UNITED STATES DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
NATIONAL MARINE FISHERIES SERVICE
Northwest Region
7600 Sand Point Way N.E., Bldg. 1
Seattle, WA 98115

NMFS Tracking No.:
2001/01363

August 18, 2003

Mark Madrid, Forest Supervisor
ATTN: Dave Burns, Forest Fish Biologist
Payette National Forest
P.O. Box 1026
102 West Lake Street
McCall, Idaho 83638

Re: Endangered Species Act Section 7 Formal Consultation and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Consultation for U.S. Forest Service Noxious Weed Control Program in the Salmon River Drainage (1 Project)

Dear Mr. Madrid:

Enclosed is a document containing a biological opinion (Opinion) prepared by NOAA's National Marine Fisheries Service (NOAA Fisheries) pursuant to section 7 of the Endangered Species Act (ESA) on the effects of the proposed U.S. Forest Service Noxious Weed Control Program in the Salmon River Drainage. In this Opinion, NOAA Fisheries concludes that the proposed action is not likely to jeopardize the continued existence of ESA-listed Snake River fall and spring/summer chinook salmon, Snake River steelhead, and designated critical habitat. As required by section 7 of the ESA, NOAA Fisheries includes reasonable and prudent measures with nondiscretionary terms and conditions that NOAA Fisheries believes are necessary to minimize the impact of incidental take associated with this action.

This document contains a consultation on essential fish habitat (EFH) pursuant to section 305(b) of the Magnuson-Stevens Fishery Conservation and Management Act (MSA) and its implementing regulations (50 CFR Part 600). NOAA Fisheries concludes that the proposed action may adversely affect designated EFH for chinook salmon. As required by section 305(b)(4)(A) of the MSA, included are conservation recommendations that NOAA Fisheries believes will avoid, minimize, mitigate, or otherwise offset adverse effects on EFH resulting from the proposed action. As described in the enclosed consultation, 305(b)(4)(B) of the MSA requires that a Federal action agency must provide a detailed response in writing within 30 days of receiving an EFH conservation recommendation.

If you have any questions regarding this letter, please contact Bill Lind of my staff in the Idaho Habitat Branch at 208-378-5697.

Sincerely,

Russell M. Strach for

D. Robert Lohn
Regional Administrator

Enclosure

cc: J. Foss - USFWS
J. Hansen - IDFG
R. Eichsteadt - NPT

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

U.S. Forest Service Noxious Weed Control Program
in the Salmon River Drainage

Idaho, Valley, Adams, and Custer Counties, Idaho

Agency: U.S. Forest Service, Payette National Forest

Consultation Conducted By: NOAA's National Marine Fisheries Service,
Northwest Region

Date Issued: August 18, 2003

Issued by: Russell M Strach for
D. Robert Lohn
Regional Administrator

Refer to: NMFS Tracking No.: 2001/01363

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APPENDICES

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1. INTRODUCTION

The Endangered Species Act (ESA) of 1973 (16 USC 1531-1544), as amended, establishes a national program for the conservation of threatened and endangered species of fish, wildlife, and plants and the habitat on which they depend. Section 7(a)(2) of the ESA requires Federal agencies to consult with NOAA's National Marine Fisheries Service (NOAA Fisheries) and U.S. Fish and Wildlife Service (FWS; together "Services"), as appropriate, to ensure that their actions are not likely to jeopardize the continued existence of endangered or threatened species or adversely modify or destroy their designated critical habitats. This biological opinion (Opinion) is the product of an interagency consultation pursuant to section 7(a)(2) of the ESA and implementing regulations found at 50 CFR 402.

The analysis also fulfills the essential fish habitat (EFH) requirements under the Magnuson-Stevens Fishery Conservation and Management Act (MSA). The MSA, as amended by the Sustainable Fisheries Act of 1996 (Public Law 104-267), established procedures designed to identify, conserve, and enhance EFH for those species regulated under a Federal fisheries management plan. Federal agencies must consult with NOAA Fisheries on all actions, or proposed actions, authorized, funded, or undertaken by the agency, that may adversely affect EFH (§305(b)(2)).

The United States Forest Service (USFS), Payette National Forest (PNF) proposes to implement a noxious weed control program. The purpose of the noxious weed control program is to control, contain or eliminate noxious weed invasion and infestations. The PNF is proposing the action according to its authority under: the Federal Noxious Weed Act as amended 1974; the Forest and Rangeland Renewable Resource Planning At of 1974; the Public Rangelands Improvement Act of 1978; the Carson-Foley Act of 1968; and Executive Order 13112 signed in 1999. The administrative record for this consultation is on file at the Idaho Habitat Branch office of NOAA Fisheries.

1.1 Background and Consultation History

The noxious weed control program was first proposed by the PNF as part of the programmatic Section 7 Watershed Biological Assessment (BA) consultations for Ongoing and New Actions on the PNF. The PNF submitted programmatic BAs for Section 7 Watersheds to NOAA Fisheries on June 6, 2001. NOAA Fisheries issued letters of concurrence on August 9, 2001, that covered some of the proposed actions, but did not include the noxious weed control program because of uncertainty regarding the potential for sublethal effects on ESA-listed anadromous fish species. Over the course of the last two years, the USFS (Regions 1, 4, and 6); Bureau of Land Management (BLM) (Idaho, Washington, and Oregon); NOAA Fisheries (Northwest Region); and the FWS (Region 1) drafted interim guidance on how to conduct ESA Section 7 consultations for actions involving the use of pesticides to control noxious weeds. In a letter dated July 17, 2002, the PNF concluded that the Noxious Weed Program is not likely to adversely affect Snake River fall and spring/summer chinook salmon (*Oncorhynchus tshawytscha*) and Snake River steelhead (*O. mykiss*), and requested concurrence from NOAA Fisheries. However, given the paucity of information about the fate and transport of herbicides within forest ecosystems, and the sublethal effects of herbicides, NOAA Fisheries determined that there is more than a negligible likelihood of adverse effects, and formal consultation (through this Opinion) is required.

