a)(2) of the ESA, each federal agency must ensure that its actions are not likely to jeopardize the continued existence of endangered or threatened species, or adversely modify or destroy their designated critical habitat. Per the requirements of the ESA, federal action agencies consult with NMFS and section 7(b)(3) requires that, at the conclusion of consultation, NMFS provides an Opinion stating how the agency's actions would affect listed species and their critical habitats. If incidental take is reasonably certain to occur, section 7(b)(4) requires NMFS to provide an ITS that specifies the impact of any incidental taking and includes non-discretionary reasonable and prudent measures (RPMs) and terms and conditions to minimize such impacts.

2.1 Analytical Approach

This Opinion includes both a jeopardy analysis and/or an adverse modification analysis. The jeopardy analysis relies upon the regulatory definition of "to jeopardize the continued existence of" a listed species, which is "to engage in an action that 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). Therefore, the jeopardy analysis considers both survival and recovery of the species.

This Opinion relies on the definition of "destruction or adverse modification," which "means a direct or indirect alteration that appreciably diminishes the value of critical habitat for the conservation of a listed species. Such alterations may include, but are not limited to, those that alter the physical or biological features essential to the conservation of a species or that preclude or significantly delay development of such features" (81 FR 7214).

The designations of critical habitat for Snake River Basin steelhead and Snake River spring/summer Chinook salmon use the term primary constituent element (PCE) or essential features. The new critical habitat regulations (81 FR 7414) replace this term with physical or biological features (PBFs). The shift in terminology does not change the approach used in conducting a ''destruction or adverse modification'' analysis, which is the same regardless of whether the original designation identified PCEs, PBFs, or essential features. In this Opinion, we use the term PBF to mean PCE or essential feature, as appropriate for the specific critical habitat.

We use the following approach to determine whether a proposed action is likely to jeopardize listed species or destroy or adversely modify critical habitat:

- Identify the rangewide status of the species and critical habitat expected to be adversely affected by the proposed action.

- Describe the environmental baseline in the action area.

- Analyze the effects of the proposed action on both species and their habitat using an "exposure-response-risk" approach.

- Describe any cumulative effects in the action area.

- Integrate and synthesize the above factors by: (1) Reviewing the status of the species and critical habitat; and (2) adding the effects of the action, the environmental baseline, and cumulative effects to assess the risk that the proposed action poses to species and critical habitat.

- Reach a conclusion about whether species are jeopardized or critical habitat is adversely modified.

- If necessary, suggest a reasonable and prudent alternatives to the proposed action.

2.2 Rangewide Status of the Species and Critical Habitat

This Opinion examines the status of each species that would be adversely affected by the proposed action. The status is determined by the level of extinction risk that the listed species face, based on parameters considered in documents such as recovery plans, status reviews, and listing decisions. This informs the description of the species' likelihood of both survival and recovery. The species status section also helps to inform the description of the species' current "reproduction, numbers, or distribution" as described in 50 CFR 402.02.

This Opinion also examines the condition of critical habitat throughout the designated area, evaluates the conservation value of the various watersheds that make up the designated area, and discusses the current function of the essential PBFs that help to form that conservation value. The designations of critical habitat for Chinook salmon (58 FR 68543) and steelhead (70 FR 52630) identify features essential to the conservation of the species (Table 4).

Table 4. Listing status, status of critical habitat designations and protective regulations, and relevant Federal Register decision notices for ESA-listed species considered in this Opinion.

Species	Listing Status	Critical Habitat	Protective Regulations
Chinook salmon *(Oncorhynchus tshawytscha)*			
Snake River spring/summer-run	T 6/28/05; 70 FR 37160	10/25/99; 64 FR 57399	6/28/05; 70 FR 37160
Sockeye salmon *(O. nerka)*			
Snake River	E 6/28/05; 70 FR 37160	12/28/93; 58 FR 68543	ESA section 9 applies
Steelhead (*O. mykiss*)			
Snake River Basin	T 1/05/06; 71 FR 834	9/02/05; 70 FR 52630	6/28/05; 70 FR 37160

Note: Listing status: 'T' means listed as threatened under the ESA; 'E' means listed as endangered.

2.2.1 Status of the Species

This section describes the present condition of the Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River sockeye salmon evolutionarily significant units (ESUs), and the Snake River Basin steelhead distinct population segment (DPS). NMFS expresses the status of a salmonid ESU or DPS in terms of likelihood of persistence over 100 years (or risk of extinction over 100 years). NMFS uses McElhaney et al.'s (2000) description of a viable salmonid population (VSP) that defines "viable" as less than a 5 percent risk of extinction within 100 years and "highly viable" as less than a 1 percent risk of extinction within 100 years. A third category, "maintained," represents a less than 25 percent risk within 100 years (moderate risk of extinction). To be considered viable an ESU or DPS should have multiple viable populations so that a single catastrophic event is less likely to cause the ESU/DPS to become extinct, and so that the ESU/DPS may function as a metapopulation that can sustain population-level extinction and recolonization processes (ICTRT 2007). The risk level of the ESU/DPS is built up from the aggregate risk levels of the individual populations and major population groups (MPGs) that make up the ESU/DPS.

Attributes associated with a VSP are: (1) Abundance (number of adult spawners in natural production areas); (2) productivity (adult progeny per parent); (3) spatial structure; and (4) diversity. A VSP needs sufficient levels of these four population attributes in order to: safeguard the genetic diversity of the listed ESU or DPS; enhance its capacity to adapt to various environmental conditions; and allow it to become self-sustaining in the natural environment (ICTRT 2007). These viability attributes are influenced by survival, behavior, and experiences throughout the entire salmonid life cycle, characteristics that are influenced in turn by habitat and other environmental and anthropogenic conditions. The present risk faced by the ESU/DPS informs NMFS' determination of whether additional risk will appreciably reduce the likelihood that the ESU/DPS will survive or recover in the wild.

2.2.1.1 Snake River Spring/Summer Chinook Salmon

The Snake River spring/summer Chinook salmon ESU was listed as threatened on April 22, 1992 (57 FR 14653). This ESU occupies the Snake River basin, which drains portions of southeastern Washington, northeastern Oregon, and north/central Idaho. Several factors led to NMFS' conclusion that Snake River spring/summer Chinook were threatened: (1) Abundance of naturally produced Snake River spring and summer Chinook runs had dropped to a small fraction of historical levels; (2) short-term projections were for a continued downward trend in abundance; (3) hydroelectric development on the Snake and Columbia Rivers continued to disrupt Chinook runs through altered flow regimes and impacts on estuarine habitats; and (4) habitat degradation existed throughout the region, along with risks associated with the use of outside hatchery stocks in particular areas (Good et al. 2005). On May 26, 2016, in the agency's most recent 5-year review for Pacific salmon and steelhead, NMFS concluded that the species should remain listed as threatened (81 FR 33468).

Life History. Snake River spring/summer Chinook salmon are characterized by their return times. Runs classified as spring Chinook salmon are counted at Bonneville Dam beginning in early March and ending the first week of June; summer runs are those Chinook adults that pass Bonneville Dam from June through August. Returning adults will hold in deep mainstem and tributary pools until late summer, when they move up into tributary areas and spawn. In general, spring-run type Chinook salmon tend to spawn in higher-elevation reaches of major Snake River tributaries in mid- through late August; and summer-run Chinook salmon tend to spawn lower in Snake River tributaries in late August and September (although the spawning areas of the two runs may overlap).

Spring/summer Chinook spawn follow a "stream-type" life history characterized by rearing for a full year in the spawning habitat and migrating in early to mid-spring as age-1 smolts (Healey 1991). 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. Juveniles rear through the summer, and most overwinter and migrate to sea 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 Chinook salmon return from the ocean to spawn primarily as 4- and 5-year-old fish, after 2 to

3 years in the ocean. A small fraction of the fish return as 3-year-old “jacks,” heavily predominated by males (Good et al. 2005).

Spatial Structure and Diversity. The Snake River ESU includes all naturally spawning populations of spring/summer Chinook in the mainstem Snake River (below Hells Canyon Dam) and in the Tucannon River, Grande Ronde River, Imnaha River, and Salmon River subbasins (57 FR 23458), as well as the progeny of 15 artificial propagation programs (70 FR 37160). The hatchery programs include the South Fork Salmon River (McCall Hatchery), Johnson Creek, Lemhi River, Pahsimeroi River, East Fork Salmon River, West Fork Yankee Fork Salmon River, Upper Salmon River (Sawtooth Hatchery), Tucannon River (conventional and captive broodstock programs), Lostine River, Catherine Creek, Lookingglass Creek, Upper Grande Ronde River, Imnaha River, and Big Sheep Creek programs. The historical Snake River ESU likely also included populations in the Clearwater River drainage and extended above the Hells Canyon Dam complex.

Within the Snake River ESU, the Interior Columbia Technical Recovery Team (ICTRT) identified 28 extant and four extirpated or functionally extirpated populations of spring/summer-run Chinook salmon, listed in Table 5 (ICTRT 2003; McClure et al. 2005). The ICTRT aggregated these populations into five MPGs: Lower Snake River, Grande Ronde/Imnaha Rivers, South Fork Salmon River, Middle Fork Salmon River, and Upper Salmon River. For each population, Table 5 shows the current risk ratings that the ICTRT assigned to the four parameters of a VSP (spatial structure, diversity, abundance, and productivity).

Spatial structure risk is low to moderate for most populations in this ESU (NWFSC 2015) and is generally not preventing the recovery of the species. Spring/summer Chinook salmon spawners are distributed throughout the ESU albeit at very low numbers. Diversity risk, on the other hand, is somewhat higher, driving the moderate and high combined spatial structure/diversity risks shown in Table 5 for some populations. Several populations have a high proportion of hatchery-origin spawners—particularly in the Grande Ronde, Lower Snake, and South Fork Salmon MPGs—and diversity risk will need to be lowered in multiple populations in order for the ESU to recover (ICTRT 2007; ICTRT 2010; NWFSC 2015).

Abundance and Productivity. Historically, the Snake River drainage is thought to have produced more than 1.5 million adult spring/summer Chinook salmon in some years (Matthews and Waples 1991), yet by the mid-1990s counts of wild fish passing Lower Granite Dam dropped to less than 10,000 (IDFG 2007). Wild returns have since increased somewhat but remain a fraction of historic estimates. Between 2005 and 2015, the number of wild adult fish passing Lower Granite Dam annually ranged from 8,808 to 30,338 (IDFG 2016). Natural origin abundance has increased in the recent past for most populations in this ESU, but the increases have not been large enough to change population viability ratings for abundance and productivity; all but one population (Chamberlain Creek) remain at high risk of extinction over the next 100 years (NWFSC 2015). Many populations in Table 5 will need to see increases in abundance and productivity in order for the ESU to recover.

Table 5. Summary of viable salmonid population parameter risks and overall current status for each population in the Snake River spring/summer Chinook salmon ESU (NWFSC 2015).

MPG	Population	VSP Risk Parameter		Overall Viability Rating
		Abundance/ Productivity	Spatial Structure/ Diversity	
South Fork Salmon River (Idaho)	Little Salmon River	*Insf. data*	Low	High Risk
	South Fork Salmon River mainstem	High	Moderate	High Risk
	Secesh River	High	Low	High Risk
	East Fork South Fork Salmon River	High	Low	High Risk
Middle Fork Salmon River (Idaho)	Chamberlain Creek	Moderate	Low	Maintained
	Middle Fork Salmon River below Indian Creek	*Insf. data*	Moderate	High Risk
	Big Creek	High	Moderate	High Risk
	Camas Creek	High	Moderate	High Risk
	Loon Creek	High	Moderate	High Risk
	Middle Fork Salmon River above Indian Creek	High	Moderate	High Risk
	Sulphur Creek	High	Moderate	High Risk
	Bear Valley Creek	High	Low	High Risk
	Marsh Creek	High	Low	High Risk
Upper Salmon River (Idaho)	North Fork Salmon River	*Insf. data*	Low	High Risk
	Lemhi River	High	High	High Risk
	Salmon River Lower Mainstem	High	Low	High Risk
	Pahsimeroi River	High	High	High Risk
	East Fork Salmon River	High	High	High Risk
	Yankee Fork Salmon River	High	High	High Risk
	Valley Creek	High	Moderate	High Risk
	Salmon River Upper Mainstem	High	Low	High Risk
	Panther Creek			*Extirpated*
Lower Snake (Washington)	Tucannon River	High	Moderate	High Risk
	Asotin Creek			*Extirpated*
Grande Ronde and Imnaha Rivers (Oregon/ Washington)	Wenaha River	High	Moderate	High Risk
	Lostine/Wallowa River	High	Moderate	High Risk
	Minam River	High	Moderate	High Risk
	Catherine Creek	High	Moderate	High Risk
	Upper Grande Ronde River	High	High	High Risk
	Imnaha River	High	Moderate	High Risk
	Lookingglass Creek			*Extirpated*
	Big Sheep Creek			*Extirpated*

Note: Shaded cells denote affected populations.

2.2.1.2 Snake River Sockeye Salmon

This ESU includes all anadromous and residual sockeye salmon from the Snake River basin in Idaho, as well as artificially propagated sockeye salmon from the Redfish Lake captive propagation program. The ESU was first listed as endangered under the ESA in 1991, and the listing was reaffirmed in 2005 (70 FR 37160). Reasons for the decline of this species include high levels of historic harvest, dam construction including hydropower development on the Snake and Columbia Rivers, water diversions and water storage, predation on juvenile salmon in the mainstem river migration corridor, and active eradication of sockeye from some lakes in the 1950s and 1960s (56 FR 58619; ICTRT 2003). On May 26, 2016, in the agency's most recent

5-year review for Pacific salmon and steelhead, NMFS concluded that the species should remain listed as endangered (81 FR 33468).

Life History. Snake River sockeye salmon adults enter the Columbia River primarily during June and July, and arrive in the Sawtooth Valley peaking in August. The Sawtooth Valley supports the only remaining run of Snake River sockeye salmon. The adults spawn in lakeshore gravels, primarily in October (Bjornn et al. 1968). Eggs hatch in the spring between 80 and 140 days after spawning. Fry remain in the gravel for 3 to 5 weeks, emerge from April through May, and move immediately into the lake. Once there, juveniles feed on plankton for 1 to 3 years before they migrate to the ocean, leaving their natal lake in the spring from late April through May (Bjornn et al. 1968). Snake River sockeye salmon usually spend 2 to 3 years in the Pacific Ocean and return to Idaho in their 4th or 5th year of life.

Spatial Structure and Diversity. Within the Snake River ESU, the ICTRT identified historical sockeye salmon production in five Sawtooth Valley lakes, in addition to Warm Lake and the Payette Lakes in Idaho and Wallowa Lake in Oregon (ICTRT 2003). The sockeye runs to Warm, Payette, and Wallowa Lakes are now extinct, and the ICTRT identified the Sawtooth Valley lakes as a single MPG for this ESU. The MPG consists of the Redfish, Alturas, Stanley, Yellowbelly, and Pettit Lake populations (ICTRT 2007). The only extant population is Redfish Lake, supported by a captive broodstock program. Hatchery fish from the Redfish Lake captive propagation program have also been outplanted in Alturas and Pettit Lakes since the mid-1990s in an attempt to reestablish those populations (Ford 2011). With such a small number of populations in this MPG, increasing the number of populations would substantially reduce the risk faced by the ESU (ICTRT 2007). The Northwest Fisheries Science Center (2015) reports some evidence of very low levels of early-timed returns in some recent years from outmigrating naturally-produced Alturas Lake smolts, but the ESU remains at high risk for spatial structure.

Currently, the Snake River sockeye salmon run is highly dependent on a captive broodstock program operated at the Sawtooth Hatchery and Eagle Hatchery. Although the captive brood program rescued the ESU from the brink of extinction, diversity risk remains high without sustainable natural production (Ford 2011; NWFSC 2015).

Abundance and Productivity. Prior to the turn of the 20th century (ca. 1880), around 150,000 sockeye salmon ascended the Snake River to the Wallowa, Payette, and Salmon River basins to spawn in natural lakes (Evermann 1896, as cited in Chapman et al. 1990). The Wallowa River sockeye run was considered extinct by 1905, the Payette River run was blocked by Black Canyon Dam on the Payette River in 1924, and anadromous Warm Lake sockeye in the South Fork Salmon River basin may have been trapped in Warm Lake by a land upheaval in the early 20th century (ICTRT 2003). In the Sawtooth Valley, the Idaho Department of Fish and Game eradicated sockeye from Yellowbelly, Pettit, and Stanley Lakes in favor of other species in the 1950s and 1960s, and irrigation diversions led to the extirpation of sockeye in Alturas Lake in the early 1900s (ICTRT 2003), leaving only the Redfish Lake sockeye. From 1991 to 1998, a total of just 16 wild adult anadromous sockeye salmon returned to Redfish Lake. These 16 wild fish were incorporated into a captive broodstock program that began in 1992 and has since expanded so that the program currently releases hundreds of thousands of juvenile fish each year in the Sawtooth Valley (Ford 2011).

With the increase in hatchery production, adult returns to Sawtooth Valley have increased, ranging from 272 to 1,579 during a recent 5-year period (2010–2014) (NWFSC 2015). The increased abundance of hatchery reared Snake River sockeye reduces the risk of immediate loss, yet levels of naturally produced sockeye returns remain extremely low (NWFSC 2015). The ICTRT's viability target is at least 1,000 naturally produced spawners per year in each of Redfish and Alturas Lakes and at least 500 in Pettit Lake (ICTRT 2007). Very low numbers of adults survived upstream migration in the Columbia and Snake Rivers in 2015 due to unusually high water temperatures. The implications of this high mortality for the recovery of the species are uncertain and depend on the frequency of similar high water temperatures in future years (NWFSC 2015).

The species remains at high risk across all four risk parameters (spatial structure, diversity, abundance, and productivity). Although the captive brood program has been highly successful in producing hatchery *O. nerka*, substantial increases in survival rates across all life history stages must occur in order to reestablish sustainable natural production (NWFSC 2015). In particular, juvenile and adult losses during travel through the Salmon, Snake, and Columbia River migration corridor continue to present a significant threat to species recovery (NMFS 2015).

2.2.1.3 Snake River Basin Steelhead

The Snake River Basin steelhead was listed as a threatened ESU on August 18, 1997 (62 FR 43937), with a revised listing as a DPS on January 5, 2006 (71 FR 834). This DPS occupies the Snake River basin, which drains portions of southeastern Washington, northeastern Oregon, and north/central Idaho. Reasons for the decline of this species include substantial modification of the seaward migration corridor by hydroelectric power development on the mainstem Snake and Columbia Rivers, and widespread habitat degradation and reduced streamflows throughout the Snake River basin (Good et al. 2005). Another major concern for the species is the threat to genetic integrity from past and present hatchery practices, and the high proportion of hatchery fish in the aggregate run of Snake River Basin steelhead over Lower Granite Dam (Good et al. 2005; Ford 2011). On May 26, 2016, in the agency's most recent 5-year review for Pacific salmon and steelhead, NMFS concluded that the species should remain listed as threatened (81 FR 33468).

Life History. Adult Snake River Basin steelhead enter the Columbia River from late June to October to begin their migration inland. After holding over the winter in larger rivers in the Snake River basin, steelhead disperse into smaller tributaries to spawn from March through May. Earlier dispersal occurs at lower elevations and later dispersal occurs at higher elevations. Juveniles emerge from the gravels in 4 to 8 weeks, and move into shallow, low-velocity areas in side channels and along channel margins to escape high velocities and predators (Everest and Chapman 1972). Juvenile steelhead then progressively move toward deeper water as they grow in size (Bjornn and Rieser 1991). Juveniles typically reside in fresh water for 1 to 3 years, although this species displays a wide diversity of life histories. Smolts migrate downstream during spring runoff, which occurs from March to mid-June depending on elevation, and typically spend 1 to 2 years in the ocean.

Spatial Structure and Diversity. This species includes all naturally-spawning steelhead populations below natural and manmade impassable barriers in streams in the Snake River basin of southeast Washington, northeast Oregon, and Idaho, as well as the progeny of six artificial propagation programs (71FR834). The hatchery programs include Dworshak National Fish Hatchery, Lolo Creek, North Fork Clearwater River, East Fork Salmon River, Tucannon River, and the Little Sheep Creek/Imnaha River steelhead hatchery programs. The Snake River Basin steelhead listing does not include resident forms of *O. mykiss* (rainbow trout) co-occurring with steelhead.

The ICTRT identified 24 extant populations within this DPS, organized into five MPGs (ICTRT 2003). The ICTRT also identified a number of potential historical populations associated with watersheds above the Hells Canyon Dam complex on the mainstem Snake River, a barrier to anadromous migration. The five MPGs with extant populations are the Clearwater River, Salmon River, Grande Ronde River, Imnaha River, and Lower Snake River. In the Clearwater River, the historic North Fork population was blocked from accessing spawning and rearing habitat by Dworshak Dam. Current steelhead distribution extends throughout the DPS, such that spatial structure risk is generally low. For each population in the DPS, Table 6 shows the current risk ratings for the parameters of a VSP (spatial structure, diversity, abundance, and productivity).

The Snake River Basin DPS steelhead exhibit a diversity of life-history strategies, including variations in fresh water and ocean residence times. Traditionally, fisheries managers have classified Snake River Basin steelhead into two groups, A-run and B-run, based on ocean age at return, adult size at return, and migration timing. A-run steelhead predominantly spend 1-year in the ocean; B-run steelhead are larger with most individuals returning after 2 years in the ocean. New information shows that most Snake River populations support a mixture of the two run types, with the highest percentage of B-run fish in the upper Clearwater River and the South Fork Salmon River; moderate percentages of B-run fish in the Middle Fork Salmon River; and very low percentages of B-run fish in the Upper Salmon River, Grande Ronde River, and Lower Snake River (NWFSC 2015). Maintaining life history diversity is important for the recovery of the species.

Diversity risk for populations in the DPS is either moderate or low. Large numbers of hatchery steelhead are released in the Snake River, and the relative proportion of hatchery adults in natural spawning areas near major hatchery release sites remains uncertain. Moderate diversity risks for some populations are thus driven by the high proportion of hatchery fish on natural spawning grounds and the uncertainty regarding these estimates (NWFSC 2015). Reductions in hatchery-related diversity risks would increase the likelihood of these populations reaching viable status.

Abundance and Productivity. Historical estimates of steelhead production for the entire Snake River basin are not available, but the basin is believed to have supported more than half the total steelhead production from the Columbia River basin (Mallet 1974, as cited in Good et al. 2005). Historical estimates of steelhead passing Lewiston Dam (removed in 1973) on the lower Clearwater River were 40,000 to 60,000 adults (Ecovista et al. 2003), and the Salmon River basin likely supported substantial production as well (Good et al. 2005). In contrast, at the time of listing in 1997, the 5-year mean abundance for natural-origin steelhead passing Lower Granite Dam, which includes all but one population in the DPS, was 11,462 adults (Ford 2011). Counts have increased since then, with between roughly 23,000 and 44,000 adult wild steelhead passing Lower Granite Dam in a recent 5-year period (2011–2015) (NWFSC 2015).

Population-specific abundance estimates exist for some but not all populations. Of the populations for which we have data, three (Joseph Creek, Upper Grande Ronde, and Lower Clearwater) are meeting minimum abundance/productivity thresholds and several more have likely increased in abundance enough to reach moderate risk. Despite these recent increases in abundance, the status of many of the individual populations remains uncertain, and four out of the five MPGs are not meeting viability objectives (NWFSC 2015). In order for the species to recover, more populations will need to reach viable status through increases in abundance and productivity.

Table 6. Summary of viable salmonid population parameter risks and overall current status for each population in the Snake River Basin steelhead DPS (NWFSC 2015). Risk ratings with "?" are based on limited or provisional data series.

MPG	Population	VSP RiskParameter		
		Abundance/ Productivity	Spatial Structure/ Diversity	Overall Viability Rating
Lower Snake River	Tucannon River	High?	Moderate	High Risk?
	Asotin Creek	Moderate?	Moderate	Maintained?
Grande Ronde River	Lower Grande Ronde	N/A	Moderate	Maintained?
	Joseph Creek	Very Low	Low	Highly Viable
	Wallowa River	N/A	Low	Maintained?
	Upper Grande Ronde	Low	Moderate	Viable
Imnaha River	Imnaha River	Moderate?	Moderate	Maintained?
Clearwater River (Idaho)	Lower Mainstem Clearwater River*	Moderate?	Low	Maintained?
	South Fork Clearwater River	High?	Moderate	High Risk?
	Lolo Creek	High?	Moderate	High Risk?
	Selway River	Moderate?	Low	Maintained?
	Lochsa River	Moderate?	Low	Maintained?
	North Fork Clearwater River			Extirpated
Salmon River (Idaho)	Little Salmon River	Moderate?	Moderate	Maintained?
	South Fork Salmon River	Moderate?	Low	Maintained?
	Secesh River	Moderate?	Low	Maintained?
	Chamberlain Creek	Moderate?	Low	Maintained?
	Lower Middle Fork Salmon R.	Moderate?	Low	Maintained?
	Upper Middle Fork Salmon R.	Moderate?	Low	Maintained?
	Panther Creek	Moderate?	High	High Risk?
	North Fork Salmon River	Moderate?	Moderate	Maintained?
	Lemhi River	Moderate?	Moderate	Maintained?
	Pahsimeroi River	Moderate?	Moderate	Maintained?
	East Fork Salmon River	Moderate?	Moderate	Maintained?
	Upper Mainstem Salmon R.	Moderate?	Moderate	Maintained?
Hells Canyon	Hells Canyon Tributaries			Extirpated

Note: Shaded cells denote affected populations.
*Current abundance/productivity estimates for the Lower Clearwater Mainstem population exceed minimum thresholds for viability, but the population is assigned moderate risk for abundance/productivity due to the high uncertainty associated with the estimate.

2.2.2 Status of Critical Habitat

In evaluating the condition of designated critical habitat, NMFS examines the condition and trends of PBFs which are essential to the conservation of the ESA-listed species because they support one or more life stages of the species. Proper function of these PBFs is necessary to support successful adult and juvenile migration, adult holding, spawning, incubation, rearing, and the growth and development of juvenile fish. Modification of PBFs may affect freshwater spawning, rearing or migration in the action area. Generally speaking, sites required to support one or more life stages of the ESA-listed species (i.e., sites for spawning, rearing, migration, and foraging) contain PBF essential to the conservation of the listed species (e.g., spawning gravels, water quality and quantity, side channels, or food) (Table 7).

Table 7. Types of sites, essential physical and biological features, and the species life stage each PBF supports.

Site	Essential Physical and Biological Features	Species Life Stage
Snake River Basin Steelhead[a]		
Freshwater spawning	Water quality, water quantity, and substrate	Spawning, incubation, and larval development
Freshwater rearing	Water quantity & floodplain connectivity to form and maintain physical habitat conditions	Juvenile growth and mobility
	Water quality and forage[b]	Juvenile development
	Natural cover[c]	Juvenile mobility and survival
Freshwater migration	Free of artificial obstructions, water quality and quantity, and natural cover[c]	Juvenile and adult mobility and survival
Snake River Spring/Summer Chinook Salmon, Fall Chinook, & Sockeye Salmon		
Spawning & Juvenile Rearing	Spawning gravel, water quality and quantity, cover/shelter (Chinook only), food, riparian vegetation, space (Chinook only), water temperature and access (sockeye only)	Juvenile and adult
Migration	Substrate, water quality and quantity, water temperature, water velocity, cover/shelter, food[d], riparian vegetation, space, safe passage	Juvenile and adult

[a] Additional PBFs pertaining to estuarine, nearshore, and offshore marine areas have also been described for Snake River steelhead and Middle Columbia steelhead. These PBFs will not be affected by the proposed action and have therefore not been described in this Opinion.
[b] Forage includes aquatic invertebrate and fish species that support growth and maturation.
[c] Natural cover includes shade, large wood, log jams, beaver dams, aquatic vegetation, large rocks and boulders, side channels, and undercut banks.
[d] Food applies to juvenile migration only.

Table 8 describes the geographical extent within the Snake River of critical habitat for each of the four ESA-listed salmon and steelhead species. Critical habitat includes the stream channel and water column with the lateral extent defined by the ordinary high-water line, or the bankfull elevation where the ordinary high-water line is not defined. In addition, critical habitat for the salmon species includes the adjacent riparian zone, which is defined as the area within 300 feet of the line of high water of a stream channel or from the shoreline of standing body of water (58 FR 68543). The riparian zone is critical because it provides shade, streambank stability, organic matter input, and regulation of sediment, nutrients, and chemicals.

Table 8. Geographical extent of designated critical habitat within the Snake River for ESA-listed salmon and steelhead.

ESU/DPS	Designation	Geographical Extent of Critical Habitat
Snake River sockeye salmon	58 FR 68543; December 28, 1993	Snake and Salmon Rivers; Alturas Lake Creek; Valley Creek, Stanley Lake, Redfish Lake, Yellowbelly Lake, Pettit Lake, Alturas Lake; all inlet/outlet creeks to those lakes
Snake River spring/summer Chinook salmon	58 FR 68543; December 28, 1993. 64 FR 57399; October 25, 1999.	All Snake River reaches upstream to Hells Canyon Dam; all river reaches presently or historically accessible to Snake River spring/summer Chinook salmon within the Salmon River basin; and all river reaches presently or historically accessible to Snake River spring/summer Chinook salmon within the Hells Canyon, Imnaha, Lower Grande Ronde, Upper Grande Ronde, Lower Snake-Asotin, Lower Snake-Tucannon, and Wallowa subbasins.
Snake River Basin steelhead	70 FR 52630; September 2, 2005	Specific stream reaches are designated within the Lower Snake, Salmon, and Clearwater River basins. Table 21 in the Federal Register details habitat areas within the DPS's geographical range that are excluded from critical habitat designation.

Spawning and rearing habitat quality in tributary streams in the Snake River varies from excellent in wilderness and roadless areas to poor in areas subject to intensive human land uses (NMFS 2015; NMFS 2017). Critical habitat throughout much of the Interior Columbia (which includes the Snake River and the Middle Columbia River) has been degraded by intensive agriculture, alteration of stream morphology (i.e., channel modifications and diking), riparian vegetation disturbance, wetland draining and conversion, livestock grazing, dredging, road construction and maintenance, logging, mining, and urbanization. Reduced summer streamflows, impaired water quality, and reduction of habitat complexity are common problems for critical habitat in non-wilderness areas. Human land use practices throughout the basin have caused streams to become straighter, wider, and shallower, thereby reducing rearing habitat and increasing water temperature fluctuations.

In many stream reaches designated as critical habitat in the Snake River basin, streamflows are substantially reduced by water diversions (NMFS 2015; NMFS 2017). Withdrawal of water, particularly during low-flow periods that commonly overlap with agricultural withdrawals, often increases summer stream temperatures, blocks fish migration, strands fish, and alters sediment transport (Spence et al. 1996). Reduced tributary streamflow has been identified as a major limiting factor for Snake River spring/summer Chinook and Snake River Basin steelhead in particular (NMFS 2017).

Many stream reaches designated as critical habitat for these species are listed on the Clean Water Act 303(d) list for impaired water quality, such as elevated water temperature (IDEQ 2011). Many areas that were historically suitable rearing and spawning habitat are now unsuitable due to high summer stream temperatures, such as some stream reaches in the Upper Grande Ronde. Removal of riparian vegetation, alteration of natural stream morphology, and withdrawal of water for agricultural or municipal use all contribute to elevated stream temperatures. Water quality in spawning and rearing areas in the Snake River has also been impaired by high levels of sedimentation and by heavy metal contamination from mine waste (e.g., IDEQ and EPA 2003; IDEQ 2001).

The construction and operation of water storage and hydropower projects in the Columbia River basin, including the run-of-river dams on the mainstem lower Snake and lower Columbia Rivers, have altered biological and physical attributes of the mainstem migration corridor. These alterations have affected juvenile migrants to a much larger extent than adult migrants. However, changing temperature patterns have created passage challenges for summer migrating adults in recent years, requiring new structural and operational solutions (i.e., cold water pumps and exit "showers" for ladders at Lower Granite and Lower Monumental dams). Actions taken since 1995 that have reduced negative effects of the hydrosystem on juvenile and adult migrants including:

- Minimizing winter drafts (for flood risk management and power generation) to increase flows during peak spring passage;
- Releasing water from storage to increase summer flows;
- Releasing water from Dworshak Dam to reduce peak summer temperatures in the lower Snake River;
- Constructing juvenile bypass systems to divert smolts, steelhead kelts, and adults that fall back over the projects away from turbine units;
- Providing spill at each of the mainstem dams for smolts, steelhead kelts, and adults that fall back over the projects;
- Constructing "surface passage" structures to improve passage for smolts, steelhead kelts, and adults falling back over the projects; and,
- Maintaining and improving adult fishway facilities to improve migration passage for adult salmon and steelhead.

2.2.3 Climate Change Implications for ESA-listed Species and their Critical Habitat

Climate change is affecting aquatic habitat and the rangewide status of Snake River spring/summer Chinook salmon and Snake River Basin steelhead. The U. S. Global Change Research Program reports average warming of about 1.3°F from 1895 to 2011, and projects an increase in average annual temperature of 3.3°F to 9.7°F by 2070 to 2099 (CCSP 2014). Climate change has negative implications for ESA listed anadromous fishes and their habitats in the Pacific Northwest (CIG 2004; Scheuerell and Williams 2005; Zabel et al. 2006; ISAB 2007). According to the Independent Science Advisory Board, these effects will cause the following:

- Warmer air temperatures will result in diminished snowpack and a shift to more winter/spring rain and runoff, rather than snow that is stored until the spring/summer melt season;
- With a smaller snowpack, watersheds will see their runoff diminished earlier in the season, resulting in lower flows in the June through September period, while more

precipitation falling as rain rather than snow will cause higher flows in winter, and possibly higher peak flows; and,

- Water temperatures are expected to rise, especially during the summer months when lower flows co-occur with warmer air temperatures.

These changes will not be spatially homogeneous across the entire Pacific Northwest. Low-lying areas are likely to be more affected. Climate change may have long-term effects that include, but are not limited to, depletion of important cold-water habitat, variation in quality and quantity of tributary rearing habitat, alterations to migration patterns, accelerated embryo development, premature emergence of fry, and increased competition among species.

Climate change is predicted to cause a variety of impacts to Pacific salmon (including steelhead) and their ecosystems (Mote et al. 2003; Crozier et al. 2008a; Martins et al. 2012; Wainwright and Weitkamp 2013). The complex life cycles of anadromous fishes, including salmon, rely on productive freshwater, estuarine, and marine habitats for growth and survival, making them particularly vulnerable to environmental variation. Ultimately, the effects of climate change on salmon and steelhead across the Pacific Northwest will be determined by the specific nature, level, and rate of change and the synergy between interconnected terrestrial/freshwater, estuarine, nearshore, and ocean environments.

The primary effects of climate change on Pacific Northwest salmon and steelhead include:

- Direct effects of increased water temperatures on fish physiology;
- Temperature-induced changes to streamflow patterns;
- Alterations to freshwater, estuarine, and marine food webs; and,
- Changes in estuarine and ocean productivity.

While all habitats used by Pacific salmon will be affected, the impacts and certainty of the change vary by habitat type. Some effects (e.g., increasing temperature) affect salmon at all life stages in all habitats, while others are habitat-specific, such as streamflow variation in freshwater, sea-level rise in estuaries, and upwelling in the ocean. How climate change will affect each stock or population of salmon also varies widely depending on the level or extent of change, the rate of change, and the unique life-history characteristics of different natural populations (Crozier et al. 2008b). For example, a few weeks' difference in migration timing can have large differences in the thermal regime experienced by migrating fish (Martins et al. 2011).

Temperature Effects. Like most fishes, salmon are poikilotherms (cold-blooded animals); therefore, increasing temperatures in all habitats can have pronounced effects on their physiology, growth, and development rates (see review by Whitney et al. 2016). Increases in water temperatures beyond their thermal optima will likely be detrimental through a variety of processes, including increased metabolic rates (and therefore food demand), decreased disease resistance, increased physiological stress, and reduced reproductive success. All of these processes are likely to reduce survival (Beechie et al. 2013; Wainwright and Weitkamp 2013; Whitney et al. 2016).

By contrast, increased temperatures at ranges well below thermal optima (i.e., when the water is cold) can increase growth and development rates. Examples of this include accelerated emergence timing during egg incubation stages, or increased growth rates during fry stages (Crozier et al. 2008a; Martins et al. 2011). Temperature is also an important behavioral cue for migration (Sykes et al. 2009), and elevated temperatures may result in earlier-than-normal migration timing. While there are situations or stocks where this acceleration in processes or behaviors is beneficial, there are also others where it is detrimental (Martins et al. 2012; Whitney et al. 2016).

Freshwater Effects. Climate change is predicted to increase the intensity of storms, reduce winter snow pack at low and middle elevations, and increase snowpack at high elevations in northern areas. Middle and lower-elevation streams will have larger fall/winter flood events and lower late-summer flows, while higher elevations may have higher minimum flows. How these changes will affect freshwater ecosystems largely depends on their specific characteristics and location, which vary at fine spatial scales (Crozier et al. 2008b; Martins et al. 2012). For example, within a relatively small geographic area (the Salmon River basin in Idaho), survival of some Chinook salmon populations was shown to be determined largely by temperature, while in others it was determined by flow (Crozier and Zabel 2006). Certain salmon populations inhabiting regions that are already near or exceeding thermal maxima will be most affected by further increases in temperature and, perhaps, the rate of the increases. The effects of altered flow are less clear and likely to be basin-specific (Crozier et al. 2008b; Beechie et al. 2013). However, flow is already becoming more variable in many rivers, and this increased variability is believed to negatively affect anadromous fish survival more than other environmental parameters (Ward et al. 2015). It is likely this increasingly variable flow is detrimental to multiple salmon and steelhead populations, and also to other freshwater fish species in the Columbia River basin.

Stream ecosystems will likely change in response to climate change in ways that are difficult to predict (Lynch et al. 2016). Changes in stream temperature and flow regimes will likely lead to shifts in the distributions of native species and provide “invasion opportunities” for exotic species. This will result in novel species interactions, including predator-prey dynamics, where juvenile native species may be either predators or prey (Lynch et al. 2016; Rehage and Blanchard 2016). How juvenile native species will fare as part of “hybrid food webs,” which are constructed from natives, native invaders, and exotic species, is difficult to predict (Naiman et al. 2012).

Estuarine Effects. In estuarine environments, the two big concerns associated with climate change are rates of sea level rise and water temperature warming (Wainwright and Weitkamp 2013; Limburg et al. 2016). Estuaries will be affected directly by sea-level rise: as sea level rises, terrestrial habitats will be flooded and tidal wetlands will be submerged (Kirwan et al. 2010; Wainwright and Weitkamp 2013; Limburg et al. 2016). The net effect on wetland habitats depends on whether rates of sea-level rise are sufficiently slow that the rates of marsh plant growth and sedimentation can compensate (Kirwan et al. 2010).

Due to subsidence, sea-level rise will affect some areas more than others, with the largest effects expected for the lowlands, like southern Vancouver Island and central Washington coastal areas (Verdonck 2006; Lemmen et al. 2016). The widespread presence of dikes in Pacific Northwest estuaries will restrict upward estuary expansion as sea levels rise, likely resulting in a near-term loss of wetland habitats (Wainwright and Weitkamp 2013). Sea-level rise will also result in greater intrusion of marine water into estuaries, resulting in an overall increase in salinity, which will also contribute to changes in estuarine floral and faunal communities (Kennedy 1990). While not all anadromous fish species are highly reliant on estuaries for rearing, extended estuarine use may be important in some populations (Jones et al. 2014), especially if stream habitats are degraded and become less productive. Preliminary data indicate that some Snake River Basin steelhead smolts actively feed and grow as they migrate between Bonneville Dam and the ocean (Beckman 2018), suggesting that estuarine habitat is important for this DPS.

Marine Effects. In marine waters, increasing temperatures are associated with observed and predicted poleward range expansions of fish and invertebrates in both the Atlantic and Pacific Oceans (Lucey and Nye 2010; Asch 2015; Cheung et al. 2015). Rapid poleward species shifts in distribution in response to anomalously warm ocean temperatures have been well documented in recent years, confirming this expectation at short time scales. Range extensions were documented in many species from southern California to Alaska during unusually warm water associated with “the blob” in 2014 and 2015 (Bond et al. 2015; Di Lorenzo and Mantua 2016) and past strong El Niño events (Pearcy 2002; Fisher et al. 2015). For example, recruitment of the introduced European green crab (*Carcinus maenas*) increased in Washington and Oregon waters during winters with warm surface waters, including 2014 (Yamada et al. 2015). Similarly, the Humboldt squid (*Dosidicus gigas*) dramatically expanded its range northward during warm years of 2004–09 (Litz et al. 2011). The frequency of extreme conditions, such as those associated with El Niño events or “blobs” is predicted to increase in the future (Di Lorenzo and Mantua 2016), further altering food webs and ecosystems.

Expected changes to marine ecosystems due to increased temperature, altered productivity, or acidification will have large ecological implications through mismatches of co-evolved species and unpredictable trophic effects (Cheung et al. 2015; Rehage and Blanchard 2016). These effects will certainly occur, but predicting the composition or outcomes of future trophic interactions is not possible with current models.

Wind-driven upwelling is responsible for the extremely high productivity in the California Current ecosystem (Bograd et al. 2009; Peterson et al. 2014). Minor changes to the timing, intensity, or duration of upwelling, or the depth of water-column stratification, can have dramatic effects on the productivity of the ecosystem (Black et al. 2015; Peterson et al. 2014). Current projections for changes to upwelling are mixed: some climate models show upwelling unchanged, but others predict that upwelling will be delayed in spring, and more intense during summer (Rykaczewski et al. 2015). Should the timing and intensity of upwelling change in the future, it may result in a mismatch between the onset of spring ecosystem productivity and the timing of salmon entering the ocean, and a shift toward food webs with a strong sub-tropical component (Bakun et al. 2015).

Columbia River anadromous fishes also use coastal areas of British Columbia and Alaska and midocean marine habitats in the Gulf of Alaska, although their fine-scale distribution and marine ecology during this period are poorly understood (Morris et al. 2007; Pearcy and McKinnell 2007). Increases in temperature in Alaskan marine waters have generally been associated with increases in productivity and salmon survival (Mantua et al. 1997; Martins et al. 2012), thought to result from temperatures that are normally below thermal optima (Gargett 1997). Warm ocean temperatures in the Gulf of Alaska are also associated with intensified downwelling and increased coastal stratification, which may result in increased food availability to juvenile salmon along the coast (Hollowed et al. 2009; Martins et al. 2012). Predicted increases in freshwater discharge in British Columbia and Alaska may influence coastal current patterns (Foreman et al. 2014), but the effects on coastal ecosystems are poorly understood.

In addition to becoming warmer, the world's oceans are becoming more acidic as increased atmospheric carbon dioxide is absorbed by water. The North Pacific is already acidic compared to other oceans, making it particularly susceptible to further increases in acidification (Lemmen et al. 2016). Laboratory and field studies of ocean acidification show that it has the greatest effects on invertebrates with calcium-carbonate shells, and has relatively little direct influence on finfish; see reviews by Haigh et al. (2015) and Mathis et al. (2015). Consequently, the largest impact of ocean acidification on salmon will likely be the influence on marine food webs, especially the effects on lower trophic levels (Haigh et al. 2015; Mathis et al. 2015). Marine invertebrates fill a critical gap between freshwater prey and larval and juvenile marine fishes, supporting juvenile salmon growth during the important early-ocean residence period (Daly et al. 2009, 2014).