1.2 Description of the Proposed Action

Proposed actions are defined in the Services' consultation regulations (50 CFR 402.02) as "all activities or programs of any kind authorized, funded, or carried out, in whole or in part, by Federal agencies in the United States or upon the high seas." Additionally, U.S. Code (16 U.S.C. 1855(b)(2)) further defines a Federal action as "any action authorized, funded, or undertaken or proposed to be authorized, funded, or undertaken by a Federal agency." Because the PNF proposes to carry out an activity that may affect listed species, it must consult under ESA section 7(a)(2).

The proposed noxious weed control program is an ongoing action that will be reinitiated on January 1, 2007. Consultation will be reinitiated before January 1, 2007, if any of the conditions occur as specified in Section II. A.5: Reinitiation of Consultation, and Section III.H: Supplemental Consultation.

The proposed action will occur on PNF administered lands in the following 4th field hydrologic unit code (HUC) subbasins (4th HUCs): Little Salmon River (HUC# 17060210), Lower Salmon (HUC# 17060209), Main Salmon-Chamberlain (HUC# 17060207), Lower Middle Fork Salmon (HUC# 17060206), and Upper Middle Fork Salmon (HUC# 17060205). The proposed action does not include weed treatment within the Frank Church River of No Return Wilderness. Weed control efforts could occur on rangelands, in timber harvest units, along roads and road rights-of-way, along trail routes, at dispersed recreation sites, on developed recreation sites, and at other disturbed sites (i.e. fires, flood events). Noxious weed control measures would be conducted as described in detail in the BA and amendment to the BA. Proposed treatments include: (1) physical control; (2) biological control; (3) cultural control; (4) chemical control;
(5) rehabilitation; seeding and planting; and (6) inventory, monitoring and reporting. Key elements of the proposed action are summarized below.

<u>1.1.1 Physical Control</u>

Manual control includes the use of hand-operated power tools and simple hand tools in manual vegetation treatments to cut, clear, mow, or prune herbaceous and woody species. In manual treatments, workers cut plants above ground level; pull, grub, or dig out plant root systems to prevent subsequent sprouting and regrowth; scalp at ground level or remove competing plants around desired vegetation; or place mulch around desired vegetation to limit the growth of competing vegetation. Mechanical control activities for noxious weed control include the use of wheeled tractors, crawler-type tractors, or specially designed vehicles with attached implements for mechanical vegetation treatments (e.g. plows, harrow, rangeland drill). Mechanical control activities typically would occur on old agricultural areas or livestock feeding sites with moderate slopes (less than 20 percent). All mechanical control activities would include associated rehabilitation measures which include seeding and planting of desirable native species.

1.1.2 Biological Control

Biological control treatments would include the use of insects, pathogens, or some combination of the two. Biological weed control activities include the release of insect agents which are parasitic and "host specific" to target noxious weeds. This activity includes the collection of beetles/ insects, and supplemental stocking of populations. Insect and pathogen methods generally are applied in conjunction with other control methods (i.e. herbicides).

1.1.3 Cultural Controls

Cultural control could include preventing weed introduction and/or minimizing rate of spread by requiring the following action on public lands:

- Clean all ground surface disturbing equipment moving into or out of weed infested areas before and after use;
- Use only certified, noxious weed-free grains, hay, or pellets for feeding domestic animals and wildlife; and inspect all feeding sites during and following use;
- Use only certified noxious weed-free seed, along with hay, straw, or mulch, or other vegetation material for site stability and revegetation projects;
- Use only noxious weed-free gravel and fill material from inspected sites;
- Vegetate disturbed areas as soon as practical, use temporary fencing when necessary to assure new seedling establishment; and
- Evaluate current and proposed vegetation management practices (i.e. livestock grazing, prescribed burning, and seeding), and implement practices to restore desired plant communities.

1.1.4 Chemical Control

The PNF Noxious Weed Control Program proposes the use of five herbicides: Transline, Rodeo, Escort, Tordon 22K, and Weedar 64. Table 1 lists the common name and typical application rate of each herbicide. Water is the only carrier or adjuvant proposed for use with the herbicides. Herbicide applications will be ground based and could include: hand-held spray nozzles attached to either backpacks, vehicles or ATVs; spot-gun sprayers mounted on vehicles or ATVs (slip tanks); hand-spreading granular formations; or wicking, wiping, dripping, painting, or injecting individual target weeds. Herbicide applications could occur within forested uplands, PACFISH interim Riparian Habitat Conservation Areas (RHCAs), and roadside ditches. In a Level 1 team meeting on May 1, 2002, Pete Grinde (PNF, New Meadows Ranger District, Range Conservationist), commented that most of the applications will be in roadside ditches. Most of the applications will use liquid formulations of the herbicides and will target the foliage of noxious weeds, though soils will also receive herbicides. The total number of acres that the PNF expects to treat each year are summarized in Table 2.

Table 1. Herbicides and use rates proposed by the Payette National Forest.

Trade Name	Common Name	Typical Application Rate (AE/ac)
Transline	Clopyralid	0.1-0.5 lb/ac
Tordon 22K	Picloram	0.25-1.0 lb/ac
Rodeo	Glyphosate	0.5-2.0 lb/ac
Escort	Metsulfuron Methyl	0.5-2.0 oz/ac
Weedar 64	2,4-D (amine only)	0.5-2.0 lb/ac

Table 2. Watershed summary of annual proposed herbicide treatments.

Section 7 Watersheds*	Estimate of Number of Acres Treated
MFSR & MSRSE	10 acres for the year 2002 and a maximum of 20 acres annually through 2006
MSRSW	10 acres for the year 2002 and a maximum of 20 acres annually through 2006
LSR	56 acres for the year 2002 and a maximum of 75 acres annually through 2006

* MFSR - Middle Fork Salmon River; MFSRSE - Main Salmon River Tributaries Southeast; MSRSW - Main Salmon River Southwest; and LSR - Little Salmon River.

1.1.1.1 Best Management Practices

Best Management Practices (BMPs) as outlined in the amendment to the BA are described in Table 3 and in more detail below. The use of the term water or waters refers to perennial, intermittent, and ephemeral stream channels, lakes, reservoirs, ponds, meadows, springs, seeps and bogs.

Table 3. Buffers, maximum wind speed, application methods, and herbicide restriction associated with aquatic habitats, riparian areas, and wetland resources on the PNF.

Buffer	Maximum Wind Speed	Herbicide Application Method	Herbicides Authorized
>100 feet from open water	8 mph	All ground spraying	All proposed herbicides authorized[1]
<100 feet from open water, but >15 feet from open water	5 mph	Spot spraying, wicking, dipping, painting, and injecting	All proposed herbicides authorized except Tordon
<15 feet	5 mph	Spot spraying, wicking, dipping, painting, and injecting	Only Rodeo authorized ("aquatic approved")

[1] Transline, Rodeo, Escort, Weedar, Tordon

A. Buffers

- Minimum buffer strips for streams and wet areas would exceed label direction and State-mandated standards;
- No herbicide storage, mixing or post-application cleaning would be authorized within RHCAs. Mixing and loading operations would take place in areas where an accidental spill would not contaminate a stream or body of water before it could be contained;
- Only individual target weed application (spot-spraying, wicking, wiping, dripping, painting, injecting) would be authorized within 100 feet of any live waters;
- No spraying of persistent herbicides (i.e. picloram (Tordon 22-K)) would be authorized within 100 feet of any live waters;
- Only very low risk, or “aquatic-approved” chemicals (glyphosate, or Rodeo) would be used within 15 feet of open water (see Effects section for risk levels); and
- Herbicide spraying within 100 feet of live water would not be authorized when wind speed exceeds 5 miles per hour (mph).

B. Wind Speed Restrictions and Weather Considerations

- No spraying would occur when wind velocity exceeds 5 mph within 100 feet of open water;

- No spraying would occur when wind velocity exceeds 8 mph;

- No spraying would occur if precipitation is occurring or is imminent (within 24 hours);

- No spraying would occur if air turbulence is sufficient to affect the normal spray pattern;

- No spraying would occur if snow or ice covers the target foliage;

- During application, weather conditions would be monitored hourly by trained personnel at spray sites. Additional weather monitoring would occur whenever a weather change may impact safe placement of the herbicide on the target area;

- No herbicide application by any method would be authorized if winds exceed 15 mph; and,

- Only aquatic-approved herbicide (glyphosate) would be authorized on wet soils.

C. Herbicide and Equipment Handling

- A spill cleanup kit would be available whenever herbicides are transported or stored;

- A spill contingency plan would be developed prior to all herbicide applications. Individuals involved in herbicide handling or application would be instructed on the spill contingency plan and spill control, containment, and cleanup procedures;

- Equipment would be designed with nozzles with large orifices to deliver a median droplet diameter of 200- to 800-microns. This droplet size is large enough to avoid excessive drift while providing adequate coverage of target vegetation;

- Equipment used for transportation, storage, or application of chemicals shall be maintained in a leak-proof condition;

- All vehicles carrying herbicides shall have a standard spill kit;

- Only the amount of herbicides that are planned to be used daily would be transported in vehicles; and,

- Regular testing for field calibration of equipment would take place to prevent gross application errors.

D. Specific Chemical Authorization

- No use of 2,4-D ester formulations would be authorized;

- No carrier or adjuvant other than water would be used;

- No more than one application of picloram would be made on a given site in any given year to reduce the potential for picloram accumulation in the soil; and

- Picloram would not be used within an annual floodplain where the water table is within six feet of the surface and soil permeability is rapid to very rapid throughout the soil profile (i.e. sandy soils).

E. Project Oversight for Herbicide Application

- The PNF would follow established USFS guidelines (CD submitted with original BAs:\support documents\toxicity);

- The PNF would have a certified herbicide applicator overseeing all herbicide application;

- Trained personnel would monitor weather conditions at spray sites during application;

- All herbicide labels and direction provided within this BA would be strictly enforced;

- The Weed Coordinator will map and identify buffers, methods of application, and herbicide restrictions that may be required for the project, and will make a pre-project review of all spray projects to provide to the Streamlining level one team by April 1, annually; and,

- Herbicide applications would only treat the minimum area necessary for the control of noxious weeds.

1.1.5 Rehabilitation, Seeding, Planting

Native vegetation would be planted that would compete with noxious weeds, restrict or prevent additional infestations, and help prevent soil erosion and further soil nutrient loss. These treatments may involve ground application of seeds and fertilizers. On areas with moderate slopes that have been farmed in the past, rangeland drills or plows may be used.

1.1.6 Inventory, Monitoring and Reporting

The PNF proposes to monitor the noxious weed treatments for their effectiveness on weed eradication on both a site-specific treatment level and on a landscape level. Site-specific monitoring will include checking sites for treatment effects to both target and non-target species. Landscape level monitoring involves tracking all noxious weed sites in the Forest Service (FS) Geographic Information System. No monitoring is proposed for the effects of the proposed action on ESA-listed fish, due to uncertainty over how to monitor, availability of funding for monitoring, and how to interpret the results. The PNF proposes to further investigate the effectiveness and practicality of conducting some level of water quality monitoring to detect levels of herbicides and possibly to establish the effectiveness of the drift buffers.