Uncertainty in Climate Predictions. There is considerable uncertainty in the predicted effects of climate change on the globe as a whole, and on the Pacific Northwest in particular. Many of the effects of climate change (e.g., increased temperature, altered flow, coastal productivity, etc.) will have direct impacts on the food webs that species rely on in freshwater, estuarine, and marine habitats to grow and survive. Such ecological effects are extremely difficult to predict even in fairly simple systems, and minor differences in life-history characteristics among stocks of salmon may lead to large differences in their response (e.g. Crozier et al. 2008b; Martins et al. 2011, 2012). This means it is likely that there will be "winners and losers," meaning some salmon populations may enjoy different degrees or levels of benefit from climate change while others will suffer varying levels of harm. Climate change is expected to impact anadromous fishes during all stages of their complex life cycle. In addition to the direct effects of rising temperatures, indirect effects include alterations in flow patterns in freshwater and changes to food webs in freshwater, estuarine, and marine habitats. There is high certainty that predicted physical and chemical changes will occur; however, the ability to predict bio-ecological changes to fish or food webs in response to these physical/chemical changes is extremely limited, leading to considerable uncertainty. In additional to physical and biological effects, there is also the question of indirect effects of climate change and whether human "climate refugees" will move into the range of salmon and steelhead, increasing stresses on their respective habitats (Dalton et al. 2013; Poesch et al. 2016).

Summary. The 20-year duration of the proposed action will likely occur while climate change-related effects are expected to become more evident within the range of the Snake River spring/summer Chinook salmon ESU and the Snake River Basin steelhead DPS. Climate change is expected to impact Pacific Northwest anadromous fishes during all stages of their complex life cycle. In addition to the direct effects of rising temperatures, indirect effects include alterations in stream-flow patterns in freshwater and changes to food webs in freshwater, estuarine, and marine habitats. There is high certainty that predicted physical and chemical changes will occur; however, the ability to predict bio-ecological changes to fish or food webs in response to these physical/chemical changes is extremely limited, leading to considerable uncertainty. As we continue to deal with a changing climate, management actions may help alleviate some of the potential adverse effects (e.g., hatcheries serving as a genetic reserve and source of abundance for natural populations, increased riparian vegetation to control water temperatures, etc.).

Climate change is expected to make recovery targets for Chinook salmon and steelhead populations more difficult to achieve. Climate change is expected to alter critical habitat by generally increasing temperature and peak flows and decreasing base flows. Although changes will not be spatially homogenous, effects of climate change are expected to decrease the capacity of critical habitat to support successful spawning, rearing, and migration. Habitat actions can address the adverse impacts of climate change on Chinook salmon and steelhead. Examples include restoring connections to historical floodplains and freshwater and estuarine habitats to provide fish refugia and areas to store excess floodwaters, protecting and restoring riparian vegetation to ameliorate stream temperature increases, and purchasing or applying easements to lands that provide important cold water habitat and cold water refugia (Battin et al. 2007; ISAB 2007).

2.3 Action Area

"Action area" means 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). For purposes of this consultation, the action area consists of all streams within the FC-RONRW boundary. This area includes the watersheds where weed control treatments could occur within the wilderness boundary and the water bodies within and downstream from the treatment area, where herbicides from the proposed action may be detectable in the water.

The action area is used by all freshwater life history stages of threatened Snake River spring/summer Chinook salmon and Snake River Basin steelhead. It also includes migratory life stages of endangered Snake River sockeye salmon. Streams within the action area are designated critical habitat for sockeye salmon, Chinook salmon, and steelhead (Table 8). Critical habitat for Snake River sockeye salmon is designated in the mainstem Salmon River and 300 feet upslope from its ordinary high water line. Designated critical habitat for the Snake River spring/summer Chinook salmon includes all river reaches presently or historically accessible to the species (64 FR 57399) as well as out to 300 feet from the ordinary high water line. Designated critical habitat for Snake River Basin steelhead includes specific reaches of streams and rivers, as published in the Federal Register (70 FR 52630). The action area, except for areas above natural barriers to fish passage, is also EFH for Chinook salmon (PFMC 1999), and is in an area where environmental effects of the proposed project may adversely affect EFH for this species.

2.4 Environmental Baseline

The "environmental baseline" includes the past and present impacts of all federal, state, 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).

NMFS describes the environmental baseline in terms of the biological requirements for habitat features and processes necessary to support all life stages of each listed species within the action area. Spring/summer Chinook salmon, sockeye salmon, and steelhead occur in the action area. Thus, for this action area, the biological requirements for spring/summer Chinook salmon, sockeye salmon, and steelhead are the habitat characteristics that support successful completion of spawning, rearing, and freshwater migration of all three species.

The environmental baseline information submitted by the Forests' in their BA provides a thorough and updated description of the environmental baseline. The following discussion on environmental baseline conditions is presented similar as it appears in the Forests' final BA, dated June 10, 2019.

Given the size of the action area, spread across roughly 3 million acres, this narrative provides a broad-scale description of the environmental baseline, with watershed-specific descriptions of natural physical characteristics, human-caused physical characteristics, cumulative watershed effects, specific stream and river characteristics, and habitat condition, trend, and limiting factors presented by project area subbasin (4th Field Hydrologic Unit). Subbasin baseline descriptions are organized under their respective river basins to assist in evaluating current conditions for an entire river basin.

Upper Salmon River Subbasin (Hydrologic Unit 17060201)

The Upper Salmon River subbasin is estimated to be 1,552,016 acres and contains 18 fifth-level hydrologic units and 81 sixth-level hydrologic units. Approximately 35 percent of the subbasin (537,376 acres) is administered by the SCNF and managed by the Challis-Yankee Fork Ranger District. Another 35 percent of the subbasin, including portions of the Sawtooth Wilderness, is administered by the Sawtooth National Forest, with 24 percent administered by the Bureau of Land Management and 2 percent in scattered state lands. The subbasin includes the communities of Challis, Clayton, and Stanley. Virtually all of the private and state inholdings lie along the Salmon River or Valley Creek corridors.

There are approximately 5,700 miles of streams in the Upper Salmon River subbasin. The largest tributary within the subbasin is the East Fork Salmon River, portions of which are administered by the SCNF. Other major tributaries to the Salmon River in this subbasin include the Yankee Fork Salmon River, Kinnikinic Creek, Slate Creek, Thompson Creek, Garden Creek, Challis Creek, Squaw Creek, Bayhorse Creek, Peach Creek, Basin Creek, and Warm Springs Creek.

The dominant land uses within the subbasin are livestock grazing, mining, developed and dispersed recreation, and fish habitat protection and restoration. Livestock grazing is widespread throughout the subbasin, and has been a constant land use for over a century. The subbasin lowlands are primarily used for cattle grazing with a few upper rangeland areas grazed by sheep. Mining is very important to the history and the economy of the subbasin with 0.45 percent of the nation's mining occurring there. There are several historic mining districts in the subbasin, including the ghost towns of Custer, Bonanza, Sunbeam, and Bayhorse.

Watershed Conditions

Watershed conditions in the subbasin are influenced by actions on both federal and non-federal land. Conditions in this subbasin have been influenced by livestock grazing (primarily cattle), irrigation dams and ditch networks, residential development, recreation, road construction and maintenance, timber harvest, noxious weed infestation and wildfire. These actions have resulted in sediment delivery to streams, altered riparian vegetation, loss of potential wood sources, altered stream channels and flows and elimination of connectivity and access.

Most subwatersheds in the subbasin exhibit low to moderate watershed vulnerability. Most watersheds exhibit low road density, but some high to very high density areas exist (USFS 2008). Valley bottom roads are present in some areas.

Placer and hydraulic mining were prevalent in the subbasin in the late 19th century. Long reaches (approximately 13 miles) of the Yankee Fork Salmon River and its tributary Jordan Creek are listed in the Idaho 303(d) impaired streams lists as "habitat altered," due to mining that occurred with a large floating dredge during the mid-20th century and the presence of dredge spoils along the riverbanks.

In recent years, the occurrence of several large wildfires has dramatically increased the subbasin's overall equivalent clearcut area (ECA). Notable among these were the Halstead Fire, which burned 72,000 acres within the subbasin in 2012, and the Lodgepole Fire, which burned 22,750 acres in 2013. Since 1998, approximately 180,000 acres have burned within the Upper Salmon River subbasin, representing about 28 percent of the total subbasin area. Mountain pine beetle (Dendroctonus ponderosae) infestations, primarily within the upper portions of the basin, have reached epidemic levels, and could fuel additional stand replacing fires in coming years.

Overall, watershed conditions in the Upper Salmon River subbasin no longer reflect natural conditions within the range of desired conditions, and are Functioning at Risk. This determination reflects a downgrade from a previous determination of Functioning Appropriately for the wilderness portions of the subbasin.

Water Quality

Surface water quality varies throughout the subbasin and is dependent on land uses, local geology, and discharge. Most surface water originates in the high mountainous areas above the principal rivers and is of high quality near the source. Some localized areas, however, have accelerated sediment impacts, increased water temperatures, and stream channel and flow alteration from roads, developed and dispersed recreation, livestock grazing, and irrigation diversions. These localized effects can be individually severe.

The subbasin includes a Total Maximum Daily Load (TMDL) for sediment for the Challis Creek Watershed (IDEQ 2003), and the current Idaho 303(d) list identifies Challis Creek, Garden Creek, Squaw Creek, Aspen Creek, Bruno Creek, Basin Creek, Valley Creek, Meadow Creek, Alturas Creek, Champion Creek, Williams Creek, Slate Creek, Big Lake Creek, Road Creek, Mosquito Creek, Warm Spring Creek, Broken Wagon Creek, the East Fork Salmon River, the Yankee Fork, and sections of the main Salmon River as impaired (IDEQ 2010). Pollution sources are linked to both Forest and private land activities within the respective drainages and include sedimentation/siltation, water temperature, and combined biota/habitat impacts. Water quality within the subbasin is Functioning at Risk because of the above factors.

Flow/Hydrology

The ECA and road densities affect flow and hydrologic characteristics. The ECA plays a major role in the ability of the watershed to hold water. While road densities are low within most subwatersheds, overall ECA has increased significantly in the subbasin in recent years in association with a number of large wildfire events.

This subbasin contains many small irrigation dams, diversions, and impoundments on both public and private land. In addition to affecting access, culverts and irrigation diversions within the subbasin have altered flow patterns. Approximately half of the subbasin's subwatersheds exhibit flow disruptions. Flow and hydrology condition within the subbasin is functioning at risk because of the human-caused flow pattern alterations and other disturbances.

Middle Salmon-Panther Subbasin (Hydrologic Unit 17060203)

The Middle Salmon-Panther subbasin is estimated to be 1,166,537 acres, with 990,978 acres (85 percent) administered by the SCNF. National Forest System (NFS) lands are managed by the Salmon-Cobalt and North Fork Ranger Districts. The subbasin contains 13 fifth-level hydrologic units and 64 sixth-level hydrologic units, and supports approximately 1,958 miles of streams. Approximately 53,435 acres of the 2.3 million acre FC-RONRW lie within the Panther Creek portion of the subbasin.

Primary streams within the subbasin include the mainstem Salmon River, Panther Creek, North Fork Salmon River, Hat Creek, McKim Creek, Iron Creek, Williams Creek, Twelvemile Creek, Carmen Creek, Tower Creek, Fourth of July Creek, Pine Creek, Indian Creek, Squaw Creek, Spring Creek, Owl Creek and Colson Creek. The mainstem Salmon River is a designated National Recreational River below the confluence of the North Fork.

The communities of Salmon, North Fork, Shoup, Gibbonsville, Elk Bend, and Ellis are located within the Middle Salmon-Panther subbasin. Most of the population is concentrated within the Salmon River Valley.

Lowland areas along the Salmon River are used mostly for agriculture, primarily livestock and hay production. Agriculture has been a major part of the economic base of the Middle Salmon-Panther subbasin since the beginning of the century and continues to be a key industry in the area. Upland areas are forested and generally encompass NFS lands.

Watershed Conditions

Watershed conditions within the Middle Salmon-Panther subbasin are influenced by actions on both federal and non-federal lands. Conditions in this watershed have been influenced by livestock grazing, past and current mining, irrigation dams and ditch networks, residential development, recreation, road construction and maintenance, timber harvest, noxious weed infestation and wildfire.

Watersheds within the subbasin generally display low to moderate watershed vulnerability, with relatively few classified as high. Outside of wilderness areas within the western portions of the Panther Creek drainage, watersheds of the subbasin typically exhibit high road densities, and many streams have roads located within their valley bottoms. Legacy mining has produced significant impacts to watershed conditions in both the Panther Creek and North Fork portions of the subbasin. The ECAs are variable across the subbasin's watersheds, ranging from only several percent to over 50 percent. Wildfire activity has been significant within the subbasin, and the occurrence of several large wildfires has dramatically increased the subbasin's overall ECA. The Clear Creek Fire burned approximately 165,000 acres in the Panther Creek drainage in 2000, and the Salt Fire burned an additional 24,500 acres within the drainage in 2011. The 2012 Mustang Fire burned 165,500 acres within mainstem Salmon River tributaries in the northern portions of the subbasin. Since 1998, approximately 394,700 acres have burned within the subbasin, representing about 37 percent of the total subbasin area.

Overall, watershed conditions in the Middle Salmon-Panther subbasin no longer reflect natural conditions within the range of desired conditions, and are functioning at risk. This determination reflects a downgrade from a previous determination of functioning appropriately.

Water Quality

Surface water quality varies throughout the subbasin and is dependent on current and past land uses, local geology, and discharge. Most surface water originates in the high mountainous areas above the principal rivers and is of high quality near the source. Some localized areas, however, have accelerated sediment impacts, chemical contamination, increased water temperatures, and stream channel and flow alteration resulting legacy mining activities, roads, developed and dispersed recreation, livestock grazing, and irrigation diversions. These localized effects have, in some cases, had widespread and severe impacts on the fisheries resources of the subbasin.

Legacy mining activities within middle reaches of Panther Creek have produced long lasting and widespread heavy metal contamination which is now being remediated through Comprehensive Environmental Response, Compensation, and Liability Act activities in the drainage. Past mining activities have also impacted habitats within portions the North Fork Salmon River, and produced a catastrophic erosion event within the Dump Creek drainage.

The Idaho 303(d) list identifies Big Deer Creek, South Fork Big Deer Creek, sections of mainstem Panther Creek, Trail Creek, Wallace Creek, Cow Creek and multiple segments of the mainstem Salmon River as impaired waters within the Middle Salmon-Panther subbasin (IDEQ 2010). Pollutant sources include metals, sediment, and general biotia/habitat condition. Pollution sources are linked to both Forest Service and private land activities within the respective drainages. A TMDL for phosphorus has been developed for Williams Lake (IDEQ 2001).

Overall water quality within the Middle Salmon-Panther subbasin is generally functioning at risk as indicated by 303(d) designations of both tributary streams and the mainstem river.

Flow/Hydrology

Overall flow/hydrology conditions within the subbasin are functioning at risk. The ECA and road densities have likely had significant influence on the subbasin's flow and hydrologic characteristics. Many areas of the subbasin exhibit high road densities, and large wildfires in 2000 (Clear Creek) and 2012 (Mustang) have dramatically increased the subbasin's overall ECA and produced observable changes to both peak and base flows. Irrigation diversions, originating on both federal and non-federal lands, are scattered throughout the subbasin and have local impacts to natural flow regimes of some tributary streams.

Upper Middle Fork Salmon River Subbasin (Hydrologic Unit 17060205)

The Upper Middle Fork Salmon River subbasin is an estimated 960,929 acres in size and contains 12 fifth-level and 52 sixth-level hydrologic units. Ninety-nine percent of the land base is comprised of NFS lands. Seventy-nine percent of the subbasin is administered by the SCNF and managed by the Middle Fork Ranger District. Twenty percent of the subbasin is administered by the Boise and Payette National Forests. The remaining one percent of the subbasin is comprised of scattered state and private lands. The vast majority of the subbasin is within the FC-RONRW. The Middle Fork Salmon River (MFSR) is a designated Wild and Scenic River corridor.

Major streams within the Upper Middle Fork Salmon River subbasin include Marsh Creek, Bear Valley Creek, Elk Creek, Elkhorn Creek, Soldier Creek, Rapid River, Pistol Creek, Indian Creek, Marble Creek, Loon Creek, Little Loon Creek and Warm Spring Creek.

Past and ongoing land uses within the subbasin include mining, road construction, irrigation diversion, grazing, timber harvest, fire exclusion/suppression, outdoor recreation, and limited amounts of development. Current activities within wilderness portions of the subbasin have largely been associated with outdoor recreation, with relatively minor private land uses in isolated inholdings.

Watershed Conditions

Because the Forest Service administers nearly all the land in the Upper Middle Fork Salmon River subbasin, watershed conditions are largely influenced by actions authorized or performed by the agency. A large part of the subbasin is designated wilderness, where watershed conditions are generally of high quality and Functioning Appropriately, and within the range of desired conditions. Watershed vulnerability ratings are low throughout the subbasin (USFS 2008). Road density levels are low due to the largely wilderness setting.

Human-caused effects are generally concentrated in non-wilderness areas in the upper portion of the subbasin. There have been localized impacts from roads, mining, past timber harvest, past livestock grazing, and recreation that have resulted in accelerated sediment, stream channel modification, and streambank degradation in some locations. Wildfires have affected both wilderness and non-wilderness lands in recent years. The 2011 Velvet Fire and 2012 Merino Fire collectively burned a total of 10,300 acres within wilderness areas of the subbasin, while the 2012 Halstead Fire burned 96,500 acres of wilderness and non-wilderness lands.

Overall, watershed conditions within the Upper Middle Fork Salmon River subbasin are functioning appropriately, while some watersheds in non-wilderness portions of the subbasin are functioning at risk.

Water Quality

Water quality in the wilderness portion of the subbasin is functioning appropriately. In the non-wilderness portion, water quality is functioning at risk with several 303(d)-listed stream segments. Water quality in these areas is affected by sedimentation from roads, historic mining, past livestock grazing, recreation use, and high naturally occurring sediment levels. Significant recovery from past impacts is occurring due to reduction or elimination of many of the past management activities that contributed to sedimentation (e.g., grazing, mining, and minor amount of timber harvest).

The current Idaho 303(d) list identifies non-wilderness and wilderness sections of Bear Valley Creek as impaired (IDEQ 2010). Pollution sources are linked to both Forest and private land activities, and include water temperature and sedimentation/siltation. Temperature TMDLs have also been developed for Marsh Creek, Knapp Creek, Beaver Creek and Winnemucca Creek (IDEQ 2008).

Flow/Hydrology

Flow/hydrology characteristics are generally Functioning Appropriately in the wilderness, and are functioning at risk in local areas outside wilderness. Overall ECA and road density values are low due to the large amount of subbasin lands within wilderness, and these factors are not having broad scale effects. Watershed-level ECAs have been raised in some areas by wildfire and human disturbance, and there are some possible increases in drainage network associated with roads outside wilderness. There are several scattered irrigation diversions within the subbasin having local flow impacts.

Lower Middle Fork Salmon River Subbasin (Hydrologic Unit 17060206)

The Lower Middle Fork Salmon River subbasin is an estimated 880,275 acres in size, with 498,749 acres (57 percent) administered by the SCNF. Portions of the western area of the subbasin, within the Big Creek drainage, are administered by the Payette National Forest. The vast majority of lands are within the FC-RONRW. Private inholdings are scatted both within and outside wilderness areas. The subbasin contains nine fifth-level hydrologic units and 47 sixth-level hydrologic units.

Major streams within the Lower Middle Fork Salmon River subbasin include Camas Creek, Sheep Creek, Brush Creek, Soldier Creek, Big Creek, Papoose Creek, Stoddard Creek, Warm Spring Creek, Wilson Creek, Waterfall Creek, Ship Island Creek, Roaring Creek and Goat Creek. The MFSR is a designated Wild and Scenic River corridor.

Past and ongoing land uses within the subbasin include mining, road construction, irrigation diversion, grazing, timber harvest, fire exclusion/suppression, outdoor recreation, and limited amounts of development. Current activities within wilderness portions of the subbasin have largely been associated with outdoor recreation.

Watershed Conditions

Because the Forest Service administers nearly all the land base in the Lower Middle Fork Salmon River subbasin, watershed conditions are largely influenced by actions authorized or performed by the agency.

A large part of the Lower Middle Fork Salmon River subbasin is designated wilderness (FC-RONRW). Watershed conditions are generally of high quality and Functioning Appropriately, and within the range of desired conditions. As with the Upper Middle Fork subbasin, watershed vulnerability ratings are low throughout the Lower Middle Fork subbasin (USFS 2008). Overall road density levels are low due to the largely wilderness setting.

Human-caused impacts are generally concentrated in non-wilderness areas in the upper portion the subbasin. Anthropogenic activities have degraded aquatic habitats and affected native fish populations on land outside of wilderness. Land uses including mining, road construction, irrigation diversions, grazing, timber harvest, outdoor recreation, and limited amounts of development have influenced habitat conditions in the watershed, as has wildfire activity. Activities in the watershed have led to elevated levels of instream sediment, altered riparian vegetation, reduced levels of large woody debris, reduced floodplain function and connectivity, altered flows/timing, degraded water quality, and blocked/disconnected fish passage. Legacy mining still dominates significant portions of the Camas Creek watershed landscape and is potentially exposing ESA-listed salmonids and their designated critical habitat to unknown levels of heavy metals, other mining-related pollutants, and acid mine drainage (Kratz et al. 2004). The Ship Island Fire and Papoose Fire collectively burned 12,300 acres of acres of wilderness lands within the Lower Middle Fork Salmon River subbasin between 2011 and 2013.

Overall, watershed conditions within the Lower Middle Fork Salmon River subbasin are Functioning Appropriately, while some watersheds in non-wilderness areas are Functioning at Risk.

Water Quality

Water quality is Functioning Appropriately in wilderness portions of the Lower Middle Fork Salmon River subbasin, and Functioning at Risk in non-wilderness areas due to elevated water temperatures in some streams. There are currently no 303(d) designated streams within the subbasin (IDEQ 2010), but temperature TMDLs have been developed for Camas Creek, Castle Creek, Silver Creek, Duck Creek, and Yellowjacket Creek (IDEQ 2008).

Flow/Hydrology

Flow/hydrology characteristics are generally Functioning Appropriately in the wilderness, and are functioning at risk in local areas outside wilderness. Overall ECA and road density values are low due to the amount of subbasin lands within wilderness, and these factors are not having broad scale effects. Watershed level ECAs have been raised in some areas by wildfire and human disturbance, and there are some possible increases in drainage network associated with roads outside wilderness. There are several scattered irrigation diversions within both wilderness and non-wilderness areas of the subbasin having local flow impacts.

Middle Salmon-Chamberlain Subbasin (Hydrologic Unit 17060207)

The Middle Salmon-Chamberlain subbasin is an estimated 1,095,737 acres in size, and contains 11 fifth-level hydrologic units and 51 sixth-level hydrologic units. Ninety-eight percent of the subbasin is public land administered by the Payette, Nez Perce, Bitterroot and Salmon-Challis National Forests. About 145,433 acres (13 percent) of the total subbasin land base is managed by the SCNF, with only a small fraction of this area, from the MFSR to Wheat Creek, lying outside wilderness.

Major streams within portions of the subbasin administered by the SCNF include Kitchen Creek, Bear Basin Creek, Butts Creek, Corn Creek, Wheat Creek, Horse Creek, and Cottonwood Creek.

Past and ongoing land uses within the subbasin include mining, road construction, irrigation and hydropower diversion, grazing, timber harvest, firewood gathering, outdoor recreation, and limited amounts of development within non-wilderness areas. Current activities within wilderness portions of the subbasin have largely been associated with outdoor recreation, with relatively minor private land uses in isolated inholdings.

Watershed Conditions

Because the Forest Service administers nearly all the land base in the Middle Salmon-Chamberlain subbasin, watershed conditions are largely influenced by actions authorized or performed by the agency.

Overall, watershed conditions within the Middle Salmon-Chamberlain subbasin are Functioning Appropriately. A large part of the subbasin lies within the FC-RONRW, where watershed conditions are generally of high quality and within the range of desired conditions. Overall road density levels are low due to the largely wilderness setting. Watershed vulnerability ratings are generally low within this subbasin (USFS 2008); however, ECAs in some watersheds in the northern portions of the subbasin have been elevated dramatically in recent years by large wildfires. Significant recent fires included the 16,900 acre Saddle Fire of 2011, and the Mustang Fire of 2012, which burned 83,500 acres within the subbasin. Within the wilderness portions of the subbasin, however, these fires reflect a minor portion of the overall subbasin area, and watershed conditions for the wilderness portions of the subbasin remain functioning appropriately.

Human-caused effects are generally concentrated in non-wilderness areas of the subbasin, where watershed conditions are also functioning at risk. Anthropogenic activities have degraded aquatic habitats and affected native fish populations in some portions of these areas. Activities including legacy mining, road building, timber harvest and limited development have impacted aquatic habitats and associated riparian areas.

Water Quality

Water quality is functioning appropriately in both wilderness and non-wilderness portions of the subbasin. A TMDL for water temperature in Crooked Creek has been developed by the Idaho Department of Environmental Quality (2002), but there are currently no 303(d) designated streams identified within the subbasin (IDEQ 2010).

Flow/Hydrology

Flow/hydrology characteristics are generally functioning appropriately within the wilderness, and Functioning at Risk due to some localized impacts in areas outside wilderness. Watershed level ECAs have been raised significantly in some areas by wildfire and human disturbance, and have likely had localized influences on flow and hydrologic characteristics. There are several irrigation and hydropower diversions within both wilderness and non-wilderness areas of the subbasin having local flow impacts.

South Fork Salmon River Subbasin (Hydrologic Unit 17060208)

The South Fork Salmon River subbasin is an estimated 840,054 acres is size and is almost entirely in federal ownership. About 98 percent of the subbasin is administered by the Payette (65 percent) and Boise (33 percent) National Forests. Less than 2 percent of the land base is in non-Forest Service ownership. The majority of the subbasin consists of inventoried roadless areas. About eight percent or 69,099 acres of the subbasin lie within the FC-RONRW.

Major streams within the South Fork Salmon River subbasin include the South Fork Salmon River, the Secesh River, the East Fork South Fork Salmon River, Johnson Creek, Sheep Creek, Elk Creek, Pony Creek, Smith Creek, Porphyry Creek, and Rooster Creek.

Past and ongoing land uses within the subbasin include timber harvest, road construction, mining, grazing, fire exclusion/suppression, outdoor recreation, and limited amounts of development. Current activities within wilderness portions of the subbasin have largely been associated with outdoor recreation.

Watershed Conditions

Because the Forest Service administers nearly all the land base in the subbasin, watershed conditions are largely influenced by actions authorized or performed by the agency.

In the wilderness (FC-RONRW) and roadless portions of the subbasin, watershed conditions are generally of high quality and Functioning Appropriately, and within the range of desired conditions. Watershed vulnerability ratings are low in these areas.

Human-caused impacts and their legacy effects are concentrated in the non-wilderness/roadless portions of the South Fork Salmon River subbasin. In those areas, anthropogenic activities including timber harvest, road construction, mining, grazing, fire exclusion/suppression, and outdoor recreation have degraded aquatic habitats and depressed native fish populations. Roads and mining, in particular, have had lasting legacy effects. Roads built to harvest timber and retained for recreation and fire suppression are the dominant sources of sediment in the subbasin. Mining has played a significant role in the subbasin. The alluvial deposits in and along the South Fork Salmon River, the East Fork South Fork Salmon River, the upper Secesh River and Johnson Creek were placer mined during the latter portion of the nineteenth century and into recent years. The most extensive mining occurred in the Upper East Fork South Fork Salmon River at the Stibnite mine site. Stibnite is now closed and has been reclaimed, but a new plan of operations is being pursued and open pit gold mining may resume in this location in the future. Collectively, anthropogenic activities have resulted in elevated levels of instream sediment, altered riparian vegetation, reduced levels of large woody debris, reduced floodplain function and connectivity, and degraded water quality.

Overall, watershed conditions within the wilderness and roadless portions of the South Fork Salmon River subbasin are Functioning Appropriately, while the watersheds that are located in the non-wilderness/roadless portions of the subbasin are Functioning at Risk.

Water Quality

Water quality is functioning appropriately in the wilderness and roadless portions of the subbasin, and functioning at risk outside of wilderness and roadless areas, primarily due to elevated levels of sediment. There are thirteen 303(d) designated streams within the subbasin: South Fork Salmon River, Johnson Creek, Rice Creek, Trail Creek, Trout Creek, Sand Creek, Warm Lake Creek, Tyndall Creek, Profile Creek, Buckhorn Creek, Lick Creek, Grouse Creek, and Elk Creek (IDEQ, 2002). The pollutant of concern is temperature for all of the listed waterbodies, and temperature and sediment for the South Fork Salmon River.

Flow/Hydrology

Flow/hydrology characteristics are generally functioning appropriately in the wilderness and roadless portions of the subbasin, and functioning at risk in areas outside of wilderness and roadless. Overall ECA and road density values are low due to the amount of subbasin lands that are within wilderness or inventoried roadless areas, and these factors are not having broad scale effects. Watershed level ECAs have been raised in some areas by wildfire and human disturbance.

Upper Selway River Subbasin (Hydrologic Unit 17060301)

The Upper Selway River subbasin is an estimated 631,296 acres in size and contains seven fifth-level hydrologic units and 28 sixth-level hydrologic units. Nearly all of the subbasin is public land administered by the Bitterroot and Nez Perce-Clearwater National Forests. There are only 53 acres of private inholdings scattered throughout the subbasin. All of the Upper Selway River subbasin is designated wilderness (FC-RONRW south of the Magruder Corridor; Selway-Bitterroot north of the Corridor) with the exception of a single, narrow road corridor (the Magruder Corridor) connecting Montana to Elk City, Idaho.

Major streams within the Upper Selway River subbasin include Bear Creek, Goat Creek, Running Creek, Whitecap Creek, Indian Creek, the Little Clearwater River, Deep Creek, Wilkerson Creek, Storm Creek, Swet Creek, Surprise Creek, and Hidden Creek.

Current activities within the subbasin are largely associated with outdoor recreation, with relatively little use occurring on the isolated private inholdings.

Watershed Conditions

Because the Forest Service administers nearly all the land base in the Upper Selway River subbasin, watershed conditions are largely influenced by actions authorized or performed by the agency.

Overall, watershed conditions within the Upper Selway River subbasin are functioning appropriately. Nearly all of the subbasin is designated wilderness. Watershed conditions are being driven by natural forces and are within the range of desired condition. Road density levels are low due to the wilderness setting. Watershed vulnerability ratings are generally low in the subbasin; however, ECAs in the FC-RONRW portion of the subbasin have been elevated dramatically over the past two decades by large wildfires. Significant large fires include the 47,500 acre Swet-Warrior Fire of 1996, the 76,634 acre Wilderness Complex in 2000, the 32,860 Selway-Salmon Wilderness Complex in 2005, the 44,167 acre Mustang/Porcupine Complexes in 2012, and the 39,452 acre Gold Pan Complex in 2013. The end result of these fires is a highly mosaic “patchy” landscape with conifer regeneration ongoing in various stages. Some patches of regeneration have experienced a re-burn since their initial fire, while others have not. Despite the widespread occurrence of fire, the overall watershed condition for the Upper Selway River subbasin is still Functioning Appropriately.

Human-caused impacts (e.g., sediment from roads, dispersed and developed campsites in riparian habitat conservation area) are generally restricted to the edges of the Magruder Corridor road system and have a negligible effect on watershed conditions at the subbasin scale.

Water Quality

Water quality is functioning appropriately in the Upper Selway River subbasin. There are currently no 303(d) designated streams identified within the subbasin (IDEQ 2010).

Flow/Hydrology

Flow/hydrology characteristics are functioning appropriately in the subbasin. Watershed level ECAs have been raised significantly in some areas by wildfire, and have likely had localized influences on flow and hydrologic characteristics. There are four small irrigation diversions associated with private inholdings that have localized flow impacts on a few tributaries to the Selway River.

Herbicide Treatment in the FCRONRW

Since the 2014 Opinion, the SCNF has provided NMFS with several annual reports (for the 2015, 2016, and 2018 spray seasons) summarizing the noxious weed management activities on FC-RONRW lands. The Forests have never come close to exceeding the 6,250 acre limit in past years. The closest that the Forests have come to the 625 acre limit within 100 feet of water is 428 acres in a single year. The total acreage treated with chemicals in the FC-RONRW in 2018 was 148.3 acres. Approximately 61.4 of those acres were treated within 100 feet of live water; and 86.9 acres were treated greater than 100 feet from live water. The 2018 treatments accounted for about 10 percent of the annual acre limit within 100 feet of live water, and about 2 percent of the 6,250 annual acre limit.

2.5 Effects of the Action

Under the ESA, "effects of the action" means the direct and indirect effects of an action on the species or critical habitat, together with the effects of other activities that are interrelated or interdependent with that action, that will be added to the environmental baseline (50 CFR 402.02). Indirect effects are those that are caused by the proposed action and are later in time, but still are reasonably certain to occur.

2.5.1 Effects on ESA-Listed Species

The proposed action involves all noxious weed treatment activities authorized, funded or carried out by the Forest Service through the 2039 treatment season. In general, there are four treatment types with potential to affect ESA-listed species: (1) Manual treatments; (2) biological treatments; (3) rehabilitation and restoration; and (4) chemical treatments. More detailed descriptions of these activities are included in Section 1.3 above. The proposed action included extensive PDC or best management practices (BMP) (Section 1.3) to avoid and/or minimize adverse effects to ESA-listed species during noxious weed management and the following assessment assumes those measures will be implemented as described for all applications.

The BA concluded that chemical weed treatments may affect, and are likely to adversely affect ESA-listed species via temporary chemical contamination of waterways. Manual, biological control, and rehabilitation and restoration treatments are less likely to result in effects to ESA-listed species. However, a summary of the potential effects from these other treatment activities' effects is also presented. The bulk of the analysis will primarily focus on the effects of chemical treatments.

2.5.1.1 Effects of Non-Herbicide Treatments

Manual Treatments. Treatments typically involve hand pulling, grubbing, hoeing, cutting with non-mechanized tools, and torching to physically remove noxious weeds from infested sites. Only minor amounts of ground disturbance occur at individual sites. Commonly, dead plant material from plants that were removed breaks down and covers the soil surface, providing a protective litter layer. However, where this does not occur, increased amounts of bare ground could result in a temporary increase in soil erosion and associated sediment delivery to stream channels, especially during a high intensity rain event. Torching would be conducted as spot applications and will not cause large areas of bare ground. Tilling the ground and other mechanical treatment methods are not included in the proposed methods, further limiting soil disturbance in riparian areas to minimize potential effects.

Treatments will only remove invasive species, retaining native vegetation important for proper ecological function and filtering of sediments that are mobilized during subsequent rainstorms. In localized instances (i.e., larger treatment areas or riparian treatment sites) the Forests will revegetate areas with noxious weed free native seed mix or root stocks if needed. This practice will further augment retained ground cover, minimizing erosion potential and reducing the need for future weed treatments. Considered together, the small size of treatments sites, their wide geographic distribution, and the protective PDC appear sufficient to avoid significant sediment delivery to action area streams. Replanting sensitive sites will support the goal of long-term native vegetation establishment and reduce the need for future weed treatments of all types – supporting natural riparian processes and water quality important for supporting anadromous fish. Removal of invasive plants from riparian areas may also reduce long-term sediment delivery, potentially providing for improved habitat conditions and thus beneficial migration, rearing, and spawning conditions for Chinook, sockeye, and steelhead. For the reasons discussed, manual invasive plant treatments proposed to occur in the action area are expected to have very minor effects on ESA-listed fish, with effects limited to minor sediment delivery in localized areas, with improved native riparian conditions and reduced long-term sediment delivery occurring over time.

Biological Control. The Forests utilizes biological control agents to annually treat approximately 100 acres. Biological release agents receive intensive evaluation by APHIS prior to approval for release. This process is intended to ensure release agents do not have unintended ecological consequences. Only APHIS-approved biological controls will be used and releases will enforce APHIS requirements or USFS policy, whichever is more restrictive. This process should ensure release agents only feed on target species. As a result, native vegetation will not be adversely affected and target noxious weed infestations will be controlled. Control of noxious weed infestations will enable native vegetation to recolonize the affected areas, promoting proper function within the treated areas. No direct effects to ESA-listed species are anticipated. As vegetation recovers there may be minor reductions in sediment delivery and riparian function improvements.

Rehabilitation and Restoration. The objective is to reestablish a desired plant community and return to conditions that foster the recovery of natural ecosystem processes. Natural revegetation is the preferred option whenever possible, but on occasion reseeding will be needed. Equipment that could be used during reseeding activities includes non-mechanized hand tools such as rakes. PDC limiting treated slopes to less than 45 percent and land type erosion hazard ratings to low or moderate would minimize erosion from occurring during and after these treatments. The small amount of acreage treated at any given time, combined with the anticipated effectiveness of proposed PDC (e.g., Minimize ground disturbing activities to the extent possible during reseeding efforts; and conducting rehabilitation and restoration activities only in areas with slope gradients less than 45 percent and areas with low or moderate land type erosion hazard ratings), are expected to ensure that effects related to these types of activities do not reach levels resulting in harm.

Direct disturbance of spawning ESA-listed fish could potentially occur if rehabilitation work occurred directly adjacent to spawning habitat and while fish were spawning. However, treatment areas will be small in size and all ground disturbing activities will only occur with non-motorized hand tools.

2.5.1.2 Effects of Herbicide Treatments

The analysis of the effects of herbicides on salmonids is evaluated in this Opinion by: (1) Assessing the likelihood that ESA-listed fish and other aquatic organisms will be exposed to the herbicides; (2) reviewing the toxicological effects of the herbicides, inert ingredients, and adjuvants on listed fish and other aquatic organisms; and (3) assessing the ecological risk qualitatively based on the exposure risk and toxicity. The analysis considers the:

- Life history stages (and any associated vulnerabilities) of the ESA-listed species present in the action area;

- Known or suspected mechanisms of toxicity for the active ingredients, EUPs, inert ingredients, or known adjuvants;

- The PDCs/BMPs, chemical application rates, amount of chemical use, location, application methods, and other factors that determine the likelihood of chemicals reaching the water; and

- Possibility for antagonistic, additive, or synergistic interactions between active ingredients within formulations and with other chemicals that may enter surface waters as a result of parallel or upstream land use activities.

Under the proposed action, the risks to salmon and steelhead from herbicides are likely to occur primarily through the direct toxicological effects of the herbicides and adjuvants on the fish, rather than physical changes in fish habitat or effects on aquatic vegetation or prey species. However, both types of effects may occur and are considered in this Opinion. Unfortunately, the toxicological effects and ecological risks to aquatic species, including ESA-listed fish, are not fully known for all of herbicides, EUPs, and adjuvants in the proposed action.

Due to concerns about the uncertainty of effects of pesticides on ESA-listed fish, the EPA was directed by the 9th Circuit Court of Appeals (Washington Toxics Coalition v. EPA, 413 F.3d 1024(2005)) to consult with NMFS on the effects of 54 pesticides used in the states of Washington, Oregon, and California. On August 1, 2008, NMFS entered into a settlement agreement to complete consultations on 37 active ingredients. All Opinions for these active ingredients were to be completed on or before June 30, 2013. To date, NMFS has completed seven biological opinions, covering 31 active ingredients (https://www.fisheries.noaa.gov/national/endangered-species-conservation/pesticide-consultations-summary-and-schedule). Of those active ingredients, two (2,4-D and triclopyr) are proposed for use by the Forests. Results of the national consultation on the registration of 2,4-D and triclopyr are incorporated in this Opinion. None of the remaining four active ingredients awaiting EPA consultation are proposed for use in this action.

2.5.1.3 Chemical Exposure Pathways and Mitigating Measures

Water can be contaminated by direct spray, drift, windblown soils, spills and leakage, or through leaching and runoff. The following analysis will discuss the potential for each to occur in detail.

Water Contamination by Direct Spray. Accidental spraying of herbicides into water is expected to occur infrequently and likely only when a ground applicator inadvertently directs the nozzle toward the water to spray weeds located near the water's edge and a portion of the spray stream misses the target plant. Direct spraying of water is likely only if the applicator is not paying attention to where the spray is going, or when weeds are sprayed near water and the nozzle is not adjusted properly. This type of contamination is unlikely to involve significant amounts of chemicals. Marker dyes will be used to monitor where sprays are landing; consequently, operators will be capable of quickly adjusting their aim away from the water if they observe the spray stream hitting the water or streambank. Berg (2004) noted that direct spray or drift of herbicide onto water sources was the most commonly noted BMP failure. However, with proper management, direct application to water can be minimized. The amount of contamination from over-spraying should be reduced by proposed PDC/BMPs that: (1) Require a certified herbicide applicator oversee all spray projects; (2) restrict herbicide application near water to spot-spraying with a single nozzle by hand; (3) require that no broadcast application methods will be used in riparian areas (the transition area between the aquatic ecosystem and the adjacent terrestrial ecosystem; identified by soil characteristics or distinctive vegetation communities that require free or unbound water); (4) check local weather conditions daily, and monitoring site-specific conditions during herbicide application; (5) select the most suitable herbicide and adjuvant (as appropriate) combination for the setting, apply the lowest effective use rates, employ spot spraying techniques in riparian areas, apply herbicide at low pressure, using the largest appropriate nozzle size and other appropriate equipment, adding drift control agents where necessary, and utilizing directional application techniques to direct herbicide away from water; and (6) no herbicide will be applied in sustained wind conditions exceeding 5 mph in riparian areas or in any wind conditions exceeding product label directions. Given these criteria, herbicides that enter water by direct spray are unlikely to reach concentrations that cause more than minor effects since relatively small volumes of herbicides would be involved.

When applied consistently and correctly, these measures are expected to limit direct spray of herbicide into surface waters to infrequent occurrences. With routine compliance with the described PDC/BMPs, direct delivery of herbicide to action area surface waters is expected to be rare, of low intensity when or if it occurs, and isolated to individual project sites that are geographically scattered. As a result, in-water herbicide concentrations resulting from direct spray of herbicide into action are waters should be minor.

The proposed action also included a conservation measure stating the Forests will "ensure that contracts and agreements include all of these design criteria as a minimum." NMFS interpreted this statement as being applicable to all permittees and contractors that apply for and are ultimately authorized to perform any type of invasive plant control on FC-RONRW lands during the 20-year consultation timeframe. Non-USFS agents may be applying herbicide on FC-RONRW lands. Although it is possible applicants may have appropriate backgrounds and knowledge to successfully apply the proposed PDC, such experience is unlikely to be possessed by all applicants. For this reason, there is at least the potential for applicants to be more likely to fail to understand or fail to apply some PDC for some sites. Such failure may lead to increased risk of direct spray. However, the Forests will have the opportunity to review all applicants' future proposals and gage their ability to appropriately implement necessary PDC, as well as evaluate appropriateness of proposed herbicides. For this reason, NMFS believes the Forests can maximize applicants' ability to successfully implement the action, but the BA (USFS 2019) likely could have better articulated how such project reviews would be conducted and identified responsible staff. Such a description would provide more certainty regarding avoidance of direct delivery to water, as well as other delivery pathways. The overall small quantity of projected chemically treated acres in riparian areas, and the anticipated small proportion of applicant treatments likely to occur results in only minor risk of contamination from this pathway. Further, reserved Forest Service discretion over applicant treatments is expected to sufficiently minimize the risk for inadvertent deliveries from this pathway.

Water Contamination by Wind Drift. Rates of contamination by wind drift are largely dependent on droplet size, elevation of the spray nozzle, wind speed, and weather conditions (heat and humidity) that can cause the water droplets to evaporate, leaving the chemicals suspended in the air (Rashin and Graber 1993). During periods when there is virtually no wind, little vertical air mixing occurs, and drift can travel long distances. The complete absence of winds can cause herbicide particles to remain suspended in air for hours. This circumstance is likely to occur infrequently in mountainous regions, due to daily convective cycles. Convective winds occur daily, except under heavy cloud cover; consequently, periods without any wind are rare during the summer weed spraying season. However, they may occur occasionally on cloudy days.