The noxious weed control program is a long-term endeavor to control weeds where and when practical. However, because there are areas of scientific and management uncertainty about the effectiveness of weed control treatments, the proposed action may be refined over time to meet the basic objective of systematically reducing noxious weed abundance, and their extent and spread throughout the action area watersheds. The proposed action will be reevaluated on a 5-year cycle (life of the BA) or if consultation is reinitiated. Information from weed inventories and results of treatments will be mapped spatially, used to assess the noxious weed program objectives, and developed into a baseline for future ESA/EFH consultations.

Project proposals (with methods, objectives of treatment, location, map of treatment area, acreage, proposed dates to be started and completed, sensitive areas, and special mitigation) for noxious weed control activities involving herbicides would be prepared annually by Weed Coordinators and submitted by April 1, for review by PNF biologists. Project proposals would be reviewed for compliance with the BA and this Opinion. The PNF biologists (Level 1) would provide a list of project descriptions and maps annually (or as identified) for informal review and approval by NOAA Fisheries and FWS Level 1 team members before the projects are implemented. All projects would be reviewed and approved by NOAA Fisheries and FWS before herbicide application occurs.

Annually, a project summary of treatments would be prepared for land treatments that took place during the past year. The report would document treatments that took place, methods used, location, map, acreage, evaluation of achievement of objectives, brief summary of environmental effects, and evaluation of compliance with the BA. This summary report would be completed by April 1, annually.

Based on annual treatment evaluations and with the likely development of new control methods and technology, changes in existing or use of new noxious weed treatments may be authorized and warranted. Any changes to the proposed action, as described in the BA, would be analyzed for impacts to listed/proposed species and critical habitat, and consultation would be reinitiated as appropriate (see section II.F: Reinitiation of Consultation).

1.3 Description of the Action Area

An action area is defined by NOAA Fisheries regulations (50 CFR Part 402) as "all areas to be affected directly or indirectly by the Federal action and not merely the immediate area involved in the action." The action area affected by the proposed action includes PNF administered lands and areas immediately downstream within the following 4^{th} HUCs: Lower Salmon (17060209), Little Salmon (17060210), Middle Salmon-Chamberlain (17060207), Lower Middle Fork Salmon (17060206), and Upper Middle Fork Salmon (17060205). The proposed action occurs within designated critical habitat for ESA-listed Snake River spring/summer chinook salmon, including all river reaches presently or historically accessible to the species within all of the above listed 4^{th} HUCs. Critical habitat for Snake River fall chinook salmon within the action area includes all river reaches presently or historically accessible to the species within the Lower Salmon 4^{th} HUC (17060209). This area serves as a migratory corridor, spawning, and rearing habitat for the salmonid Evolutionarily Significant Units (ESUs) listed below in Table 4.

2. ENDANGERED SPECIES ACT - BIOLOGICAL OPINION

The objective of this Opinion is to determine whether the PNFs' Noxious Weed Control Program is likely to jeopardize the continued existence of the Snake River fall and spring/summer chinook salmon and Snake River steelhead or destroy or adversely modify designated critical habitat for Snake River fall and spring/summer chinook salmon.

2.1 Evaluating the Proposed Action

The standards for determining jeopardy and destruction or adverse modification of critical habitat are set forth in section 7(a)(2) of the ESA. In conducting analyses of habitat-altering actions under section 7 of the ESA, NOAA Fisheries uses the following steps of the consultation regulations and when appropriate combine them with the Habitat Approach[1] (NMFS 1999): (1) Consider the biological requirements and status of the listed species; (2) evaluate the relevance of the environmental baseline in the action area to the species' current status; (3) determine the effects of the proposed or continuing action on the species, and whether the action is consistent with any available recovery strategy; and (4) determine whether the species can be expected to survive with an adequate potential for recovery under the effects of the proposed or continuing action, the effects of the environmental baseline, and any cumulative effects, and considering measures for survival and recovery specific to other life stages. In completing this step of the analysis, NOAA Fisheries determines whether the action under consultation, together with all cumulative effects when added to the environmental baseline, is likely to jeopardize the ESA-listed species or result in the destruction or adverse modification of critical habitat. If jeopardy or adverse modification are found, NOAA Fisheries may identify reasonable and prudent alternatives for the action that avoid jeopardy and/or destruction or adverse modification of critical habitat.

The fourth step above (jeopardy/adverse modification analysis) requires a two-part analysis. The first part focuses on the action area and defines the proposed action's effects in terms of the species' biological requirements in that area (i.e., effects on essential features). The second part focuses on the species itself. It describes the action's effects on individual fish, populations, or both, and places that impact in the context of the ESU as a whole. Ultimately, the analysis seeks to determine whether the proposed action is likely to jeopardize a listed species' continued existence or destroy or adversely modify its critical habitat.

2.1.1 Biological Requirements in the Action Area

The first step NOAA Fisheries uses when applying ESA section 7(a)(2) to the listed ESUs considered in this Opinion includes defining the species' biological requirements within the action area. Biological requirements are population characteristics necessary for the listed ESUs to survive and recover to naturally reproducing population sizes at which protection under the ESA would become unnecessary. The listed species' biological requirements may be described as characteristics of the habitat, population or both (McElhany *et al*. 2000). See Table 4 for a listing of the interim recovery targets established by NOAA Fisheries for ESA-listed fish species potentially affected by the proposed action (NMFS 2002). Interim recovery targets are also available at the following website: <*http://www.nwr.noaa.gov/1habcon/habweb/habguide/appendix_b.pdf*>

[1] The Habitat Approach is intended to provide guidance to NOAA Fisheries staff for conducting analyses, and to explain the analytical process to interested readers. As appropriate, the Habitat Approach may be integrated into the body of Opinions. NOAA Fisheries staff are encouraged to share the Habitat Approach document with colleagues from other agencies and private entities who are interested in the premises and analysis methods.

Table 4. Interim Recovery Targets Established for ESA-listed Fish Species under NOAA Fisheries Jurisdiction (NMFS 2002).

ESU/Spawning Aggregation	Interim Recovery Target
Snake River Spring/summer Chinook Salmon	41,900
Snake River Fall Chinook Salmon	2,500
Snake River Steelhead	53,700

For actions that affect freshwater habitat, NOAA Fisheries may describe the habitat portion of a species' biological requirements in terms of a concept called properly functioning condition (PFC). The PFC is defined as the sustained presence of natural[2] habitat-forming processes in a watershed that are necessary for the long-term survival of the species through the full range of environmental variation (NMFS 1999). The PFC, then, constitutes the habitat component of a species' biological requirements. Although NOAA Fisheries is not required to use a particular procedure to describe biological requirements, it typically considers the status of habitat variables in a matrix of pathways and indicators (MPI) (NMFS [1996] Table 1) that were developed to describe PFC in forested montane watersheds. In the PFC framework, baseline environmental conditions are described as "properly functioning," "at risk," or "not properly functioning."

The Federal action would occur within designated critical habitat delineated for these chinook salmon ESUs. Freshwater critical habitat can include all waterways, substrates, and adjacent riparian areas[3] below longstanding, natural impassable barriers (i.e., natural waterfalls in existence for at least several hundred years) and dams that block access to former habitat (see citations in Table 5).

Essential features of critical habitat for the listed species are: (1) Substrate, (2) water quality, (3) water quantity, (4) water temperature, (5) water velocity, (6) cover/shelter, (7) food (juvenile only), (8) riparian vegetation, (9) space, and (10) safe passage conditions. All of these essential features of critical habitat are included in a NMFS (1996) analysis framework called *Making Endangered Species Act Determinations of Effect for Individual or Grouped Actions at the Watershed Scale*. The PNF used this Matrix to evaluate the environmental baseline condition, and effects of the action on essential habitat features for affected ESA-listed fish species.

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[2]The word "natural" in this definition is not intended to imply "pristine," nor does the best available science lead us to believe that only pristine wilderness will support salmon.

[3]Riparian areas adjacent to a stream provide the following functions: shade, sediment delivery/filtering, nutrient or chemical regulation, streambank stability, and input of large woody debris and fine organic matter.

2.1.2 Status and Generalized Life History of the Listed Species

In this step, NOAA Fisheries also considers the current status of the listed species within the action area, taking into account population size, trends, distribution, and genetic diversity. To assess the current status of the listed species, NOAA Fisheries starts with the determinations made in its decision to list the species and also considers any new data that is relevant to the species' status. Please refer to Appendix A of the following website for the general life history of the listed species: <http://www.nwr.noaa.gov/1habcon/habweb/habguide/appendix_a_june2001.pdf>

The Federal action has been found, by NOAA Fisheries, likely to adversely affect the ESA-listed species and designated critical habitat identified below in Table 5. Based on the life histories of these ESUs, it is likely that incubating egg, juvenile, smolt, and adult life stages of each of these listed species would be adversely affected by the Federal action.

Table 5. References for Additional Background on Listing Status, Protective Regulations, and Life History for the ESA-Listed Species Considered in this Consultation.

Species ESU	Status	Critical Habitat Designation	Protective Regulations	Life History
Chinook salmon (*O. tshawytscha*)				
Snake River fall	Threatened; April 22, 1992; 57 FR 14653[4]	December 28, 1993; 58 FR 68543	July 10, 2000; 65 FR 42422	Waples *et al.* 1991b; Healey 1991
Snake River spring/summer	Threatened; April 22, 1992; 57 FR 14653	October 25, 1999, 64 FR 57399[5]	July 10, 2000; 65 FR 42422	Matthews and Waples 1991; Healey 1991
Steelhead (*O. mykiss*)				
Snake River Basin	Threatened; August 18, 1997; 62 FR 43937	N/A	July 10, 2000; 65 FR 42422	Busby et al. 1996

4 Also see, June 3, 1992, 57 FR 23458, correcting the original listing decision of by refining ESU ranges.

5 This corrects the original designation of December 28, 1993, 58 FR 68543 by excluding areas above Napias Creek Falls, a naturally impassable barrier.

2.1.2.1 Snake River Fall Chinook Salmon

The Snake River fall chinook salmon ESU, listed as threatened on April 22, 1992, (67 FR 14653), includes all natural populations of fall chinook salmon in the mainstem Snake River below Hell's Canyon Dam, and the Tucannon, Grande Ronde, Imnaha, Salmon, and Clearwater Rivers. Fall chinook from the Lyons Ferry Hatchery are included in the ESU but are not listed. Critical habitat was designated for Snake River fall chinook salmon on December 28, 1993, (58 FR 68543).

The historic distribution of fall chinook salmon is limited on the PNF, occurring only in large mainstem rivers and tributaries to the Snake and Salmon Rivers. The current distribution of fall chinook salmon potentially affected by the proposed action is located along the lower/middle main Salmon River, from the mouth upstream to approximately its confluence with French Creek. Counts of returning wild fall chinook salmon at Lower Granite Dam from 1975 through 1980 averaged 600 fish per year (Waples et al. 1991). From 1985 to 1999 an average of 459 naturally produced fall chinook salmon reached Lower Granite Dam (USDI BLM 2000). In recent years, two fall chinook satellite hatchery facilities have been operated on the Snake River to increase the numbers of fall chinook salmon. The facilities are used to acclimate and release one-year smolts from Lyons Ferry hatchery.