Unfavorable weather conditions can cause high wind drift rates and are likely to be encountered occasionally during periods with low relative humidity and high temperatures, which are common from July through September. During hot and dry conditions, herbicides can quickly volatize when the carrier (i.e., water) evaporates. Even under the most extreme heat and low humidity, volatilization is unlikely to be a significant cause of wind drift, since there is little opportunity for the spray to evaporate because it will be applied by hand with the nozzle close to the target. The proposed action does not include specifications for controlling droplet size during hot and dry conditions, but droplet size is not likely to be a significant factor affecting wind drift when herbicides are applied by hand and with a single nozzle. Drift control agents can also be used where necessary, along with utilizing directional application techniques to direct herbicide away from water.

Occasional wind drift may occur when herbicides are applied near streams. Spray streams typically include a range of droplet sizes, and a fraction of the droplets are small enough to be blown off-target and into the water. The majority of the material in a spray stream delivered by hand with a marker dye typically hits the target, leaving only a fraction of the chemicals subject to drift. Based on proposed design criteria water contamination from wind drift will be further reduced by: (1) Ensuring a certified herbicide applicator oversees all spray projects; (2) restricting herbicide application near water to spot-spraying with a single nozzle by hand;

(3) no herbicide shall be applied in sustained wind conditions exceeding 5 mph in riparian areas or in any wind conditions exceeding product label directions; (4) applicators obtaining a weather forecast prior to initiating spraying to ensure no precipitation or wind events could occur during or immediately after spraying that could allow drift into surface waters; and (5) applicators using a marker dye to provide the operator the ability to accurately see the locations where herbicides are applied.–Considering these factors, the risk of contaminating surface waters from wind-drift contamination is minimal for this action.

Water Contamination by Wind-Blown Soils. Certain herbicides may readily bind to mineral or organic particles in soils, and the contaminated soils can subsequently be blown into streams. Wind-blown soil particles can be a significant source of water contamination in agricultural settings, where large contiguous blocks of land are tilled to bare soils and exposed to winds. This is not the case in the wilderness, where bare soil areas are typically small in the action area and interspersed with abundant vegetative cover which blocks wind and filters dust. These conditions make the contamination risk from wind-blown soils low. In addition, only a small fraction of herbicide contaminants reside on the soil surface, and contaminated soils may not be subjected to winds before herbicides become immobilized through absorption by plants or downward movement into the soils.

The sites most susceptible to generate contamination from wind-blown soils are recently burned areas, trails, drainage ditches, and undeveloped recreation sites next to streams. When treated with herbicides, these types of locations can produce small amounts of herbicide-laden dust that could be blown into streams. However, the amount of contamination from this pathway is likely to be very small and immeasurable unless extensively burned areas are sprayed in the same year in which the fire occurred. Historically, the Forests have not treated extensively burned areas with herbicide. In general, a reseeding strategy is implemented to treat burned areas. Considering these factors, the risk of contaminating surface waters from wind-blown soil contamination is minimal for this action.

Water Contamination by Spills and Leakage. Most of the herbicides in the proposed action will be applied in a liquid solution, which requires transferring liquids from one container to another and field mixing of chemicals. Liquids are prone to spills through leaky spray equipment or containers and when mixing or transferring chemicals from one container to another. In general, minor amounts of herbicide leakage are likely to occur throughout the spray season from dripping while using spray equipment, but this type of leakage would occur at concentrations far below the target application rate, and it would not cause any meaningful increase in water contamination over the amount expected based on the target rate.

Chemical contamination of water involving larger amounts of herbicides from spilled or leaking containers is likely to be an uncommon event because aleak or spill must occur and the spilled chemicals must reach the water. The likelihood of a spill is difficult to predict, but is constrained by PDC/BMPs that limit the amounts of chemicals that are transported at any given time (i.e., transport only the quantity of herbicide and adjuvants needed for a project; secure containers being transported in such a way to prevent the likelihood of spills; make periodic checks en route to help avoid spillage; carry herbicides and adjuvants in watertight, floatable containers when supplies need to be carried over water by boat, raft or other watercraft.). Spilled chemicals reaching water will be restricted by limiting storage and mixing of chemicals to locations where a spill would be too distant from water to reach it before it could be cleaned up (i.e., wherever possible, chemicals will be mixed and loaded at a distance greater than 100 feet from water and where spilled materials will not flow into groundwater, wetlands or streams; and MSDS, safety plan, spill prevention plan and cleanup kits will be available to applicators, per the requirements of FSH 2109.). Consequently, spills and leakage from handling are not likely to be more than a minimal source of water contamination.

Water Contamination from Leaching and Runoff. Post-application rain events can potentially mobilize herbicides deposited on plant tissues or soils through leaching and runoff. The highest herbicide contamination from surface runoff typically occurs in brief pulses during and shortly after storm events; especially when applied close to streams, dry intermittent or ephemeral stream beds, or to drainage ditches that flow directly into stream channels. Runoff from intense rainfall events is believed to be the primary mechanism allowing herbicides to reach streams in the action area. A large percentage of an applied herbicide could be moved off site if intense rainfall occurs shortly after herbicides are applied. These storms tend to be isolated and high intensity events should affect only a fraction of the action area at any given time. Weed infestations are widely scattered across the action area. If the Forests applied herbicides to the annual maximum 6,250 acres, a limited application compared to the approximately 2.3 million acres of wilderness lands, less than 0.27 percent of FC-RONRW managed lands would receive herbicide applications annually. A maximum of 625 treatments acres will occur near water and will be spread out over a large geographical area. Therefore, only a very small proportion of the anadromous part of the action area would be treated annually at any given time, and in any given year (0.027 percent or less).

Physical properties of the herbicides (i.e., movement in groundwater, soil half-life, water solubility, etc.) and environmental conditions (i.e., soil type, precipitation rates, wind, etc.) are the primary variables influencing herbicide movement from runoff. Some of these variables and estimated environmental concentration (EEC), which are a product of water contamination rates (WCRs) multiplied by application rate, are shown in Table 9. Herbicides with a low sorption coefficient and a long half-life, such as chlorsulfuron, dicamba, metsulfuron-methyl, and picloram have the greatest ability to leach through soils and reach groundwater or surface water, with picloram possessing the greatest risk.

Contaminants are filtered out of the water to varying degrees by sorption onto plants, debris, and soils encountered in the flow path. In general, the amount of filtering increases with the distance surface runoff travels before reaching a stream and with increasing dispersal of the flow. Herbicides can be completely removed from surface runoff that trickles a long distance though vegetation and organic debris, while little or no filtering might occur in runoff that is quickly concentrated into a channel or drainage ditch. Due to the effects of filtering and greater residence time, vegetative buffers are generally effective in controlling water contamination (Berg 2004).

Herbicides mobilized by subsurface flows can enter a nearby stream by contaminating the hyporheic aquifer that mixes with surface flows in a stream, or they can contaminate more distant streams if the runoff reaches deeper aquifers and then emerge as springs. Contamination by subsurface flows through the hyporheic zone occurs more gradually over an extended period of time following a storm, with timeframes highly variable, typically ranging from hours to weeks, depending on the soil type, physical properties of the aquifer, and distance from the point of contamination to the stream.

As herbicides move through groundwater, some of the herbicides are lost to chemical breakdown and metabolism by plants and other organisms. Contaminants may also be filtered out of subsurface runoff as the runoff percolates through the soil. Soils with large fractions of fine particles and organic materials have a large filtering capacity and coarse soils that lack organic material have little or no filtering capacity. At some point, distance from water also becomes the dominant influence on the amount of water soluble chemicals that reach a stream, and in general, the greater the distance between treated areas and the water table or nearest stream channel, the smaller the potential for water contamination. Stream buffers and/or spot spraying in riparian areas are used in the proposed action to take advantage of the filtering capacity provided by plants, debris and soils.

A portion of the herbicides applied to the action area may be affected by the processes described above and may contaminate water in concentrations high enough to cause adverse effects to ESA-listed fish or other aquatic organisms. However, water contamination will be minimized by: (1) Applicators obtaining a weather forecast prior to initiating spraying to ensure no precipitation or wind events could occur during or immediately after spraying that could allow runoff or drift into surface waters; (2) limiting the application of certain chemicals (i.e., picloram, clopyralid, chlorsulfuron, imazapic, etc.), both in distance from streams and number of applications, to minimize the potential of waterditch contamination and applying no-spray buffer zones along all streams and ponded waterbodies for those herbicides not labeled for aquatic or streamside application; (3) not using broadcast application methods in riparian areas; (4) using spot spraying techniques in riparian areas, applying herbicide at low pressure, using the largest appropriate nozzle size and other appropriate equipment, adding drift control agents where necessary; (5) selecting the most suitable herbicide and adjuvants (as appropriate) combination for the target species and environmental setting and applying the lowest effective use rates; and (6) not storing, mixing, or cleaning herbicides within 100 feet of any live waters or over shallow groundwater areas. Considering application of these PDC, we believe that surface runoff or leaching will occur infrequently, be of small magnitude, be widely spaced geographically, and produce very low herbicide concentrations.

Reduction of Exposure Risks. In addition to the PDC/BMPs mentioned in the previous sections, the Forests proposed several additional measures as part of their general weed management program to minimize the potential for contaminating action area waterbodies. Key criteria include the following:

Only low risk application methods will be used near water. This approach substantially eliminates the risk of directly spraying chemicals into water or dry stream channels. Applicators are required to use more risk-averse application methods in sites that are close to stream channels. Because of reduced risk with increasing distance from water, fewer safeguards are required as distance from waterbodies increase. Key provisions include using more precise herbicide application methods in stream side areas, such as wicking, wiping, or hand spraying with a single nozzle.

Dyes will be used to more accurately control the application rate. Dyes also reduce the likelihood of spraying chemicals directly into water by showing the applicator exactly where the herbicide is being sprayed and limits over application by identifying treated areas.

Taken together, the entire suite of PDC/BMPs described in the proposed action address, to varying degrees, the mechanisms that enable chemicals to reach stream channels (i.e., drift, spillage, direct overspray, leaching, and runoff). For this reason they are likely to keep herbicides from reaching water in appreciable amounts in the vast majority of circumstances. However, the efficacy of the PDC/BMPs is not documented to the extent that risks can be quantified. For example, some PDC logically prevent water contamination (i.e., in field mixing provisions), while others are assumed to be reliable precautions, but less than 100 percent effective under all possible circumstances. Consequently, the precautionary measures in the proposed action are likely to eliminate routine risks under the vast majority of application circumstances, but they do not completely eliminate the risk of chemical contamination. For these reasons the following sections discuss potential exposure and biological response scenarios for the proposed herbicides and adjuvants.

2.5.1.4 Exposure to Herbicide

Weed treatments typically occur sometime between the start of emergence (about April) and can extend until dormancy (until November at some elevations). During this period, all life stages of Chinook salmon, steelhead, and sockeye salmon could potentially be exposed to herbicides, including incubating eggs, rearing juveniles, and migrating/holding adults. As previously described, the Forests have proposed multiple measures to minimize or avoid water contamination from herbicides, effectively targeting all potential exposure pathways. In addition, the proposed action estimates no more than 6,250 acres of FC-RONRW managed lands will be treated with herbicides annually; with no more than 625 of those acres being treated within 100 feet of live water in a single year. The Forests have never come close to exceeding the 6,250-acre limit in past years. The closest that the Forests have come to the 625 acre limit within 100 feet of water is 428 acres in a single year (USFS 2019).

The total acreage treated with chemicals in the FC-RONRW in 2018 was 148.3 acres. Approximately 61.4 of those acres were treated within 100 feet of live water; and 86.9 acres were treated greater than 100 feet from live water. The 2018 treatments accounted for about 10 percent of the annual acre limit within 100 feet of live water, and about 2 percent of the 6,250 annual acre limit.

Available data on water quality monitoring by the USFS for past weed treatments are limited, but suggest that PDC similar to those in the proposed action successfully minimize the occurrence of water contamination and the concentration of chemicals in the water when contamination occurs (Berg 2004). Even with implementation of the numerous PDC and the small size of the treatment area relative to managed area, there is uncertainty that herbicide delivery to surface water can be eliminated given the potential for human error and/or unpredicted post-treatment storm events. Subsections on exposure pathways (above), support NMFS' conclusion that potential for herbicides to reach streams exists, although that potential diminishes as the distance from the herbicide application to surface water increases and as application methods change. Water contamination (and subsequent fish exposure to herbicides) is most likely to occur under the proposed action in occasional circumstances: (1) where chemicals are applied close to water, in dry channels, along hydrologically connected trails, or on coarse alluvial soils and are mobilized by surface runoff or leaching after application; (2) when operator errors occur, such as spilling chemicals during transportation, or accidentally spraying herbicides into water; or (3) when unexpected weather conditions occur during or shortly after spraying. Beyond these occasional circumstances, chemical contamination of water from the proposed action is unlikely to occur given the properties of the herbicides, the small amounts of chemicals used, the short persistence in soil and lack of soil mobility for most active ingredients, and implementation of PDC that minimize or avoid water contamination (such as use of hand application and wind speed criteria), and limited acreage treated within the action area.

The EDRR and adaptive management decision tree (Figure 1) is considered a key element in avoiding or minimizing effects as well as modifying treatment methods during and/or after initial effort to treat a site. The effects analyses in this Opinion are based upon the assumption that all of the protective measures listed in proposed action will be implemented without exception wherever the action could affect ESA-listed fish or critical habitat. To most effectively avoid/minimize potential effects to listed fish or designated critical habitat, all PDC designed to reduce the likelihood of surface runoff or groundwater contamination need to be applied in or immediately upstream from watersheds currently occupied by ESA-listed fish species or designated as critical habitat.

Site-specific estimates of fish exposure are not known since the exact treatment locations, the amount and type of chemicals that will be applied, water volume of contaminated streams, and weather conditions are not known ahead of time. Thus, a quantitative estimate of fish exposure to individual proposed herbicides was evaluated for a generalized “worst-case” scenario.

Worst-case scenarios include development of the EECs for each active ingredient. The EECs are a product of the WCRs found in the most recent Forest Service Risk Assessments prepared by the Syracuse Environmental Research Associates, Inc. (SERA) and the maximum label application rates. Although a variety of models can be used to estimate contamination rates of surface water after pesticide applications, SERA relies heavily upon the Groundwater Loading Effects of Agricultural Management Systems (GLEAMS) model. The GLEAMS model is a root zone model that can be used to examine the fate of chemicals in various types of soils under different meteorological and hydrogeological conditions. The WCRs extrapolated from SERA were generally GLEAMS model results; however, some WCRs selected for this Opinion came from field studies. A discussion about how the EEC was derived for each of the active ingredients is included in Appendix A. Herbicide WCRs and EECs are summarized in Table 9, along with their persistence/mobility in soils, and some general physical property information.

Our worst-case scenario differs from the Forests’ assessment. We applied WCRs found in the most recent SERA document multiplied by the max label application rate. The Forests (USFS 2019) used LC_{50} divided by 10 and the maximum allowable toxicant concentrations (MATC) values compared against NOEL/NOEC/NOAEL[1] values to develop risk quotients. The MATC represents the acute toxicity value (96-hour LC_{50}) of either rainbow trout or Daphnia *spp.* (a type of water flea), whichever is less, to a specific herbicide divided by 25.

[1] NOEL – No observable effect level; NOEC – No observable effect concentration; NOAEL – no observed adverse effect level.

The NOEL/NOEC/NOAEL values usually exceed calculated MATC values. The LC_{50} is typically used as an assessment of acute dose-response; while the NOEL/NOEC/NOAEL is used to assess chronic risk. The NOECor NOAEL is the highest concentration of toxicant to which organisms are exposed in a full life-cycle or partial life-cycle test that causes no observable effect on survival, growth, or reproduction of the test population. NMFS' use of the WCR values provided in the SERA Risk Assessment produces higher EECs due to use of upper bounds of WCR modeling results. The Forests indicated that the EEC is an extreme level that is unlikely to occur during implementation of the proposed action and should be viewed as a worst-case situation.

NMFS' approach is conservative, and likely overstates EEC and potentially inflates the potential risk to species exposed during the action for most applications. However, given the current status and baseline conditions for the affected species, and unknowns regarding site-specific environmental conditions at application sites, we elected to present the worst-case scenario in this Opinion. This is consistent with recent consultations on similar actions NMFS has completed (NMFS 2012; NMFS 2013; NMFS 2016b; NMFS 2016c) and ensures consideration of maximum potential exposure of ESA-listed species considered in the analysis.

2.5.1.5 Toxicological Effects of Herbicides

Herbicides (including the active ingredient, inert ingredients, and adjuvants) can potentially harm[2] fish directly or indirectly. Herbicides can directly affect fish by killing them outright or causing sublethal changes in behavior or physiology. Herbicides indirectly affect fish by altering their environment (Scholz et al. 2005). Environmental alterations may include changes in cover, shade, runoff, and availability of prey species.

Direct Effects. Herbicide exposure may directly result in one or more of the toxicological endpoints identified below. These endpoints are generally considered to be important for the fitness of salmonids and other fish species. They include:

- Direct mortality at any life history stage;
- An increase or decrease in growth;
- Changes in reproductive behavior;
- A reduction in the number of eggs produced, fertilized, or hatched;
- Developmental abnormalities, including behavioral deficits or physical deformities;
- Reduced ability to osmoregulate or adapt to salinity gradients;
- Reduced ability to tolerate shifts in other environmental variables (e.g., temperature or increased stress);

- An increased susceptibility to disease;
- An increased susceptibility to predation; and
- Changes in migratory behavior.

The ecological significance of sublethal toxicological effects to individual fish depends on the degree to which essential behavior patterns are impaired, and the number of individuals exposed to harmful effects. Sublethal effects could compromise the viability and genetic integrity of wild populations if the effects are widespread across an entire DPS or ESU, or if localized exposures result in the concentrated loss of fish in a geographic area occupied by a local population with unique genetic traits. Scholz et al. (2000) and Moore and Waring (1996) indicated that environmentally relevant exposures to diazinon can disrupt olfactory capacity in the context of survival and reproductive success of Chinook salmon, both of which are key management considerations under the ESA (Scholz et al. 2000). The likelihood of population effects from sublethal effects of the chemicals in the proposed action are largely undocumented, but appreciable population effects can be ruled out if the potential exposure to harmful effects is limited to small numbers of fish and a spatial pattern that is not likely to cause the loss of a unique genetic stock.

Table 9. Physical properties, application rates, and EECs for herbicides proposed for use.

Active Ingredient	Persistence in Soil (days)[1]	Mobile in Soil	Typical Application Rate (lb a.e./Acre)[2]	Max Label Application Rate (lb a.e./Acre)	WCRs (mg a.e./L)[3]	Upper Bound Peak EEC (mg a.e./L)[4]
2,4-D amine	10 Low	Yes, but degrades quickly	0.5–1.5	4.0	0.44	1.76
Clopyralid	40 Moderate	No	0.28–0.5	0.5	0.07	0.035
Aminopyralid	103 Days (High)	Yes	0.078–0.11	0.11	0.6	0.066
Chlorsulfuron	40 (28–42) Low-Mod	No	0.02–0.09	0.12	0.2	0.024
Dicamba[5]	7–42 Low-Mod	Yes	0.25–2	2	0.01	0.02
Aquatic Glyphosate	47 Moderate	No	0.26–3.7	8.0	0.083	0.332
Imazapic	7–150 Low-High	No	0.1–0.19	0.19	0.01	0.002
Imazapyr	2,150 (313–2,972) High	Yes	0.5–1.0	1.5	0.009	0.0135
Imazamox	81 Days	Yes	0.25–0.5	0.5	0.04	0.02
Metsulfuron-methyl	30 (7–28) Low	No	0.04–0.11	0.15	0.01	0.0003
Picloram[6]	90 (20–300) Mod-High	Yes	0.5–0.75	1.0	0.18	0.18
Sulfometuron-methyl	20–28 Low	No	0.09–0.23	0.37	0.02	0.008

Active Ingredient	Persistence in Soil (days)[1]	Mobile in Soil	Typical Application Rate (lb a.e./Acre)[2]	Max Label Application Rate (lb a.e./Acre)	WCRs (mg a.e./L)[3]	Upper Bound Peak EEC (mg a.e./L)[4]
Triclopyr triethylamine salt (TEA) (Garlon 3A)	30 Low	Yes	4.5–6.0	9.0	Acid: 0.24 TCP: 0.028	Acid: 2.16 TCP: 0.252
Fluroxypyr	7–23[7] Low 3.2–185[8] Low-High	Yes	0.35–0.4	0.48	0.08	0.04

[1] Soil half-life values for herbicides are from Herbicide Handbook (Ahrens 1994). Pesticides considered non-persistent are those with a half-life of less than 30 days; moderately persistent herbicides are those with a half-life of 30 to 100 days; pesticides with a half-life of more than 100 days are considered persistent.
[2] Typical application rates are those used by the Forests; maximum application rates are those on the product labels unless otherwise noted.
[3] The WCRs were obtained from the most recent SERA risk assessments: https://www.fs.fed.us/foresthealth/protecting-forest/integrated-pest-management/pesticide-management/pesticide-risk-assessments.shtml
[4] The EECs were derived by multiplying the maximum label application rate by the WCR.
[5] Dicamba application rates on the Forests are 1 lb per acre for broadcast applications and no more than 2 lb per acre per year on a treatment area, per label requirements for rangeland uses.
[6] Maximum application rate for picloram is 1 lb a.e./acre; rates may be higher for smaller portions of the acre, but the total use on the acre cannot exceed 1 lb a.e. per acre per year.
[7] SERA 2009
[8] Bureau of Land Management 2014

Weis et al. (2001) reviewed published literature on consequences of changes in behavior of fish from exposure to contaminants and noted studies reporting impaired growth and population declines from altered feeding behavior and impaired predator avoidance. Potential sublethal effects, such as those leading to a shortened lifespan, reduced reproductive output, or other deleterious biological outcomes are a potential threat to ESA-listed species from the proposed action. Anadromous fish in the Snake River are exposed to multiple physiological sublethal stressors with apparent cumulative effects (e.g., Ebel et al. 1975; Matthews et al. 1986; Coutant 1999). Cumulative exposure to multiple sublethal stressors associated with the Snake River hydropower system has been attributed to delayed mortality in Snake River salmon (Budy et al. 2002). Mortality resulting from a history of multiple physiologically sublethal stressors is referred to as "ecological death" (Kruzynski et al. 1994; Kruzynski and Birtwell 1994). Cumulative effects of multiple stressors are thought to be the cause of declines in some fish populations, even though the effects of any single stressor appeared to be minor (Korman et al. 1994). Although exposure to pesticides is not a reported factor in delayed mortality of fish, one can reasonably assume that physiological stress created from sublethal exposure to herbicides, or potential impacts to primary production, could contribute to reduced growth of juveniles. Baldwin et al. (2009) found evidence that juvenile growth reductions caused by exposure to popular pesticides (organophosphate and carbamate classes), led to modeled reductions in population productivity. The proposed action does not include the same class of pesticides, and herbicide exposure is likely to be of lower duration and less chronic than evaluated by Baldwin et al. (2009), suggesting the proposed action will not lead to widespread growth reductions and decreased productivity.

Additional uncertainty with the available toxicity information exists in translating laboratory assays to field conditions and an individual organism's response. Traditional chemical evaluations have not routinely considered fish in impaired baseline habitat settings, which exist in portions of the action area (see Section 2.4). This uncertainty contributes to our conclusion that some sublethal effects are likely to occur from the proposed action. However, proposed PDC are expected to greatly reduce both the frequency and magnitude of exposures to infrequent and brief events in isolated locations.

The likelihood of adverse indirect effects is dependent on environmental concentrations, bioavailability of the chemical, and persistence of the herbicide in salmon habitat. For most herbicides, including those in the proposed action, there is little information available on environmental effects such as negative impacts on primary production, nutrient dynamics, or the trophic structure of macroinvertebrate communities. Most available information on potential environmental effects must be inferred from laboratory assays; although a few observations of environmental effects are reported in the literature.

Indirect Effects. In addition to effects of direct exposure on ESA-listed fish, indirect effects of pesticides can occur through their effects on the aquatic environment and non-target species. The likelihood of adverse indirect effects is dependent on environmental concentrations, bioavailability of the chemical, and persistence of the herbicide in salmon habitat. For most herbicides, including those in the proposed action, there is little information available on environmental effects such as negative impacts on primary production, nutrient dynamics, or the trophic structure of macroinvertebrate communities. Most available information on potential environmental effects must be inferred from laboratory assays; although a few observations of environmental effects are reported in the literature.

Due to the paucity of information, there are uncertainties associated with the following factors: (1) The fate of herbicides in streams; (2) the resiliency and recovery of aquatic communities; (3) the site-specific foraging habits of salmonids and the vulnerability of key prey taxa; (4) the effects of pesticide mixtures that include adjuvants or other ingredients that may affect species differently than the active ingredient; and (5) the mitigating or exacerbating effects of local environmental conditions. Where uncertainties cannot be resolved using the best available scientific literature, the benefit of the doubt should be given to the threatened or endangered species in question [H.R. Conf. Rep. No. 697, 96th Cong., 1st Sess. (1979)].

Indirect effects of contaminants on ecosystem structure and function are a key factor in determining a chemical's cumulative risk to aquatic organisms (Preston 2002). Moreover, aquatic plants and macroinvertebrates are generally more sensitive than fish to acute toxic effects of herbicides. Therefore, chemicals can potentially affect the structure of aquatic communities at concentrations below thresholds for direct impairment in salmonids. Because the integrity of the aquatic food chain is an essential biological requirement for salmonids, the possibility that herbicide applications will limit the productivity of streams and rivers is an unknown risk of the proposed action.

Juvenile Pacific salmon feed on a diverse array of aquatic invertebrates, with terrestrial insects, aquatic insects, and crustaceans comprising the large majority of the diets of fry and parr in all salmon species (Higgs et al. 1995). Prominent taxonomic groups in the diet include *Chironomidae* (midges), *Ephemeroptera* (mayflies), *Plecoptera* (stoneflies), *Tricoptera* (caddisflies), and *Simuliidae* (blackfly larvae); as well as amphipods, harpacticoid copepods, and daphniids. Chironomids in particular are an important component of the diet of nearly all freshwater salmon fry (Higgs et al. 1995). In general, insects and crustaceans are more acutely sensitive to the toxic effects of environmental contaminants than fish or other vertebrates. However, with a few exceptions (e.g., daphnids), the impacts of pesticides on salmonid prey taxa have not been widely investigated.

Factors affecting prey species are likely to affect the growth of salmonids, which is largely determined by the availability of prey in freshwater systems (Mundie 1974). Food supplementation studies (e.g., Mason 1976) have shown a clear relationship between food abundance and the growth rate and biomass yield of juveniles in streams. Therefore, herbicide applications that kill or otherwise reduce the abundance of macroinvertebrates in streams can also reduce the energetic efficiency for growth in salmonids. Less food can also induce density-dependent effects, such as increased competition among foragers as prey resources are reduced (Ricker 1976). These considerations are important because juvenile growth is a critical determinant of freshwater and marine survival (Higgs et al. 1995). A study on size-selective mortality in Chinook salmon from the Snake River (Zabel and Williams 2002) found that naturally reared wild fish did not return to spawn if they were below a certain size threshold when they migrated to the ocean. There are two primary reasons mortality is higher among smaller salmonids. First, fish that have a slower rate of growth suffer size-selective predation during their first year in the marine environment (Parker 1971; Healey 1982; Holtby et al. 1990). Growth-related mortality occurs late in the first marine year and may determine, in part, the strength of the year class (Beamish and Mahnken 2001). Second, salmon that grow more slowly may be more vulnerable to starvation or exhaustion (Sogard 1997).

It is possible that the action may also cause detrimental effects when non-target plants are killed by herbicides. Herbicide spraying in riparian areas can kill non-target plants that provide streambank stability, shade, and cover for fish. Spraying can also increase surface runoff by creating areas of bare soil devoid of any vegetation. This is particularly true for non-selective herbicides that kill all plants, such as glyphosate. However, non-target species killed by herbicides tend to be mostly forbs, grasses, and legumes, which are capable of reestablishing themselves within a few growing seasons. Although shrubs and trees are also susceptible to herbicide effects, the quantity of herbicide applied during spot spraying in riparian areas is not likely to kill mature shrubs or trees that have matured beyond the pole stage. In addition, the Forests will conduct site assessments prior to spraying to determine the most appropriate spray patterns, herbicides, and application rates to use to further reduce the potential for injuring or killing non-target species.

In contrast to the potential for non-target plant impacts to negatively influence fish growth and survival, eradication or control of invasive plants may produce some beneficial effects to fish. Beneficial effects may result from reestablished desirable vegetation and associated potential for reduced sediment delivery as infested site acreage is reduced (Herron et al. 2001; Larson 2003). Over the 15-year action period, these types of beneficial effects could be significant, particularly in localized areas where existing infestations are high and currently impairing riparian health and function or contributing to stream sedimentation, both of which can reduce salmonid survival and or growth. Invasive plant expansion is recognized as a threat in NMFS' draft recovery plan, so successful treatment of existing and prevention of future infestations should assist with reducing threats to the recovery and survival of affected fish populations, even if only to a minor level.

The majority of active ingredients proposed for use will not remain persistent in soils beyond the season of application, with only a few persisting for more than 50 days (Table 2). Although Aminopyralid, Imazapic, Imazamox, Imazapyr, and Picloram tend to be most persistent, only Imzapyr is expected to persist beyond the year of application. Imazapyr is proposed for both upland and riparian treatments, but only the aquatic formulation of Imazapyr is proposed for use. Imazapyr will not be applied directly to water, to areas where surface water is present, or to areas below the OHWM. Even though persistent in soils, it is not expected to affect fish long-term as it is rapidly degraded by sunlight in solution should it reach action area waters; and it is considered practically non-toxic to fish with LC_{50} values of >100 milligrams acid equivalent per liter (mg a.e./L) (Appendix A).

Available Information. Available information on the toxicological effects of each of the active ingredients and EUPs proposed for use is summarized in Appendix A. Table 10 summarizes toxicity information for active ingredients and surfactants, respectively.

As previously discussed, many of the proposed EUPs contain significant proportion of inert ingredients and many recommend adding a surfactant. The designation as "inert" does not mean an additive is chemically inactive, and it does not convey any information about the toxicity of the ingredient (Tu et al. 2003). In addition to increasing the herbicidal effect of the active ingredient, inert ingredients can have toxic properties in and of themselves. Because many manufacturers consider inert ingredients to be proprietary, they do not list specific chemicals. In some cases the toxicity of the inert ingredient may be greater than the toxicity of the active ingredient (Solomon and Thompson 2003).

Table 10. Toxicity of active ingredients and surfactants proposed for use within the FC-RONRW.

Active Ingredient	Rainbow Trout 96-hour LC_{50} (mg a.e./L)[1]	Lowest Sublethal Effect Threshold (mg a.e./L)[1]	Daphnid 48-hour LC_{50} milligrams per Liter (mg/L)
2,4-D amine	162	4.78	25
Aminopyralid	>100[6]	NOEC = 50	>100
Clopyralid	103.5	5.175	225
Chlorsulfuron	40	NOEC = 30	>100
Dicamba[8]	28	1.4	100
Glyphosate (more toxic forms)	1	NOAEC = 0.05	1.5–62
Glyphosate (less toxic formulations)	10	NOAEC = 0.5 - 21	>200
Imazamox	>115	NOEC = 89.2	115 mg a.e./L (96-hour)
Imazapic	100	NOEC = 100	>100
Imazapyr	115	NOAEC = 10.4	350
Metsulfuron-methyl	150	NOEC = 10	>150
Picloram	4.8	NOEC = 0.19	48
Sulfometuron-methyl	148	NOEC = 7.3	>150
Triclopyr TEA	Acid: 117 TCP: 1.5	Acid: NOAEC = 20 TCP: 0.075	Acid: 132.9 TCP: 10.9
Fluroxypyr	16	NOEC = 0.06	100

Active Ingredient	Rainbow Trout 96-hour LC_{50} (mg a.e./L)[1]	Lowest Sublethal Effect Threshold (mg a.e./L)[1]	Daphnid 48-hour LC_{50} milligrams per Liter (mg/L)
Surfactant			
Activator 90	2.02	NA	2.02
LI 700	17–130[3,4]	NA	170–190[3,4]
MSO	485	NA	>100[5]
R11	3.8–6[2,4]	NA	5.7–19[2,4]
Spreader 90	3.35	NA	7.3 (96-hour)[5]
Syl-Tac	>5[5]	NA	>5[5]
Colorants			
Highlight	NA	NA	NA
Bullseye	NA	NA	35,747
Insight	NA	NA	NA

[1] Lowest available LC_{50} values for salmonids, obtained from the most recent SERA risk assessments. For triclopyr, the values presented are for the formulated product and a metabolite. For Glyphosate a range was presented, please see SERA 2011a due to extreme variability between EUP, adjuvants, and studies. For fluroxypyr, 14.3 was for bluegill, and was the only LC_{50} for a known formulation. However it is above the solubility of fluroxypyr in water and actual concentration in water will be much lower than nominal concentration, making adverse effects unlikely.
[2] McLaren-Hart Environmental Engineering Corporation 1995.
[3] LI 700 MSDS.
[4] Smith et al. 2004.
[5] Bakke 2003.
[6] No mortality in any study, maximum concentration evaluated was 100 mg a.e./L.

Manufacturers of many of the EUPs proposed for use by the Forests recommend the addition of a surfactant. There are numerous surfactants available on the market, some of which have reported toxicity (i.e., acute lethality) information. Toxicity (LC_{50}) values reported for surfactants commonly added to glyphosate field solutions typically range from 1 to 10 mg/L (SERA 2011a). However, there are some surfactants that are considered slightly toxic (LC_{50} values ranging from >10 to 100 mg/L) to practically non-toxic (LC_{50} values greater than 100 mg/L). As identified previously, the Forests have proposed to only use the following surfactants: Activator 90, Spreader 90, LI 700, Syl-Tac, R11, and MSO. The toxicity of these surfactants ranges from 2 mg/L to 130 mg/L. These surfactants are not hazardous nor are they categorized by EPA as List 1 (inert ingredients of toxicological concern) or List 2 (potentially toxic other ingredients/high priority for testing inerts) compounds when used as intended and label directions are followed (CH2MHILL 2004). The toxicities of mixtures of these surfactants with the EUPs are largely unknown. Mitchell et al. (1987) tested the toxicity of Rodeo with and without a surfactant. Without the surfactant, the 96-hour LC_{50} for rainbow trout was 429 mg a.e./L. With the surfactant X-77, the 96-hour LC_{50} ranged from 96.4 mg a.e./L (rainbow trout) to 180.2 mg acide equivalent per liter (a.e./L) (Chinook salmon). The addition of X-77 altered the toxicity of the formulation by up to four times. Knowing that the addition of a surfactant can increase the toxicity of an EUP is taken into account when evaluating the risk to ESA-listed species.

2.5.1.6 Assessment of Ecological Risk

Assessing the potential ecological risk associated with the use of pesticides is a complicated task. This is in part because there are numerous active ingredients, EUPs, adjuvants, and mixtures that can be applied on the ground. There is also limited available information upon which to evaluate the risk of pesticide use on the survival and recovery of endangered species. A general discussion about each of these is below and is followed by a discussion of the assessment methodology and summary.

Active Ingredients, End-Use Products, and Adjuvants. Toxicological effects of herbicides must be assessed before they can be registered for use. The EPA registers several types of chemicals and chemical formulations under Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA): pure (or nearly pure) active ingredients (also known as the technical grade active ingredient [TGAI]), manufacturing use products (MUP), and EUP. The TGAI is the component which kills/controls the target species. A MUP is any pesticide product other than an EUP. It may consist of just the technical grade active ingredient, or a combination of the TGAI along with inert ingredients such as stabilizers or solvents. An EUP is a product that may be any combination of TGAIs, MUPs, and additional inert ingredients. An EUP may not be used to manufacture or formulate other pesticide products, and it must have a label specifying the directions for use. Each TGAI registered by EPA may have more than one registered MUP, which in turn may have more than one registered EUP. By 1997, approximately 890 active ingredients and more than 20,000 EUPs were registered under FIFRA (Aspelin and Grube 1999).

The MUPs or EUPs containing the same active ingredient may have different toxicities to aquatic organisms. This is because they have different formulations (i.e., different proportion of active ingredient, different inert ingredient composition, or different proportions of each inert ingredient). The EPA's registration process does not require that all EUPs be tested for their toxicity. Rather, EUPs that are similar in their formulation and their use may be "batched." Batching the registration process allows manufacturer(s) to select a representative EUP and conduct toxicity tests on that single EUP. The other EUPs within that batch are assumed to have similar toxicities; therefore, there are some EUPs whose toxicological effects have not been directly tested.

Because not all EUPs have been directly tested, evaluating the risk to ESA-listed fish requires NMFS to examine whether EUPs can be considered similar. There are a few ways to assess the similarity of EUPs. The first way is to examine their labels to see the form of active ingredient, proportion of active ingredient, and type/composition of inert ingredients. In most cases, obtaining a list of the inert ingredients in products is extremely difficult because the manufacturer considers that information to be proprietary. As such, NMFS relies heavily on any available information in the USFS risk assessments prepared by SERA. A second way to examine similarity is to compare their EPA registration numbers. If the registration numbers are identical, the EUPs may be regarded as equivalent. For example, the EPA registration number for Renovate 3 is 62719-37-67690. The EPA registration number for Garlon 3A is 62719-37. The first two number groups consist of the company identification number followed by the product code. The third number, in the case of Renovate 3, is the company code for SePRO (which manufactures Renovate 3). Thus, Renovate 3 is a repackaging of Garlon 3A. This means the two formulations are equivalent, thus, the toxicity of Garlon 3A is representative of the toxicity of Renovate 3. A third way to evaluate the similarity of EUPs is to examine their respective MSDS. If the toxicity information included on the MSDS are identical, then it is likely EPA allowed data on one EUP to be used to support the registration of other EUPs. In evaluating the effects of the proposed action on ESA-listed species, NMFS considered the similarity of EUPs proposed for use. Table 11 summarizes the available information for EUPs proposed for use by the Forests.

In addition to not having toxicity information for all EUPs, there is generally little, if any, toxicity information available for adjuvants and EUP mixtures. Adjuvants are generally defined as any substance added separately to a pesticide EUP (typically as part of a spray tank mixture) to enhance the activity of the herbicide or assist with the application. Adjuvants most commonly used on the FC-RONRW-are surfactants and dyes. When a formulation doesn't contain a surfactant, the product label will often indicate that a nonionic surfactant must or should be added to the field solution prior to application. Some surfactants are toxic by themselves and have been documented to increase the toxicity of formulations in comparison to technical grade active ingredients (SERA 2011; Stark and Walthall 2003). The increase in toxicity is not necessarily additive, but depends on the type of surfactant used as well as the proportion of surfactant in the formulation or tank mixture. Even in light of this, the surfactants used in formulations are not always known and there is a paucity of toxicity testing on many of the surfactants.

Table 11. Characteristics of end-use products that may be used on the FC-RONRW.

Active Ingredient	End Use Product[1]	EPA Registration Number	Manufacturer	% a.i. (*other a.i.s in EUP*)	% Other Ingredients	Rainbow trout LC_{50} (mg a.e./L)[2]
2,4-D amine	2,4-D Amine 4[3]	1381–103	WinField Solutions	47.3	Not Otherwise Specified (NOS) (52.7%)	Not Reported
	Weedar 64[4]	71368–1	Nufarm Americas	46.8	NOS (53.2%)	250 (a.i.); 208 (a.e.)
	Weedestroy AM-40[5]	228–145	Nufarm Americas	46.8	NOS (52.7%)	250 (a.i.); 208 (a.e.)
	DMA 4	62719–3	Dow AgroSciences	46.3	53.7%	Not Reported
	Clean Amine	34704–120	Loveland Products	46.5	53.5%	Not Reported
Aminopyralid	Milestone	62719–519	Dow AgroSciences	40.6	Water (59.4%)	>100
Chlorsulfuron	Telar XP	352–651	Du Pont	75	NOS (25%)	>122
Clopyralid	Transline	62719–259	Dow AgroSciences	40.9	Isopropanol (5%) Polyglycol (1%) NOS (53.1%)	103.57[6]
Dicamba	Banvel	66330–276	Arysta Lifescience North America	48.2	NOS (51.8%)	>1,000 (a.i.); >350 (a.e.)
	Vanquish	100–884	Syngenta	56.8	NOS (43.2%)	135.4 (a.i.); 52.1 (a.e.)
Aquatic Glyphosate	Rodeo	62719–324	Dow AgroSciences	53.8	Water (46.2%)	>2,500 (a.i.); >430 (a.e.)
	Glypro	62719–324	DowAgro Sciences	53.8	Water (46.2%)	>2,500 (a.i.); >430 (a.e.)
	Accord Concentrate	62719–324	DowAgro Sciences	53.8	Water (46.2%)	>2,500 (a.i.); >430 (a.e.)
	Foresters	228–381	Riverdale	53.8	Water (46.2%)	780 a.e.
	AquaMaster	524–343	Monsanto	53.8	Water (46.2%)	>1,000 (a.i.); >172 (a.e.)
	AquaNeat Aquatic Herbicide	228–365	Nufarm Americas	53.8	Water (46.2%)	86
Imazamox	Clearcast	241–437	BASF Corporation	12.1	NOS (87.9%)	Not Reported
Imazapic	Plateau	241–365	BASF Corporation	23.6	NOS (76.4%)	>100 (a.i.); >94 (a.e.)
Imazapyr	Habitat	241–426	SePro	28.7	NOS (71.3%)	>100 (a.e)
Metsulfuron methyl	Escort XP	352–439	Du Pont	60	NOS (40%)	>150
Picloram	Tordon 22K	62719–6	Dow AgroSciences	24.4	Polymer (1.7%) NOS (73.9%)	Not Reported
Sulfometuron methyl	Oust XP	352–601	Du Pont	75	NOS (25%)	>148

Active Ingredient	End Use Product[1]	EPA Registration Number	Manufacturer	% a.i. (*other a.i.s in EUP*)	% Other Ingredients	Rainbow trout LC_{50} (mg a.e./L)[2]
Triclopyr TEA	Garlon3A	62719-37	Dow AgroSciences	44.4	Ethanol (2.1%) NOS (50.5%)	286
Fluroxypyr	Vista XRT (Ultra)	62719-586	Dow AgroSciences	45.52	NOS 54.5%	>100

[1]Other brands not listed in the table, but contain identical EPA registration numbers may be added or substituted in the future after Level 1 Team review.
[2]The LD_{50}/LC_{50} is the dose/concentration of a chemical that kills 50 percent of the test organisms. [3]This product is also sold under the following names: 2,4-D Lo V (Universal Crop Protection Alliance) and 2,4-D Amine. [4]This product is sold under the following additional names: Base camp Amine 4 (by Wilbur-Ellis Company) and tenkoz amine 4 2,4-d (by Tenkoz, Inc). [5]This product is sold under 20 different product names (PAN 2012). [6]Value is for the 3,6-dichloropicolinic acid and not the EUP. The Rainbow trout LC_{50} for the Monoethanolamine salt (a.i.) of clopyralid is 700 mg a.e./L.

Limitations of Available Information. Although toxicity data are needed to register active ingredients and EUPs, much of the available information still only addresses toxicity of the active ingredient, and does not address all individual EUPs (including their inert ingredients) and adjuvants. Furthermore, much of the available toxicity information focuses on direct lethality from the active ingredient and little published chronic toxicity data is available to assess the risk of herbicides on fish (Fairchild et al. 2009a; Fairchild et al. 2009b). This may be due to their limited toxicity in available acute tests (ENTRIX 2003). The lethal endpoint has little predictive value for assessing whether pesticide exposure will cause sublethal neurological and behavioral disorders in wild salmon (Scholz et al. 2000; Weis 2014). Many of the toxicological endpoints previously listed have not been investigated for the herbicides used in the proposed action (Stehr et al. 2009). Although Stehr et al. (2009) did not find evidence of early life history developmental impacts on salmonid surrogates exposed to six forestry herbicides (each proposed in this action) at higher concentration than anticipated under this action, there is little information available on the sublethal effects (e.g., feeding, spawning, or migration) or ecological effects (e.g., effects on fish behavior, prey composition or availability) of the active ingredients, EUPs, and tank mixtures (Baldwin et al. 2009; Weis 2014).