The Snake River component of the fall chinook run has been increasing during the past few years as a result of the hatchery and supplementation efforts in the Snake and Clearwater River basins. Greater than 15,000 adult fall chinook were counted past the two lower projects with about 12,400 counted above Lower Granite Dam. These adult returns are about triple the 10-year average at these Snake River projects (Fish Passage Center 2002). Detailed information on the current range-wide status of Snake River chinook salmon under the environmental baseline, is described in the chinook salmon status review (Myers et al. 1998).

NOAA Fisheries estimates that the median population growth rate (lambda) for the Snake River fall chinook ESU as a whole, from 1980-1997, ranges from 0.94, assuming no reproduction by hatchery fish in the wild, to 0.86, assuming that hatchery fish reproduce in the river at the same rate as wild fish (Tables B-2a and B-2b in McClure et al. 2000). The proportion of hatchery fish in the Snake River fall chinook population has been increasing with time; consequently, growth rates for the wild fall chinook population are overestimated unless corrected for hatchery influence. The degree of hatchery influence is unknown. NOAA Fisheries estimated the risk of absolute extinction considering a range of assumptions about the relative effectiveness of hatchery fish. At the low end, assuming that hatchery fish spawning in the wild have not reproduced (i.e., hatchery effectiveness = 0), the risk of absolute extinction within 100 years is 0.40 for Snake River chinook (Table B-5 in McClure et al. 2000). At the high end, assuming that the hatchery fish spawning in the wild have been as productive as wild-origin fish (hatchery effectiveness = 100%), the risk of absolute extinction within 100 years is 1.00 (Table B-6 in McClure et al. 2000).

2.1.2.2 Snake River Spring/summer Chinook Salmon

The Snake River spring/summer chinook salmon ESU, listed as threatened on April 22, 1992, (67 FR 14653), includes all natural-origin populations in the Tucannon, Grande Ronde, Imnaha, and Salmon Rivers. Some or all of the fish returning to several hatchery programs are also listed, including those returning to the Tucannon River, Imnaha, and Grande Ronde hatcheries, and to the Sawtooth, Pahsimeroi, and McCall hatcheries on the Salmon River.

Historically, the Snake River drainage is thought to have produced more than 1.5 million adult spring/summer chinook salmon in some years during the late 1800s (Matthews and Waples 1991). By the 1950s the abundance of spring/summer chinook had declined to an annual average of 125,000 adults. Adult returns counted at Lower Granite Dam reached all-time lows in the mid-1990s (<8,000 adult returns), and numbers have begun to increase since 1997. Habitat degradation is common in the range of this ESU. Spawning and rearing habitats are likely impaired by factors such as tilling, water withdrawals, timber harvest, grazing, mining, and alteration of floodplains and riparian vegetation. Mainstem Columbia River and Snake River hydroelectric developments have altered flow regimes and estuarine habitat, and disrupted migration corridors. Competition between natural indigenous stocks of spring/summer chinook salmon and spring/summer chinook salmon of hatchery origin has likely increased due to an increasing proportion of naturally-reproducing fish of hatchery origin.

Snake River wild spring/summer chinook salmon runs, as counted at the Lower Granite dam, have dwindled from an average of about 60,000 adults in the early to mid-1960s to a few thousand in recent years. Over the last 10 years (1992 to 2001), which includes the year of listing (1992), returns of wild/natural fish ranged from 183 in 1994, to 12,475 in 2001 and averaged 3,314. The estimated smolt production capacity of 10 million smolts for rivers in Idaho, coupled with historic smolt-to-adult return rates of two percent to six percent, indicate Idaho could produce wild/natural runs of 200,000 to 600,000 adults (Fish Passage Center 2002). The recent low numbers are reflected throughout the entire distribution of the chinook salmon subpopulations scattered throughout the Grande Ronde, Imnaha, and Salmon River Basins. Approximately five percent of upper Salmon River juvenile fish passing Lower Granite Dam eventually return to the upper Salmon River as adults (Bjornn et al 1996).

Even though in 2001 and 2002 there were record returns (hatchery and natural origin combined), natural origin fish numbers are in general very low in comparison to historic levels (Bevan et al 1994). Average returns of adult Snake River spring/summer chinook salmon (averaging 3,314 over the last 10 years) are also low in comparison to interim target species recovery levels of 41,900 for the Snake River Basin (NMFS 2002). The low returns amplify the importance that a high level of protection be afforded to each adult chinook salmon, particularly because a very small percentage of salmon survive to the life stage of a returning, spawning adult, and because these fish are in the final stage of realizing their reproductive potential (approximately 2,000 to 4,000 progeny per spawning pair of adults).

NOAA Fisheries estimates that the median population growth rate (lambda) for the Snake River spring/summer chinook ESU as a whole, from 1980-1997, ranges from 0.96, assuming no reproduction by hatchery fish in the wild, to 0.80, assuming that hatchery fish reproduce in the river at the same rate as wild fish (Tables B-2a and B-2b in McClure et al. 2000). The proportion of hatchery fish in the Snake River spring/summer chinook population has been increasing with time; consequently, growth rates for the wild spring/summer chinook population are overestimated unless corrected for hatchery influence. The degree of hatchery influence is unknown. NOAA Fisheries estimated the risk of absolute extinction considering a range of assumptions about the relative effectiveness of hatchery fish. At the low end, assuming that hatchery fish spawning in the wild have not reproduced (i.e., hatchery effectiveness = 0), the risk of absolute extinction within 100 years is 0.40 for Snake River chinook (Table B-5 in McClure et al. 2000). At the high end, assuming that the hatchery fish spawning in the wild have been as productive as wild-origin fish (hatchery effectiveness = 100%), the risk of absolute extinction within 100 years is 1.00 (Table B-6 in McClure et al. 2000).

2.1.2.3 Snake River Steelhead

The Snake River steelhead ESU, listed as threatened on August 18, 1997, (62 FR 43937), includes all natural-origin populations of steelhead in the Snake River basin of Southeast Washington, northeast Oregon, and Idaho. None of the hatchery stocks in the Snake River basin are listed, but several are included in the ESU. Critical habitat for Snake River steelhead was administratively withdrawn on April 30, 2002, therefore critical habitat is not designated at this time.

Natural runs of Snake River steelhead have been declining in abundance over the past decades. Some of the significant factors in the declining populations are mortality associated with the many dams along the Columbia and Snake Rivers, losses from harvest, loss of access to more than 50% of their historic range, and degradation of habitat used for spawning and rearing. Possible genetic introgression from hatchery stocks is another threat to Snake River steelhead since wild fish comprise such a small proportion of the population. Additional information on the biology, status, and habitat elements for Snake River steelhead are described in Busby et al. (1996).

The 2000 and 2001 counts at Lower Granite Dam indicate a two-year increase in returning adult spawners. Adult returns (hatchery and wild) in 2001 were the highest in 25 years and 2000 counts were the sixth highest on record (Fish Passage Center 2001a). Increased levels of adult returns are likely a result of favorable ocean and instream flow conditions for these cohorts. Although steelhead numbers have dramatically increased, wild steelhead comprise only 10% to 20% of the total returns since 1994. Consequently, the large increase in fish numbers does not reflect a change in steelhead status based on historic levels. Recent increases in the population are not expected to continue, and the long-term trend for this species indicates a decline.

Survival of downstream migrants in 2001 was the lowest since 1993. Low survival was due to record low water run-off, and elimination of spills from the Snake River dams to meet hydropower demands (Fish Passage Center 2001b). Average downstream travel times for steelhead nearly doubled and were among the highest observed since recording began in 1996. Consequently, wide fluctuations in population numbers are expected over the next few years when adults from recent cohorts return to spawning areas. Detailed information on the current range-wide status of Snake River steelhead, under the environmental baseline, is described in the steelhead status review (Busby et al. 1996), status review update (BRT 1997), and the draft Clearwater Subbasin Summary (CBFWA 2001).

NOAA Fisheries estimates that the median population growth rate (lambda) for the Snake River steelhead ESU as a whole, from 1980-1997, ranges from 0.91, assuming no reproduction by hatchery fish in the wild, to 0.70, assuming that hatchery fish reproduce in the river at the same rate as wild fish (Tables B-2a and B-2b in McClure et al. 2000). The proportion of hatchery fish in the Snake River steelhead population has been increasing with time; consequently, growth rates for the wild steelhead population are overestimated unless corrected for hatchery influence. The degree of hatchery influence is unknown. NOAA Fisheries estimated the risk of absolute extinction for the A and B runs, considering a range of assumptions about the relative effectiveness of hatchery fish. At the low end, assuming that hatchery fish spawning in the wild have not reproduced (i.e., hatchery effectiveness = 0), the risk of absolute extinction within 100 years is 0.01 for A run steelhead and 0.93 for B run fish (Table B-5 in McClure et al. 2000). At the high end, assuming that the hatchery fish spawning in the wild have been as productive as wild-origin fish (hatchery effectiveness = 100%), the risk of absolute extinction within 100 years is 1.00 for both runs (Table B-6 in McClure et al. 2000).

2.1.3 Environmental Baseline in the Action Area

The environmental baseline is defined as: "the past and present impacts of all Federal, state, or private actions and other human activities in the action area, including the anticipated impacts of all proposed Federal projects in the action area that have undergone section 7 consultation and the impacts of state and private actions that are contemporaneous with the consultation in progress" (50 CFR 402.02). In step 2, NOAA Fisheries' evaluates the relevance of the environmental baseline in the action area to the species' current status. In describing the environmental baseline, NOAA Fisheries evaluates essential habitat features of designated critical habitat and the listed Pacific salmonid ESUs affected by the proposed action.

In general, the environment for listed species in the Columbia River Basin (CRB), including those that migrate past or spawn upstream from the action area, has been dramatically affected by the development and operation of the Federal Columbia River Power System. Storage dams have eliminated mainstem spawning and rearing habitat, and have altered the natural flow regime of the Snake and Columbia Rivers, decreasing spring and summer flows, increasing fall and winter flow, and altering natural thermal patterns. Power operations cause fluctuation in flow levels and river elevations, affecting fish movement through reservoirs, disturbing riparian areas and possibly stranding fish in shallow areas as flows recede. The eight dams in the migration corridor of the Snake and Columbia Rivers kill or injure a portion of the smolts passing through the area. The low velocity movement of water through the reservoirs behind the dams slows the smolts' journey to the ocean and enhances the survival of predatory fish (Independent Scientific Group 1996; National Research Council 1996). Formerly complex mainstem habitats in the Columbia and Snake Rivers have been reduced, for the most part, to single channels, with floodplains reduced in size, and off-channel habitats eliminated or disconnected from the main channel (Sedell and Froggatt 1984; Independent Scientific Group 1996; and Coutant 1999). The amount of large woody debris in these rivers has declined, reducing habitat complexity and altering the rivers' food webs (Maser and Sedell 1994).

Other human activities that have degraded aquatic habitats or affected native fish populations in the CRB include stream channelization, elimination of wetlands, construction of flood control dams and levees, construction of roads (many with impassable culverts), timber harvest, splash dams, mining, water withdrawals, unscreened water diversions, agriculture, livestock grazing, urbanization, outdoor recreation, fire exclusion/suppression, artificial fish propagation, fish harvest, and introduction of non-native species (Henjum *et al.* 1994; Rhodes *et al.* 1994; National Research Council 1996; Spence *et al.* 1996; and Lee *et al.* 1997). In many watersheds, land management and development activities have: (1) reduced connectivity (i.e., the flow of energy, organisms, and materials) between streams, riparian areas, floodplains, and uplands; (2) elevated fine sediment yields, degrading spawning and rearing habitat; (3) reduced large woody material that traps sediment, stabilizes streambanks, and helps form pools; (4) reduced vegetative canopy that minimizes solar heating of streams; (5) caused streams to become straighter, wider, and shallower, thereby reducing rearing habitat and increasing water temperature fluctuations; (6) altered peak flow volume and timing, leading to channel changes and potentially altering fish migration behavior; and (7) altered floodplain function, water tables and base flows (Henjum *et al.* 1994; McIntosh *et al.* 1994; Rhodes *et al.* 1994; Wissmar *et al.* 1994; National Research Council 1996; Spence *et al.* 1996; and Lee *et al.* 1997).