Most toxicity studies are performed in laboratory settings, with strictly controlled environmental conditions. Baseline environmental conditions are not controlled. Test conditions also lack exposure to predators, reduced competition (Jones et al. 2011), some pathogens and other natural hazards. Some authors have suggested laboratory bioassays may provide poor predictions of a toxicant's effects on natural populations when considered at an ecosystem level (Kimball and Levin 1985; Cairns 1983) while others, when evaluating metals toxicity, suggest there may be little differences between lab and natural condition impacts on fish (Larson et al. 1985). Laetz et al. (2014) found elevated temperature alone increased toxicity of mixed pesticides. In the event field conditions are more stressful to fish or their prey, herbicide exposure may have different effects than produced under laboratory conditions. It is also possible that environmental factors such as sediment availability or sunlight may bind or degrade herbicides at different rates than under laboratory conditions and thus present lower toxicity. Given there is some uncertainty in extrapolating laboratory effects of herbicide exposures directly to field conditions, some level of caution is warranted to ensure protection of ESA-listed fish.

Even in light of this uncertainty, NMFS believes that outright lethality from the use of herbicides in the FCRONRW is unlikely to occur. This is because the laboratory EECs represent worse-case scenarios and environmental concentrations are expected to actually be much less than these estimates due to implementation of BMPs. Furthermore, only a small proportion of the action area will be treated, thus any potential water contamination will be small in magnitude, infrequent, and short in duration. However, NMFS cannot say with any certainty that ESA-listed fish will not be harmed through sublethal effects or indirectly through toxic effects on other aquatic organisms. The risk of sublethal effects from water contamination by herbicides cannot be ignored, based on the available information. Water contamination by herbicides could occur in occasional circumstances, and sublethal effects of herbicides or their adjuvants can occur within the range of concentrations likely to occur under the proposed action.

2.5.2 Effects on Designated Critical Habitat

The action area contains designated critical habitat for Snake River spring/summer Chinook salmon, Snake River Basin steelhead, and Snake River sockeye salmon. Critical habitat within the action area has an associated combination of PBFs essential for supporting freshwater rearing, migration, and spawning for Chinook salmon and steelhead; and freshwater migration for sockeye salmon.

The critical habitat elements most likely to be affected by the proposed action include water quality, riparian vegetation, natural cover/shelter, and forage/food. Modification of these PBFs may affect freshwater spawning, rearing or migration in the action area. Proper function of these essential features is necessary to support successful adult and juvenile migration, adult holding, spawning, incubation, rearing, and the growth and development of juvenile fish.

The proposed weed treatment areas are scattered in patches of various size. Potential effects of weed spraying on designated critical habitat will vary at each location depending on the size of the treatment area, the chemicals used, method of application, distance from water, and vegetative characteristics of the treatment areas. If chemicals were to reach the water in an appreciable amount, a variety of biological effects could occur, including harmful effects on listed fish or other aquatic organisms due to direct exposure to the chemicals or indirectly from changes in the biotic community. In general, most instream effects of herbicides are short-lived, discreet events associated with spills, drift, or runoff events. Following a contamination event, critical habitat elements are likely to return to normal within a few hours to a few days. None of the chemicals proposed for use would result in long-term alteration of critical habitat through water contamination.

Long-term changes in critical habitat features are possible as a result of changes in terrestrial vegetation, but the effects are subtle and poorly understood. Herbicide use may affect critical habitat beneficially by restoring natural plant communities and reducing erosion (Herron et al. 2001; Larson 2003); or affect it adversely, where desired plant species are killed unintentionally.

The use of herbicides can potentially affect physical watershed and stream functions through the removal of vegetation and exposing bare soil. With spot spraying, injecting, painting, dipping, and wicking, as proposed, the potential for significant increases in erosion or water yield is limited because treatments will consist of small, scattered areas, and vegetation will likely be reestablished within a few months to a year. The potential for non-target mortality of riparian plants is negligible with these proposed application methods.

Noxious weed control measures will reduce weed competition between weeds and native plant species. Herbicide spraying in riparian areas will be minimal and will primarily be associated with spot spraying of small patches of noxious weeds or individual plants. No measurable adverse effects to peak/base flow, water yield, or sediment yield are likely to occur from implementation of noxious weed control and rehabilitation measures. Removal of solid stands of noxious weed vegetation by chemical treatment may result in short-term, negligible increases in surface erosion that would diminish as vegetation reoccupies the treated site. Only spot spraying, wicking, dipping, painting, and injecting would be used within 100 feet of live water.

This precautionary measure significantly reduces risks associated with spraying of non-target riparian vegetation and the likelihood of accidentally spraying herbicides directly into the water. Given the above considerations, the potential for adverse effects on physical characteristics of critical habitat (other than water quality) is negligible due to treatment areas being scattered in small blocks, precautionary measures to keep chemicals out of the water, and limited potential for herbicides to adversely alter riparian vegetation or other vegetation characteristics that affect critical habitat.

The potential for adverse effects on biological components of critical habitat (such as invertebrates and aquatic plants) from water contamination by herbicides is negligible at spatial scales of an individual stream or larger, but local adverse effects on biological components are possible in treatment areas where herbicides reach water. Adverse effects on the biological component of critical habitat that are most likely to occur are sublethal toxicological effects on listed species from direct exposure to chemicals. Effects of salmonid exposure to herbicides were evaluated in this Opinion in the previous section. Secondarily, there is a risk of adverse effects on aspects of the biological community that supports listed fish. However, secondary effects to biological communities that support listed fish are unlikely given the limited circumstances where herbicides are likely to enter water and the small amount of herbicides used at any particular location.

Although the incidence of herbicides reaching water in an appreciable amount under the proposed action is likely to be infrequent, herbicides are capable of altering the biotic composition of aquatic species when they reach water. A notable concern is the potential for impacts on benthic algae. Benthic algae are important primary producers in aquatic habitats and are thought to be the principal source of energy in many mid-sized streams (Minshall 1978; Vannote et al. 1980; Murphy 1998). Herbicides can cause significant shifts in the composition of benthic algal communities at concentrations in the low parts per billion (Hoagland et al. 1996). Moreover, based on the data available, herbicides have a high potential to elicit significant effects on aquatic microorganisms at concentrations that may occur with normal usage under the label instructions (DeLorenzo et al. 2001) that do not include many of the precautionary measures (such as buffers, wind restrictions, application methods, etc.) that are part of the proposed action. In most cases, the sensitivities of algal species to herbicides and their response to herbicides are not known. Herbicides have the potential to decrease or increase algal production and, by extension, alter the trophic support for stream ecosystems. However, the community response to changes in the algal community is unpredictable. Limited information is available on the ecological effects of the herbicides in streams, making it difficult to predict the degree of ecological risk to salmon and steelhead from alteration of the biological community. In general, human activities that modify the physical or chemical characteristics of streams often lead to changes in the trophic system that ultimately reduce salmonid productivity (Bisson and Bilby 1998). Consequently, herbicides have the potential to affect salmonid productivity through their effects on the biotic community. However, the potential for alteration of the biotic community under the proposed action is limited due to precautionary measures intended to keep herbicides from reaching water.

The herbicides may either increase or decrease growth in various algae and microphyte species, and the growth effects for a given species may go either direction, depending on the concentration. Similarly, certain aquatic invertebrates may decrease in number from direct exposure to herbicides or reduction in food sources, while other species may increase in number, in response to changes in primary productivity or community composition. Available tools and information cannot reliably predict such complex responses for this specific action; consequently, there is considerable uncertainty about ecological effects of the action.

Although changes in the biological community appear likely from herbicides that reach the water, adverse effects of herbicides on primary production and the invertebrate community are likely to be limited in size to stream reaches in the vicinity of application areas where herbicides may reach water in appreciable concentrations. The weed program includes numerous safeguards intended to eliminate or minimize water contamination; and any water contamination from herbicides is not likely to persist due to the small amount of chemicals proposed for use at any given application site and the dispersed use of chemicals. Therefore, the proposed action is unlikely to appreciably diminish the conservation value of designated critical habitat within action area streams.

2.6 Cumulative Effects

"Cumulative effects" are those effects of future state or private activities, not involving federal activities, that are reasonably certain to occur within the action area of the federal action subject to consultation (50 CFR 402.02). 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.

The action area consists almost entirely of undeveloped Federal lands with a few state and private in-holdings. State and private in-holdings within the FC-RONRW comprise less than one-tenth of 1 percent of the total land area. Most of the state and private lands are ranches or historic mining areas. The ranches are used primarily for commercial recreational activities such as hunting, hiking, fishing, and boating. Commercial recreation and mining activities both require federal permits if the activities affect resources on federal lands. Consequently, many of the activities on state and private lands are subject to ESA consultations on the federal permits. Other than the continued operation of the ranches and mines, there are no known state or private activities that are reasonably certain to occur. Cumulative effects are likely to continue at the same level as what currently exists.

2.7 Integration and Synthesis

The Integration and Synthesis section is the final step in our assessment of the risk posed to species and critical habitat as a result of implementing the proposed action. In this section, we add the effects of the action (Section 2.5) to the environmental baseline (Section 2.4) and the cumulative effects (Section 2.6), taking into account the status of the species and critical habitat (Section 2.2), to formulate the agency's Opinion as to whether the proposed action is likely to: (1) Reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing its numbers, reproduction, or distribution; or (2) appreciably diminishes the value of designated or proposed critical habitat for the conservation of the species.

As described above, acute lethality of ESA-listed fish is unlikely as a result of the proposed action. However, there is potential for delayed mortality and sublethal effects to occur. Although not all-inclusive, a variety of sublethal effects on fish are reported for the herbicides evaluated in this Opinion. The effects were documented at concentrations that are likely to occur occasionally due to implementing the action.

Cumulative effects are expected to be maintained at current levels. Baseline conditions in the action area are largely pristine due to its wilderness designation. The populations affected in ESA-listed species (ESUs/DPSs) do not currently meet VSP criteria. Climate change is expected to make recovery targets for salmon and steelhead populations more difficult to achieve but these effects are likely offset to some degree by the high quality habitat found in the action area.

In spite of uncertainties regarding toxic effects of the herbicides, the effects will not be substantial enough to negatively influence VSP criteria at the population scale. This is due to the sub-lethal nature of these effects. Any harm that might occur from sublethal effects is expected to affect only a very small portion of the action area. Any water contamination that does occur will be isolated and of short duration (e.g., spikes in herbicide concentration following a rainfall). For these reasons, the proposed action is unlikely to reduce the likelihood of the survival and recovery of the species considered.

The proposed action will have localized short-term effects on riparian vegetation through the intentional eradication of weeds and any incidental mortality of desired riparian plants exposed to herbicides. In the long term, weed control will help restore ecological functions of riparian communities where those functions have been impaired by invasion of exotic plants. Incidental losses of desired plants will be sporadic and localized in riparian areas since herbicides applied in riparian areas will be applied to individual plants, primarily by backpack sprayer, wicking, painting, wiping, etc.

(1) In aquatic environments, herbicides are capable of altering the biotic composition of aquatic species when they reach water, thereby affecting salmonid productivity. Limited information is available on the ecological effects of the herbicides in streams, making it difficult to predict the degree of ecological risk to salmon and steelhead from alteration of the biological community. Although changes in the biological community appear likely from herbicides that reach the water, adverse effects of herbicides on primary production and the invertebrate community are likely to be limited in size to stream reaches where herbicides actually reach water. The weed program includes numerous safeguards intended to eliminate or minimize water contamination; any water contamination from herbicides is not likely to persist due to the small amount of chemicals proposed for use at any given application site, the type of herbicides applied, and the dispersed use of chemicals across the landscape and over time. Considering this, the proposed action is unlikely to appreciably diminish the conservation value of designated critical habitat for the following reasons: Any significant chemical contamination is likely to be infrequent, dispersed, and of short duration due to restrictions on the specific herbicides allowed in riparian areas, methods for applying the herbicides, and amount of herbicide applied to any given area which preclude large or persistent herbicide concentrations from occurring.

(2) Where chemical contamination occurs, concentrations are not likely to reach levels that alter the biological community in a manner that would appreciably alter PBFs of designated critical habitat at spatial and temporal scales relevant to conservation of the species.

Coupling the potential effects of the proposed action with the baseline condition and cumulative effects within the action area, NMFS concludes that the proposed action is not likely to appreciably diminish the function and conservation role of the PBFs within the action area. Because the function and conservation value of PBFs will not be appreciably reduced in the action area, they will also not be appreciably reduced at the designation scale.

2.8 Conclusion

After reviewing the current status of the listed species and their designated critical habitat, the environmental baseline within the action area, the effects of the proposed action, and cumulative effects, it is NMFS' Opinion that the proposed action is not likely to jeopardize the continued existence of Snake River spring/summer Chinook salmon, Snake River sockeye salmon, and Snake River Basin steelhead. NMFS has also determined that the action is not likely to destroy or adversely modify their designated critical habitats.

2.9 Incidental Take Statement

Section 9 of the ESA and federal regulations pursuant to section 4(d) of the ESA prohibit the take of endangered and threatened species, respectively, without a 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 regulation to include significant habitat modification or degradation that actually kills or injures fish or wildlife by significantly impairing essential behavioral patterns, including breeding, spawning, rearing, migrating, feeding, or sheltering (50 CFR 222.102). On an interim basis, NMFS interprets "harass" to mean "Create the likelihood of injury to wildlife by annoying it to such an extent as to significantly disrupt normal behavioral patterns which include, but are not limited to, breeding, feeding, or sheltering." "Incidental take" is defined by regulation as takings that result from, but are not the purpose of, carrying out an otherwise lawful activity conducted by the federal agency or applicant (50 CFR 402.02). Section 7(b)(4) and section 7(o)(2) provide that taking that is incidental to an otherwise lawful agency action is not considered to be prohibited taking under the ESA if that action is performed in compliance with the terms and conditions of this ITS.

2.9.1 Amount or Extent of Take

The proposed action is reasonably certain to result in incidental take of ESA-listed species. NMFS is reasonably certain the incidental take described here will occur because: (1) Recent and historical surveys indicate that listed species occur in the action area; and (2) the proposed action is likely harm individual listed salmon and steelhead through sublethal exposure to herbicides. These effects may occur as a result failure of PDC to keep chemicals from entering water (i.e., human error), unexpected sublethal effects that have not been reported in the scientific literature, additive or synergistic effects of herbicides from multiple sources in the action area, or fish or invertebrate response to herbicide exposures at lower exposures due to other environmental stresses. Herbicides applied by the Forests are not expected to reach streams in concentrations that kill listed fish, and most take resulting from the action is likely to be short-term sublethal effects that are harmful to fish or secondary effects from brief changes in water quality or food availability. Aquatic application of herbicides is not proposed to occur where it could affect anadromous fish; therefore, no take is exempted from aquatic applications.

Despite the use of best scientific and commercial data available, NMFS cannot quantify the specific amount of incidental take of individual fish or incubating eggs for this action. The amount of take from the proposed action depends on the circumstances at the specific times and locations that weed treatments will occur, such as rainfall, wind, humidity, and proximity of invasive plants to individual fish or redds and physical conditions that influence drift or runoff. Because circumstances causing take are likely to arise, but cannot be quantitatively predicted from available information, the extent of incidental take is described, pursuant to 50 CFR 402.14.

Similarly, it is difficult for NMFS to quantify the extent of take for the action as proposed. However, the acreage where herbicides are applied within 100 feet of live water is a reasonable surrogate for describing the extent of take. In the wilderness, chemicals are most likely to reach streams when they are applied to riparian areas, hydrologically connected trails, floodplains, or ephemeral drainage features in close proximity to water. Consequently, the acreage treated within 100 feet of live water is used to describe the extent of incidental take in this ITS, and it represents those areas where chemicals are most likely to reach the stream, and harm listed fish.

In many places where herbicides are applied within 100 feet of water, take is not anticipated to occur. The choice of the particular herbicides and the BMPs in the proposed action are likely to minimize the frequency and severity of incidental take in those places where chemicals reach the stream. However, the BMPs do not completely eliminate incidental take since herbicides will be used in sites where they are likely to reach water where listed fish are present. There is no practical alternative to using proximity to water and treatment acreage as a surrogate measure of take without knowing ahead of time the densities of fish present, precise locations where herbicides will be used, and without consideration of weather following herbicide application, along with site-specific features affecting herbicide transport. Based on a maximum annual treatment of 6,250 acres of weed infestations, up to 10 percent of which is likely to occur within 100 feet of live water, the extent of incidental take is anticipated to be no more than 625 acres of herbicide application within the riparian area annually.

Take is expected to occur when products with unknown highly toxic inert ingredients are used within 100 feet of streams and in occasional circumstances where precipitation transports other herbicides into water in concentrations where sublethal or lethal effects are likely. Such circumstances are expected to occur only in sites where weeds are sprayed in the vicinity of small stream channels that are occupied by listed fish, and showers or thunderstorms deliver a pulse of herbicides to the occupied stream channels.

The extent of take authorized in this Opinion is that resulting from the application of herbicides on 625 acres within 100 feet of any stream and 6,250 acres total within the action area per year.

2.9.2 Effect of the Take

In the Opinion, NMFS determined that the amount or extent of anticipated take, coupled with other effects of the proposed action, is not likely to result in jeopardy to the species or destruction or adverse modification of critical habitat.

2.9.3 Reasonable and Prudent Measures

"Reasonable and prudent measures" are nondiscretionary measures that are necessary or appropriate to minimize the impact of the amount or extent of incidental take (50 CFR 402.02). The Forests have the continuing duty to regulate the activities covered in this ITS where discretionary federal involvement or control over the action has been retained or is authorized by law. The protective coverage of section 7(o)(2) will lapse if the USFS fails to exercise their discretion to require adherence to terms and conditions of the ITS, or to exercise that discretion as necessary to retain the oversight to ensure compliance with these terms and conditions. Similarly, if any applicant fails to act in accordance with the terms and conditions of the ITS, protective coverage will lapse.

NMFS believes that full application of conservation measures included as part of the proposed action, together with use of the RPMs and terms and conditions described below, are necessary and appropriate to minimize the likelihood of incidental take of ESA-listed species due to completion of the proposed action.

The USFS shall minimize incidental take by:

1. Implementing precautionary measures that keep herbicides, adjuvants, and other chemicals out of water.

2. Accurately projecting and tracking applications within 100 feet of anadromous waters in the action area to ensure extent of take limits are not accidentally exceeded.

3. Reporting the activities proposed for treatment in the upcoming spray season and those activities and monitoring results from the previous season to NMFS annually.

To be exempt from the prohibitions of section 9 of the ESA, the USFS, and their cooperators, including any applicant, must fully comply with conservation measures described as part of the proposed action and the following terms and conditions that implement the RPMs described above. Partial compliance with these terms and conditions may invalidate this take exemption, result in more take than anticipated, and lead NMFS to a different conclusion regarding whether the proposed action will result in jeopardy or the destruction or adverse modification of critical habitats.

2.9.4 Terms and Conditions

The terms and conditions described below are non-discretionary, and the USFS or any applicant must comply with them in order to implement the RPMs (50 CFR 402.14). The USFS or any applicant has a continuing duty to monitor the impacts of incidental take and must report the progress of the action and its impact on the species as specified in this ITS (50 CFR 402.14). If the agency to whom a term and condition is directed does not comply with the following terms and conditions, protective coverage for the proposed action would likely lapse.

1. To implement RPM 1, the USFS shall:

a. Review the FC-RONRW herbicide spill plan with all applicators prior to herbicide applications. The spill plan shall be made available for inspection on request by NMFS.

b. Require any permittee have an appropriate spill cleanup kit on-site throughout any permitted herbicide application.

c. Reduce the likelihood of surface runoff or groundwater contamination by successfully applying all PDC described in this Opinion.

d. Select herbicides for use within 300 feet of water that have the lowest toxicological profile to anadromous fish and still able to meet desired treatment objectives.

e. Use added precaution when applying herbicides near streams or trails and drainage ditches that drain directly into streams, regardless of application method. Herbicides approved for use within riparian areas, and use of the least toxic surfactant compatible with the EUP, shall be the product of choice under appropriate site conditions.

f. To the extent practicable, avoid the use of picloram, clopyralid, chlorsulfuron, and metsulfuron-methyl within annual floodplains and where soil permeability is expected to be high (e.g., silt loam and sand soils). Where necessary to achieve weed management objectives use application methods with the lowest potential for off-site transport (e.g., large droplet size, hand held sprayer, etc.).

g. Establish a 15-foot herbicide application buffer width, near water, for ground based applications of chlorsulfuron.

h. When adding adjuvants to herbicides, only use the Washington State aquatic-certified adjuvants (Ecology 2016 or latest list) in riparian areas as proposed in the BA. When adding new adjuvants, the Forests shall notify NMFS, via the Level 1 process, of the newly proposed adjuvant prior to its use. NMFS will then evaluate the product to ensure effects are likely to be consistent with this Opinion's analysis.

i. Ensure all chemical storage, chemical mixing, transportation, and post-application equipment cleaning is completed in such a manner as to prevent the potential contamination of any riparian area, perennial or intermittent waterway, ephemeral waterway, hydrologically connected road ditch, or wetland.

j. Maintenance and calibration of spray equipment should occur regularly through the spray season to ensure proper application rates.

k. For treatments within 300 feet of anadromous waters, the Forest shall present Pesticide Use Proposal's with planned EUPs (name, active ingredient, registration number) to the Level 1 Team prior to use of any new EUPs not listed in this Opinion. The Level 1 Team shall agree that EUPs, and their effects to ESA-listed fish and critical habitats, are substantially similar to formulations identified in this Opinion. Should a concern be raised regarding an EUPs potential to affect ESA-listed fish and critical habitats, the Level I team member raising the concern will provide documentation, research or best available science supporting why the EUP should not be used. All parties shall work to conclude discussions in a timely manner so as not to delay treatments. If the USFS proposes to add a new active ingredient not identified in the BA and analyzed in this Opinion the action agencies will request consultation for that new chemical.

2. To implement RPM 2, above, the USFS shall:

   a. Maintain daily application logs. Information recorded or which may be produced from the logs will include the following:

      (1) The number of acres treated within 100 feet of streams and greater than 100 feet from streams.

      (2) The product names, herbicide formulations, including adjuvants and surfactants used.

      (3) The herbicide application rate.

      (4) The application method.

      (5) The wind speed and air temperature at the time of application.

   b. Require similar daily application logs and data (term and condition 3.a) be maintained and submitted by any USFS-authorized herbicide applicator (e.g., contractor, permittee) within the action area.

   c. Daily application logs shall be retained by the USFS and be available for review by NMFS, if requested.

   d. Summarize the information from daily logs in an annual report organized by 5th or 6th field hydrologic unit codes for the total acreage treated for each herbicide applied within 100 feet of streams and greater than 100 feet from streams.

3. To implement RPM 3, above, the USFS shall:

   a. Annually, report to NMFS by April 1, the following: (1) Acres of applied herbicide treatments within 100 feet of streams and great than 100 feet of streams in the action area (including any acres treated by permittees); (2) any spills and spill response that may have occurred; and (3) a statement affirming the Forests (including any contractors and permittees) successful implementation of the action, all PDC, and mandatory terms and conditions. Submit the report to:

National Marine Fisheries Service
NMFS Tracking Number: WCRO-2019-01915
800 East Park Boulevard
Plaza IV, Suite 220
Boise, Idaho 83712-7743

b. NOTICE: If a steelhead or salmon becomes sick, injured, or killed as a result of project-related activities, and if the fish would not benefit from rescue, 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. If the fish in question appears capable of recovering if rescued, photograph the fish (if possible), transport the fish to a suitable location, and record the information described above. Adult fish should generally not be disturbed unless circumstances arise where an adult fish is obviously injured or killed by proposed activities, or some unnatural cause. The finder must contact NMFS Law Enforcement at (206) 526-6133 as soon as possible. The finder may be asked to carry out instructions provided by Law Enforcement to collect specimens or take other measures to ensure that evidence intrinsic to the specimen is preserved.

2.10 Conservation Recommendations

Section 7(a)(1) of the ESA directs federal agencies to use their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of the threatened and endangered species. Specifically, conservation recommendations are suggestions regarding discretionary measures to minimize or avoid adverse effects of a proposed action on listed species or critical habitat or regarding the development of information (50 CFR 402.02). The following recommendations are discretionary measures that NMFS believes are consistent with this obligation and therefore should be carried out by the Forests:

1. The Forests should use herbicides and surfactants with the least toxicity to ESA-listed fish and other non-target organisms whenever possible.

2. The Forests should maintain and where appropriate increase efforts to prevent invasive terrestrial and aquatic plant introductions.

3. To mitigate the effects of climate change on ESA-listed salmonids, follow recommendations by the Independent Scientific Advisory Board (2007) to plan now for future climate conditions by implementing protective tributary and mainstem habitat measures. In particular, implement measures to protect or restore riparian buffers, wetlands, and floodplains; remove stream barriers; and to ensure late summer and fall tributary streamflows.

Please notify NMFS if the Forests carries out these recommendations so that we will be kept informed of actions that minimize or avoid adverse effects and those that benefit listed species or their designated critical habitats.

2.11 Reinitiation of Consultation

This concludes formal consultation for the USFS FC- RONRW Weeds Management Program. As 50 CFR 402.16 states, 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 amount or extent of incidental taking specified in the ITS is exceeded; (2) new information reveals effects of the agency action that may affect listed species or critical habitat in a manner or to an extent not considered in this Opinion; (3) the agency action is subsequently modified in a manner that causes an effect on 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 action.

3. MAGNUSON-STEVENS FISHERY CONSERVATION AND MANAGEMENT ACT ESSENTIAL FISH HABITAT RESPONSE

Section 305(b) of the MSA directs federal agencies to consult with NMFS on all actions or proposed actions that may adversely affect EFH. The MSA (section 3) defines EFH as "those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity." Adverse effect means any impact that reduces quality or quantity of EFH, and may include direct or indirect physical, chemical, or biological alteration of the waters or substrate and loss of (or injury to) benthic organisms, prey species and their habitat, and other ecosystem components, if such modifications reduce the quality or quantity of EFH. Adverse effects on EFH may result from actions occurring within EFH or outside of it and may include site-specific or EFH-wide impacts, including individual, cumulative, or synergistic consequences of actions (50 CFR 600.810). Section 305(b) also requires NMFS to recommend measures that can be taken by the action agency to conserve EFH.

This analysis is based, in part, on the EFH assessment provided by the Forests and descriptions of EFH for Pacific Coast salmon (PFMC 2014) contained in the fishery management plans developed by the Pacific Fishery Management Council and approved by the Secretary of Commerce.

3.1 Essential Fish Habitat Affected by the Project

The proposed action and action area for this consultation are described in the ESA portion of this document. The action area includes areas designated as EFH for spawning, rearing, and migration life-history stages of Chinook salmon. Existing habitat areas of particular concern present in the action area include: complex channel and floodplain habitat, spawning habitat, thermal refugia, and submerged aquatic vegetation.

3.2 Adverse Effects on Essential Fish Habitat

The effects of the proposed action on anadromous fish habitat are described in the habitat effects section of the Opinion. Briefly, the effects analysis found that herbicide spraying is "likely to adversely affect" habitat quality for Chinook salmon in instances where herbicides would be applied in drainages containing or upstream from, occupied habitat.

The likelihood of contamination is minimized in the proposed action through the use of extensive PDC/BMPs. However, water contamination cannot be avoided since the likelihood of contamination is partly dependent on the weather at the time of, and following herbicide application, herbicide properties, and is subject to human error. The proposed action will adversely affect the quality of EFH as a result of incidental water contamination that may occur from application of herbicides in the vicinity of streams designated as EFH. Water contamination from herbicides is expected to occur (albeit infrequently) when unanticipated precipitation carries the herbicides to water through overland flow, percolation, or in shallow ground water; and/ or in the unlikely event that herbicides fall directly in the water from spray drift or by accidentally directing the spray stream into water.

For an herbicide to have an adverse effect on EFH, the chemical must be of sufficient concentration or present for adequate duration in water to cause a reduction in the quantity or quality of EFH. Anticipated changes in water quality caused by the action can reasonably be expected to cause adverse effects to some Chinook salmon. Thus, effects to the water quality of EFH in the action area will be adverse since the nature of the effect is adequate to influence the species negatively given the small amounts of chemical proposed for use in any given area, maximum possible concentrations of the herbicides can seldom reach thresholds where toxic effects are likely to occur. In the limited circumstances where toxic thresholds are reached, the effects are likely to be sublethal and herbicide concentrations are likely to rapidly drop with increasing distance from the treatment area due to dispersion of the herbicides and increasing stream discharge. Most of the herbicides proposed for use break down chemically in a matter of months, although clopyralid and picloram may be present in the environment for longer periods. Under the worst possible contamination scenario, herbicides are not likely to reach lethal concentrations, sublethal concentrations would likely occur in only a few treatment locations, and sublethal water quality levels will exist only briefly. For these reasons adverse effects to EFH are likely to occur infrequently and last for brief periods of time before water quality returns to pre-application conditions.

3.3 Essential Fish Habitat Conservation Recommendations

NMFS believes that the following Conservation Recommendations are necessary to avoid, mitigate, or offset the impact that the proposed action has on EFH. These Conservation Recommendations are a non-identical set of the ESA Terms and Conditions.

1. The USFS should review the USFS herbicide spill plan with all applicators, including permittees, prior to herbicide applications. The plan should meet requirements of the USFS directives (FSH 2109.14, Chapter 60) regardless of party performing the application.

2. The Forests should require any permittee/applicator have an appropriate spill cleanup kit on-site throughout any permitted herbicide application.

3. To most effectively avoid/minimize potential effects to EFH, all PDC described in the proposed action and summarized in this Opinion should be consistently applied in the action area to reduce the likelihood of surface runoff, drift, or other groundwater contamination pathways.

4. The Forests should use added precaution when applying herbicides near streams or on trails and drainage ditches that drain directly into streams. Herbicides approved for use within riparian areas, and use of the least toxic surfactant compatible with the EUP, should be the product of choice under appropriate site conditions.

5. The Forests should use added precaution or limit use of herbicides with high persistence in soil (e.g., picloram, clopyralid, chlorsulfuron, and metsulfuron-methyl) within annual floodplains and where soil permeability is expected to be high (e.g., silt loam and sand soils). Where necessary to achieve weed management objectives, use application methods with the lowest potential for off-site transport (e.g., large droplet size, hand spraying, etc.).

6. The Forests should only use the Washington State aquatic-certified adjuvants in riparian areas as identified in the BA and this Opinion. In order to add new Washington State aquatic-certified adjuvants, the SCNF should present documentation (e.g., toxicity information, ingredient information, and registration information) to the Level 1 Team demonstrating that the potential the effects of the ingredients have been tested on salmonids and have been found to be of low toxicity and are similar to those adjuvants considered in this Opinion.

7. The EUPs other than those evaluated in this Opinion and identified in Table 3 should not used without additional MSA EFH consultation or Level 1 review demonstrating that the potential effects of using the EUP are similar to those considered in this Opinion. The Level 1 Team should be notified about all additions or substitutions of EUPs. The Forests should present Pesticide Use Proposal's with planned EUPs (name, active ingredient, registration number) to the Level 1 Team prior to use. The Level 1 Team should agree that EUPs, and their effects to ESA-listed fish and critical habitats, are substantially similar to formulations identified in this Opinion. Should a concern be raised regarding an EUPs potential to affect ESA-listed fish and critical habitats, the Level I team member raising the concern should provide documentation, research or best available science supporting why the EUP should not be used. All parties should work to conclude discussions in a timely manner so as not to delay treatments.

NMFS expects that full implementation of these EFH Conservation Recommendations would protect, by avoiding or minimizing the adverse effects described in Section 3.2 above on all EFH present in the FC-RONRW and potentially affected by plant treatments over the 20-year consultation term.

3.4 Statutory Response Requirement

As required by section 305(b)(4)(B) of the MSA, the Forests must provide a detailed response in writing to NMFS within 30 days after receiving an EFH Conservation Recommendation. Such a response must be provided at least 10 days prior to final approval of the action if the response is inconsistent with any of NMFS' EFH Conservation Recommendations unless NMFS and the federal agency have agreed to use alternative timeframes for the federal agency response. The response must include a description of measures proposed by the agency for avoiding, minimizing, mitigating, or otherwise offsetting the impact of the activity on EFH. In the case of a response that is inconsistent with the Conservation Recommendations, the federal agency must explain its reasons for not following the recommendations, including the scientific justification for any disagreements with NMFS over the anticipated effects of the action and the measures needed to avoid, minimize, mitigate, or offset such effects (50 CFR 600.920(k)(1)).

In response to increased oversight of overall EFH program effectiveness by the Office of Management and Budget, NMFS established a quarterly reporting requirement to determine how many conservation recommendations are provided as part of each EFH consultation and how many are adopted by the action agency. Therefore, we ask that in your statutory reply to the EFH portion of this consultation, you clearly identify the number of conservation recommendations accepted.

3.5 Supplemental Consultation

The USFS must reinitiate EFH consultation with NMFS if the proposed action is substantially revised in a way that may adversely affect EFH, or if new information becomes available that affects the basis for NMFS' EFH Conservation Recommendations (50 CFR 600.920(l)).

4. DATA QUALITY ACT DOCUMENTATION AND PRE-DISSEMINATION REVIEW

The DQA specifies three components contributing to the quality of a document. They are utility, integrity, and objectivity. This section of the Opinion addresses these DQA components, documents compliance with the DQA, and certifies that this Opinion has undergone pre-dissemination review.

4.1 Utility

Utility principally refers to ensuring that the information contained in this consultation is helpful, serviceable, and beneficial to the intended users. The intended users of this Opinion are the USFS. Other interested users could include permittees or contractors interested in the conservation of the affected ESUs/DPS. Individual copies of this Opinion were provided to the USFS as well as the Nez Perce and Shoshone-Bannock Tribes. The format and naming adheres to conventional standards for style.

4.2 Integrity

This consultation was completed on a computer system managed by NMFS in accordance with relevant information technology security policies and standards set out in Appendix III, 'Security of Automated Information Resources,' Office of Management and Budget Circular A-130; the Computer Security Act; and the Government Information Security Reform Act.

4.3 Objectivity

Information Product Category: Natural Resource Plan

Standards: This consultation and supporting documents are clear, concise, complete, and unbiased; and were developed using commonly accepted scientific research methods. They adhere to published standards including NMFS' ESA Consultation Handbook, ESA regulations, 50 CFR 402.01 et seq., and the MSA implementing regulations regarding EFH, 50 CFR 600.

Best Available Information: This consultation and supporting documents use the best available information, as referenced in the References section. The analyses in this Opinion and EFH consultation contain more background on information sources and quality.

Referencing: All supporting materials, information, data and analyses are properly referenced, consistent with standard scientific referencing style.

Review Process: This consultation was drafted by NMFS staff with training in ESA MSA implementation, and reviewed in accordance with West Coast Region ESA quality control and assurance processes.

5. REFERENCES

Ahrens, W.H. 1994. Herbicide Handbook. Seventh Edition. Weed Society of Americ~Champaign, Illinois. 352 pages.

Asch, R. 2015. Climate change and decadal shifts in the phenology of larval fishes in the California Current ecosystem. PNAS:E4065-E4074, 7/9/2015.

Aspelin, A.A. and A.H. Grube. 1999. Pesticide industry sales and usage. 1996 and 1997 market estimates. 733-R-99-001. Office of Pesticide Programs, U.S. Environmental Protection Agency, Washington, D.C. Available: http://www.epa.goy/oppOOOOl/pestsales/. (May 2012).

Bakke D. 2003. USPS. Human and Ecological Risk Assessment of Nonylphenol Polyethoxylate-based (NPE) Surfactants in Forest Service Herbicide Applications. Internal Report, Pacific Southwest Region (Region 5).

Bakun, A., B. A. Black, S. J. Bograd, M. García-Reyes, A. J. Miller, R. R. Rykaczewski, and J. Sydeman. 2015. Anticipated Effects of Climate Change on Coastal Upwelling Ecosystems. Current Climate Change Reports 1:85-93. DOI: 10.1007/s40641-015-0008-4, 3/7/2015.

Baldwin, D.H., J.A. Spromberg, T.K. Colier, and N.L. Scholz. 2009. A fish of many scales: extrapolating sublethal pesticide exposures to the productivity of wild salmon populations. Ecological Applications, 19 (8), pp. 2004-2015.

Battin, J., and coauthors. 2007. Projected impacts of climate change on salmon habitat restoration. Proceedings of the National Academy of Sciences of the United States of America 104(16):6720-6725.

Beamish, R. J. and C. Mahnken. 2001. A critical size and period hypothesis to explain natural regulation of salmon abundance and the linkage to climate and climate change. Progress in Oceanography 49:423-437.

Beckman, B. 2018. Estuarine growth of yearling Snake River Chinook salmon smolts. Progress report. Northwest Fisheries Science Center, Seattle, Washington, 7/3/2018.

Beechie, T., H. Imaki, J. Greene, et al. 2013. Restoring Salmon Habitat for a Changing Climate. River Research and Application 29:939-960.

Berg, N. 2004. Assessment of Herbicide Best Management Practices: Status of Our Knowledge of BMP Effectiveness. Final Report, March 2004, Pacific Southwest Research Station, USDA Forest Service, Albany, CA.

Bisson, P. A. and R. E. Bilby. 1998. Organic matter and trophic dynamics. Pages 373 - 398 in: Naiman, R. J. and R. E. Bilby, editors. River ecology and management: lessons from the Pacific coastal ecoregion. Springer-Verlag, New York, NY.

Bjornn, T. C. and D. W. Reiser. 1991. Habitat requirements of salmonids in streams. Pages 83–138 in W.R. Meehan, editor. Influences of forest and rangeland management on salmonid fishes and their habitats. American Fisheries Society, Special Publication 19. Bethesda, Maryland.

Bjornn, T. C., D. R. Craddock, and D. R. Corley. 1968. Migration and survival of Redfish Lake, Idaho, sockeye salmon, Oncorhynchus nerka. Transactions of the American Fisheries Society. 97:360-373.

Black, B., J. Dunham, B. Blundon, J. Brim Box, and A. Tepley. 2015. Long-term growth-increment chronologies reveal diverse influences of climate forcing on freshwater and forest biota in the Pacific Northwest. Global Change Biology 21:594-604. DOI: 10.1111/gcb.12756.

Bograd, S., I. Schroeder, N. Sarkar, X. Qiu, W. J. Sydeman, and F. B. Schwing. 2009. Phenology of coastal upwelling in the California Current. Geophysical Research Letters 36:L01602. DOI: 10.1029/2008GL035933.

Bond, N. A., M. F. Cronin, H. Freeland, and N. Mantua. 2015. Causes and impacts of the 2014 warm anomaly in the NE Pacific. Geophysical Research Letters 42:3414–3420. DOI: 10.1002/2015GL063306.

Budy, P., G.P. Thiede, N. Bouwes, C.E. Petrosky, H. Schaller. 2002. Evidence linking delayed mortality of Snake River salmon to their earlier hydrosystem experience. North American Journal of Fisheries Management 22:35-51.

Cairns, J. Jr. 1983. Are single species toxicity tests alone adequate for estimating environmental hazard? Hydrobiologia. Vol. 100, Issue 1, pgs. 47-57.

Chapman, D., W. Platts, D. Park and M. Hill. 1990. Status of Snake River sockeye salmon. Final Report to PNUCC, June 26. Don Chapman Consultants Inc.: Boise, Idaho. 96 p.

Cheung, W., N. Pascal, J. Bell, L. Brander, N. Cyr, L. Hansson, W. Watson-Wright, and D. Allemand. 2015. North and Central Pacific Ocean region. Pages 97-111 in N. Hilmi, D. Allemand, C. Kavanagh, and et al, editors. Bridging the Gap Between Ocean Acidification Impacts and Economic Valuation: Regional Impacts of Ocean Acidification on Fisheries and Aquaculture. DOI: 10.2305/IUCN.CH.2015.03.en.

CH2MHILL. 2004. Programmatic Biological Assessment of Effects of the Noxious Weed Management Program on Lands Administered by the Salmon-Challis National Forest on Federally Listed, Proposed, and Candidate Plant, Wildlife, and Fish Species and Critical Habitat. Prepared for the USFS, Salmon-Challis National Forest, Salmon, Idaho. April 2004.

Climate Change Science Program (CCSP). 2014. Climate Change Impacts in the United States. Third National Climate Assessment. U.S. Global Change Research Program. DOI:10.7930/J0Z31WJ2.

Climate Impacts Group (CIG). 2004. Overview of Climate Change Impacts in the U.S. Pacific Northwest, 7/29/2004.

Coutant, C. C. 1999. Perspectives on temperature in the Pacific Northwest's fresh waters. Oak Ridge National Laboratory, Report ORNL/TM-1999/44, Oak Ridge, Tennessee.

Crozier, L. and R. W. Zabel. 2006. Climate impacts at multiple scales: evidence for differential population responses in juvenile Chinook salmon. Ecology 75:1100-1109. DOI: 10.1111/j.1365-2656.2006.01130.x.

Crozier, L. G., R. W. Zabel, and A. F. Hamlet. 2008a. Predicting differential effects of climate change at the population level with life-cycle models of spring Chinook salmon. Global Change Biology 14:236-249. DOI: 10.1111/j.1365-2486.2007.01497.x.

Crozier, L. G., A. P. Hendry, P. W. Lawson, T. P. Quinn, et al. 2008b. Potential responses to climate change for organisms with complex life histories: evolution and plasticity in Pacific salmon. Evolutionary Applications 1:252-270. DOI: 10.1111/j.1752-4571.2008.00033.x.

Dalton, M., P. W. Mote, and A. K. Stover. 2013. Climate change in the Northwest: implications for our landscapes, waters and communities. Island Press, Washington, D.C.

Daly, E. A., R. D. Brodeur, and L. A. Weitkamp. 2009. Ontogenetic Shifts in Diets of Juvenile and Subadult Coho and Chinook Salmon in Coastal Marine Waters: Important for Marine Survival? Transactions of the American Fisheries Society 138(6):1420-1438.

Daly, E. A., J. A. Scheurer, R. D. Brodeur, L. A. Weitkamp, B. R. Beckman, and J. A. Miller. 2014. Juvenile Steelhead Distribution, Migration, Feeding, and Growth in the Columbia River Estuary, Plume, and Coastal Waters. Marine and Coastal Fisheries: Dynamics, Management, and Ecosystem Science 6(1):62-80.

DeLorenzo, M. E., G. I. Scott, and P. E. Ross. 2001. Toxicity of pesticides to aquatic microorganisms: A review. Environmental Toxicology and Chemistry 20:84-98.

Di Lorenzo, E. and N. Mantua. 2016. Multi-year persistence of the 2014/15 North Pacific marine heatwave. Nature Climate Change 1-7. DOI:10.1038/nclimate3082, 7/11/2016.

Ebel, W. J., H. L. Raymond, G. E. Monan, W. E. Farr, and G. K Tanoka. 1975. Effects of atmospheric gas saturation caused by dams on salmon and steelhead trout of the Snake and Columbia rivers. U.S. National Marine Fisheries Service, Seattle, Washington.

Ecovista, Nez Perce Tribe Wildlife Division, and Washington State University Center for Environmental Education. 2003. Draft Clearwater Subbasin Assessment, Prepared for Nez Perce Tribe Watersheds Division and Idaho Soil Conservation Commission. 463 p. http://www.nwcouncil.org/fw/subbasinplanning/clearwater/plan/Default.htm

ENTRIX. 2003. Ecological Risk Assessment of the Proposed Use of the Herbicide Imazapyr to Control Invasive Cordgrass (Spartina spp.) in Estuarine Habitat of Washington State, prepared for Washington State Department of Agriculture, October 30, 2003. 160 pp. http://www.ecy.wa.gov/programs/wq/pesticides/final_pesticide_permits/noxious/risk_assessment_Imazapyr.pdf

Everest, F. H. and D. W. Chapman. 1972. Habitat selection and spatial interaction by juvenile Chinook salmon and steelhead trout in two Idaho streams. Journal of the Fisheries Research Board of Canada 29(1):91-100.

Fairchild, J. F., A. Allert, K. P. Feltz, K. J. Nelson, and J. A. Valle. 2009a. An ecological risk assessment of the acute and chronic effects of the herbicide clopyralid to rainbow trout (*Oncorhynchus mykiss*). Archives of Environmental Contamination and Toxicology 57:725-731.

Fairchild, J. F., K. P. Feltz, L. C. Sappington, A. L. Allert, K. J. Nelson, and J. Valle. 2009b. An ecological risk assessment of the acute and chronic toxicity of the herbicide picloram to the threatened bull trout (*Salvelinus confluentus*) and the rainbow trout (*Oncorhynchus mykiss*). Archives of Environmental Contamination and Toxicology 56(4):761-769.

Fisher, J., W. Peterson, and R. Rykaczewski. 2015. The impact of El Niño events on the pelagic food chain in the northern California Current. Global Change Biology 21: 4401-4414. DOI: 10.1111/gcb.13054, 7/1/2015.

Ford, M. J. (ed.). 2011. Status review update for Pacific salmon and steelhead listed under the Endangered Species Act: Pacific Northwest. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-NWFSC-113, 281 p. http://www.westcoast.fisheries.noaa.gov/publications/status_reviews/salmon_steelhead/multiple_species/5-yr-sr.pdf

Foreman, M., W. Callendar, D. Masson, J. Morrison, and I. Fine. 2014. A Model Simulation of Future Oceanic Conditions along the British Columbia Continental Shelf. Part II: Results and Analyses. Atmosphere-Ocean 52(1):20-38. DOI: 10.1080/07055900.2013.873014.

Gargett, A. 1997. Physics to Fish: Interactions Between Physics and Biology on a Variety of Scales. Oceanography 10(3):128-131.

Good, T. P., R. S. Waples, and P. Adams (editors). 2005. Updated status of federally listed ESUs of West Coast salmon and steelhead. U.S. Dept. Commerce, NOAA Tech. Memo. NMFS-NWFSC-66, 598 p.

Haigh, R., D. Ianson, C. A. Holt, H. E. Neate, and A. M. Edwards. 2015. Effects of Ocean Acidification on Temperate Coastal Marine Ecosystems and Fisheries in the Northeast Pacific. PLoS ONE 10(2):e0117533. DOI:10.1371/journal.pone.0117533, 2/11/2015.

Healey, M. C. 1982. Timing and relative intensity of size-selective mortality of juvenile chum salmon (Oncorhynchus keta) during early sea life. Canadian Journal of Fisheries and Aquatic Sciences 39:952-957.

Healey, M. C. 1991. Life history of chinook salmon (Oncorhynchus tshawytscha). Pages 80 in C. Groot, and L. Margolis, editors. Pacific salmon life histories. University of British Columbia Press, Vancouver, Canada.

Herron, G. J., R. L. Sheley, B. D. Maxwell, and J. S., Jacobsen. 2001. Influence of Nutrient Availability on the Interaction Between Spotted Knapweed and Bluebunch Wheatgrass. Restoration Ecology. Vol. 9 No. 3, pp. 326–331

Higgs, D. A., J. S. MacDonald, C. D., Levings, and B. S. Dosanjh. 1995. Nutrition and feeding habits in relation to life history stage. Pages 161-315 in: Groot, C., L. Margolis and W.C. Clark, editors. Physiological ecology of Pacific Salmon. UBC Press, Vancouver, Canada.

Hoagland, K. D., J. P. Carder, and R. L. Spawn. 1996. Effects of organic toxic substances. Pages 469- 496 *in*: Stevenson, R. J., M. L. Bothwell, and R. L. Lowe, editors. Algal ecology: Freshwater Benthic Ecosystems. Academic Press. New York, NY.

Hollowed, A. B., N. A. Bond, T. K. Wilderbuer, W. T. Stockhausen, Z. T. A'mar, R. J. Beamish, J. E. Overland, et al. 2009. A framework for modelling fish and shellfish responses to future climate change. ICES Journal of Marine Science 66:1584-1594. DOI:10.1093/icesjms/fsp057.

Holtby, L.B, Andersen, B.C., and R.K. Kadowaki. 1990. Importance of smolt size and early ocean growth in inter-annual variability in marine survival of coho salmon (Oncorhynchus kisutch). Canadian Journal of Fisheries and Aquatic Sciences 47:2181-2194.

Idaho Department of Environmental Quality (IDEQ). 2001. Middle Salmon River-Panther Creek Subbasin Assessment and TMDL. March 2001.

IDEQ. 2002. Middle Salmon River-Chamberlain Creek Subbasin Assessment and Crooked Creek TMDL. Revised Final, December, 2002.

IDEQ. 2003. Upper Salmon River Subbasin Assessment and TMDL. January 2003.

IDEQ. 2008. Middle Fork Salmon River Subbasin Assessment and TMDLs. November 2008.

IDEQ. 2010. 2010 Integrated 303(d)/305(b) Report. Section 5: Impaired Waters: Lakes and Rivers (303(d) list)

IDEQ. 2011. Idaho's 2010 Integrated Report, Final. IDEQ: Boise, Idaho. 776 p.

IDEQ and U.S. Environmental Protection Agency (EPA). 2003. South Fork Clearwater River Subbasin Assessment and Total Maximum Daily Loads. IDEQ: Boise, Idaho. 680 p.

Idaho Department of Fish and Game. 2007. Mountain Whitefish Conservation and Management Plan for the Big Lost River Drainage, Idaho. May 2007.

IDFG. 2007. Annual returns to Lower Granite Dam, Idaho Department of Fish and Game data provided to NMFS by Peter Hassemer, December 2007. IDFG: Boise, Idaho.

IDFG. 2016. "Hatchery and Wild Chinook Salmon Return to Lower Granite Dam." Follow Idaho Salmon Home (F.I.S.H) website, http://216.206.157.62/idaho/web/apps/MAIN_LGRchinadultreturn_spsu_wild.php, accessed 6-3-16.

Interior Columbia Technical Recovery Team (ICTRT). 2003. Working draft. Independent populations of Chinook, steelhead, and sockeye for listed evolutionarily significant units within the Interior Columbia River domain. NOAA Fisheries. July.

ICTRT. 2007. Viability Criteria for Application to Interior Columbia Basin Salmonid ESUs, Review Draft March 2007. Interior Columbia Basin Technical Recovery Team: Portland, Oregon. 261 pp. https://www.nwfsc.noaa.gov/research/divisions/cb/genetics/trt/trt_documents/ictrt_viability_criteria_reviewdraft_2007_complete.pdf

ICTRT. 2010. Status Summary – Snake River Spring/Summer Chinook Salmon ESU. Interior Columbia Technical Recovery Team: Portland, Oregon.

Independent Scientific Advisory Board (ISAB). 2007. Climate change impacts on Columbia River Basin fish and wildlife. ISAB Climate Change Report, ISAB 2007-2, Northwest Power and Conservation Council, Portland, Oregon.

Jones, D.K., J.I. Hamond, and R.A. Relyea. 2011. Competitive stress can make the herbicide Roundup more deadly to larval amphibians. Environmental Toxicology and Chemistry 30: 446-454.

Jones, K. K., T. J. Cornwell, D. L. Bottom, L. A. Campbell, and S. Stein. 2014. The contribution of estuary-resident life histories to the return of adult Oncorhynchus kisutch. Journal of Fish Biology 85:52–80. DOI:10.1111/jfb.12380.

Kennedy, V. S. 1990. Anticipated Effects of Climate Change on Estuarine and Coastal Fisheries. Fisheries 15(6):16-24.

Kimball, K. D. and S. A. Levin. 1985. Limitations of Laboratory Bioassays: The Need for Ecosystem-Level Testing. BioScience, Vol 35, Issue 3, pgs. 165-171.

Kirwan, M. L., G. R. Guntenspergen, A. D'Alpaos, J. T. Morris, S. M. Mudd, and S. Temmerman. 2010. Limits on the adaptability of coastal marshes to rising sea level. Geophysical Research Letters 37:L23401. DOI: 10.1029/2010GL045489, 12/1/2010.

Korman, J., D. R. Marmorek, G. L. Lacroix, P. G. Amiro, J. A. Ritter, W. D. Watt, R. E. Cutting, and D. C. E. Robinson. 1994. Development and evaluation of a biological model to assess regional-scale effects of acidification on Atlantic salmon (Salmo salar). Canadian Journal of Fisheries and Aquatic Sciences 51:662–680.

Kratz, K., L. Croft, V. Kozakiewicz, D. Bambrick, C. Tortorici, C. Toole,R. Tweeten, D. Guy, B. Meyer, N. Munn, B. Ries, and L. Zuckerman. 2004. Habitat Improvements as Offsite Mitigation for FCRPS Operations: A Qualitative Assessment. U.S. Department of Commerce, NMFS, Portland, Oregon.

Kruzynski, G. M. and I. K. Birtwell. 1994. A predation bioassay to quantify the ecological significance of sublethal responses of juvenile Chinook salmon (*Oncorhynchus tshawytscha*) to the antisapstain fungicide TCMTB. Canadian Journal of Fisheries and Aquatic Sciences 51:1780-1790.

Kruzynski, G. M., I. K. Birtwell, and G. L. Chew. 1994. Behavioural approaches to demonstrate the ecological significance of exposure of juvenile Pacific salmon (genus *Oncorhynchus*) to the antisapstain fungicide TCMTB. Journal of Aquatic Ecosystem Health 3:113-127.

Laetz, C. A., D. H. Baldwin, V. Hebert, J. D. Stark, N. L. Scholz. 2014. Elevated temperatures increase the toxicity of pesticide mixtures to juvenile coho salmon. Aquatic Toxicology, 146:38-44.

Larson, A., C. Haux, and M. Sjöbeck. 1985. Fish physiology and metal pollution: Results and experiences from laboratory and field studies. Ecotoxicology and Environmental Safety. Volume 9, Issue 3, pp. 250-281.

Larson, D. 2003. Native weeds and exotic plants: relationships to disturbance in mixed-grass prairie. Plant Ecology 169: 317–333.

Lemmen, D. S., F. J. Warren, T. S. James, and C. S. L. Mercer Clarke (Eds.). 2016. Canada's Marine Coasts in a Changing Climate. Ottawa, ON: Government of Canada.

Limburg, K., R. Brown, R. Johnson, B. Pine, R. Rulifson, D. Secor, K. Timchak, B. Walther, and K. Wilson. 2016. Round-the-Coast: Snapshots of Estuarine Climate Change Effects. Fisheries 41(7):392-394, DOI: 10.1080/03632415.2016.1182506.

Litz, M. N., A. J. Phillips., R. D. Brodeur, and R. L. Emmett. 2011. Seasonal occurrences of Humboldt Squid in the northern California Current System. California Cooperative Oceanic Fisheries Investigations Report. December 2011 Vol. 52. 97-108.

Lucey, S. and J. Nye. 2010. Shifting species assemblages in the Northeast US Continental Shelf Large Marine Ecosystem. Marine Ecology Progress Series, Marine Ecology Progress Series 415:23-33. DOI: 10.3354/meps08743.

Lynch, A. J., B. J. E. Myers, C. Chu, L. A. Eby, J. A. Falke, R. P. Kovach, T. J. Krabbenhoft, T. J. Kwak, J. Lyons, C. P. Paukert, and J. E. Whitney. 2016. Climate Change Effects on North American Inland Fish Populations and Assemblages. Fisheries 41(7):346-361. DOI: 10.1080/03632415.2016.1186016, 7/1/2016.

Mantua, N. J., S. Hare, Y. Zhang, et al. 1997. A Pacific interdecadal climate oscillation with impacts on salmon production. Bulletin of the American Meteorological Society 78:1069-1079, 1/6/1997.

Martins, E. G., S. G. Hinch, D. A. Patterson, M. J. Hague, S. J. Cooke, K. M. Miller, M. F. Lapointe, K. K. English, and A. P. Farrell. 2011. Effects of river temperature and climate warming on stock-specific survival of adult migrating Fraser River sockeye salmon (*Oncorhynchus nerka*). Global Change Biology 17(1):99–114. DOI:10.1111/j.1365-2486.2010.02241.x.

Martins, E. G., S. G. Hinch, D. A. Patterson, M. J. Hague, S. J. Cooke, K. M. Miller, D. Robichaud, K. K. English, and A. P. Farrell. 2012. High river temperature reduces survival of sockeye salmon (*Oncorhynchus nerka*) approaching spawning grounds and exacerbates female mortality. Canadian Journal of Fisheries and Aquatic 69:330–342. DOI: 10.1139/F2011-154.

Mason, J. C. 1976. Response of underyearling coho salmon to supplemental feeding in a natural stream. Journal of Wildlife Management. 40:775-788.

Mathis, J. T., S. R. Cooley, N. Lucey, S. Colt, J. Ekstrom, T. Hurst, C. Hauri, W. Evans, J. N. Cross, and R. A. Feely. 2015. Ocean acidification risk assessment for Alaska's fishery sector. Progress in Oceanography 136:71-91.

Matthews, G. M., D. L. Park, S. Achord, and T .E. Ruehle. 1986. Static seawater challenge test to measure relative stress levels in spring Chinook salmon smolts. Transactions of the American Fisheries Society 115: 236–244.

Matthews, G. M., R. S. Waples. 1991. Status Review for Snake River Spring and Summer Chinook Salmon. U.S. Dept. of Commerce, NOAA Tech. Memo., NMFS-F/NWC-200. https://www.nwfsc.noaa.gov/publications/scipubs/techmemos/tm201/

McClure, M., T. Cooney, and ICTRT. 2005. Updated population delineation in the interior Columbia Basin. May 11, 2005 Memorandum to NMFS NW Regional Office, Co-managers, and other interested parties. NMFS: Seattle, Washington. 14 p.

McElhany, P., M. H. Ruckelshaus, M. J. Ford, T. C. Wainwright, and E. P. Bjorkstedt. 2000. Viable salmonid populations and the recovery of evolutionarily significant units. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-NWFSC-42, Seattle, Washington, 156 p.

Minshall, G. W. 1978. Autotrophy in stream ecosystems. BioScience 28:767-771.

Moore, A. W. and C. P. Waring. 1996. Sublethal effects of the pesticide Diazinon on olfactory function in mature male Atlantic salmon parr. Journal of Fish Biology 48:758-775.

Morris, J. F. T., M. Trudel, J. Fisher, S. A. Hinton, E. A. Fergusson, J. A. Orsi, and J. Edward V. Farley. 2007. Stock-Specific Migrations of Juvenile Coho Salmon Derived from Coded-Wire Tag Recoveries on the Continental Shelf of Western North America. American Fisheries Society Symposium 57:81-104.

Mote, P. W., E. A. Parson, A. F. Hamlet, et al. 2003. Preparing for Climatic Change: The Water, Salmon, and Forests of the Pacific Northwest. Climatic Change 61:45-88.

Mundie, J. H. 1974. Optimization of the salmonid nursery stream. Journal of the Fisheries Research Board of Canada 31: 1827-1837.

Murphy, M. L. 1998. Primary production. Pages 144-168 in: Naiman R. J. and R. E. Bilby, editors. River ecology and management: lessons from the Pacific coastal ecoregion. Springer-Verlag, New York, NY.

Naiman, R. J., J. R. Alldredge, D. A. Beauchamp, P. A. Bisson, J. Congleton, C. J. Henny, N. Huntly, R. Lamberson, C. Levings, E. N. Merrill, W. G. Pearcy, B. E. Rieman, G. T. Ruggerone, D. Scarnecchia, P. E. Smouse, and C. C. Wood. 2012. Developing a broader scientific foundation for river restoration: Columbia River food webs. Proceedings of the National Academy of Sciences of the United States of America 109(52):21201-21207.

National Marine Fisheries Service (NMFS). 2012. Endangered Species Act Section 7(a)(2) Biological Opinion and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Consultation for the Sawtooth National Forest’s Programmatic Noxious and Invasive Weed Management Program; Upper Salmon River Subbasin HUC 17060201; Custer and Blaine Counties, Idaho (One Project). November 26, 2012. NMFS No.: 2012/01793.

NMFS. 2013. Reinitiation for the Biological Assessment of Effects of Treatment of Noxious Weeds in the Frank Church-River of No Return Wilderness Management – Salmon-Challis, Boise, Payette, Nez Perce and Bitterroot National Forests; Hydrologic Unit Codes: Salmon River 17060201, 17060203, 17060205, 17060206, 17060207, and 17060208, and Upper Selway River 17060301; Custer, Idaho, Lemhi, and Valley Counties, Idaho. (One Action). October 13, 2009. NMFS No.: 2004/01889; 2009/03999.

NMFS. 2015. ESA Recovery Plan for Snake River Sockeye Salmon (Oncorhynchus nerka), June 8, 2015. NOAA Fisheries, West Coast Region. 431 p. http://www.westcoast.fisheries.noaa.gov/publications/recovery_planning/salmon_steelhead/domains/interior_columbia/snake/snake_river_sockeye_recovery_plan_june_2015.pdf

NMFS. 2016. Draft Proposed ESA Recovery Plan for Snake River Idaho Spring/Summer Chinook Salmon and Steelhead Populations. October. West Coast Region, Portland Oregon.

NMFS. 2016b. Endangered Species Act Section 7(a)(2) Biological Opinion, and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Consultation for the Salmon-Challis National Forest Invasive Plant Treatment Program (Non-wilderness Areas), Upper Salmon River (17060201); Pahsimeroi River (17060202); Middle Salmon-Panther (17060203); Lemhi River (17060204); Upper Middle Fork Salmon (17060205); Lower Middle Fork Salmon (17060206); Middle Salmon-Chamberlain (17060207); Little Lost River (17040217); Big Lost River (17040218); Lemhi and Custer Counties, Idaho. May 17, 2016. WCR-2015-3863.

NMFS. 2016c. Endangered Species Act Section 7(a)(2) Biological Opinion, and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Response for the Proposed Poison Ivy Control at Snake River Recreation and Administrative Sites Project. July 22, 2016. NMFS No.: 2016-4766.

NMFS. 2017. ESA Recovery Plan for Snake River Spring/Summer Chinook & Steelhead. NMFS. http://www.westcoast.fisheries.noaa.gov/publications/recovery_planning/salmon_steelhead/domains/interior_columbia/snake/Final%20Snake%20Recovery%20Plan%20Docs/final_snake_river_spring-summer_chinook_salmon_and_snake_river_basin_steelhead_recovery_plan.pdf

Northwest Fisheries Science Center (NWFSC). 2015. Status review update for Pacific salmon and steelhead listed under the Endangered Species Act: Pacific Northwest. 356 p.

Pacific Fishery Management Council (PFMC). 1999. Description and identification of essential fish habitat for the Coastal Pelagic Species Fishery Management Plan. Appendix D to Amendment 8 to the Coastal Pelagic Species Fishery Management Plan. Pacific Fishery Management Council, Portland, Oregon. December.

Parker, R. R. 1971. Size selective predation among juvenile salmonid fishes in a British Columbia inlet. Journal of the Fisheries Research Board of Canada 28:1503-1510.

Pearcy, W. G. 2002. Marine nekton off Oregon and the 1997–98 El Niño. Progress in Oceanography 54:399–403.

Pearcy, W. G. and S. M. McKinnell. 2007. The Ocean Ecology of Salmon in the Northeast Pacific Ocean-An Abridged History. American Fisheries Society 57:7-30.

Pesticide Action Network (PAN). 2012. Pesticide Database. Available: www.pesticideinfo.org. (July 2012)

Peterson, W., J. Fisher, J. Peterson, C. Morgan, B. Burke, and K. Fresh. 2014. Applied Fisheries Oceanography Ecosystem Indicators of Ocean Condition Inform Fisheries Management in the California Current. Oceanography 27(4):80-89. 10.5670/oceanog.2014.88.

Poesch, M. S., L. Chavarie, C. Chu, S. N. Pandit, and W. Tonn. 2016. Climate Change Impacts on Freshwater Fishes: A Canadian Perspective. Fisheries 41:385-391.

Preston, B. L. 2002. Indirect effects in aquatic ecotoxicology: implications for ecological risk assessment. Environmental Management 29:311-323.

Rashin, E. and C. Graber. 1993. Effectiveness of best management practices for aerial application of forest practices. Ecology Publication No. 93-81, Prepared for the Timber/Fish/Wildlife Cooperative Monitoring Evaluation and Research Committee. Olympia, Washington.

Rehage J. S. and J. R. Blanchard. 2016. What can we expect from climate change for species invasions? Fisheries 41(7):405-407. DOI: 10.1080/03632415.2016.1180287.

Ricker, W.E. 1976. Review of the rate of growth and mortality of Pacific salmon in salt water, and noncatch mortality caused by fishing. Journal of the Fisheries Research Board of Canada. 33:1483-1524.

Rykaczewski, R., J. P. Dunne, W. J. Sydeman, et al. 2015. Poleward displacement of coastal upwelling-favorable winds in the ocean's eastern boundary currents through the 21st century. Geophysical Research Letters 42:6424-6431. DOI:10.1002/2015GL064694.

Scheuerell, M. D. and J. G. WIlliams. 2005. Forecasting climate-induced changes in the survival of Snake River spring/summer Chinook salmon (Oncorhynchus tshawytscha). Fisheries Oceanography 14(6):448–457.

Scholz, N. L., J. P. Incardona, C. M. Stehr, and T. L. Linbo. 2005. Evaluating the effects of forestry herbicides on early development of fish using the zebrafish phenotypic screen. FS-PIAP FY 03-04 Final Report, November 18, 2005.

Scholz, N. L., N. K. Truelove, B. L. French, B. A. Berejikian, T. P. Quinn, E. Casillas, and T. K. Collier. 2000. Diazinon disrupts antipredator and homing behaviors in Chinook salmon (Oncorhynchus tshawytscha). Canadian Journal of Fisheries and Aquatic Sciences 57(9):1911-1918.

Syracuse Environmental Research Associates, Inc. (SERA). 2009. Fluroxypyr - Human Health and Ecological Risk Assessment FINAL REPORT. SERA TR-052-12-03a. Prepared for: USDA USFS.

SERA. 2011. Glyphosate – human health and ecological risk assessment – final report. SERA TR-052-22-03b. Manlius, New York. Prepared for: USFS, Southern Region, Atlanta, Georgia. Available: www.fs.fed.us/foresthealth/pesticide/risk.shtml. (May 2012).

Smith, B. C., C. A. Curran, K. W. Brown, J. L. Cabarrus, J. B. Gown, J. K. Mcintyre, E. E. Moreland, V. L. Wong, J. M. Grassley, and C. E. Grue. 2004. Toxicity of Four Surfactants to juvenile rainbow trout: Implications for use over water. Bulletin of Environmental Contamination and Toxicology 72:647-654.

Sogard, S. M. 1997. Size-selective mortality in the juvenile stage of teleost fishes: a review. Bulletin of Marine Science 60:1129-1167.

Solomon, K. R. and D. G. Thompson. 2003. Ecological Risk Assessment for Aquatic Organisms from Over-Water Uses of Glyphosate. May 2003

Spence, B., G. Lomnicky, R. Hughes, and R. P. Novitski. 1996. An ecosystem approach to salmonid conservation. TR-4501-96-6057. ManTech Environmental Research Services Corp.: Corvallis, Oregon.

Stark, J. D. and W. K. Walthall. 2003. Agricultural adjuvants: Acute Mortality and Effects on Population growth Rate of Daphnia Pulex after chronic exposure. Environmental Toxicology and Chemistry, Vol. 22, No. 12, 6 pgs.

Stehr, C. M., T. L. Linbo, D. H. Barldwin, N. L. Sholz, and J. P. Incardona. 2009. Evaluating the Effects of Forestry Herbicides on Fish Development Using Rapid Phenotypic Screens. North American Journal of Fisheries Management. 29:975-984.

Sykes, G. E., C. J. Johnson, and J. M. Shrimpton. 2009. Temperature and Flow Effects on Migration Timing of Chinook Salmon Smolts. Transactions of the American Fisheries Society 138:1252-1265

Tu, M., Hurd, C., and J. M. Randall. 2003. Weed Control Methods Handbook: Tools & Techniques for Natural Areas. The Nature Conservancy. June 2003.

U.S. Forest Service (USFS). 1995. Forest Service Handbook

USFS. 1996. Forest Service Noxious Weed Strategy. USDA National Forest Service, Washington, D.C. March 1996.

USFS. 1999. Frank Church River of No Return Noxious Weed Treatments Final Environmental Impact Statement. Intermountain and Northern Regions: Bitterroot, Nez Perce, Payette, and Salmon-Challis National Forests.

USFS. 2000. Flathead National Forest Noxious and Invasive Weed Control Environmental Assessment. 2000.

USFS. 2008. Salmon-Challis National Forest Hydrology Department Files. Travel Analysis 2008 Hydro Revision. Prepared by D. Deschaine, SCNF hyrologist.

USFS. 2019. Final Biological Assessment on the Weed Management Program for the Frank Church River of No Return Wilderness Area. Salmon-Challis, Payette, Nez Perce-Clearwater, and Bitterroot National Forests June 10, 2019. 132 pgs.

Wainwright, T. C. and L. A. Weitkamp. 2013. Effects of Climate Change on Oregon Coast Coho Salmon: Habitat and Life-Cycle Interactions. Northwest Science 87(3):219-242.

Ward, E. J., J. H. Anderson, T. J. Beechie, G. R. Pess, and M. J. Ford. 2015. Increasing hydrologic variability threatens depleted anadromous fish populations. Global Change Biology 21(7):2500-2509.

Weis, J. S. 2014. Delayed Behavioral Effects of Early Life Toxicant Exposures in Aquatic Biota. Toxics, volume 2, pp. 165-187.

Weis, J. S., G. Smith, T. Zhou, C. Santiago-Bass, and P. Weis. 2001. Effects of contaminants on behavior: biochemical mechanisms and ecological consequences. BioScience 51(3):209-217.

Whitney, J. E., R. Al-Chokhachy, D. B. Bunnell, C. A. Caldwell, et al. 2016. Physiological Basis of Climate Change Impacts on North American Inland Fishes. Fisheries 41(7):332-345. DOI: 10.1080/03632415.2016.1186656.

Vannote, R. L., G. W. Minshall, K. W. Cummins, J. R. Sedell, and C.E. Cushing. 1980. The river continuum concept. Canadian Journal of Fisheries and Aquatic Sciences 37:130-137.

Verdonck, D. 2006. Contemporary vertical crustal deformation in Cascadia. Tectonophysics 417(3):221-230. DOI: 10.1016/j.tecto.2006.01.006.

Yamada, S., W. T. Peterson, and P. M. Kosro. 2015. Biological and physical ocean indicators predict the success of an invasive crab, Carcinus maenas, in the northern California Current. Marine Ecology Progress Series 537:175-189. DOI: 10.3354/meps11431.

Zabel, R. W. and J. G. Williams. 2002. Selective mortality in Chinook salmon: what is the role of human disturbance? Ecological Applications 12(1): 173-183.

Zabel, R. W., M. D. Scheuerell, M. M. McClure, et al. 2006. The Interplay Between Climate Variability and Density Dependence in the Population Viability of Chinook Salmon. Conservation Biology 20(1):190-200, 2/1/2006.

APPENDIX A

Toxicological Effects of Herbicides Proposed for Use In the FC-RONRW Noxious Weed Treatment Program

Aminopyralid

Exposure. The half-life of aminopyralid in soils ranges from 32 to 533 days, with a typical time of 103.5 days (EPA 2005b). Microbes and sunlight break it down and, in aquatic systems; the primary route of degradation is through sunlight (photolysis), with laboratory experiments yielding a product half-life of 0.6 days. In another experiment, aminopyralid photolyzed moderately slowly on a soil surface, with a half-life of 72 days. A laboratory Freundlich absorption isotherm study with eight United States and European soils yielded absorption values at 1.05 to 24.3 milliliter per gram (mL/g), which shows that aminopyralid is weakly sorbed to soil (EPA Aminopyralid Fact Sheet August 10, 2005). This also represents moderate mobility in the environment with a moderate potential to leach through soils and into groundwater. Aminopyralid is "rainfast" within 2 hours, leaving less potential for runoff during a rain event. Aminopyralid does not bioaccumulate through the food chain and is absorbed through the leaves and the roots where it is transported to other parts of the plant. Fish and aquatic insect exposure to aminopyralid occurs primarily through direct contact with contaminated surface waters.

Syracuse Environmental Research Associates, Inc. (SERA) (2007) identified a peak estimated rate of contamination of ambient water associated with the normal application of aminopyralid as 0.6 milligrams acid equivalent/L (mg a.e./L) at an application rate of 1 pound acid equivalents/acre (lb a.e./acre). Typical application rates for aminopyralid in the proposed action range from 0.078 to 0.11 lb a.e./acre, and the maximum label application rate is 0.11 lb a.e./acre. At the maximum application rate of 0.11 lb. a.e./acre, the peak concentrations of aminopyralid in ambient water, using the modeled water contamination rate in SERA (2007), would be 0.066 mg a.e./L. Considering the best management practices (BMPs) that will be implemented, it is likely that water concentrations of aminopyralid will be far less than that estimated from modeling performed by SERA.

End Use Products. Aminopyralid is a pyridine carboxylic acid herbicide and the current market products containing it include Milestone, Milestone VM, and Forefront HL. Both of the Milestone formulations contain the triisopropanolamine (TIPA) salt of aminopyralid (21.1 percent a.e.). These formulations contain no inert ingredients other than water and TIPA. Forefront HL also contains the TIPA salt of aminopyralid, but only as a minor component (4.28 percent a.e.) as the formulation also contains 34.25 percent a.e. 2,4-D. Forefront consists of 50.5 percent other ingredients and was not assessed in SERA (2007). Aminopyralid is considered in the same class of herbicides as clopyralid and picloram, which are described below, but lacks the carcinogen hexachlorobenzene or other chlorinated benzenes.

Toxicity: Fish. Because aminopyralid is a relatively new pesticide, very little information is available regarding its toxicological effects to endangered species act (ESA)-listed fish or other aquatic species. The information on the toxicity of aminopyralid comes from studies that have been submitted to U.S. Environmental Protection Agency (EPA) as part of the registration package for the chemical. The toxicity studies performed to date have used the technical grade aminopyralid; no toxicity studies in fish are available for the TIPA formulation of aminopyralid. In the available studies, aminopyralid has been shown to be practically non-toxic to fish and aquatic invertebrates, and slightly toxic to algae and aquatic vascular plants (EPA 2005b). Aminopyralid is not expected to bioaccumulate in fish tissue (EPA 2005b).

The SERA (2007) summarized several acute exposure studies that reported no mortality to organisms exposed to aminopyralid in concentrations up to 100 mg/L. Aminopyralid has a low order of acute toxicity to aquatic animals, with acute No Observed Effects Concentration (NOEC) values falling within a narrow range of 50 mg a.e./L to 100 mg a.e./L, depending on the fish species. Only one of the studies documented sublethal effects in trout. In the study conducted by Marino et al. (2001a in SERA 2007), approximately seven percent of rainbow trout (*Oncorhynchus mykiss*) exposed to 100 mg a.e./L for 96 hours experienced a partial loss of equilibrium. However, this result was not statistically significant relative to the control group using the Fisher Exact test ($p = 0.2457$). As such, the Environmental Fate and Effects Division of EPA classified the 100 mg/L exposure as a NOEC. National Marine Fisheries Service (NMFS) has used this value as the lowest sulblethal effect threshold.

Only one chronic toxicity study is available for aminopyralid, and it involves the fathead minnow (*Pimephales promelas*) (Marino et al. 2003 in SERA 2007). The NOEC of 1.36 mg a.e./L was derived from this egg to fry study. In this study, the percent larval survival and growth (wet weight and length) were significantly ($p < 0.05$) reduced at 2.44 mg a.e./L relative to controls. Sublethal effects such as pale coloration, immobility, deformed or underdeveloped bodies, and scoliosis (curvature of the spine) were also observed at concentrations at or exceeding 2.44 mg a.e./L. The EPA (2005b) classified the lowest observed effects concentration (LOEC) as 2.44 mg a.e./L.

The sublethal effects of aminopyralid and its end-use products (EUPs) on ESA-listed fish are unknown. Due to the relatively low toxicity and low application rates for aminopyralid, the estimated risks to fish and aquatic invertebrates from U.S. Forest Services' (USFS) use patterns or accidental exposure are estimated to be low. However, due to this chemical's fairly new emergence on the market, the overall effects whether sublethal or lethal are uncertain. Future research may reveal additional effects associated with the use of this herbicide.

Toxicity: Other Aquatic Organisms. Aminopyralid has been shown to be practically non-toxic to aquatic invertebrates, and slightly toxic to algae and aquatic vascular plants (EPA 2005b). Similar to fish, acute toxicity values for amphibians and aquatic invertebrates fall within 50 mg a.e./L to 100 mg a.e./L. *Daphnia magna* did not exhibit mortality or sublethal effects when exposed to a measured 98.6 mg a.e./L concentration for a 48-hour exposure period (Marino et al. 2001b in SERA 2007). Aquatic invertebrates are much less sensitive to chronic exposures to aminopyralid than fish. In a daphnid study, no adverse effects on adults, offspring, or reproductive parameters were observed in concentrations up to 102 mg a.e./L. As such, EPA (2005b) classified 102 mg a.e./L as the NOEC. In a separate study using midges, the NOEC was 130 mg a.e./L based on mean measured water column test concentrations and 82 mg a.e./L based on pore water concentrations.

Algae and aquatic macrophytes are only somewhat more sensitive than fish and aquatic invertebrates with NOEC values for algae in the range of 6 mg a.e./L to 23 mg a.e./L and a single NOEC of 44 mg a.e./L for an aquatic macrophyte. No chronic toxicity tests were reported (SERA 2007).

2,4-D (amine salt only)

Exposure. The herbicide 2,4-D is highly soluble in water, but it rapidly degenerates in most soils, and is rapidly taken up in plants. 2,4-D ranges from being mobile to highly mobile in sand, silt, loam, clay loam, and sandy loam (EPA 2005a). Consequently, 2,4-D may readily contaminate surface waters when rains occur shortly after application, but is unlikely to be a ground-water contaminant due to the rapid degradation of 2,4-D in most soils and rapid uptake by plants. Most reported 2,4-D ground-water contamination has been associated with spills or other large sources of 2,4-D release. 2,4-D may remain active for 1 to 6 weeks in the soil and will degrade to half of its original concentration in several days. Soils high in organic matter will bind 2,4-D the most readily. 2,4-D is degraded in soil by microorganisms and degradation is more rapid under warm, moist conditions. Some forms of 2,4-D evaporate from the soil.

Transport of 2,4-D into rivers by storm runoff is likely to occur from rain events within or shortly following the spray season, based on documented studies. The Washington Department of Ecology collected 32 stream samples downstream from a helicopter application of 2,4-D conducted according to Washington State BMPs. 2,4-D was found in all samples collected and in highest concentrations following a rainstorm the day after the spraying (Rashin and Graber 1993). In a national study of surface water quality, 2,4-D was found in 19 of 20 basins sampled throughout the United States (USGS 1998). In the U.S. Geological Survey (1998) study, 2,4-D was found in 12 percent of agricultural stream samples, 13.5 percent of urban stream samples, and in 9.5 percent of the samples from rivers draining a variety of land uses.

The SERA (2006a) identified a peak estimated rate of contamination of ambient water associated with the normal application of 2,4-D as 0.44 mg a.e./L at an application rate of 1 lb a.e./acre. Typical application rates for 2,4-D in the proposed action range from 0.5 to 1.5 lb a.e./acre, and the maximum label application rate is 4 lb a.e./acre. At the maximum application rate of 4 lb. a.e./acre, the peak concentrations of 2,4-D in ambient water, using the modeled water contamination rate in SERA (2006a), would be 1.76 mg a.e./L. Considering the BMPs that will be implemented, it is likely that water concentrations of 2,4-D will be far less than that estimated from modeling performed by SERA.

End-Use Products. The herbicide 2,4-D is available in a variety of chemical forms (e.g., esters, amine salts, and acids) with different toxicities to fish. For this consultation, the Forests are proposing to use the amine salt forms of 2,4-D and has specifically requested the use of Amine 4 (various manufacturers), Weedar 64, Weedestroy, AM-40, Clean Amine, and DMA 4. The active ingredient in these products is the 2,4-D dimethylamine salt (DMA).

Both Weedar 64 and Weedestroy (and their substantially similar products as identified on the Pesticide Action Network pesticide database) consist of approximately 47 percent 2,4-D DMA. Unspecified, inert ingredients comprise the remaining 53 percent of the product. 2,4-D Amine 4 and its substantially similar products include 47.3 percent 2,4-D DMA and 52.3 percent of unspecified inert ingredients. The most recent SERA risk assessment (2006a) included these products in the effects analysis.

Toxicity: Fish. The Forests' are not proposing to use the ester formulation, which is more toxic to fish than the other forms. Instead, the Forests' propose to use the amine form of 2,4-D, which has the lowest toxicity among the various 2,4-D formulations. Toxicities for the acid and amine salts of 2,4-D indicated that both forms are practically non-toxic to freshwater or marine fish, with LC_{50}s ranging from more than 80.24 mg a.e./L to 2,244 mg a.e./L (EPA 2005a). Of the EPA-required studies, the most sensitive results were obtained for rainbow trout exposed to the TIPA salt (96-hour LC_{50} of 162 mg a.e./L). The comparable most tolerant results of the EPA-required studies were obtained with rainbow trout exposed to the DMA salt (96-hour LC_{50} of 830 mg a.e./L). These values are similar to the LC_{50} values of 362 mg a.e./L (Martinez-Tabache et al. 2004) and 358 mg a.e./L (Alexander et al. 1985) obtained when rainbow trout were exposed to 2,4-D acid.

Most of the potential sublethal effects from exposure to 2,4-D have not been investigated for endpoints important to the overall health and fitness of salmonids. Exposure to 2,4-D has been reported to cause changes in schooling behavior, red blood cells, reduced growth, impaired ability to capture prey, and physiological stress (NLM 2012; Gomez 1998). Tierney et al. (2006) found modifications in electro-olfactogram response when exposing juvenile coho salmon (*O. kisutch*) to 100 mg a.e./L of 2,4-D. Little et al. (1990) examined behavior of rainbow trout exposed for 96 hours to sublethal concentrations of 2,4-D acid and observed inhibited spontaneous swimming activity (at 5 mg/L), swimming stamina (at 50 mg/L), predator avoidance (50 mg/L), and prey capture (5 mg/L).

Early life-state tests evaluating the effects of various forms of 2,4-D on growth and larval survival of the fathead minnow were submitted to EPA as part of the registration process. For the acid and salts, the reported NOECs for survival and reproduction ranged from 14.2 mg a.e./L (DMA) to 63.4 mg a.e./L (2,4-D acid). The LOEC values associated with these results are 23.6 mg a.e./L (length) and 102 mg a.e./L (larval survival), respectively (SERA 2006a).

Toxicity: Other Aquatic Organisms. The EPA (2005a) classifies the acid and amine salts of 2,4-D as slightly toxic to practically non-toxic to aquatic invertebrates. *Daphnia* was the most sensitive species of freshwater species exposed to the 2,4-D acid, with a 48-hour LC_{50} of 25 mg a.e./L (Alexander et al. 1985). When *Daphnia* were exposed to the DMA of 2,4-D, the reported 48-hour LC_{50} values range from 153[3] mg a.e./L (Alexander et al. 1985) to 642.8 mg a.e./L (EPA 2005a). Some chronic studies (21-day) have been conducted to evaluate the effects of 2,4-D formulations on survival and reproduction. Ward (1991) reported a 21-day LC_{50} of 75.7 mg a.e./L for *Daphnia* from exposure to the DMA form of 2,4-D. A NOEC was not reported.

2,4-D is an effective herbicide that adversely affects aquatic plants. Based on the data available, it appears that the vascular plants are more than two orders of magnitude more sensitive than the non-vascular plants (EPA 2005a). The SERA (2006a) reported the 5-day effect concentration where 50% of the organisms exhibited toxic effects (EC_{50}s) (algal cell growth) for 2,4-D acids and salts as ranging from 3.88 mg a.e./L (a corresponding NOEC of 1.41 mg a.e./L) to 156 mg a.e./L (a corresponding NOEC of 56.32 mg a.e./L). The most sensitive species was *Navicula*

[3] The SERA risk assessment (2006) reports a LC_{50} of 184 mg a.e./L; however, the Alexander et al. (1985) paper specifically states that results are reported as the technical product and not as acid equivalents. NMFS used a conversion factor of 0.831 to convert from technical product to acid equivalents.

pelliculosa (a freshwater diatom), and the least sensitive species was a freshwater blue-green alga, *Anabaena flos-aquae*. Aquatic macrophytes appear to have a greater range of toxicity values, with target species having lower tolerances. Roshon et al. (1999) reported 14-day EC_{50} toxicity values for common water milfoil (*Myriophyllum sibiricum*), a target species, of 0.018 mg/L (shoot growth) and 0.013 mg/L (root length). Sprecher et al. (1998) report no effects on sago pondweed (*Potamogeton pectinatus*), a non-target species, at concentrations of up to 2 mg/L of WEEDAR 64.

NMFS Pesticide Registration Opinion. Chemical concentrations examined in the in the 2011 registration biological opinion (Opinion) (NMFS Tracking # 2004/02673) did not vary drastically from those summarized here. The 2,4-D registration Opinion reported acute toxicity data for rainbow trout ranging from 162 mg a.e./L (2,4-D triisopropanolamine salt) to 2,244 mg a.e./L (2,4-D isoproylamine). For the 2,4-D DMA, the acute toxicity information ranged from >100 mg a.e./L to 807 mg a.e./L. Information presented in the 2011 Opinion for EPAs registration of 2,4-D does not suggest a different endpoint as being more appropriate than that which was used in this Opinion.

The registration Opinion concluded there was no overlap between the estimated environmental concentrations (EECs) for forestry uses and the fish and invertebrate toxicity endpoints for amine, salt, and acid forms of 2,4-D. Generally, the toxicity endpoints were several orders of magnitude higher than the EECs. There was some overlap with the algal and aquatic vascular plant endpoints with the floodplain estimate EEC. The registration Opinion concluded that use of 2,4-D in terrestrial applications was not likely to result in mortality of fish; however, it may result in some sublethal effects.

CHLORSULFURON

Exposure. Chlorsulfuron has a soil half-life of 1 to 3 months, with a typical half-life of 40 days. Soil microbes break down chlorsulfuron and can break it down faster in warm, moist soils (WSDOT 2006). Alternatively, EPA (2005b) describes soil half-life ranging from 14 to 320 days. The WSDOT reported that chlorsulfuron has a high potential to contaminate groundwater, with contamination potentially resulting from application drift, surface runoff, and/or leaching through soil into groundwater (WSDOT 2006). EPA (2005b) also describes chlorsulfuron as likely to be persistent and highly mobile in the environment, transported to non-target areas by surface runoff and/or spray drift.

The SERA (2004a) identified a peak estimated rate of contamination of ambient water associated with the normal application of chlorsulfuron at 0.2 mg a.e./L at an application rate of 1 lb a.e./acre. The maximum application rate listed in the proposed action is 2.6 ounces/acre, which is equivalent to 0.12 lbs a.e./acre; consequently, maximum peak exposure would be approximately 0.024 mg a.e./L. For longer-term exposures, average estimated rate of contamination of ambient water associated with the normal application of chlorsulfuron is 0.0006 (0.0001 to 0.0009) mg a.e./L at an application rate of 1 lb a.e./acre.

End-Use Products. The product formulation of chlorsulfuron proposed for use is Telar XP. Telar DF was listed as the product formulation proposed for use in the 2007 Opinion; however, that product is no longer being manufactured. Telar XP is the same formulation as Telar DF; however, the granule shape has been changed to provide better mixability. The manufacturer requested a new registration number for Telar XP for internal tracking purposes. Telar XP contains 25 percent inert ingredients that have not been disclosed publicly. None of the inert ingredients are classified as toxic by the EPA (SERA 2004a).

Toxicity: Fish. EPA (2005b) describe chlorsulfuron as "practically non-toxic" to fish, and it does not bioaccumulate in fish (WSDOT 2006). The 96-hour LC_{50} value for rainbow trout has been reported as greater than 250 parts per million (Smith et al. 1979). Although full dose-response curves have not been generated (due to limited water solubility of chlorsulfuron), fish do not appear to be susceptible to chlorsulfuron toxicity. The LC_{50} values in most species exceed the limit of solubility for chlorsulfuron (SERA 2004a). Grande et al. (1994) exposed brown trout (*Salmo trutta*) to Glean (a product formulation consisting of 75 percent chlorsulfuron) and reported a 96-hour LC_{50} of 40 mg/L. Because the formulated product was tested, we cannot rule out the possibility that some of the toxicity may be due to the inert ingredients. There was not a paired study done on chlorsulfuron alone.

Pierson (1991) is the only study available regarding the toxicity of long-term (77 days) exposure of chlorsulfuron to fish or fry. Survival of rainbow trout embryos and alevins was not affected at concentrations up to 900 mg/L. However, fingerlings experienced 40 percent mortality at 900 mg/L. No mortality of fingerlings occurred in groups that were exposed to concentrations less than 900 mg/L (Pierson 1991). The NOEC for growth (as measured at the end of the study) was determined to be 32 mg/L and the LOEC was reported as 66 mg/L (Pierson 1991).

These studies indicate that outright mortality from exposure to the active ingredient is unlikely from the proposed action since peak estimated exposure from SERA (2004a) is about three orders of magnitude lower than the reported LC_{50} for brown trout and approximately four orders of magnitude lower than the reported LC_{50} for rainbow trout. Because there are limited studies available, there is substantial uncertainty surrounding potential sublethal effects of chlorsulfuron and Telar XP. There is no assurance that the proposed action will not cause lethal or sublethal effects to ESA-listed fish if the fish are exposed to the product in any appreciable amount.

Toxicity: Other Aquatic Organisms. The effects of chlorsulfuron on aquatic plants and invertebrates are limited to assays reported for *Daphnia* and several species of plants. Chlorsulfuron is described by EPA (2005b) as "practically non-toxic" to aquatic invertebrates, with 48-hour LC_{50} values for *Daphnia* greater than 100 mg/L. Chlorsulfuron does not bioaccumulate in aquatic invertebrates (WSDOT 2006).

The SERA (2004a) summarized standard toxicity bioassays in *Daphnia* (Goodman 1979; Ward and Boeri 1989) and mysids (Ward and Boeri 1991) to assess the effects of chlorsulfuron on aquatic invertebrates. Mysids and daphnia had similar LC_{50} values. The 96-hour LC_{50} and NOEC (for lethality) values for *Mysidopsis bahia* were reported as 89 mg/L and 35 mg/L, respectively (Ward and Boeri 1991). The reported 48-hour LC_{50} value in *Daphnia pulex* ranged between 32 and 100 mg/L, and the reported NOEC (for lethality) was 32 mg/L (Hessen et al. 1994). *D. magna* appear to be more resistant to chlorsulfuron toxicity based on a 48-hour LC_{50} value range of >100 to 370.9 mg/L. The reported NOEC for lethality was 10 mg/L (Goodman

1979). For reproductive effects, a NOEC of 20 mg/L was reported in a 21-day exposure study in *D. magna* (Ward and Boeri 1989).

Studies have demonstrated that aquatic plants are far more sensitive than aquatic animals to chlorsulfuron, with studies occurring for both algae and aquatic macrophytes. Study results summarized by SERA (2004a) revealed substantial differences in the response of algae and various cyanobacteria to chlorsulfuron. However, due to the many variations in experimental protocols, including the duration of exposure and the specific variables used to determine EC_{50} values, identifying the species most sensitive and most resistant to chlorsulfuron is difficult. *Selenastrum capricornutum* is fairly sensitive to chlorsulfuron toxicity, with reported EC_{50} values ranging from 0.05 mg/L to 0.8 mg/L (Abdel-Hamid 1996; Blasberg et al. 1991; Fairchild et al. 1997; Kallqvist and Romstad 1994). *Selenastrum* is an algal species that occurs in lakes and ponds, and it is used as a toxicity test species because it is sensitive to toxins. *Selanstrum* is generally not found in mountain streams and rivers, but it is a general indicator of potential algal responses in freshwater habitats. Results of a standard toxicity bioassay in *S. capricornutum* yield a NOEC of 0.01 mg/L (exposure duration of 120 hours) (Blasberg et al. 1991), which is consistent with the NOEC of <0.019 mg/L reported by Fairchild et al. (1997). Fairchild et al. 1997 also reported an LOEC in *S. capricornutum* of 0.019 mg/L. *Cryptomonas pyrenoidifera*, another freshwater algal species, has an EC_{50} of 213 mg/L (Nystrom et al. 1999). The longest chlorsulfuron exposure duration for laboratory studies in algae was 92 hours; with no laboratory studies with longer exposure durations identified.

Chlorsulfuron can cause changes in phytoplankton communities at concentrations as low as 0.010 mg/L (Kallqvist et al. 1994). A decrease in biomass development was observed following exposure to chlorsulfuron concentrations of 0.010 mg/L for 13 days. A dose-dependent decrease in species diversity (based on the Shannon-Weiner diversity index) was also observed, with the lowest values recorded on the second and last days of the exposure period. With these low concentrations where changes have been observed, the proposed use of chlorsulfuron is likely to alter the algal communities in locations where it reaches water. However, any community effect is likely to be transient, and localized, since exposure is likely to occur through discrete runoff events or spillage with limited duration, and any such incidents are likely to be widely scattered.

Only three studies were identified by SERA (2004a) regarding the toxicity of chlorsulfuron to aquatic macrophytes: a 96-hour exposure study and a 7-day exposure study in duckweed (Fairchild et al. 1997; Peterson et al. 1994); and a 4-week exposure study in sago pondweed (Coyner et al. 2001). The 96-hour EC_{50} value for growth inhibition based on biomass in duckweed is reported as 0.0007 mg /L, with an NOEC value of 0.0004 mg/L and an LOEC of 0.0007 mg/L (Fairchild et al. 1997). Exposure of duckweed to 0.02 mg/L for 7 days resulted in 86 percent inhibition of growth (Peterson et al. 1994). Results of the 4-week exposure in sago pondweed yield an LC_{50} value of 0.001 mg/L, with 100 percent plant death following a 96-hour exposure to 0.002 mg/L (Coyner et al. 2001). No field studies assessing the effects of chlorsulfuron in aquatic plants have been identified.

Very little information is available regarding the toxicity of chlorsulfuron degradation products to aquatic plants or algae. Based on a single study described by SERA (2004a), comparing chlorsulfuron and two chlorsulfuron degradation products in *Chlorella pyrenoidosa,* chlorsulfuron breakdown products appear to be considerably less toxic than chlorsulfuron; EC_{50} values for the degradation products are at least 100-fold greater than for chlorsulfuron (Wei et al. 1998).

CLOPYRALID

Exposure. Clopyralid's half-life in the environment averages 1 to 2 months and ranges up to 1-year. It is degraded almost entirely by microbial metabolism in soils and aquatic sediments, and is not degraded by sunlight or hydrolysis. Clopyralid is highly soluble in water, does not adsorb to soil particles, is not readily decomposed in some soils, and may leach into ground water. Clopyralid is extremely stable in anaerobic sediments, with no significant decay noted over a 1-year period (Hawes and Erhardt-Zabik 1995; Tu et al. 2001). Because clopyralid does not bind with sediments readily, it can be persistent in an aquatic environment, where clopyralid half-life ranges from 8 to 40 days (Tu et al. 2001). Clopyralid is stable in water over a pH range of 5 to 9 (Woodburn 1987), and the rate of hydrolysis in water is extremely slow with a half-life of 261 days (Concha and Shepler 1994).

Clopyralid does not bind tightly to soil and has a high potential for leaching. While clopyralid will leach under conditions that favor leaching (e.g., sandy soil, a sparse microbial population, and high rainfall), the potential for leaching or runoff is functionally reduced by the relatively rapid microbial degradation of clopyralid in soil (Baloch-Haq et al. 1993; Bergstrom et al. 1991; Bovey and Richardson 1991). A number of field lysimeter studies and the long-term field study by Rice et al. (1997) indicate that leaching and subsequent contamination of groundwater are likely to be minimal. This conclusion is also consistent with a short-term monitoring study of clopyralid in surface water after aerial application (Leitch and Fagg 1985).

The SERA (2004b) estimated peak rates of contamination of ambient water associated with the normal application of clopyralid to be 0.07 mg a.e./L at an application rate of 1 lb a.e./acre. For longer-term exposures, average estimated rate of contamination of ambient water associated with the normal application of clopyralid is 0.007 (0.001 to 0.013) mg a.e./L at an application rate of 1 lb a.e./acre.

End-Use Products. Clopyralid is available in two forms (acid and amine salt). The Forests' only propose to use Transline, which contains 40.9 percent clopyralid as the monoethanolamine salt. It also contains 59.1 percent inert ingredients. Two of the inert ingredients include: isopropyl alcohol (five percent) and a polyglycol (1 percent), neither of which are classified by EPA as toxic. Transline is currently produced by DowAgroSciences.

Toxicity: Fish. Little information is reported for toxic effects of clopyralid. The acid and amine forms of clopyralid have different toxicities to fish. The monoethanolamine salt of clopyralid appears to have lower toxicity compared to the acid formulation present in some other products. Toxicity of the acid formulation of clopyralid for a 96-hour LC_{50} is reported in SERA (2004b) to be 103.5 mg a.e./L, using an unspecified life stage of rainbow trout. For the monoethanolamine salt form used in the proposed action, SERA (2004b) reported a 96-hour LC_{50} of 700 mg a.e./L. Fairchild et al. (2008) exposed rainbow trout and bull trout to chlopyralid and reported 96-hour LC_{50} values of 700 mg a.e./L and 802 mg a.e./L, respectively. The authors also used accelerated life testing procedures in EPA's Acute-to-Chronic Estimation with Time-Concentration-Effect Models program to estimate chronic lethal concentrations resulting in 1 percent mortality (LC_1) at 30-days. The reported chronic LC_1 was 477 mg a.e./L, with a 95 percent confidence interval of 53 mg a.e./L to 900 mg a.e./L.

Only one longer-term toxicity study for clopyralid was available. Fairchild et al. (2009) conducted 30-day chronic toxicity tests with juvenile rainbow trout. No mortality was observed at the highest concentrations tested (273 mg a.e./L). They found no significant effects on growth of juvenile trout after 15 days of exposure to clopyralid at concentrations up to 256 mg a.e./L. However, both length and weight of trout were significantly affected after exposure to clopyralid for 30-days, with a calculated LOEC of 136 mg a.e./L. The 30-day NOEC value was reported as 68 mg a.e./L. No other longer-term toxicity studies are available on the toxicity of clopyralid.

Toxicity: Other Aquatic Organisms. Toxic effects on aquatic invertebrates are reported only for *Daphnia*, which has an LC_{50} of 350 mg a.e./L for the monoamine salt and 225 mg a.e./L for the acid LC_{50} (SERA 2004b). Results from a single, standard chronic reproduction bioassay exposing *Daphnia* to the monoethanolamine salt of clopyralid indicate a NOEC value of 23.1 mg a.e./L (SERA 2004b). If other invertebrates respond similarly to *Daphnia*, then lethal effects on aquatic invertebrates are unlikely.

Aquatic plants are more sensitive to clopyralid than fish or aquatic invertebrates (SERA 2004b). The EC_{50} for growth inhibition in duckweed, an aquatic macrophyte, is 89 mg/L. However, at lower concentrations, in the range of 0.01 to 0.1 mg/L, growth of other aquatic macrophytes is stimulated (Forsyth et al. 1997). From information reported in SERA (2004b) it appears that there could be potential losses in primary productivity from algae killed by clopyralid, based on an EC_{50} for algae of 6.9 mg/L. However, concentrations lethal to algae are unlikely to occur unless clopyralid is directly added to water, or if a rainfall washes the chemical into a stream shortly after it is applied.

DICAMBA

Exposure. Dicamba is highly mobile in and poorly adsorbed by most soil types. It is also highly soluble in water, so its transport is influenced by precipitation. At low rainfall rates, dicamba dissipation had a half-life of about 20 days. At high rainfall rates using modeled runs, virtually all the dicamba was washed from the soil. The environmental fate of dicamba has been extensively studied. In general, dicamba is very mobile in most soil types, with the only reported exception being peat, to which dicamba is strongly adsorbed (Grover and Smith 1974). For many soil types, the extent of soil adsorption is positively correlated with and can be predicted from the organic matter content and exchangeable acidity of the soil (Johnson and Sims 1993). In a monitoring study by Scifres and Allen (1973), dicamba levels in the top 6 inches of soil dissipated at a rate of about 0.22 day-1 (t1/2=3.3 days) over the first 2 weeks following application. After 14 days no dicamba was detected, with the limit of detection of 0.01 mg/kg, in the top 6 inches of soils. The rates of dissipation in clay and loam were essentially identical.

Available monitoring data indicate that ambient water may be contaminated with dicamba after standard applications of the product. The range of average to maximum dicamba levels in water, reported in a monitoring study by Waite et al. (1992), are from approximately 0.1 to 0.4 microgram per liter (µg/l). SERA (2004c) estimated peak rates of contamination of ambient water associated with the normal application of dicamba to range from less than 0.00001 mg a.e./L to 0.0005 mg a.e./L at an application rate of 1 lb a.e./acre. The estimated water contamination rate for an accidental direct spray of a stream was reported as 0.01 mg a.e./L. Because dicamba has been detected in surface water at concentrations higher than those modeled by groundwater loading effects of agricultural management systems (GLEAMS), SERA (2004c) opted to use the 0.01 mg a.e./L as the peak water contamination rate in their risk assessment.

End-Use Products. Dicamba is available as a diglycolamine (DGA) salt and DMA. The products proposed for use include Banvel and Vanquish. Banvel is formulated with the DMA of dicamba, with roughly 52percent inert ingredients. Vanquish is the DGA salt of dicamba, and contains approximately 43 percent inert ingredients.

Toxicity: Fish. There is wide variation in the reported acute toxicity of dicamba to fish, with 96-hour LC_{50} values ranging from 28 mg/L (rainbow trout) to 465 mg/L (mosquito fish [*Gambusia affinis*]). Although limited data are available, salmonids appear to be more sensitive to dicamba than other freshwater fish. Rainbow trout had the lowest reported 96-hour LC_{50} value. The reported 96-hour LC_{50} value for cutthroat trout (*O. clarki*) was more than 50 mg/L (Woodward 1982). For coho salmon, reported 48- and 144-hour LC_{50} values were 120 mg/L and more than 109 mg/L, respectively (Bond et al. 1965; Lorz et al. 1979). In a study by Lorz et al. (1979), yearling coho mortality was observed at 0.25 mg/L during a seawater challenge test which simulates their migration from rivers to the ocean.

There are limited studies on sublethal effects from acute or chronic exposures. The only study providing histopathologic evaluation is that of Lorz et al. (1979) using coho salmon. In this study, non-lethal concentrations of dicamba at a concentration of 100 mg/L were associated with histopathological changes in the liver but not in the kidneys or gills. Acute NOEC values have been reported for bluegill sunfish (*Lepomis macrochirus*) (56 mg/L in Vilkas 1977a; 100 mg/L in McAllister et al. 1985a), rainbow trout (56 mg/L in McAllister et al. 1985b), and sheepshead minnow (*Cyprinodon variegatus*) (>180 mg/L from Vilkas 1977b). However, these NOEC values are based on relatively gross endpoints – i.e., no mortality and no behavioral changes. A significant issue with these values is the fact that some reported NOEC values are greater than values reported to cause an adverse effect. For example, as noted above, McAllister et al. (1985b) report an NOEC of 56 mg/L in rainbow trout. While this is consistent with the LC_{50} value of 320 mg/L reported by Bond et al. (1965) in rainbow trout, Johnson and Finley (1980) report an LC_{50} of 28 mg/L in rainbow trout. These sorts of discrepancies are not uncommon with compounds for which many studies are conducted at different times by several different laboratories. The reported NOEC values for dicamba will not be used directly in this Opinion because they may not fully encompass sublethal toxicity and because some of the reported NOEC values exceed other reports of concentrations that are associated with lethality.

Toxicity: Other Aquatic Organisms. The range of toxicity values of dicamba to aquatic invertebrates suggests wide variation among species. The lowest reported 48-hour LC_{50} is

5.8 mg/L for *Gammarus lacustris* (Sanders 1969). While *Daphnia magna,* a common test species, appears to be relatively tolerant to dicamba with reported 48-hour LC_{50} values from 100 mg/L to >1000 mg/L (Johnson and Finley 1980; Forbis et al. 1985). *Daphnia pulex* is much more sensitive with a 48-hour LC_{50} value of 11 mg/L (Hurlbert 1975). As with fish, no longer-term studies are available on the lethal and sublethal toxicity of dicamba to aquatic invertebrates.

Algae species are more sensitive to dicamba than aquatic animals (SERA 2004c). The most sensitive species on which data are available is the freshwater algae, *Anabaene flos-aquae,* with a 5-day EC_{50} of 0.061 mg/L (Hoberg 1993a). The aquatic macrophyte, *Lemna gibba,* had reported 14-day NOEC and LOEC values of 0.25 mg/L and 0.51 mg/L, respectively (Hoberg 1993b). A higher 4-day NOEC of 100 mg/L was reported for *Lemna minor* (Fairchild et al. 1997). Whether this value reflects a true difference in species sensitivity or whether is simply reflects a shorter duration of exposure is unknown.

GLYPHOSATE

Exposure. Glyphosate strongly binds to most soils, but dissolves easily in water. Glyphosate remains unchanged in the soil for varying lengths of time, depending on soil texture and organic matter content. The half-life of glyphosate can range from 3 to 249 days in soil and from 35 to 63 days in water (USFS 2000a). Soil microorganisms break down glyphosate and the potential for leaching is low due to the soil adsorption. However, glyphosate can move into surface water when the soil particles to which it is bound are washed into streams or rivers (EPA 1993). Studies examined glyphosate residues in surface water after forest application in British Columbia with and without no-spray streamside zones. With a no-spray streamside zone, very low concentrations were sometimes found in water and sediment after the first heavy rain (USFS 2000a). Although glyphosate is chemically stable in pure aqueous solutions, it is degraded relatively fast by microbial activity, and water levels are further reduced by the binding of glyphosate to suspended soil particulates in water and dispersal (SERA 2011a).

Biodegradation represents the major dissipation process. After glyphosate was sprayed over two streams in the rainy coastal watershed of British Columbia, glyphosate levels in the streams rose dramatically after the first rain event, 27 hours after application, and fell to undetectable levels in 96 hours (NLM 2012). The highest residues were associated with sediments, indicating that they were the major sink for glyphosate. Residues persisted throughout the 171-day monitoring period. Suspended sediment is not a major mechanism for glyphosate transport in rivers, but glyphosate sprayed in roadside ditches could readily be transported as suspended sediment and cause acute exposures following rain events.

The SERA (2011a) estimated peak rates of contamination of ambient water associated with the normal application of glyphosate to be 0.083 mg a.e./L at an application rate of 1 lb a.e./acre. For longer-term exposures, average estimated rate of contamination of ambient water associated with the normal application of glyphosate is 0.00019 (0.000088 to 0.0058) mg a.e./L at an application rate of 1 lb a.e./acre. Peak contamination rates in a stream after a direct spray were modeled to be 0.091 mg a.e/L.

End-Use Products. Glyphosate is available in a variety of formulations that contain the ammonium, DMA, isopropylamine (IPA), or potassium salts of glyphosate. Some formulations contain only one of these salts as an aqueous solution (e.g., Accord, AquaNeat, and Rodeo), and other formulations (e.g., Roundup®) contain surfactants. The Forests' proposes to use products that are formulated as salts in water with no added surfactants. Products that appear to fit these criteria include: Rodeo, Accord Concentrate, GlyPro, AquaMaster, AquaNeat Aquatic Herbicide, and Foresters. All of these EUPs have the same proportion of the IPA salt of glyphosate. Manufacturers of these EUPs recommend that a surfactant be added to the formulation in a tank mix prior to application.

Toxicity: Fish. The EPA (1993) classified glyphosate (technical grade) as slightly toxic to practically non-toxic to fish. The rainbow trout 96-hour LC_{50} values for glyphosate acid and the IPA salt of glyphosate range from 10 mg a.e./L to 240 mg a.e/L. Wan et al. (1989) found the toxicity of glyphosate is affected by pH. The authors tested the toxicity of glyphosate to various salmonids (rainbow trout, coho salmon, chum salmon [*O. keta*], Chinook salmon [*O. tshawytscha*], and pink salmon [*O. gorbuscha*]) in water with pH values ranging from 6.3 to 8.2. Rainbow trout were the most sensitive to pH variance, with 96-hour LC_{50} values ranging from 10 mg a.e./L (pH 6.3) to 197 mg a.e./L (pH 8.2).

The various formulations of glyphosate have different toxicities to fish (rainbow trout 96-hour LC_{50} values ranging from 1.3 mg a.e./L to 429 mg a.e./L), which highlights the role of inert ingredients in toxicity (SERA 2011a). Of the glyphosate formulations tested, both Rodeo and Accord (and other equivalent formulations) are the least toxic. These formulations consist of only the active ingredient and water; however, the manufacturer recommends the EUP be mixed with a surfactant prior to applying the herbicides. Mitchell et al. (1987) tested the toxicity of Rodeo with and without a surfactant. Without the surfactant, the 96-hour LC_{50} for rainbow trout was 429 mg a.e./L. With the surfactant X-77, the 96-hour LC_{50} ranged from 96.4 mg a.e./L (rainbow trout) to 180.2 mg a.e./L (Chinook salmon).

The most toxic formulation tested was Roundup® Original and its apparently equivalent formulations (Honcho, Gly Star Plus, and Cornerstone). These Roundup® formulations contain glyphosate IPA and the polyoxyethyleneamine (POEA) surfactant MON 0818. The reported range of 96-hour LC_{50} values for Roundup® formulations that appear to contain this POEA surfactant is 0.96 mg a.e./L to 10 mg a.e./L (SERA 2011a). Other formulations with the trade name of Roundup® have been found to be much less toxic (i.e., rainbow trout LC_{50} of 800 mg a.e./L for Roundup® Biactive) than standard Roundup® formulations (SERA 2011a). The decreased toxicity of these formulations is likely due to the use of different surfactants.

Of the numerous surfactants that may be used in glyphosate EUPs, POEA is the most important class of surfactants. The POEA is commonly used to designate surfactants used in some glyphosate formulations; however, it is not a single surfactant. POEA surfactants are mixtures, and not all POEA surfactants used in glyphosate formulations are equivalent (even among formulations provided by the same manufacturer). Data indicate that some POEA surfactants may be equally toxic or more toxic than glyphosate itself.

The surfactant MON0818® in Roundup® formulations has been studied most extensively (of all the surfactants used with glyphosate products). It is considered highly toxic to fish (typical LC_{50} values ranging from about 1 to 3 mg/L). MON0818® is more toxic than glyphosate by factors of 3.1 (i.e., pink salmon at pH 6.3) to 135.6 (i.e., pink salmon at pH of 8.2) (SERA 2011a). Unlike technical grade glyphosate, the toxicity of MON0818® increases with increasing pH. Toxicity (LC_{50}) values reported for other surfactants added to glyphosate field solutions typically range from 1 to 10 mg/L (SERA 2011a). However, there are some surfactants that are considered slightly toxic (LC_{50} values ranging from >10 to 100 mg/L) to practically non-toxic (LC_{50} values greater than 100 mg/L). The surfactants Agri-Dex, LI 700 and Geronol CF/AR have LC_{50} values greater than 100 mg/L (McLaren/Hart 1995).

As noted previously, the surfactants can substantially alter the toxicity of a formulation (e.g., toxicity increased four times when X-77 was added to Rodeo). The USFS identified the following surfactants for use: Activator 90, Spreader 90, LI-700, Syl-Tac, R-11, and MSO. Three of these surfactants have reported rainbow trout 96-hour LC_{50} values: Activator 90 (2.0 mg/L); R-11 (3.8 mg/L); LI-700 (130 mg/L). The toxicity of X-77 is reported to be similar to that of R-11. As such, we will assume that the Rodeo/R-11 mixture has a similar toxicity to that of the Rodeo/X-77 (LC_{50} of 96.4 mg a.e./L) mixture for this Opinion.

Information on sublethal effects of glyphosate and glyphosate formulations is extremely limited and not available for many of the endpoints important to the overall health and fitness of salmonids. Xie et al. (2005) exposed juvenile rainbow trout to 0.11 mg a.e./L[4] glyphosate for 7 days and did not observe any significant increase in vitellogenin concentrations. The authors also exposed juvenile trout to mixtures of glyphosate (0.11 mg a.e./L) and either the surfactant R-11 (0.06 mg/L) or TPA (0.02 mg/L) for 7 days and observed some increases in vitellogenin concentrations. However, those increases were not statistically significant. No other studies evaluating sublethal effects to salmonids from acute exposures to technical grade glyphosate were found.

There have been some acute studies performed using Roundup® formulations. Morgan et al. (1991) reported that trout do not exhibit avoidance responses to glyphosate formulations (Vision with 15 percent surfactant and Vision with 10 percent surfactant) at concentrations less than the 96-hour LC_{50}. However, behavioral changes such as changes in coughing and ventilation rates, changes in swimming, loss of equilibrium, and changes in coloration were observed at concentrations as low as 50 percent of the LC_{50} over exposures of up to 96 hours (some erratic swimming behavior was observed after just 24 hours). In this study, rainbow trout exposed to concentrations of up to 6.75 mg a.e./L of Vision (with 15 percent surfactant) did not exhibit abnormal behavior during the exposure period. Similarly, no abnormal behavior was observed in fish exposed to concentrations of up to 18.75 mg a.e./L of Vision (with 10 percent surfactant). Tierney et al. (2007) reported that rainbow trout may be able to sense glyphosate (Roundup® formulation) at about 0.076 mg a.e./L (as measured by olfactory-mediated behavioral and neurophysiological responses) during 30 minute exposure periods, but will not exhibit an

[4] The SERA (2011a), reported the exposure concentration as 1.25 mg a.e./L; however, following review of the original publication (Xie et al. 2005), it appears as though 0.11 mg a.e./L glyphosate was measured, and the 1.25 mg a.e./L concentration was applicable to the chemical triclopyr.

avoidance response at this concentration. Rather, avoidance responses were exhibited at concentrations that were close to those causing acute lethality.

One full life-cycle study assessing the chronic toxicity of technical grade glyphosate has been performed using the fathead minnow. In this study, no adverse effects to survival or reproduction occurred at exposures up to 25.7 mg a.e./L (the highest concentrations tested). Morgan and Kiceniuk (1992) conducted a long-term study (2 months) exposing rainbow trout to Vision at concentrations up to 0.046 mg a.e./L. No mortality or signs of toxicity were observed during the exposure period, and the authors did not find any evidence or pathology or changes in growth. The authors noted a decrease in the frequency of wigwag behavior in exposed trout at 0.0045 mg a.e./L; however, this effect was not observed at higher exposure concentrations (0.043 mg a.e./L). Because the change in wigwag behavior did not have a clear dose-response relationship, the authors were uncertain about its biological significance. No other chronic studies using salmonids were located.

Toxicity: Other Aquatic Organisms. The EPA (1993) classified glyphosate (technical grade) as slightly toxic to practically non-toxic to aquatic invertebrates. The 48-hour EC_{50} values for aquatic invertebrates exposed to glyphosate or glyphosate IPA generally range from 50 to 650 mg a.e./L (SERA 2011a). For *Daphnia magna*, studies provided to EPA in support of the registration for glyphosate reported EC_{50} values ranging from 128 to 647 mg a.e./L. Pereira et al. (2009) reported an extremely high acute EC_{50} (more than 2,000 mg a.e./L). Even though this result is much higher than any previously reported EC_{50} values, the test protocol used appeared to be relatively standard (SERA 2011a).

As expected, Rodeo has similar toxicities to the active ingredient and is much less toxic than formulations that contain surfactants. For aquatic invertebrates, the LC_{50} values for Rodeo range from 86 mg a.e./L[5] to more than 2,000 mg a.e./L. Simenstad et al. (1996) found no significant differences in the short term (28 days post treatment) or long term (119 days post treatment) between benthic communities of algae and invertebrates on untreated mudflats and mudflats treated with Rodeo® and the surfactant X-77 spreader.

Similar to fish, Roundup® and similar formulations of glyphosate are much more toxic to aquatic invertebrates than glyphosate, glyphosate IPA, and Rodeo. Toxicity values for most Roundup® formulations range from approximately 1.5 to 62 mg a.e./L. In a study of avoidance behavior, Folmar (1978) noted that mayflies avoided Roundup® at concentrations of 10 mg/L; however, no effect was noted at concentrations of 1 mg/L. Hildebrand et al. (1980) found that Roundup® treatments of an experimental pond at concentrations up to 196 lbs/acre did not significantly affect the survival of *Daphnia*.

Glyphosate is highly toxic to all types of terrestrial plants and is used to kill floating and emergent aquatic vegetation. Differences in species sensitivities to glyphosate acid are apparent for both algae (EC_{50} values from about 2 to 600 mg a.e./L) and aquatic macrophytes (EC_{50} values

[5] Henry et al. (1994) reported an LC_{50} value of 218 mg formulation/L for *Daphnia magna*. It appears as though SERA (2011a) erroneously reported this value as milligrams acid equivalent per liter (mg a.e./L). The formulation used contained 53.5 percent IPA salt of glyphosate. The ratio of glyphosate acid to the IPA salt in the formulation is 0.74. Thus, a toxicity value of 218 mg formulation/L equates to 86.3 mg a.e./L (218 * 0.535 * 0.74).

from 10 to near 200 mg a.e./L). The toxicity of Rodeo (no surfactant) to the algae *Ankistrodesmus* sp. was reported to be 29 mg a.e./L (Gardner et al. 1997). Perkins (1997) found Rodeo to be much more toxic to the aquatic macrophyte water milfoil (14-day EC_{50} of 0.84 mg a.e./L) and *Lemna gibba* (7.6 mg a.e./L).

IMAZAMOX

Exposure. Imazamox has an average soil half-life of 81 days (SERA 2010). This estimated soil half-life is quite conservative based on SERA information indicating reports have shown that soil half-lives for imazamox range from 12 days to 30 days. All exposure assessments associated with terrestrial applications are based on the maximum application rate of 0.5 lb a.e./acre. All exposure assessments for aquatic applications are based on the maximum target concentration of 0.5 mg a.e./L. Simulations of imazamox runoff were conducted for clay, loam, and sand at annual rainfall rates, with the average penetration depth of imazamox estimated at about 40 inches with a range of about 12 to 60 inches (SERA 2010).

End-Use Products. Imazamox is available in acid and ammonium salt forms. The USFS proposes to use Clearcast, which is formulated with 12.1 percent of the ammonium salt of imazamox. No other EUPs are proposed for use.

Toxicity: Fish. (This excerpt is taken directly form SERA 2010). Data on the toxicity of imazamox appears to be essentially nontoxic to fish in acute exposure assays. LC_{50} values for fish were not determined in the standard acute toxicity studies on imazamox that are required for pesticide registration, and the reported acute no oberved adverse effect concentration (NOAECs) range from 94.2 mg/L in sheepshead minnows to 122 mg/L in rainbow trout. All of the acute NOAECs are the highest concentrations tested in the acute toxicity studies. Thus, the differences in NOAECs do not imply differences in species sensitivities. Just as mammalian toxicity studies on imazamox fail to determine acutely toxic doses, aquatic toxicity studies fail to determine imazamox concentrations that are acutely toxic to fish—i.e., acute lowest observed adverse effect concentration (LOAECs).

Chronic toxicity studies in fish are not listed among the registrant-submitted studies on imazamox, and the EPA ecological risk assessments (EPA/OPP 2008a, 2008b) note that chronic risks to fish cannot be assessed because chronic toxicity data on fish are not available. This is an unusual situation, particularly for a pesticide that is labeled for aquatic applications. For some herbicides, the EPA/OPP will waive chronic toxicity studies in fish because of the low acute toxicity of the herbicide to fish. While the acute toxicity of imazamox appears to be very low, there is no explicit indication in the EPA ecological risk assessments (EPA/OPP 2008a, 2008b) that chronic studies on fish were waived. The review of imazamox by the European Commission (2002) provides a very brief summary of 37 two longer-term studies in rainbow trout—i.e., a 28-day NOEC of 122 mg/L and a 96-day NOEC 38 of 11.8 mg/L. No further information, including reference citations, for these studies is provided. As noted in Section 4.1.2.4 Terrestrial Invertebrates of the SERA document, registrants often submit studies to European regulatory groups that are not submitted to the EPA/OPP. Longer-term studies in fish, however, are typically required by the EPA/OPP, and it is not clear why these studies appear to have been submitted to the European Commission but not to the U.S. 43 EPA/OPP.

Toxicity: Other Aquatic Organisms. (This excerpt is taken directly form SERA 2010). For imazamox, the toxicity data on aquatic invertebrates are similar to the data on fish. As summarized in Appendix 6 of the SERA 2010 document, there are two standard registrant-submitted acute toxicity studies on aquatic invertebrates, and as with fish, LC_{50} values for imazamox were not determined. At the highest concentrations tested, imazamox caused no mortality and no signs of toxicity in either *Daphnia magna* with an NOAEC of 115 mg a.e./L or mysid shrimp with an NOAEC of 89.3 mg a.e./L.

Similar again to the fish data on imazamox, chronic toxicity studies in aquatic invertebrates are listed among the registrant-submitted studies on imazamox, and the EPA ecological risk assessments note that chronic risks to aquatic invertebrates cannot be assessed because chronic toxicity data are not available. There is no indication in the EPA risk assessments that the requirements for chronic toxicity testing in aquatic invertebrates were waived.

Finally, similar to the situation with the data on fish, the review by the European Commission notes a 21-day NOAEC for *Daphnia magna* of 137 mg/L. No other study details are provided, including whether the units are in active ingredient or acid equivalents, and the source for the information is not cited. It is likely that the 21-day NOAEC is from a standard reproduction study in *Daphnia* (SERA 2010).

While Clearcast is labeled for the control of aquatic macrophytes, it is not specifically labeled for the control of algae. As with imazapyr and imazapic, imazamox appears to be less toxic to algae than to aquatic macrophytes. This generalization, however, is somewhat tenuous in that extensive data are not available on the toxicity of imazamox to algae. The limited data on the toxicity of imazamox to algae is noted in the EPA ecological risk assessments on imazamox which indicate that the maximum application rate for imazamox is 112.5 parts per billion (ppb) for aquatic weed control (EPA/OPP 2008a). As detailed in Section 2.4.2 in SERA 2010, the current maximum target concentration for imazamox in aquatic weed control is 500 ppb. In any event, current data on the effect of imazamox on algae do not encompass the concentrations of 112.5 ppb or 500 ppb (SERA 2010).

IMAZAPIC

Exposure. Imazapic has an average soil half-life of 120 days, with degradation primarily occurring through soil microbial metabolism (Tu et al. 2001). Imazapic is moderately persistent in soils, and has not been found to move laterally with surface water (generally moving only 6 to 12 inches laterally but can leach to depths of 18 inches in sandy soils). Although the extent to which imazapic is degraded by sunlight is believed to be minimal when applied to terrestrial plants, it is rapidly degraded by sunlight in aqueous solutions (half-life of 1 to 2 days). Imazapic is water soluble and is not degraded hydrolytically in aqueous solution (Tu et al. 2001).

Simulations of imazapic runoff were conducted for both clay, loam, and sand at annual rainfall rates from 5 to 250 inches and the typical application rate of 0.1 lb a.e./acre (SERA 2004d). Based on the modeling, under arid conditions (i.e., annual rainfall of about 10 inches or less), no runoff is expected and degradation, not dispersion, accounts for the decrease of imazapic concentrations in soil. At higher rainfall rates, plausible offsite movement of imazapic may reach up to 3.5 percent of the applied amounts in clay soils. In very arid environments substantial contamination of water is unlikely. In areas with increasing levels of rainfall, exposures to aquatic organisms are more likely to occur. Thus, the anticipated water contamination rates (WCRs) (concentration of imazapic in ambient water per lb a.e./acre applied) associated with runoff encompass a very broad range, from 0 to 0.002 mg/L, depending on rainfall rates and soil type (SERA 2004d).

In their risk assessment, SERA (2004d) utilized a peak estimated rate of contamination of ambient water associated with the normal application of imazapic of 0.01 mg a.e./L at an application rate of 1 lb a.e./acre. Typical application rates for imazapic in the proposed action range from 0.1 to 0.19 lb a.e./acre, and the maximum label application rate is 0.19 lb a.e./acre. At the maximum application rate of 0.19 lb. a.e./acre, the peak concentrations of imazapic in ambient water, using the modeled water contamination rate in SERA (2004d), would be 0.002 mg a.e./L. Considering the BMPs that will be implemented, it is likely that water concentrations of imazapic will be far less than that estimated here.

End-Use Products. Imazapic is available in acid and ammonium salt forms. The USFS proposes to use Plateau, which is formulated with 23.6 percent of the ammonium salt of imazapic. No other EUPs are proposed for use.

Toxicity: Fish. The ammonium salt form of imazapic is less toxic than the acid form. Fish appear to be relatively insensitive to imazapic exposures, with LC_{50} values >100 mg/L for both acute toxicity and reproductive effects (SERA 2004d). In acute toxicity studies, all tested species (channel catfish, bluegill, sunfish, trout, and sheepshead minnow) evidenced 96-hour LC_{50} values of >100 mg/L. The low toxicity of imazapic to fish is probably related to a very low rate of uptake of this compound by fish. In a 28-day flow-through assay, the bioconcentration of imazapic was measured at 0.11 L/kg (Barker et al. 1998) indicating that the concentration of imazapic in the water was greater than the concentration of the compound in fish. No studies are reported in the SERA assessment (2004d) for sublethal effects of imazapic to listed fish. Barker et al. (1998) observed no effects on reproductive parameters in a 32-day egg and fry study using fathead minnow.

Even though imazapic itself appears to be only moderately toxic to fish, based on the LC_{50}, Plateau contains roughly 76 percent inert ingredients that are not identified by the manufacturer. With many herbicides, the inert ingredients may be more toxic to fish and other aquatic organisms than the active ingredient. While toxicity tests are reported for imazapic, there is no apparent information regarding the toxicity to salmon and trout for the product formulation in Plateau, which includes imazapic and unspecified inert ingredients. Although none of the inert ingredients contained in Plateau are classified as toxic by the EPA (SERA 2004d), no studies are available lending insight into how the inerts may affect the toxicity of Plateau. Consequently the toxic effects of salmon or trout exposure to Plateau are unknown.

Toxicity: Other Aquatic Organisms. Similar to fish, there is relatively little information about the effects of imazapic on aquatic organisms in the natural environment. No adverse effects to *Daphnia* or mysid shrimp were observed at nominal concentrations of imazapic of up to

100 mg/L in 96-hour studies (SERA 2004d); however, the report did not specify if the analysis included any sublethal endpoints. Additionally, no adverse effects were noted in a life-cycle study that exposed *Daphnia* to concentrations up to 100 mg/L.

Effects of imazapic on aquatic plants is highly variable. *Lemna gibba*, a freshwater macrophyte, is the most sensitive aquatic plant reported in the literature, with an EC_{50} value based on decreased frond counts of 0.0061 mg/L. Algae were less sensitive than macrophytes (reported LC_{50} values >0.045 mg/L), and responses included both growth inhibition and growth stimulation (SERA 2004d).

IMAZAPYR

Exposure. The persistence of imazapyr in soil has a reported soil half-lives ranging from a 313 to 2,972 days, with an overall average of 2,150 days. Imazapyr is rapidly degraded by sunlight in aquatic solutions. In soils, however, there is little or no photodegradation of imazapyr, is slowly degraded by microbial metabolism, and can be relatively persistent in soils. As of September 2003, imazapyr (tradename Habitat®) is conditionally registered for use in aquatic areas, including brackish and coastal waters, to control emerged, floating, and riparian/wetland species.

End-Use Products. Imazapic is available an IPA salt. The USFS proposes to use Habitat, which is formulated with 28.7 percent of the IPA salt of imazapyr. No other EUPs are proposed for use.

Toxicity: Fish. Based on LC_{50} values of >100 mg a.e./L, imazapyr acid is classified as practically non-toxic to fish (SERA 2011d). Based on acute bioassays in both bluegills and trout, the IPA salt of imazapyr is also practically non-toxic to fish.

Toxicity data are also available on one formulation of imazapyr, Arsenal herbicide. This formulation consists of the IPA salt of imazapyr (27.8 percent 40 a.i, 22.6 percent a.e.) and 72.2 percent inerts which include an unspecified solvent. The 96-hour LC_{50} of 41 Arsenal Herbicide is about 41 mg a.e./L in bluegills and 21 mg 42 a.e./L in trout (SERA 2011d). The SERA (SERA 2011d) document describes a study in trout, which notes the higher toxicity of the formulation relative to imazapyr and IPA salt of imazapyr. The substantially lower LC_{50} values of Arsenal Herbicide, expressed in acid equivalents, suggests that the inerts in the formulation contribute to its greater toxicity, as discussed further in the dose-response assessment (SERA 2011d). At this time, the USFS does not propose to use the Arsenal herbicide.

Effective aquatic applications of imazapyr will cause oxygen depletion in the water column secondary to rotting vegetation. The event will occur after the application of any effective aquatic herbicide and may kill fish as well as other aquatic organism. While hypoxia in fish due to oxygen depletion in water is identified as an endpoint of concern for fish and other aquatic organisms, potential hazards to fish associated with hypoxia should be minimal, if label directions are followed and only partial sections of standing bodies of water are treated at one time (SERA 2011d).

Toxicity: Other Aquatic Organisms. The acute toxicity data on aquatic invertebrates are similar to the data on fish. Both imazapyr acid and IPA salt of imazapyr are classified as practically non-toxic to *Daphnia magna* as well as saltwater invertebrates—i.e., oysters and pink shrimp (SERA 2011d).

METSULFURON-METHYL

Exposure. The persistence of metsulfuron-methyl in soil is highly variable; reported soil half-lives range from a 14 to 180 days, with an overall average of 30 days. The rate of metsulfuron-methyl degradation depends on factors like temperature, rainfall, pH, organic matter, and soil depth. Metsulfuron-methyl in the soil is broken down to non-toxic and non-herbicidal products by soil microorganisms and chemical hydrolysis. Degradation will occur more rapidly under acidic conditions, and in soils with higher moisture content and higher temperature (Extoxnet 1996).

The mobility of metsulfuron methyl ranges from moderate to highly mobile (NLM 2012). Off-site movement of metsulfuron-methyl is governed by the binding of metsulfuron-methyl to soil, the persistence in soil, as well as site-specific topographical, climatic, and hydrological conditions. The adsorption of metsulfuron-methyl to soil varies with the amount of organic matter present in the soil, soil texture, and pH. Adsorption to clay is low. In general, metsulfuron-methyl absorption to a variety of different soil types will increase as the pH decreases. Metsulfuron-methyl dissolves easily in water. There is a potential for metsulfuron-methyl to contaminate ground waters at very low concentrations. Metsulfuron-methyl readily leaches through silt loam and sand soils.

Fate and transport simulations reported in SERA (2004e) were conducted for clay, loam, and sand at annual rainfall rates ranging from 5 to 250 inches and the typical application rate of 0.03 lb a.e./acre. In all soil types under arid conditions (i.e., annual rainfall of about 10 inches or less), substantial contamination of surface water is unlikely. In areas with increasing levels of rainfall, peak WCRs of about 0.0001 to 0.002 mg a.e./L (per application of 1 lb a.e./acre) can be anticipated, under worst-case conditions, at rainfall rates ranging from 15 to 250 inches per year. SERA (2004e) also estimated the water contamination rate associated with an accidental direct spray to be 0.010 mg/L at an application rate of 1 lb a.e./acre. For this Opinion, the higher water contamination rate was multiplied by the maximum label application rate to estimate the EEC (i.e., 0.010 mg a.e./L x 0.15 lb a.e./acre = 0.002 mg a.e./L).

End-Use Products. There are several formulations of metsulfuron-methyl registered for use; however, the USFS proposes to only use the formulation Escort XP. Escort XP is the same formulation as Escort, which was analyzed in the 2007 Opinion; however, the granule shape has been changed to provide better mixability. Escort XP is manufactured by DuPont and is comprised of 60 percent metsulfuron methyl and 40 percent inert ingredients (SERA 2004e). The inert ingredients include sodium naphthalene sulfonate-formaldehyde condensate, a mixture of a sulfate of alkyl carboxylate and sulfonated alkyl naphthalene (sodium salt), polyvinyl pyrrolidone, trisodium phosphate, and sucrose.

Both trisodium phosphate and sucrose are generally recognized as safe compounds and are approved as food additives. Although none of the remaining inerts are categorized by EPA as being of toxicological concern (List 1) or as being potentially toxic or as having a high priority for testing (List 2), there is insufficient information available to assess their potential toxicity to fish. Polyvinyl pyrrolidone is marketed as a disinfectant for fish aquaria and treatment of certain fish infections; consequently, the product is not likely to be toxic to listed trout at environmental concentrations encountered in the proposed action.

The label for Escort XP recommends the use of a non-ionic surfactant, except in certain circumstances. There is limited information on the toxicity of surfactants (refer to the discussion included in Section 2.4.1.3 of the Opinion and the “glyphosate” section of this appendix).

Toxicity: Fish. Based on available studies, metsulfuron-methyl appears to have a low toxicity to and does not bioaccumulate in fish. The reported rainbow trout 96-hour LC_{50} values for metsulfuron-methyl range from more than 150 mg a.e./L to more than 1,000 mg a.e/L (SERA 2004e). The lowest concentration at which rainbow trout mortality was observed is 100 mg/L; however, in the same study, no mortality was observed in rainbow trout exposed to 1,000 mg/L (Hall 1984). Because of the lack of a dose-response relationship, Hall (1984) asserts that the mortality in the 100 mg/L exposure group was probably incidental rather than treatment related. This Opinion uses a LC_{50} of 150 mg a.e./L to evaluate the potential for use of metsulfuron-methyl to adversely affect ESA-listed fish.

Debilitating sublethal effects (i.e., erratic swimming, rapid breathing, and lying on the bottom of the test container) were observed by Muska and Hall (1982) after exposure to 150 mg/L for 24 hours. In tests with rainbow trout, no significant long-term effects (90-day exposure) were observed by Kreamer (1996) on hatch rate, last day of hatching, first day of swim-up, larval survival, and larval growth at concentrations up to 4.7 mg/L. However, concentrations greater than 8 mg/L resulted in small but significant decreases in hatching and survival of fry.

Indirect Effects on Aquatic Organisms. Toxicity studies on aquatic invertebrates are reported only for *Daphnia*. For acute exposures, the range of EC_{50} values for immobility ranges from more than 150 mg/L to 720 mg/L. For chronic exposures, the NOEC of 17 mg/L for growth inhibition is used, although higher chronic NOECs, ranging from 100 to150 mg/L, have been reported for survival, reproduction and immobility (SERA 2004e). The only effect reported by Hutton (1989) in a 21-day *Daphnia* study was a decrease in growth at concentrations as low as 5.1 mg/L, but decreased growth at concentrations less than 30 mg/L was not statistically significant. In aquatic invertebrates, decreased growth appears to be the most sensitive endpoint. Wei et al. (1999) report that neither metsulfuron-methyl nor its degradation products are acutely toxic to *Daphnia* at concentrations that approach the solubility of the compounds in water at pH 7.

The available data suggest that metsulfuron-methyl, like other herbicides, is much more toxic to aquatic plants than to aquatic animals. Macrophytes appear more sensitive to metsulfuron-methyl than algae (SERA 2004e). There are substantial differences in sensitivity to effects of metsulfuron-methyl among algal species, but all EC_{50} values reported in SERA (2004e) are above 0.01 mg/L, and some values are substantially higher. Toxicity in algae increases with lower pH, most probably because of decreased ionization leading to more rapid uptake. At a concentration of 0.003 mg/L, metsulfuron-methyl was associated with a six percent to 16 percent inhibition (not statistically significant) in algal growth rates for three species but stimulation of growth was observed in *Selenastrum capricornutum* and the aquatic macrophyte, duckweed (SERA 2004e). Wei et al. (1998; 1999) assayed the toxicity of metsulfuron-methyl degradation products in *Chlorella pyrenoidosa* and found that the acute toxicity of the degradation products was about two to three times less than that of metsulfuron-methyl itself in a 96-hour assay. One field study cited in SERA (2004c) on the effects of metsulfuron-methyl in algal species found that concentrations of metsulfuron-methyl as high as 1 mg/L are associated with only slight and transient effects on plankton communities in a forest lake.

PICLORAM

Exposure. Picloram is relatively persistent and can remain effective in the soil for up to 3 years after application. Picloram is resistant to biotic and abiotic degradation processes and has a field half-life of 20 to 300 days. Picloram is highly soluble in water and can readily leach through some soil types. Ismail and Kalithasan (1997) found that picloram moves rapidly out of the top 2 inches of soil with a half-life of about 4 to 10 days. Somewhat longer half-lives of 13 to 23 days have been reported by Krzyszowska et al. (1994), who also noted that picloram is degraded more rapidly under anaerobic than aerobic conditions and also degrades more rapidly at lower application rates.

The SERA (2011b) identified a peak estimated rate of contamination of ambient water associated with the normal application of picloram as 0.011 (0.001 to 0.18) mg a.e./L at an application rate of 1 lb a.e./acre. Typical application rates for picloram in the proposed action range from 0.5 to 0.75 lb a.e./acre, and the maximum application rate is 1 lb a.e./acre. The estimated peak water contamination rate of picloram in ambient water normalized to an application rate of 1 lb a.e./acre is 0.18 mg a.e./L (SERA 2011b).

Multiplying the maximum application rate by the peak water contamination rate results in an EEC of 0.18 mg a.e./L. Considering the BMPs that will be implemented (e.g., no application of picloram within riparian vegetation or 50 feet from water's edge, whichever is greater) it is likely that water concentrations of picloram will be far less than that modeled by SERA. The most likely scenario where picloram will enter the stream is where weeds are treated on floodplains with a high water table and highly permeable soils.

End-Use Products. The proposed action includes the use of Tordon 22K, which contains the potassium salt of picloram (24.4 percent weight per volume). The remaining 75.6 percent of the formulations consist of inert ingredients. One inert is listed as a polymer of ethylene oxide, propylene oxide, and di-sec-butyl-phenol (CAS No. 69029-39-6).

Toxicity: Fish. The EPA (1995) classified picloram acid and picloram potassium salt as moderately toxic to freshwater fish with reported rainbow trout LC_{50}s of 5.5 mg a.e./L and 13 mg a.e./L, respectively. SERA (2011b) reported a variety of 96-hour LC_{50} values for rainbow trout, which ranged from 5.5 mg a.e./L to 41 mg a.e./L. These tests used either technical grade picloram, picloram acid, or the picloram potassium salt. The 96-hour LC_{50} of 5.5 mg a.e./L was obtained by Batchelder (1974) in a test of the technical grade picloram. Earlier production of picloram contained impurities, which have been minimized in more recent production of picloram. As such, the 5.5 mg a.e./L might not be representative of current toxicity. Fairchild et al. (2009) reported an 96-hour LC_{50} of 36 mg a.e./L for juvenile rainbow trout. The authors did not observe any mortality at a concentration of 12 mg a.e./L.

Fish size or life stage can sometimes be an important factor in the toxicity of pesticides. Mayer and Ellersieck (1986) studied the toxicity of picloram on yolk sac rainbow trout fry, swim up fry, and advanced fry. They found LC_{50}s of 8 mg a.e./L, 8 mg a.e./L, and 11 mg a.e./L (yolk sac fry, swim up fry, and advanced fry, respectively), which demonstrates little difference in sensitivity among the various stages tested.

Most of the potential sublethal effects for picloram have not been investigated in regard to toxicological endpoints that are important to the overall health and fitness of salmonids (e.g., growth, life history, mortality, reproduction, adaptability to environment, migration, disease, predation, or population viability). Of the very little research that has been conducted on the potential sublethal effects of picloram on aquatic life, the focus has primarily been on growth. Woodward (1979) found that picloram concentrations greater than 0.61 mg/L decreased growth of cutthroat trout, and a similar finding was reported by Mayes (1984). Exposure regimes where the maximum exposure concentration did not exceed 0.29 mg a.e./L had no adverse effects on the survival and growth of cutthroat trout fry (Woodward 1979). In a study of lake trout, picloram concentrations of 0.04 mg a.e./L reduced the rate of yolk sac absorption, as well as fry survival, weight, and length (Woodward 1976). Mayes et al. (1987) reported that picloram concentrations of 0.9 mg a.e./L reduced the length and weight of rainbow trout larvae and concentrations of 2 mg a.e./L reduced survival of the larval fish. The authors reported the lowest NOEC as 0.55 mg a.e./L. Fairchild et al. (2009) reported a LOEC for growth of juvenile rainbow trout of 2.37 mg a.e./L, and a NOEC of 1.18 mg a.e./L. For juvenile bull trout, Fairchild et al. (2009) reported a LOEC for growth of 1.18 mg a.e./L and a NOEC of 0.6 mg a.e./L. Yearling coho salmon exposed to nominal concentrations of 5 mg a.e./L for 6 days suffered "extensive degenerative changes" in the liver and wrinkling of cells in the gills (Lorz et al. 1979).

Toxicity: Other Aquatic Organisms. Although picloram is toxic to salmonids, it is not as toxic to *Daphnia* or algae at the same concentrations. For *Daphnia*, the reported acute (48-hour) LC_{50} values range from 48 mg a.e./L to 173 mg a.e./L (SERA 2011b). Chronic studies using reproductive or developmental parameters in *Daphnia* reported a NOEC of 11.8 mg a.e./L and a LOEC of 18.1 mg a.e./L (Gersich et al. 1984). Boeri et al. (2002) studied the effects of picloram acid on *Daphnia* reproductive endpoints and reported a NOEC of 6.79 mg a.e./L and a LOEC of 13.5 mg a.e./L. No toxicity studies involving the exposure of *Daphnia* to Tordon 22K are readily available.

The toxicity of picloram to aquatic plants varies substantially among different species. Based on the available toxicity bioassays, the most sensitive species is *Navicula pelliculosa*, a freshwater diatom, with an EC_{50} (i.e., the concentration causing 50 percent inhibition of a process for growth) of 0.93 mg a.e./L and a NOEC of 0.23 mg a.e./L. The least sensitive aquatic plants appear to be from the genus *Chlorella* (another group of freshwater algae), with EC_{50} values greater than 160 mg a.e./L (Baarschers et al. 1988). The macrophyte *Lemna gibba* (duckweed) has a reported 14-day EC_{50} of 47.8 mg a.e./L and a 14-day NOEC of 12.2 mg a.e./L (Kirk et al. 1994). Other studies on the toxicity of picloram to macrophytes were not used in the 2011 risk assessment (SERA) because the test agent was not specified, the reporting units were not clear, or the test agent was a formulation of picloram not used by the USFS.

Effects on Non-Target Plants. While most grasses are resistant to picloram, it is highly toxic to many broad-leafed plants. Crop damage from irrigation water contaminated by picloram has been documented by the EPA (EPA 1995; USFS 2000b). Picloram is persistent in the environment, and may exist at levels toxic to plants for more than a year after application at normal rates. In normal applications, non-target plants may be exposed to chemical concentrations many times the levels that have been associated with toxic effects. Picloram's mobility allows it to pass from the soil to nearby, non-target plants. It can also move from target plants, through roots, down into the soil, and into nearby non-target plants. Given this capability, an applicator does not have to spray the buffer zone in order to affect the riparian vegetation. Spray drift may also kill plants some distance away from the area being treated. The proposed no application of picloram within riparian vegetation or 50 feet from water's edge, whichever is greater, should reduce the unintended mortality of streamside trees, shrubs and other broadleaf plants.

SULFOMETURON-METHYL

Exposure. Sulfometuron-methyl can be moderately persistent in soils, with reported half-lives ranging from 10 to 170 days (SERA 2004f). Sulfometuron-methyl readily biodegrades in aerobic soil conditions, with reported half-lives of 12 to 25 days for various soil conditions (e.g., pH levels and moisture content). Sulfometuron-methyl does not bind strongly to soils and it is slightly soluble in water. Depending on soil conditions, sulfometuron-methyl can be mobile and may be transported to off-site soil by runoff or percolation. The potential for leaching depends on soil conditions such as organic matter content, moisture, and soil pH. Under acid conditions, sulfometuron-methyl hydrolyzes quickly and has less potential for movement.

At least one percent of the applied sulfometuron-methyl applied to an area could run off from the application site to adjoining areas after a moderate rain, based on studies of runoff from 3.3 inches of total rainfall (1.7 inches per hour for 2 hours) by Hubbard et al. (1989) and from 0.47 to 1.18 inches of rainfall by Wauchope et al. (1990). Losses could be much greater and might approach 50 percent in cases of extremely heavy rain and a steep soil slope (SERA 2004f).

Using the root zone model GLEAMS, SERA (2004f) estimated the peak WCRs of streams associated with the normal application (1 lb a.e./acre) of sulfometuron-methyl as ranging from 0.00006 to 0.02 mg a.e./L. Neary and Michael (1989) applied sulfometuron methyl in the form of dispersible granules at a rate of 0.36 lbs/acre to a study site in Florida. They monitored nearby surface water for chemical contamination for up to 203 days after treatment. The maximum concentration of sulfometuron methyl was reported as 0.07 mg/L. Normalizing this water concentration to an application rate of 1 lb/acre gives a water contamination rate of 0.02 mg a.e./L. At the proposed maximum application rate of 0.378 lbs a.e./acre, the expected levels of sulfometuron methyl (under conditions similar to those in the Neary and Michael [1989] study) in surface water would be 0.008 mg a.e./L.

End-Use Products. The only commercial formulation of sulfometuron-methyl that the USFS proposes to use is Oust XP. Oust XP is the same formulation as Oust DF, which was analyzed in the 2007 Opinion; however, the granule shape has been changed to provide better mixability. Oust XP is manufactured by DuPont and is comprised of 75 percent sulfometuron-methyl and 25 percent inert ingredients (SERA 2004f). The inert ingredients include sucrose, sodium salt of naphthalene-sulfonic acid formaldehyde condensate, polyvinyl pyrrolidone, sodium salt of sulfated alkyl carboxylated and sulfated alkyl naphthalene, and hydroxypropyl methylcellulose. None of these inert ingredients are classified by EPA as toxic. The toxicity of Oust XP appears to be similar to that of technical grade sulfometuron methyl; providing further support that the inerts are not very toxic.

Toxicity: Fish. Sulfometuron-methyl does not appear to be highly toxic to fish; however, investigations of acute toxicity have been hampered by the limited water solubility of sulfometuron-methyl. Furthermore, the available studies have focused on lethal endpoints rather than sublethal ones. In the available studies, none of the fish died from acute exposure to sulfometuron-methyl, even at the highest concentration tests. As such, NOEC values (based on lethality) were placed at the highest concentrations tested: 7.3 mg a.e./L for fathead minnow (Muska and Driscoll 1982) and 148 mg a.e./L for rainbow trout (Brown 1994). Only one study regarding chronic toxicity of sulfometuron-methyl to fish has been performed. Muska and Driscoll (1982) did not observe any effects on fathead minnow embryo hatch, larval survival, or larval growth over 30-day exposure periods where concentrations of sulfometuron ranged up to 1.17 mg a.e./L.

Toxicity: Other Aquatic Organisms. Sulfometuron-methyl also appears to be relatively non-toxic to aquatic invertebrates, based on acute bioassays in daphnids, crayfish, and field-collected species of *Diaptomus*, *Eucyclops*, *Alonella*, and *Cypria*. The absolute LC_{50} values reported in SERA (2004f) for daphnids, crayfish, and the aquatic invertebrates are above 601 mg a.e./L, some by more than a factor of 10. A couple of studies using daphnids as the test species did not test concentrations high enough to cause lethality (i.e., 48-hour LC_{50} values of >12.5 mg/L and >150 mg/L). One daphnid reproduction study noted a reduction in the number of neonates at 24 mg/L, but not at 97 mg/L or at any of the lower concentrations tested (Baer 1990). This study did not have a clear dose-response effect.

Aquatic plants appear more sensitive than aquatic animals to the effects of sulfometuron-methyl, although there appear to be substantial differences in sensitivity among species of macrophytes and unicellular algae. The macrophytes, however, appear to be generally more sensitive. The 14-day NOEC (growth inhibition as measured by frond count) for duckweed exposed to technical grade sulfometuron-methyl was reported as 0.00021 mg a.e./L (Kannuck and Sloman 1995). For algae, the most sensitive algal species tested was *Selenastrum capricornutum*, with a 72-hour NOEC of 0.0025 mg/L and a 72-hour EC_{50} of 0.0046 mg a.e./L, based on a reduction in cell density relative to controls (Hoberg 1990). The most tolerant algal species tested was *Navicula pelliculosa*, with a 120-hour NOEC of 0.37 mg/L (Thompson 1994). The EC_{50} values for other freshwater algal species are generally greater than 10 μg/L, depending on the endpoint assayed (Landstein et al. 1993), but still fall in a range of concentrations that are likely to occur after a rainfall.

Effects on Non-Target Plants. The toxicity of sulfometuron-methyl to terrestrial plants was studied extensively and is well characterized. Assays using an application rate of 0.00892 lbs a.i./acre show high toxicity to seedlings of several broadleaf plants and grasses, either pre-emergence or post-emergence. Moreover, adverse effects were observed in most plants tested at application rates of 0.00089 lbs a.i./acre (SERA 2004f). This application rate is a factor of about 100- to 300-fold less than the application rate that the USFS would typically use. Concern for the sensitivity of non-target plant species is further increased by field reports of substantial and prolonged damage to crops or ornamentals after the application of sulfometuron methyl in both an arid region, presumably due to the transport of soil contaminated with sulfometuron methyl by wind, and in a region with heavy rainfall, presumably due to the wash-off of sulfometuron methyl contaminated soil (SERA 2004f).

TRICLOPYR

Exposure. Triclopyr herbicides in the proposed action contain one of two forms of triclopyr, either the triethylamine salt (TEA) or the butoxyethyl ester (BEE). In both soil and aquatic environments, both the ester and amine salt formulations of triclopyr rapidly convert to the triclopyr acid and other degradates. In various soil types, the half-life of BEE has been reported to be 3 hours, and the half-life of TEA has been reported to range from 6 to 14 days. Triclopyr acid is further degraded by soil microorganisms to the metabolites trichloropyridinol (TCP) and trichloromethoxypyridine. In aerobic soils, triclopyr acid has a half-life of 8 to 18 days. The TCP is more persistent than triclopyr acid, with a soil half-life ranging from 40 to 95 days (Knuteson 1999).

In water, triclopyr TEA dissociates to the acid very rapidly (i.e., within one minute), and triclopyr BEE hydrolyzes to the acid in less than a day in natural waters with a pH of 6.7 (EPA 1998). The primary degradation mechanism for triclopyr acid in water is photolysis, with a half-life of 1-day. TCP is more persistent in aquatic environments, having a half-life of 4 to 10 days (Petty et al. 2003). Triclopyr and TCP are not strongly adsorbed to soil particles and have the potential to be mobile, thus there is a chance that application of triclopyr near aquatic environments can result in surface water contamination.

SERA (2011c) estimated peak WCRs (normalized to an application rate of 1 lb/acre) for three forms of triclopyr in stream water using a variety of methods. The WCRs were derived from various modeling efforts and from field studies pairing triclopyr application with surface water monitoring. For triclopyr BEE, stream WCRs ranged from 0.00 to 0.03 mg a.e./L. The upper bound of the peak WCR (i.e., 0.03 mg a.e./L) was the water concentration documented in a stream by Smith and McCormack (1988) after an application of triclopyr BEE. While this concentration may have been due to an accidental spray, this concentration was close to that estimated by SERA (2011c) due to drift. Thus, it has been selected as the appropriate peak WCR for this Opinion. For triclopyr acid, stream WCRs ranged from 0.00 to 0.24 mg a.e./L. The upper bound of the peak WCR (i.e., 0.24 mg a.e./L) was derived from EPA modeling efforts using PRZM/EXAMS (EPA 2009). For the metabolite TCP, modeled WCRs ranged from

0.00 to 0.03 mg TCP/L after application of triclopyr BEE and from 0.00 to 0.02 mg TCP/L after application of triclopyr TEA.

Maximum application rates in the proposed action are 9.0 lbs/acre for triclopyr TEA. Multiplying the maximum application rate by the WCRs gives an EEC of: 2.16 mg a.e./L for triclopyr acid and 0.18 mg TCP/L after application of the TEA formulation. Typical application rates are likely to generate EEC's at least 4 1/2 to 9 times lower. Because triclopyr TEA near instantaneously dissolves to the acid, SERA (2011c) did not determine an EEC for that form of triclopyr.

End-Use Products. Triclopyr herbicides included in the proposed action contain only the use of the TEA form of triclopyr. The number of triclopyr EUPs that may be used by the USFS has increased over the years. In 1996, only two EUPs were available for use, Garlon 3A (TEA formulation) and Garlon 4 (BEE formulation). Since then, 17 additional EUPs (from eight different companies) have been approved at a national level for forests to consider using. As outlined in the biological assessment, the USFS only proposes to use the TEA Garlon product Garlon 3A.

Although triclopyr TEA EUPs are equivalent to one another in that they contain 44.4 percent triclopyr TEA, their overall formulations may be different. The liquid formulations of 44.4 percent triclopyr TEA specify other ingredients as either ethanol (Garlon 3A, Renovate 3, and Tahoe 3A) or ethylenediaminetetraacetic acid (EDTA), which is a chelating agent (Triclopyr 3A, Triclopyr 3SL). Triclopyr 3SL also contains ethylene glycol. As described in Section 2.4.1.3, Renovate 3 is identical to Garlon 3A, so its toxicity is expected to be the same as those reported for Garlon 3A and triclopyr TEA. Based on information contained in the MSDS and summarized in Table 14, the toxic effects of these EUPs should be similar, and have been analyzed accordingly in this Opinion.

The EUPs evaluated in this Opinion contain varying types and amounts of inert ingredients. Identified inert ingredients include ethylene glycol, ethanol, and EDTA. Wan et al. (1987) determined that both Garlon 3A and Garlon 4 were significantly less toxic ($p<0.01$) to salmonids than their respective active ingredients triclopyr TEA and triclopyr BEE, suggesting that inert ingredients used in formulating these products do not increase toxicity. The product labels recommend that a surfactant be added to the product prior to most applications. Some surfactants are more toxic than others. Toxicity of some surfactants proposed for use has been addressed in Section 2.4.1.3 of this Opinion.

Toxicity: Fish. Both forms of triclopyr degrade into triclopyr acid and other degradates in the environment. Triclopyr acid is further degraded into TCP and other metabolites. The other metabolites (e.g., butoxyethanol and triethanolamine) are not being evaluated further because they are rapidly dissipated by microbial degradation. The TCP is of concern because it has been shown to be more toxic than the other forms of triclopyr to many groups of non-target organisms (SERA 2011c).

Lethal Effects. Data on the toxicity of triclopyr and its various forms has been collected since as early as 1973. Wan et al. (1987) completed the most extensive comparative study on the toxicity of the various forms and metabolites of triclopyr. This study summarizes a series of static bioassays on several species of salmonids that were conducted over a 4-month period in 1986 and a 2-month period in 1987. Wan et al. (1987) reported 96-hour LC_{50} values for triclopyr acid, triclopyr ester (BEE), Garlon 3A, Garlon 4, and TCP, which are summarized in Table A-1. The authors found triclopyr ester (BEE) was the most toxic chemical tested, followed in decreasing toxicity to salmonids by Garlon 4, TCP, triclopyr acid, and Garlon 3A.

Table A-1. Acute Toxicity of Triclopyr and Related Compounds to Various Species of Salmonids[1]. Results are expressed as mg a.e./L, unless otherwise noted.

Fish Species	Triclopyr TEA (Garlon 3A)	Triclopyr BEE (Garlon 4)	Triclopyr BEE (technical grade)	Triclopyr Acid	TCP (mg TCP/L)
Coho salmon	167	1.0	1.0	9.6	1.8
Chum salmon	96.1	0.82	0.3	7.5	1.8
Sockeye salmon	112	0.67	0.4	7.5	2.5
Rainbow trout	151	1.3	1.1	7.5	1.5
Chinook salmon	99	1.3	1.1	9.7	2.1
Pink salmon	-	0.58	0.5	5.3	2.7

[1] Source: Wan et al. 1987. All bioassays conducted at 46.4 to 50°F, 10 fish/concentration. Static with aeration. LC_{50} based on measured, rather than nominal concentrations. Photoperiod and lighting conditions not specified.

The BEE form of triclopyr is exponentially more toxic to fish when compared to the TEA form. The salmonid LC_{50} values for triclopyr BEE (technical grade and as formulated Garlon 4) ranged from 0.19 mg a.e./L to 1.9 mg a.e./L (SERA 2011c). The lowest LC_{50} value was for coho salmon alevins (Mayes et al. 1986). The Wan et al. (1987) study is supported by more recent flow-through toxicity assays on Garlon 4 with reported acute LC_{50} values for salmonids of 0.79 to 1.76 mg/L (Kreutzweiser et al. 1994) and 0.84 mg/L (Johansen and Geen 1990).

Wan et al. (1987) found that Garlon 3A, a formulation of triclopyr TEA, was about 170 times less toxic (significant at $p<0.01$) to salmonids than the Garlon 4 formulation. Triclopyr TEA LC_{50} values for salmonids reportedly range from 75.4 mg a.e./L to 273.7 mg a.e./L (SERA 2011c;). EPA classified triclopyr TEA as practically non-toxic to freshwater fishes (EPA 1998).

Based upon available information, the triclopyr acid appears to be approximately 11 times less toxic to salmonids than the triclopyr BEE. Based upon information in all available literature, the salmonid LC_{50} values for triclopyr acid range from 5.3 mg a.e./L to 117 mg a.e./L (SERA 2011c; Patrick Durkin, personal communication). Six of the seven LC_{50} values included in this range came from the Wan et al. (1987) study, and they appear to be outliers not only with respect to the higher LC_{50} value from Batchelder (1973), but also with respect to all 17 LC_{50} values on triclopyr TEA. According to SERA, the results from Wan et al. (1987) cannot be attributed to experimental factors or methods, and the study cannot be dismissed as irrelevant. While one would expect the acid form to be more toxic than the salt form, the extreme difference (more than an order of magnitude) noted above is suspect (Patrick Durkin, personal communication). Because of this, neither SERA (2011c) nor EPA (2009) included the data in their assessments. Giving deference to toxicological experts, this Opinion utilizes 117 mg a.e./L as the lethal concentration for triclopyr acid.

The TCP (an environmental metabolite of triclopyr acid), is substantially more toxic in fish than either triclopyr acid or triclopyr TEA, and is similar to the toxicity of triclopyr BEE. Salmonid

TCP LC_{50} values from two separate studies (Wan et al. 1987; Gorzinski et al. 1991) range from 1.5 mg TCP/L to 12.6 mg TCP/L. Six of the seven salmonid LC_{50} values for TCP are from Wan et al. (1987), and all are approximately five times lower than the value obtained by Gorzinski et al. (1991). There is no clear explanation as to why these two experiments had such vastly different results. It may reflect experimental variability or other unknown factors rather than any differences in species sensitivity (SERA 2011c). This Opinion uses the lowest value (i.e., 1.5 mg TCP/L) as the lethal concentration for TCP.

Sublethal Effects. A few acute and chronic studies examining sublethal effects have been performed on triclopyr BEE, triclopyr TEA, and the metabolite TCP. Similar to the lethality studies, results from the sublethal effects studies indicate that triclopyr BEE was the most toxic and triclopyr TEA was the least toxic.

An early life-stage study conducted with triclopyr BEE in rainbow trout yielded a NOEC of 0.017 mg a.e./L and a LOEC (based on larval length and weight) of 0.035 mg a.e./L (Weinberg et al. 1994). Johansen and Geen (1990) studied the sublethal effects of Garlon 4 on rainbow trout using flow-through systems. The authors noted fish were more docile (than the controls) at concentrations of 0.32 to 0.43 mg a.e./L, which are about a factor of 2 below the 96-hour LC_{50} determined in this study. At levels $\leq$0.1 mg a.e./L, rainbow trout were hypersensitive to photoperiod changes over 4-day periods of exposure. This is reasonably consistent with the threshold for behavioral changes in rainbow trout for Garlon 4 of 0.26 mg a.e./L reported by Morgan et al. (1991).

For triclopyr TEA, a 28 day egg-to-fry study was performed using fathead minnows (Mayes et al. 1984; Mayes 1990). In these studies, fathead minnow eggs were exposed to concentrations of 26, 43, 65, 104, 162, and 253 mg a.i./L. The survival of fathead minnows (embryo-larval stages) was significantly reduced at 253 mg/L compared with control animals. At 162 mg/L, there was a slight decrease in body length. The authors reported a NOEC of 32.2 mg a.e./L and a LOEC (length) of 50.2 mg a.e./L. Morgan et al. (1991) examined behavior changes in rainbow trout after a 0.5-hour exposure to Garlon 3A. The authors reported a threshold for behavioral changes of 63.6 mg a.e./L and a threshold for avoidance response of 254 mg a.e./L.

Marino et al. (2003) conducted an egg-to-fry study, exposing rainbow trout to TCP. The authors exposed rainbow trout to 0.586, 0.106, 0.178, 0.278, 0.479, and 0.825 mg TCP/L in a flow-through system. Observations were made for 33 days post-hatch of the water control embryos. The authors reported a NOEC for fry weight and growth of 0.178 mg TCP/L, and a LOEC of 0.278 mg TCP/L.

Although TCP is much more toxic than triclopyr TEA, field monitoring cited in SERA (2003d) indicates that TCP residues in soil and water occur at concentrations much lower than the application rate of the active ingredient. Given the high toxicity of TCP and the uncertainty of exposure risk to this metabolite, the potential for adverse effects to listed fish is uncertain. Garlon 4 is not proposed for use in this Opinion.

Toxicity: Other Aquatic Organisms. Based on acute lethality, aquatic invertebrates appear to be about equally or somewhat less sensitive than fish to the various forms of triclopyr. Acute

LC_{50} values for triclopyr acid and triclopyr TEA range from about 100 to about 6,400 mg a.e./L. Gersich et al. (1982) conducted a chronic daphnid study and reported a NOEC of 25.95 mg a.e./L. Triclopyr BEE was substantially more toxic to aquatic invertebrates, with LC_{50} values ranging from 0.19 to 20 mg a.e./L (SERA 2011c). Some of the studies reported NOEC (for lethality), and those ranged from 0.12 mg a.e./L to 1.2 mg a.e./L. Increases in invertebrate drift have been documented at triclopyr BEE concentrations of 0.6 to 0.95 mg/L (Kreutzweiser et al. 1995; Thompson et al. 1995), but no other effects such as changes in stream invertebrate abundance were noted. In a chronic study, Chen et al. (2008) reported concentration-related decreases in *Simocephalus vetulus* (a cladoceran) at triclopyr BEE concentration of 0.25 mg a.e./L and 0.5 mg a.e./L. Only two studies examining the toxicity of TCP on aquatic invertebrates were available. One study reported an acute LC_{50} of 10.9 mg TCP/L (EPA 2009). The second study reported a NOEC of 0.058 mg TCP/L, based on a decrease in mean number of young/adult (Machado 2003).

Similar to aquatic organisms, algae are more sensitive to triclopyr BEE than to triclopyr TEA. For triclopyr BEE, the EC_{50} values for growth inhibition in algae range from about 0.073 to 5.9 mg a.e./L. For triclopyr TEA and triclopyr acid, the EC_{50} values for the same endpoint in algae range from about 0.49 to 80 mg a.e./L. The TCP toxicity falls between the other forms, with a reported EC_{50} value of 1.8 mg TCP/L.

For aquatic macrophytes, triclopyr TEA is more toxic to dicots than to monocots, with EC_{50} values ranging from 0.04 to 0.56 mg a.e./L and 6.06 to 15.8 mg a.e./L, respectively. In fact, triclopyr TEA appears to be more toxic to dicots than triclopyr BEE (EC_{50} values ranging from 1.49 to 4.62 mg a.e./L). No studies were available regarding the toxicity of TCP.

NMFS Pesticide Registration Opinion. Chemical concentrations examined in the in the 2011 registration Opinion (NMFS Tracking # 2004/02673) did not vary drastically from those summarized here. The triclopyr registration Opinion used the following rainbow trout LC_{50} values as assessment endpoints for triclopyr: 0.470 mg a.e./L for BEE, 79.2 mg a.e./L for TEA, and 177 mg a.e./L for triclopyr acid. Information presented in the 2011 Opinion for EPAs registration of triclopyr does not suggest a different endpoint as being more appropriate than that which was used in this Opinion.

The registration Opinion concluded there was no overlap between the peak farm pond EECs for forestry uses (at 6 lb a.e./acre) and the fish and invertebrate toxicity endpoints for triclopyr BEE. Floodplain estimates for triclopyr BEE overlapped with all acute assessment endpoints at the application rate of 8 lb a.e./acre. For triclopyr TEA, none of the peak concentrations and assessment endpoints overlapped.

Fluroxypyr

Exposure: Because surface runoff potential is high, and potential for loss on eroded soil is low, Fluroxypyr has the potential to leach to groundwater. Fluroxypyr is moderately mobile in and poorly adsorbed by most soil types. At single point concentration of 0.066 mg/L in soil, fluroxypyr is very mobile in silt loam, sandy loam, loam, and silty clay (Lehmann et al. 1988). In field studies, Bergstrom et al. (1990) found concentrations in soil were either undetectable or at very low levels within 3 months after the application of fluroxypyr. A typical half-life for fluroxypyr in the soils is 36 days. Fluroxypyr is broken down by microbes and sunlight. Water contamination rates are complex as fluroxypyr is poorly soluble and the upper bound of measured concentrations exceeds the water solubility. SERA (2009) reported peak WCR of 0.08 mg/L per lb/acre applied.

End Use Products. Two formulations of fluroxypyr are specifically considered in the SERA (2009) risk assessment: Vista Specialty Herbicide and Vista XRT. Both of these formulations contain the 1-methylheptyl ester of fluroxypyr as well as two listed inerts: naphthalene and 1 methyl-2-pyrrolidinone. The Vista XRT (the formulation proposed for use in the proposed action) formulation contains a greater concentration of the fluroxypyr ester and much lower concentrations of the listed inerts. Fluroxypyr is the only active ingredient (26.2 percent) in the herbicide Vista. According to the product label, Vista also contains 73.8 percent other ingredients (unspecified).

***Toxicity Fish*:** Fluroxypyr does not bioconcentrate through the food chain. The SERA Risk Assessment (2009) classifies fluroxypyr-acid as slightly toxic to practically nontoxic to fish, based on acute toxicity. Acute toxicity tests evaluated by the EPA indicate that fluroxypyr is slightly toxic to practically non-toxic to freshwater fish. For bluegill sunfish, a 96-hour LC_{50} >14.3 mg/L was reported. For rainbow trout, 96-hour LC_{50} values ranged from 13.4 mg/L to >100 mg/L. In studies by Wan et al. (1992), the 96-hour LC_{50} for Chinook salmon ranged from 10 to 17 mg/L and sockeye salmon (*O. nerka*) ranged from 10 to 15 mg/L. However, the Wan et al. (1992) study was for unidentified formulations with no relation to proposed formulations. Those results were not applied in the SERA (2009) assessment. The NOEC values for fish used in the 2009 risk assessment are taken as 0.060 mg a.e./L for sensitive species, including salmonids, and 0.49 mg a.e./L for tolerant species.

The fluroxypyr literature does not include chronic fish bioassays. The EPA/OPP waived the requirement for chronic testing because of the low acute toxicity of both fluroxypyr acid and fluroxypyr-MHE to fish. Fluroxypyr has limited solubility in water and SERA (2009) indicated that solvents used in toxicity studies may influence reported toxicity levels. In addition the only LC_{50} values found were for unidentified formulations that were not covered by the Forest Service risk assessment (SERA 2009). They even state, "available fish bioassays submitted to the EPA suggest the unlikelihood of adverse effects resulting from fluroxypyr exposure, and the development of a hazard quotient is unwarranted." This is likely due to nominal concentrations (a product of volume herbicide added to volume water) are always several orders of magnitude higher than measured concentrations of fluroxypyr, because of the solubility properties. SERA (2009) stated that, "it is not clear that fluroxypyr-MHE would cause adverse effects under any plausible set of conditions."

Toxicity Other Aquatic Organisms: Results from toxicity testing conducted on Daphnia magna indicate that fluroxypyr is practically non-toxic to this species of invertebrate. The 48-hour EC_{50} for this toxicity test was >100 mg/L. However, the EPA did not require a chronic study in aquatic invertebrates because of the low toxicity of fluroxypyr acid to this group of organisms. Jones (1984) conducted a standard life cycle study which was submitted to the EPA in support of the registration of fluroxypyr. This study reports an effect on reproduction parameters but does not identify an LOEC based on immobility at 100 mg. The NOEC reported in the study is 56 mg/L. Eastern oysters, however, appear to be more sensitive to fluroxypyr-MHE. Based on the EC_{50} of 0.068 mg ester/L [measured concentration equivalent to 0.042 mg a.e./L] for shell deposition (Boeri et al. 1996), fluroxypyr-MHE is classified as very highly toxic (EPA/OPP 1998). The Forest Service risk assessment elected not to base toxicity values on estimates of an EC_{50} or LOEC. The EPA/OPP (2004), on the other hand, uses EC_{50} values, but interprets risk with levels of concern of 0.5 for acute risk and 0.05 for endangered species. To maintain compatibility with the EPA, the Forest Service risk assessments divided the EC_{50} by a factor of 20 to approximate a NOEC. Using this approach, the EC_{50} of 0.068 mg/L is used to estimate a NOEC of 0.0034 mg/L. This concentration is below the reported LOEC by a factor of about 15 [0.05 mg/L ÷0.0034 mg/L ≈ 14.71]. When the NOEC of 0.0034 mg/L is converted to acid equivalents, the acute toxicity value used for sensitive species of aquatic invertebrates is about 0.002 mg a.e./L [0.0034 mg a.i./L x 0.694 a.e/a.i. = 0.00236].

Fluroxypyr-MHE is much more toxic to aquatic plants than to aquatic animals, as is true for most herbicides. For algae, the available NOEC values range from 0.03 mg a.i./L (*Anabaena flos-aquae* (Milazzo et al. 1996a) to 0.199 mg a.i./L (*Selenastrum Capricornutum*) (Milazzo et al. 1996b). When converted to acid equivalents (0.694 a.e./a.i.), these concentrations correspond to about 0.021–0.14 mg a.e./L and are used for sensitive and tolerant species, respectively.

For macrophytes, only two studies are available, and both were conducted using *Lemna gibba*, duckweed (Kirk et al. 1996; Kirk et al. 1998). The 7-day NOECs from these studies are 0.412 mg/L (Kirk et al. 1998) and 1.22 mg/L (Kirk et al. 1996). Kirk et al. (1998) also report a 14-day NOEC of 0.437 mg/L. The study by Kirk et al. (1996) is classified as Core by EPA/OPP (1998). The later study by Kirk et al. (1998) is not cited in EPA/OPP (1998) and may not have been available at the time the risk assessment was prepared. For the current risk assessment, the lowest NOEC, 0.412 mg/L reported by Kirk et al. (1998) is used for tolerant species. When adjusted for acid equivalents, the toxicity value is about 0.29 mg a.e./L [0.412 mg a.i./L x 0.694 a.e./a.e. = 0.285928 mg a.e./L]. No dose-response assessment is proposed for sensitive species of aquatic macrophytes.

References

Abdel-Hamid, M.I. 1996. Development and application of a simple procedure for toxicity testing using immobilized algae. Water Sci Technol. 33(6):129-138. (Cited in SERA 2004a).

Alexander, H.C., F. M. Gersich, and M. A. Mayes. 1985. Acute toxicity of four phenoxy herbicides to aquatic organisms. Bulletin of Environmental Contamination and Toxicology 35:314-321.

Baarschers, W. H., J. G. Donnelly, and H. S. Heitland. 1988. Microbial toxicity of triclopyr and related herbicides. Environmental Toxicology 3(2):127-136.

Baer, K. 1990. Chronic Toxicity of IN T5648-40 to *Daphnia magna*. Lab Project Number: 614-90: MR-4581-804. Unpublished study prepared by Haskell Laboratory. 260 pages. MRID 41672806. (Cited in SERA 2004f).

Baloch-Haq, R., M. Snel, R. Jackson, et al. 1993. Comparison of the dissipation characteristics of clopyralid as determined by laboratory, field and lysimeter studies, Brighton crop protection conference, weeds. Proceedings of an international conference, Brighton, United Kingdom, pp. 783-788. (Cited in SERA 2004b).

Barker, C., K. Drottar, and H. Krueger. 1998. Toxicity of AC 263222 during the early life-stages of the fathead minnow (*Pimephelas promelas)*: Lab Project Number: 2CO 98-134: 954-98-134:130A-120. Unpublished study prepared by Wildlife International, Ltd. 93 pages. MRID No. 44728202. (Cited in SERA 2004d).

Batchelder, T. L. 1973. Acute Fish toxicity of Dowco 233 (3,5,6- Trichloro-2-pyridyloxyacetic acid) and two derivatives. Report WCL-73014. Submitted by Dow Chemical U.S.A., Midland, Michigan. MRID 49637. (Cited in SERA 2011c).

Batchelder, T. 1974. Acute fish toxicity of picloram (dry TORDON acid). Waste Control Department, The Dow Chemical Company, Midland, MI. No. File: 005101-016.pdf. DER for MRID 00112016 dated 7/16/82. (Cited in SERA 2011b).

Berg, N. 2004. Assessment of Herbicide Best Management Practices: Status of Our Knowledge of BMP Effectiveness. Final Report, March 2004, Pacific Southwest Research Station, USDA Forest Service, Albany, CA.

Bergstrom, L. F., A. S. Mcgibbon, S. R. Day, and M. Snel. 1990. Leaching Potential and Decomposition of Fluroxypyr in Swedish Soils Under Field Conditions. Pestic Sci. 29 (4): 405-418.

Bergstrom, L., A. McGibbon, S. Day and M. Snel. 1991. Leaching potential and decomposition of clopyralid in Swedish soils under field conditions. Environmental Toxicology and Chemistry 10(5):563-571.

Blasberg, J., S. Hicks, and J. Stratton. 1991. Acute Toxicity of Chlorsulfuron to *Selenastrum capricornutum Printz*: Final Report. Lab Project Number: AMR-2081-91: 39427. Unpublished study prepared by ABC Laboratories, Inc. 33 p. MRID 42186801. (Cited in SERA 2004a).

Boeri, R., P. Kowalski, and T. Ward. 1996. Fluroxypyr 1-Methylheptyl Ester: Acute Flow-Through Mollusc Shell Deposition Test. Laboratory Report ID:646-DO. Unpublished study prepared by T. R. Wilbury Labs., Inc. 13 p. MRID No. 44080310.

Boeri, R., D. Wyskiel, and T. Ward. 2002. Picloram acid: life cycle study in the Daphnid, *Daphnia magna*: Lab Project number 2391-DO: 021029. Unpublished study prepared by T.R. Wilbury Laboratories, Inc. 46 pages. MRID 45760301. (Cited in SERA 2011b).

Bond, C. E., J. D. Fortune, and F. Young. 1965. Results of preliminary bioassays with kurosal-SL and dicamba. Progressive Fish Culturalist 27(1):49-51.

Bovey, R. W. and C. W. Richardson. 1991. Dissipation of clopyralid and picloram in soil and seep flow in the Blacklands of Texas. Journal of Environmental Quality 20(3):528-531.

Brown, M. R. 1994. Static, acute, 96-hour LC_{50} of DPX-T5648-42 (sulfometuron methyl) to rainbow trout, *Oncorhynchus mykiss*: (Limit test). Laboratory Report No. 589-94. Unpublished study prepared by E.I. du Pont de Nemours and Co., Haskell Laboratory for Toxicology and Industrial Medicine, Newark, DE. 28 pages. MRID 43501802. (Cited in SERA 2004f).

Chen, C. Y., K. M. Hathaway, D. G. Thompson, and C. L. Folt. 2008. Multiple stressor effects of herbicide, pH, and food on wetland zooplankton and a larval amphibian. Ecotoxicology and Environmental Safety 71(1):209-218.

Concha, M. and K. Shepler. 1994. Photodegradation of (carbon 14) - clopyralid in buffered aqueous solution at ph 7 by natural sunlight: Lab Project Number: 451W: ENV94048 (Unpublished study prepared by PTRL West, Inc.). 65 pp. MRID No.43891401. (Cited in SERA 2004b).

Coyner, A., G. Gupta, and T. Jones. 2001. Effect of chlorsulfuron on growth of submerged aquatic macrophyte *Potamogeton pectinatus* (sago pondweed). Environmental Pollution 111(3):453-455. (Cited in SERA 2004a).

Crozier, L. G., R. W. Zabel, and A. F. Hamlet. 2008a. Predicting differential effects of climate change at the population level with life-cycle models of spring Chinook salmon. Global Change Biology 14:236-249. DOI: 10.1111/j.1365-2486.2007.01497.x.

Crozier, L. G., A. P. Hendry, P. W. Lawson, T. P. Quinn, et al. 2008b. Potential responses to climate change for organisms with complex life histories: evolution and plasticity in Pacific salmon. Evolutionary Applications 1:252-270. DOI: 10.1111/j.1752-4571.2008.00033.x. 10.

European Commission. 2002. Review report for the active substance imazamox. Health and Consumer Protection Directorate-General, Directorate E – Food Safety. Available at: http://ec.europa.eu/food/plant/protection/evaluation/newactive/imazamox.pdf. [Internet] (Cited in SERA 2010).

Extoxnet. 1996. Pesticide Information Profiles: Metsulfuron-methyl. Extension Technology Network. Available: http://extoxnet.orst.edu/. (May 2012).

Fairchild, J. F., A. Allert, K. P. Feltz, K. J. Nelson, and J. A. Valle. 2009a. An ecological risk assessment of the acute and chronic effects of the herbicide clopyralid to rainbow trout (*Oncorhynchus mykiss*). Archives of Environmental Contamination and Toxicology 57:725-731.

Fairchild, J. F., K. P. Feltz, L. C. Sappington, A. L. Allert, K. J. Nelson, and J. Valle. 2009b. An ecological risk assessment of the acute and chronic toxicity of the herbicide picloram to the threatened bull trout (*Salvelinus confluentus*) and the rainbow trout (*Oncorhynchus mykiss*). Archives of Environmental Contamination and Toxicology 56(4):761-769.

Fairchild, J. F., A. Allert, L. S. Sappington, K. J. Nelson, and J. Valle. 2008. Using accelerated life testing procedures to compare the relative sensitivity of rainbow trout and the federally listed threatened bull trout to three commonly used rangeland herbicides (picloram, 2,4-D, and clopyralid). Environmental Toxicology and Chemistry 27(3):623-630.

Fairchild, J. F., D. S. Ruessler, P. S. Haverland, and A. R. Carlson. 1997. Comparative sensitivity of *Selenastrum capricornutum* and *Lemna minor* to sixteen herbicides. Archive of Environmental Contamination and Toxicology 32(4):353-457. (Cited in SERA 2004a, c).

Folmar, L.C. 1978. Avoidance chamber responses of mayfly nymphs exposed to eight herbicides. Bulletin of Environmental Contamination and Toxicology 19(3):312-318.

Forbis, A., D. Burgess, and L. Georgie. 1985. Acute toxicity of CN 10-6471 [(Banvel herbicide)] to Daphnia magna: Report No. 33173. Unpublished study prepared by Analytical Biochemistry Laboratories, Inc. 37 pages. MRID 001513152. (Cited in SERA 2004c).

Forsyth, D. J., P. A. Martin, and G. G. Shaw. 1997. Effects of herbicides on two submersed aquatic macrophytes, *Potamogeton pectinatus* L. and *Myriophyllum sibiricum* Komarov, in a prairie wetland. Environmental Pollution 95(2):259–268.

Gardner S. C., C. E. Grue, J. M. Grassley, L. A. Lenz, J. M. Lindenauer, and M. E. Seeley. 1997. Single species algal (*Ankistrodesmus*) toxicity tests with Rodeo and Garlon 3a. Bulletin of Environmental Contamination and Toxicology 59(3):492-499.

Gersich, F., D. Hopkins, and D. Milazzo. 1984. The acute and chronic toxicity of technical picloram (4-Amino-3,5,6-trichloropicolinic acid) to *Daphnia magna* Straus: ES-690. Unpublished study prepared by Dow Chemical USA. 16 pages. MRID 00151783. (Cited in SERA 2011b).

Gersich, F., C. Mandoza, and D. Hopkins. 1982. The acute and chronic toxicity of triclopyr triethylamine salt solution to *Daphnia magna* Straus: ES-583. Unpublished study prepared by Environmental Sciences Research, Dow Chemical U.S.A. 19 pages. MRID 00151959. (Cited in SERA 2011c).

Gomez, L., J. Masot, S. Martinez, E. Duran, F. Soler, and V. Roncero. 1998. Acute 2,4-D poisoning in tench (*Tinca* L.): Lesions in the hematopoietic portion of the kidney. Archive of Environmental Contamination and Toxicology 35:479-483.

Goodman, N. C. 1979. 48-Hour LC50 to *Daphnia magna*. Haskell Laboratory Report No. 152-79. (Unpublished study received Jun 16, 1980 under 352-105; submitted by E.I. du Pont de Nemours & Co., Wilmington, Del.; CDL:099462-H). MRID 00035262. (Cited in SERA 2004a).

Gorzinski, S., K. Lehr, and D. Piasecki. 1991. Garlon 4 herbicide: static acute 96-hour toxicity to the rainbow trout, *Oncorhynchus mykiss* Walbaum: Lab Project Number: ES-DR-022406168-4. Unpublished study prepared by DowElanco. 33 pages. MRID 41971603. (Cited in SERA 2011c).

Grande, M., S. Andersen, and D. Berge. 1994. Effects of pesticides on fish: experimental and field studies. Norwegian Journal of Agricultural Sciences Suppl. 13:195-209. MRID 45202909. (Cited in SERA 2004a).

Grover, R. and A.E. Smith. 1974. Adsorption studies with the acid and dimethylamine forms of 2,4-D and dicamba. Canadian Journal of Soil Science 54:179-786.

Hall, C. 1984. 96-hour LC_{50} to rainbow trout: benzoic acid, 2-N-(4-Methoxy-6-methyl- 1,3,5-triazin-2-yl)-N-methylamino-carbonyl-amino-sulfonyl, methyl ester. Laboratory Report No. 545-84. Unpublished study prepared by Haskell Laboratory for Toxicology and Industrial Medicine. 5 pages. MRID No. 00149672. (Cited in SERA 2004e).

Hawes, K. and S. Erhardt-Zabik. 1995. The anaerobic aquatic metabolism of clopyralid. Lab Project Number: ENV93076, Unpublished study prepared by DowElanco North American Environmental Chemistry Lab. MRID No.43891404. 42 pp. (Cited in SERA 2004a).

Healey, M. C. 1982. Timing and relative intensity of size-selective mortality of juvenile chum salmon (*Oncorhynchus keta*) during early sea life. Canadian Journal of Fisheries and Aquatic Sciences 39:952-957.

Henry, C. J.; K. F. Higgins; and K. J. Buhl. 1994. Acute toxicity and hazard assessment of Rodeo, X-77 Spreader, and Chem-Trol to aquatic invertebrates. Archives of Environmental Contamination and Toxicology 27:392-399.

Hessen, D. O., T. Kallqvist, M. I. Abdel-Hamid, and D. Berge. 1994. Effects of pesticides on different zooplankton taxa in mesocosm experiments. Norwegian Journal of Agriculture Science Suppl. 0(13):153-161 (Cited in SERA 2004a).

Hildebrand, L. D., D. S. Sullivan, and T. P. Sullivan. 1980. Effects of Roundup herbicide on populations of *Daphnia magna* in a forest pond. Bulletin of Environmental Contamination and Toxicology 25(3):353-357.

Hoberg, J. 1990. Sulfometuron methyl - toxicity to the freshwater green alga *Selenastrum capricornutum*. Lab Project No. AMR- 1893-90. Unpublished study prepared by Springborn Laboratories, Inc. 24 pages. MRID 41680102. (Cited in SERA 2004f).

Hoberg J. 1993a. Dicamba technical: toxicity to the freshwater alga, *Anabaena flos-aquae*: Final Report: Lab Project Number: 93-3-4702: 10828.0892.6139.420: 100/92/02. Unpublished study prepared by Springborn Labs, Inc. 605 pages. MRID 42774109. (Cited in SERA 2004c).

Hoberg, J. 1993b. Dicamba technical: toxicity to the duckweed, *Lemna Gibba*: Final Report: Lab Project Number: 93-3-4665: 10828.0892.6140.410: 100/92/06. Unpublished study prepared
by Springborn Labs, Inc. 61 pages. MRID 42774111. (Cited in SERA 2004c).

Hubbard, R. K., R. G. Williams, M. D. Erdman, and L. R. Marti. 1989. Chemical transport from coastal plain soils under simulated rainfall. II. Movement of cyanizine, sulfometruon-methyl, and bromide. Transactions of the American Society of Agricultural Engineers 32(4):1250-1257.

Hurlbert, S. H. 1975. Secondary effects of pesticides on aquatic ecosystems. Residue Review 57:81-148.

Hutton, D. 1989. Chronic toxicity of DPX-T6376-74 to *Daphnia magna*: Lab Project Number: 4581-654: 833-88: HLR 833-88. Unpublished study prepared by DuPont Haskell Lab for Toxicology and Industrial Medicine. 39 pages. MRID No. 43490601. (Cited in SERA 2004e).

Ismail, B. S. and K. Kalithasan. 1997. Effects of repeated application on persistence and downward movement of four herbicides in soil. Australian Journal of Soil Research 35(3):503-513.

Johansen, J. A. and G. H. Geen. 1990. Sublethal and acute toxicity of the ethylene glycol butyl ether ester formulation of triclopyr to juvenile Coho salmon (*Oncorhynchus kisutch*). Archives of Environmental Contamination and Toxicology 19(4):610-616.

Johnson, W. W. and M. T. Finley. 1980. Handbook of Acute Toxicity of Chemicals to Fish and Aquatic Invertebrates. Resource Publication 137. U.S.FWS, Washington, D.C. 98 pages.

Johnson, R. M. and J. T. Sims. 1993. Influence of surface and subsoil properties on herbicide sorption by Atlantic coastal plain soils. Soil Science 155(5):339-348.

Jones, N. 1984. Prolonged Toxicity of Dowco 433 Acid to Daphnia magna: LSR/Aquatox Schedule No. AFT/84/010. Unpublished study prepared by Aquatox Limited. 38 p. MRID No. 40244521.

Kallqvist, T. and R. Romstad. 1994. Effects of agricultural pesticides on planktonic algae and cyanobacteria examples of interspecies sensitivity variations. Norwegian Journal of Agricultural Sciences Suppl. 0(13):117-131. (Cited in SERA 2004a)

Kallqvist, T., M. I. Abdel-Hamid, and D. Berge. 1994. Effects of agricultural pesticides on freshwater plankton communities in enclosures. Norwegian Journal of Agricultural Sciences Suppl. 0(13):133-152. (Cited in SERA 2004a)

Kannuck, R. and T. Sloman. 1995. Sulfometuron methyl (DPX-T5648): Influence on growth and reproduction of *Lemna gibba* G3. Lab Project No. AMR 2902-94. Unpublished study prepared by Stine-Haskell Research Center, E.I. du Pont de Nemours and Co, Newark, DE. 41pages. MRID 43538503. (Cited in SERA 2004f).

Kirk, H., D. Milazzo, and L. Humbert. 1994. The toxicity of tordon K herbicide formulation to the aquatic plant, duckweed, *Lemna gibba* L.G-3: Lab Project Number: DECO/ES/2764. Unpublished study prepared by The Environmental Toxicology and Chemistry Research Lab. 35 pages. MRID 43230311. (Cited in SERA 2011b).

Kirk, H. D., D. P. Milazzo, J. M. Hugo, and M. D. Martin. 1996. Fluroxypyr 1-Methylheptyl Ester: The Toxicity to the Aquatic Plant, Duckweed Lerona gibba L. G-3. Unpublished Study. Environmental Toxicology & Chemistry Research Laboratory, The Dow Chemical Company, Midland, MI. DowElanco, Indianapolis, IN. MRID No. 44080338.

Kirk, H., M. Gilles, and D. Rick et al. 1998. Phytotoxicological Evaluation of Starane 180 Emulsifiable Herbicide Concentrate (Formulation EF-1463) Exposed Aquatic Plant, Duckweed, Lemna gibba L. G-3: Lab Project Number: 981132. Unpublished study prepared by Dow Chemical Company. 75 p. MRID No. 44744001.

Knuteson, J. 1999. Review of Environmental Fate of 3,5,6-Trichloro-2-pyridinol. (TCP): Laboratory, Terrestrial and Aquatic Field Studies: Lab Project Number: GH-C 4875. Unpublished study prepared by Dow AgroSciences LLC. 75 pages. MRID 44766301. (Cited in SERA 2011c).

Kreamer, G. 1996. Early life-stage toxicity of DPX-T6376-141 (Metsulfuron Methyl) to rainbow trout, (*Oncorhynchus mykiss*): Lab Project Number: HLR 182-96: 10352: 182-96. Unpublished study prepared by DuPont Haskell Lab for Toxicology and Industrial Medicine. 400 pp. MRID No. 44122801. (Cited in SERA 2004e).

Kreutzweiser, D. P., S. B. Holmes, and D. C. Eichenberg. 1994. Influence of exposure duration on the toxicity of triclopyr ester to fish and aquatic insects. Archives of Environmental Contamination and Toxicology 26(1):124-129.

Kreutzweiser, D. P, D. G. Thompson, S. S. Capell, D. R. Thomas, and B. Staznik. 1995. Field evaluation of triclopyr ester toxicity to fish. Archives of Environmental Contamination and Toxicology 28 (1):18-26.

Krzyszowska, A. J., R. D. Allen and G. F. Vance. 1994. Assessment of the fate of two herbicides in Wyoming rangeland soil: column studies. Journal of Environmental Quality 23(5):1051-1058.

Landstein, D., S. Arad, Z. Barak, and D.M. Chipman. 1993. Relationships among the herbicide and functional sites of acetohydroxy acid synthase from *Chlorella emersonii*. Planta 191(1):1-6.

Lehmann, R. 1988. Formation of Fluroxypyr from Fluroxypyr-MHE by Soil Catalysis: Lab Project Number: GH-C 2068. Unpublished study prepared by Dow Chemical U.S.A. 26p. MRID No. 42137316.

Leitch, C. and P. Fagg. 1985. Clopyralid herbicide residues in streamwater after aerial spraying of a *Pinus radiata* plantation. New Zealand Journal of Forestry Science 15(2):195-206.

Little, E. E., R. D. Archeski, B. A. Flerov, and V.I. Kozlovskaya. 1990. Behavioral indicators of sublethal toxicity in rainbow trout. Archive of Environmental Contamination and Toxicology 19(3):380-385.

Lorz, H. W., S. W. Glenn, R. H. Williams, C. M. Kunkel, L. A. Norris, and B. R. Loper. 1979. Effects of selected herbicides on smolting of coho salmon. EPA-600/3-79-071. EPA, Environmental Research Laboratory, Corvallis, Oregon.

Machado, M. 2003. Triclopyr metabolite 3,5,6-TCP -- Full life-cycle toxicity test with water fleas, *Daphnia magna*, under static-renewal conditions: Lab Project Number: 12550.6286: 021300. Unpublished study prepared by Springborn Smithers Laboratories. 66 pages. MRID 45861301. (Cited in SERA 2011c).

Marino, T., E. McClymont, and C. Hales. 2003. Revised report for 3,5,6-Trichloro-2-Pyridinol (TCP): evaluation of the toxicity to the early life stages of the rainbow trout, *Oncorhynchus mykiss* Walbaum. Project Number: 011150, 011150R. Unpublished study prepared by The Dow Chemical Co. 69 pages. MRID 46033201. (Cited in SERA 2011c).

Martinez-Tabche, L., E. Madrigal-Bujaidar, and T. Negrete. 2004. Genotoxicity and lipoperoxidation produced by paraquat and 2,4-Dischlorophenoxyacetic acid in the gills of rainbow trout (*Oncorhynchus mikiss*). Bulletin of Environmental Contamination and Toxicology 73:146-152.

Martins, E. G., S. G. Hinch, D. A. Patterson, M. J. Hague, S. J. Cooke, K. M. Miller, M. F. Lapointe, K. K. English, and A. P. Farrell. 2011. Effects of river temperature and climate warming on stock-specific survival of adult migrating Fraser River sockeye salmon (*Oncorhynchus nerka*). Global Change Biology 17(1):99–114. DOI:10.1111/j.1365-2486.2010.02241.x.

Mayer, F. L. and M. R. Ellersieck. 1986. Manual of acute toxicity: interpretation and data base for 410 chemicals and 66 species of freshwater animals. Resource Publication Number 160. U.S. FWS, Washington, DC. 505 pages.

Mayes, M. A. 1984. The acute toxicity of picloram potassium salt and picloram triisopzopanolamine salt to aquatic organisms. Environmental Toxicology and Chemistry 3(2):263-269.

Mayes, M. 1990. Dow Chemical USA phase 3 summary of MRID 00151958. The acute and chronic toxicity of triclopyr [(3,5,6-Trichloro-2-Pyridinyl)oxy) acetic acid] triethylamine salt solution to the fathead (*Pimephales promelas* Rafinesque): minnow. Project ID: ES-582. Prepared by Dow Chemical CO. 11 pages. MRID 92189012. (Cited in SERA 2011c).

Mayes, M. A., P. G. Murphy, D. L. Hopkins, F. M. Gersich, and F. A. Blanchard. 1986. The toxicity and metabolism of triclopyr butoxyethyl ester: coho salmon. Toxicologist 6:26 (Abstract Only). (Cited in SERA 2011c).

Mayes, M. A., D. L. Hopkins, and D. C. Dill. 1987. Toxicity of picloram (4-amino-3,5,6-trichloropicolinic acid) to life stages of the rainbow trout. Bulletin of Environmental Contamination and Toxicology 38:653-600.

McAllister, W., J. Bowman, and P. Cohle. 1985a. Aucte toxicity of CN 10-6471 [(Banvel herbicide)] to bluefill sunfish. (*Lepomis macrochirus*): Report No. 33171. Unpublished study prepared by Analytical Biochemistry Laboratories, Inc. 53 pages. MRID 00153150. (cited in SERA 2004c).

McAllister W., J. Bowman, and P. Cohle. 1985b. Acute Toxicity of CN 10-6471 [(Banvel Herbicide)] to Rainbow Trout. (*Salmo gairdneri*): Report No. 33172. Unpublished study prepared by Analytical Biochemistry Laboratories, Inc. 58 pages. MRID 00153151. (cited in SERA 2004c).

McLaren/Hart Environmental Engineering Corporation. 1995. Use of the Registered Aquatic Herbicide Fluridone (Sonar) and the Use of the Registered Aquatic Herbicide Glyphosate (Rodeo and Accord) in the State of New York - Final Generic Environmental Impact Statement. (prepared for Dow- Elanco and Monsanto).

Milazzo, D. P., J. A. Miller, J. M. Hugo, and M. D. Martin. 1996a. Fluroxypyr 1-Methylheptyl Ester: The Toxicity to the Blue-Green Alga, Anabaena flos-aquae. Unpublished Study. Environmental Toxicology & Chemistry Research Laboratory, The Dow Chemical Company, Midland, MI DowElanco, Indianapolis, IN. Laboratory Report ID: ES-3073. MRID No. 44080336.

Milazzo, D. P., J. M. Hugo, and M. D. Martin. 1996b. Fluroxypyr 1-Methylheptyl Ester: The Toxicity to the Freshwater Green Alga, Selenastrum capricornutum Printz. Unpublished Study. Environmental Toxicology & Chemistry Research Laboratory, The Dow Chemical Company, Midland, MI. DowElanco, Indianapolis, IN. MRID No. 44803340.

Mitchell, D. G., P. M. Chapman, and T. J. Long. 1987. Acute toxicity of Roundup and Rodeo herbicides to rainbow trout, chinook, and coho salmon. Bulletin of Environmental Contamination and Toxicology 39(6):1028-1035.

Morgan M. J. and J. W. Kiceniuk. 1992. Response of rainbow trout to a two month exposure to vision a glyphosate herbicide. Bulletin of Environmental Contamination and Toxicology 48(5):772-780.

Morgan J. D., G. A. Vigers, A. P. Farrell, D. M. Janz, and J. F. Maville. 1991. Acute avoidance reactions and behavioral responses of juvenile rainbow trout (*Oncorhynchus mykiss*) to Garlon 4, Garlon 3A and Vision Herbicides. Environmental Toxicology and Chemistry 10(1):73-79.

Murphy, M. L. 1998. Primary production. Pages 144-168 *in:* Naiman R. J. and R. E. Bilby, editors. River ecology and management: lessons from the Pacific coastal ecoregion. Springer-Verlag, New York, NY.

Muska, C. and R. Driscoll. 1982. Early life stage toxicity of INT-5648-18 to fathead minnow. Haskell Laboratory Report No. 765-82. Unpublished study received March 28, 1983, under 352-401; submitted by Haskell Laboratory for Toxicology and Industrial Medicine, E.I. du Pont de Nemours and Co. Inc., Newark, DE. MRID 00126600. (Cited in SERA 2004f).

Muska, C. and C. Hall. 1982. 96-hour LC_{50} to rainbow trout: Laboratory Report No. 515-82. Unpublished study received Feb 14, 1983 under 352-EX-111. CDL:071434-B. Submitted by E.I. du Pont de Nemours & Co., Inc., Wilmington, Delaware. MRID No. 00125816. (Cited in SERA 2004e).

National Library of Medicine (NLM). 2012. Data Query of the Hazardous Substances Database. National Library of Medicine, Toxnet. Available: http://toxnet.nlm.nih.gov/. (May 2012).

National Marine Fisheries Service (NMFS). 2012. Endangered Species Act Section 7(a)(2) Biological Opinion and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Consultation for the Sawtooth National Forest's Programmatic Noxious and Invasive Weed Management Program; Upper Salmon River Subbasin HUC 17060201; Custer and Blaine Counties, Idaho (One Project). November 26, 2012. NMFS No.: 2012/01793.

NMFS. 2013. Reinitiation for the Biological Assessment of Effects of Treatment of Noxious Weeds in the Frank Church-River of No Return Wilderness Management – Salmon-Challis, Boise, Payette, Nez Perce and Bitterroot National Forests; Hydrologic Unit Codes: Salmon River 17060201, 17060203, 17060205, 17060206, 17060207, and 17060208, and Upper Selway River 17060301; Custer, Idaho, Lemhi, and Valley Counties, Idaho. (One Action). October 13, 2009. NMFS No.: 2004/01889; 2009/03999.

NMFS. 2015. Snake River Sockeye Recovery Plan (Oncorhynchus nerka). June 8, 2015. Available at: http://www.westcoast.fisheries.noaa.gov/publications/recovery_planning/salmon_steelhead/domains/interior_columbia/snake/snake_river_sockeye_recovery_plan_june_2015.pdf

NMFS. 2016. Draft Proposed ESA Recovery Plan for Snake River Idaho Spring/Summer Chinook Salmon and Steelhead Populations. October. West Coast Region, Portland Oregon.

NMFS. 2016b. Endangered Species Act Section 7(a)(2) Biological Opinion, and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Consultation for the Salmon-Challis National Forest Invasive Plant Treatment Program (Non-wilderness Areas), Upper Salmon River (17060201); Pahsimeroi River (17060202); Middle Salmon-Panther (17060203); Lemhi River (17060204); Upper Middle Fork Salmon (17060205); Lower Middle Fork Salmon (17060206); Middle Salmon-Chamberlain (17060207); Little Lost River (17040217); Big Lost River (17040218); Lemhi and Custer Counties, Idaho. May 17, 2016. WCR-2015-3863.

NMFS. 2016c. Endangered Species Act Section 7(a)(2) Biological Opinion, and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Response for the Proposed Poison Ivy Control at Snake River Recreation and Administrative Sites Project. July 22, 2016. NMFS No.: 2016-4766.

Neary, D. G. and J. L. Michael. 1989. Herbicides - protecting long-term sustainability and water quality in forest ecosystems. New Zealand Journal of Forestry Science 26:241-264.

Nystrom B., B. Bjornsater, and H. Blanck. 1999. Effects of sulfonylurea herbicides on non-target aquatic micro-organisms: Growth inhibition of micro-algae and short-term inhibition of adenine and thymidine incorporation in periphyton communities. Aquatic Toxicology 47(1):9-22.

Parker, R. R. 1971. Size selective predation among juvenile salmonid fishes in a British Columbia inlet. Journal of the Fisheries Research Board of Canada 28:1503-1510.

Pereira, J. L., S. C. Antunes, B. B. Castro, C. R. Marques, A. M. M. Goncalves, F. Goncalves, and R. Pereira. 2009. Toxicity evaluation of three pesticides on non-target aquatic and soil organisms: commercial formulation versus active ingredient. Ecotoxicology 18:455-463.

Perkins, M. J. 1997. Effects of two formulations of glyphosate and triclopyr on four non-target aquatic species: *Xenopus laevis*, *Myriophyllum sibiricum*, *Lemna gibba* and *Tubifex*. MS Thesis, University of Guelph, Canada. 110 pages. (Cited in SERA 2011a).

Peterson, H. G., C. Boutin, P. A. Martin, K. E. Freemark, N. J. Ruecker, and M. J. Moody. 1994. Aquatic phyto-toxicity of 23 pesticides applied at expected environmental concentrations. Aquatic Toxicology 28(3-4):275-292.

Petty, D. G., K. D. Getsinger, and K. B. Woodburn. 2003. A review of the aquatic environmental fate of triclopyr and its major metabolites. Journal of Aquatic Plant Management 41:69-75.

Pierson, K. 1991. Flow-Through 77 Day Toxicity of DPX-W4189-170 to Embryo and Larval Rainbow Trout, Oncorhynchus mykiss. Lab Project Number: 494-91: MR-4581-866. Unpublished study prepared by E.I. du Pont de Nemours and Co. MRID 41976405. (Cited in SERA 2004a).

Rashin, E. and C. Graber. 1993. Effectiveness of best management practices for aerial application of forest practices. Ecology Publication No. 93-81, Prepared for the Timber/Fish/Wildlife Cooperative Monitoring Evaluation and Research Committee. Olympia, Washington.

Rice, P. M., J. C. Toney, D. J. Bedunah, and C. E. Carlson. 1997. Plant community diversity and growth form responses to herbicide applications for control of *Centaurea maculosa*. Journal of Applied Ecology 34(6):1397-1412.

Roshon, R. D., J. H. Mccann, D. G. Thompson, and G. R. Stephenson. 1999. Effects of seven forestry management herbicides on *Myriophyllum sibiricum*, as compared with other nontarget aquatic organisms. Canadian Journal of Forest Research 29(7):1158-1169.

Sanders, H. O. 1969. Toxicity of pesticides to the crustacean *Gammarus lacustris*. USDI Technical Paper 25. Bureau of Sport Fisheries and Wildlife, U.S. FWS, Columbia, MO.

Scifres, C .J. and T.J . Allen. 1973. Dissipation of dicamba from grassland soils of Texas. Weed Science 21(5):393-396.

Simenstad, C.A., J. R. Cordell, L. Tear, L. A. Weitkamp, F. L. Paveglio, K. M. Kilbride, K. L. Fresh, and C. E. Grue. 1996. Use of Rodeo and X-77 spreader to control smooth cordgrass (*Spartina alterniflora*) in a southwestern Washington estuary: 2. Effects on benthic microflora and invertebrates. Environmental Toxicology and Chemistry 15(6):969-978.

Smith, B. C., C. A. Curran, K. W. Brown, J. L. Cabarrus, J. B. Gown, J. K. Mcintyre, E. E. Moreland, V. L. Wong, J. M. Grassley, and C. E. Grue. 2004. Toxicity of Four Surfactants to juvenile rainbow trout: Implications for use over water. Bulletin of Environmental Contamination and Toxicology 72:647-654.

Smith, E. J. 1979. 96-Hour LC50 to Rainbow Trout. Haskell Laboratory Report No. 386-79. (Unpublished study received Jun 16, 1980 under 352-105; submitted by E.I. du Pont de Nemours & Co., Wilmington, Del.; CDL:099462-E). MRID 00035259. (Cited in SERA 2004a).

Smith, C. T. and M. L. McCormack. 1988. Watershed losses of triclopyr after aerial application to release spruce-fir. Proceedings – Annual Meeting of Northeast Weed Science Society 42:104-108.

Sprecher, S. L., K. D. Getsinger, and A. B. Stewart. 1998. Selective effects of aquatic herbicides on sago pondweed. Journal of Aquatic Plant Management 36: 64-68.

Syracuse Environmental Research Associates, Inc. (SERA). 2004a. Chlorsulfuron – Human Health and Ecological Risk Assessment – Final Report. SERA TR 04-43-18-01c. Fayetteville, New York. Prepared for: USFS Forest Health Protection, Arlington, VA. Available: http://www.fs.fed.us/foresthealth/pesticide/pdfs/112104_chlorsulf.pdf. (May 2012).

SERA. 2004b. Clopyralid – Human Health and Ecological Risk Assessment - Final Report. SERA TR 04-43-17-03c. Fayetteville, New York. Prepared for: USFS Forest Health Protection, Arlington, VA. Available: www.fs.fed.us/foresthealth/pesticide/pdfs/120504_clopyralid.pdf. (May 2012).

SERA. 2004c. Dicamba – Human Health and Ecological Risk Assessment – Final Report. SERA TR 04-43-17-06d. Fayetteville, New York. Prepared for: USFS, Forest Health Protection, Arlington, VA. Available: www.fs.fed.us/foresthealth/pesticide/pdfs/112404_dicamba.pdf. (May 2012).

SERA. 2004d. Imazapic – human health and ecological risk assessment – final report. SERA TR 04-43-17-04b. Fayetteville, New York. Prepared for: USFS, Forest Health Protection, Arlington, VA. Available: www.fs.fed.us/foresthealth/pesticide/pdfs/122304_Imazapic.pdf. (May 2012).

SERA. 2004e. Metsulfuron methyl – human health and ecological risk assessment – final report. SERA TR 04-43-17-01b. Fayetteville, New York. Prepared for: USFS, Forest Health Protection, Arlington, VA. Available: www.fs.fed.us/foresthealth/pesticide/pdfs/120904_Metsulfuron.pdf. (May 2012).

SERA. 2004f. Sulfometuron methyl – human health and ecological risk assessment – final report. SERA TR 03-43-17-02c. Fayetteville, New York. Prepared for: USFS, Forest Health Protection, Arlington, VA. Available: http://www.fs.fed.us/foresthealth/pesticide/pdfs/121404_Sulfometuron.pdf. (May 2012).

SERA. 2006a. 2,4-D – human health and ecological risk assessment – final report. Fayetteville, New York. Prepared for USDA Forest Service, Forest Health Protection, Arlington, VA. Available: http://www.fs.fed.us/foresthealth/pesticide/pdfs/093006_24d.pdf. (March 2012).

SERA. 2007. Aminopyralid – human health and ecological risk assessment – final report. SERA TR 052-04 Manlius, NY. Prepared for: USDA USFS, Forest Health Protection, http://www.fs.fed.us/foresthealth/pesticide/pdfs/062807_Aminopyralid.pdf (June 2007).

SERA. 2009. Fluroxypyr - Human Health and Ecological Risk Assessment FINAL REPORT. SERA TR-052-12-03a. Prepared for: USDA USFS.

SERA. 2010. Imazamox – human health and ecological risk assessment – final report. SERA TR 052-24-02a. Manlius, NY. Prepared for: USDA USFS, Forest Health Protection, Available: http://www.fs.fed.us/foresthealth/pesticide/pdfs/052-24-02a_Imazamox.pdf (December 2010).

SERA. 2011a. Glyphosate – human health and ecological risk assessment – final report. SERA TR-052-22-03b. Manlius, New York. Prepared for: USFS, Southern Region, Atlanta, Georgia. Available: www.fs.fed.us/foresthealth/pesticide/risk.shtml. (May 2012).

SERA. 2011b. Picloram – human health and ecological risk assessment – final report. SERA TR 052-27-03a. Fayetteville, New York. Prepared for USFS, Southern Region, Atlanta, Georgia. Available: http://www.fs.fed.us/foresthealth/pesticide/pdfs/Picloram_SERA_TR-052-27-03a.pdf. (May 2012).

SERA. 2011c. Triclopyr – human health and ecological risk assessment – final report. SERA TR 052-25-03a. Fayetteville, New York. Prepared for: USFS, Southern Region, Atlanta, Georgia. Available: http://www.fs.fed.us/foresthealth/pesticide/risk.shtml. (May 2012).

SERA. 2011d. Imazapyr – human health and ecological risk assessment – final report. SERA TR 052-29-03a. Manlius, NY. Prepared for: USDA USFS, Forest Health Protection, Available: http://www.fs.fed.us/foresthealth/pesticide/pdfs/Imazapyr_TR-052-29-03a.pdf (December 2011).

Thompson, S. 1994. Sulfometuron methyl (DPX-T5648): Influence on growth and reproduction of three select algal species: Lab Project Number: AMR 2892-93: 112A-121. Unpublished study prepared by Wildlife International Ltd, Easton, MD and Haskell Laboratory for Toxicology and Industrial Medicine, E.I. du Pont de Nemours and Co, Inc., Newark, DE. 58 pages. MRID 43538502. (Cited in SERA 2004f).

Thompson, D. G., D. P. Kreutzweiser, S. S. Capell, D. R. Thomas, B. Staznik, and T. Viinikka. 1995. Journal of Environmental Toxicology and Chemistry 14(8):1307-1317.

Tierney, K. B., P. S. Ross, H. E. Jarrard, K. R. Delaney, and C. J. Kennedy. 2006. Changes in juvenile coho salmon electro-olfactogram during and after short-term exposure to current-use pesticides. Environmental Toxicology and Chemistry 25(10):2809-2817.

Tierney K. B., C. R. Singh, P. S. Ross, and C. J. Kennedy. 2007. Relating olfactory neurotoxicity to altered olfactory-mediated behaviors in rainbow trout exposed to three currently-used pesticides. Aquatic Toxicology 81(1):55-64.

Tu, M., C. Hurd, and J.M . Randall. 2001. Weed Control Methods Handbook: Tools and Techniques for Use in Natural Areas. The Nature Conservancy. Version: April 2001. Available: http://www.invasive.org/gist/products/handbook/methods-handbook.pdf. (May 2012).

U.S. Bureau of Land Management (BLM). 2014. Fluroxypyr Ecological Risk Assessment Final. March 2014. Washington D.C.

U.S. Environmental Protection Agency (EPA). 1993. Registration Eligibility Decision for Glyphosate. EPA 738-R-93-014. Office of Prevention, Pesticides, and Toxic Substances. 291 pages.

EPA. 1995. Registration Eligibility Decision for Picloram. EPA 738-R95-019. Office of Prevention, Pesticides, and Toxic Substances. 301 pages.

EPA. 1998. Registration Eligibility Decision for Trichlopyr. EPA 738-R-98-011. Office of Prevention, Pesticides, and Toxic Substances. 285 pages.

EPA. 2005a. Reregistration Eligibility Decision for 2,4-D. EPA 738-R-05-002. Office of Prevention, Pesticides, and Toxic Substances. 304 pages.

EPA. 2005b. Reregistration Eligibility Decision for Chlorsulfuron. EPA Office of Prevention, Pesticides, and Toxic Substances. 90 pages.

EPA. 2009. Risks of Triclopyr Use to Federally Threatened California Red-legged Frog (*Rana aurora draytonii*). Including Appendices A through Q. Available: http://www.epa.gov/espp/litstatus/effects/redleg-frog/. (May 2012).

U.S. Environmental Protection Agency/Office of Pesticide Programs (EPA/OPP). 1998. Review for Section 3 Registration of Fluroxypyr Acid (Chemical # 128959) and Fluroxypyr MHE (Chemical # 128968-5). Report dated June 8, 1998.

EPA/OPP. 2008a. Environmental Fate and Effects Science Chapter: Environmental Fate and Ecological Risk Assessment: Registration of New Use Imazamox for the proposed new use for the control of weed in Clearfield rice (imidazolinone-tolerant rice). Document dated 9/24/2008. USEPA PC Code: 12971. Environment Risk Branch II Team: I. Abdel-Saheb, and M. Davy, Environmental Fate and Effects Division, (Cited in SERA 2010).

EPA/OPP. 2008b. Ecological risk assessment evaluating Imazamox (PC 129171) for the proposed new use for the control of vegetation in and around aquatic and noncropland sites. DP Barcode: 339560. Document dated 2/14/2008. Copy courtesy of U.S. EPA/OPP.

U.S. Forest Service (USFS). 2000a. Glyphosate: herbicide information profile. USFS, Pacific Northwest Region. Available: www.fs.fed.us/r6/weeds/Gly-Nov-17-2000.pdf. (May 2012).

USFS. 2000b. Picloram: Herbicide Information Profile. Pacific Northwest Region Forest Service. Available at: http://www.fs.fed.us/r6/nr/fid/pubsweb/piclo.pdf. Accessed March 26, 2012.

U.S. Geological Survey (USGS). 1998. Data on pesticides in surface and ground water of the United States. Revised October 23, 1998. Results of the National Water Quality Assessment Program (NAWQA).

Vannote, R. L., G. W. Minshall, K. W. Cummins, J. R. Sedell, and C.E. Cushing. 1980. The river continuum concept. Canadian Journal of Fisheries and Aquatic Sciences 37:130-137.

Vilkas, A. 1977a. The acute toxicity of banvel technical to the bluegill sunfish. Union Carbide Environmental Services. November 21, 1977. Review. 2 pages. Accession No. 232965. (Cited in SERA 2004c).

Vilkas, A. 1977b. The acute toxicity of banvel technical to the sheepshead minnow. Union Carbide Environmental Services. December 14, 1977. Review. 1 page. Accession No. 232965. (Cited in SERA 2004c).

Waite, D. T., R. Grover, N. D. Westcott, H. Sommerstad, and L. Kerr. 1992. Pesticides in groundwater, surface water, and spring runoff in a small Saskatchewan watershed. Environmental Toxicology and Chemistry 11(6):741-748.

Wan, M. T., D. J. Moul, and R. G. Watts. 1987. Acute toxicity to juvenile Pacific salmonids of Garlon 3A, Garlon 4, triclopyr, triclopyr ester, and their transformation products 3,5,6-trichloro-2-pyridinol and 2-methoxy-3,5,6-trichloropyridine. Bulletin of Environmental Contamination and Toxicology 39(4):721-728.

Wan M. T., R. G. Watts, and D. J. Moul. 1989. Effects of different dilution water types on the acute toxicity to juvenile pacific salmonids and rainbow trout of glyphosate and its formulated products. Bulletin of Environmental Contamination and Toxicology 43(3):378-385.

Wan, M. T., R. G. Watts, and D.J . Moul. 1992. Effects of Different Dilution Water Types on the Acute Toxicity to Juvenile Pacific Salmon and Rainbow Trout of Fluroxypyr Formulated Product XRM-5084. Bull Environ Contam Toxicol. 49 (6): 914-921.

Ward, G. 1991. 2,4-D, Dimethylamine Salt: Chronic Toxicity to the Water Flea, *Daphnia magna*, Under Flow-through Test Conditions: Lab Project Number: J9002003D. Unpublished study prepared by Toxikon Environmental Sciences. 45 p. MRID 41835210. (Cited in SERA 2006a).

Ward, T. and R. Boeri. 1989. Chronic Toxicity of DPX-W4189-94 to Daphnia magna. Lab Project Number: 4581-655: 87-89. Unpublished study prepared by E.I. du Pont de Nemours and Co. 27 p. MRID 41976404. (Cited in SERA 2004a).

Ward, T. and R. Boeri. 1991. Static Acute Toxicity of DPX-W4189-165 (Chlorsulfuron) to the Mysid, Mysidopsis bahia. Lab Project Number: MR-4581-866: 523-91: 9131-DU. Unpublished study prepared by EnviroSystems Division. 31 p. MRID 41976402. (Cited in SERA 2004a).

Washington State Department of Transportation (WSDOT). 2006. WSDOT Chlorsulfuron Roadside Vegetation Management Herbicide Factsheet. Available: www.wsdot.wa.gov/maintenance/pdf/chlorsulfuron.pdf. (May 2012).

Washington State Department of Ecology and Agriculture (Washington State). 2016. Summary of Aquatic Acute Toxicity Data for Spray Adjuvants Allowed for Use on Aquatic Sites in Washington. Rev. November 23, 2016. Pesticide Management Division, Registration Services Program.

Wauchope, R. D., R. G. Williams, and L. R. Marti. 1990. Runoff of sulfometuron-methyl and cyanazine from small plots: Effects of formulation and grass cover. Journal of Environmental Quality 19(1):119-125.

Wei, L., H. Yu, Y. Sun, J. Fen, and L. Wang. 1998. The effects of three sulfonylurea herbicides and their degradation products on the green algae *Chlorella pyrenoidosa*. Chemosphere 37(4):747-751. (Cited in SERA 2004a, e).

Wei, L., H. Yu, J. Cao, Y. Sun, J. Fen, and L. Wang. 1999. Determination and prediction of partition coefficient and toxicity for sulfonylurea herbicides and their degradation products. Chemosphere 38(7):1713-1719. (Cited in SERA 2004e).

Weinberg, J., J. Hugo and M. Martin. 1994. Evaluation of the toxicity of triclopyr butoxyethyl ester to the early life stages of the rainbow trout, *Oncorhynchus mykiss* Walbaum: Lab Project Number: DR/0133/7242: DECO/ES/2699. Unpublished study prepared by The Environmental Toxicology & Chemistry Research Lab. 47 pages. MRID 43230201. (Cited in SERA 2011c).

Weis, J. S. 2014. Delayed Behavioral Effects of Early Life Toxicant Exposures in Aquatic Biota. Toxics, volume 2, pp. 165-187.

Woodburn, K. 1987. Clopyralid – The hydrolysis of clopyralid in sterile, buffered water: lab Project Number: LEH060893. (Unpublished study prepared by DowElanco). MRID No. 41971101. (Cited in SERA 2004b).

Woodward, D. F. 1976. Toxicity of the herbicides dinoseb and picloram to cutthroat (*Salmo clarki*) and lake trout (*Salvelinus namaycush*). Journal of the Fisheries Research Board of Canada 33:1671-1676.

Woodward, D. F. 1979. Assessing the hazard of picloram to cutthroat trout. Journal of Range Management 32(3):230-232.

Woodward, D. F. 1982. Acute toxicity of mixtures of range management herbicides to cutthroat trout. Journal of Range Management 35(4):539-540.

Xie L., K. Thrippleton, M. A. Irwin, G. S. Siemering, A. Mekebri, D. Crane, K. Berry, and D. Schlenk. 2005. Evaluation of estrogenic activities of aquatic herbicides and surfactants using an rainbow trout vitellogenin assay. Toxicological Sciences 87(2):391