Stream channelization, construction of flood control dams and levees, construction of roads (many with impassable culverts), timber harvest, mining, water withdrawals, unscreened water diversions, agriculture, livestock grazing, outdoor recreation, fire exclusion, artificial fish propagation, fish harvest, and introduction of non-native species have adversely affected listed species or their habitat in much of the Snake River basin.

To address problems inhibiting salmonid recovery in CRB tributaries, the Federal resource and land management agencies developed the *All H Strategy* (Federal Caucus 2000). Components of the *All H Strategy* commit these agencies to increased coordination and a fast start on protecting and restoring habitat for salmon and steelhead.

2.1.3.1 Lower Salmon River Subbasin

The Lower Salmon River Subbasin includes the Salmon River from its mouth upstream to its confluence with French Creek (Rivermile [RM] 104.8). This reach of the Salmon River is characterized by a steep rocky canyon where the channel alternates between large pools and boulder dominated rapids with a gradient of approximately 0.2%. Mean annual discharge for the Salmon River is estimated at 11,210 cubic feet per second (cfs) at the U.S. Geological Survey (USGS) White Bird gage station (RM 53.7; 0.1 miles upriver from White Bird Creek). The period of record is 1910 to 1999 (ongoing). The lowest flow recorded was 1,000 cfs (January 4, 1995) and the maximum flow recorded was 130,000 cfs (June 17, 1974). Minimum mean monthly flow was estimated at 4,242 cfs in January, and maximum mean monthly flow is 38,650 cfs in June. The average high mean daily flow during spring runoff is 44,800 cfs, and the average low mean daily flow in late summer is 4,340 cfs.

The Salmon River flows into the Snake River at RM 188.2. The subbasin includes a total of 793,600 acres, and the PNF manages approximately eleven percent (87,926 acres) of the area. Private lands comprise the majority of the subbasin, followed by USFS, BLM, and State Lands. Elevations within the subbasin range from 916 feet at the mouth to over 8,000 feet. Private land uses include livestock grazing, timber harvest, recreation, agriculture, communities, and residences. Historically, mining was a major land use along the Salmon River and in the Florence area. Public lands are limited to blocks of USFS lands in the mid and upper portions of the watersheds from White Bird up the Salmon River. The PNF lands cover the northern tip of the subbasin. Land uses on public lands include timber harvest, livestock grazing, roads, mining, and recreation.

The canyon grasslands are primarily a broad extension of the Pacific bunchgrass formation. The dominant habitat types are bluebunch wheatgrass and Idaho fescue. Sand dropseed and red three-awn have become disclimax species on some river benches, bars, and toeslope areas. Yellow starthistle and annual grasses (i.e. cheatgrass) are common invaders of canyon grasslands when they are in poor to fair ecological condition. In locations where a suitable seed source has been established, yellow starthistle is also invading canyon grasslands and displacing native vegetation that is presently in good ecological condition.

The mainstem Salmon River is used primarily by ESA-listed salmonids as an upstream and downstream passage corridor. Fall chinook salmon use the mainstem Salmon River for spawning and rearing; although such use is at very low levels. Spring/summer chinook salmon and steelhead use the mainstem Salmon River to a limited extent for rearing. Steelhead will use accessible tributaries for spawning and rearing. Spring/summer chinook salmon use White Bird Creek and Slate Creek for spawning and rearing, and also use the mouth area or lower reaches of accessible tributaries for juvenile rearing.

Overall, the water quality of the Salmon River is generally good. However, summer water temperatures in portions of the subbasin are elevated above those that might naturally occur, and sometimes well above the lethal limit for salmon and steelhead. Temperatures recorded at the USGS White Bird stream gauge in the Salmon River ranged from 16.5°C to 28.0°C during July, from 1976 to 1991. A combination of erodible soils, natural fires, periodic intense climatic events, and development of road systems have resulted in substantial natural and unnatural erosion and delivery of sediment to the Salmon River. Suspended sediment concentrations and turbidity in the river also become high enough that visibility in the water column is only a few inches. These conditions most often occur in the spring and summer from rainfall events and runoff from snow melt. Such occurrences can last several weeks. Sediment delivery events also occur as a result of summer rainstorms and may last over a week. Suspended sediment concentrations in the Salmon River near White Bird Creek ranged from 2 mg/L to 65 mg/L from 1976-91, except in May, when suspended sediment concentrations ranged from 6 mg/ to 503 mg/L. A suspended sediment concentration of 25 mg/L was suggested as a standard which would provide high protection to aquatic organisms (Thurston et al., 1979). Most of the time the Salmon River would meet this standard, however, during spring run-off, rain on snow events, or intense summer rainstorms, suspended sediment concentration can significantly exceed this amount.

The river has tremendous capability to transport sediment ranging in size from sand to large cobbles. A general observation of the river bed does not indicate that deposition of fine sediment is a serious problem. The riverbed appears to be largely composed of cobble and boulder material which would seem to offer abundant cover for salmonids. Although interstitial deposition of fines is evident, certain habitats such as pool tailouts, appear to be relatively free of fine sediment deposition. During a 1993 survey at RM 65.7, the BLM estimated cobble embeddedness in the Salmon River to be 26.3% and surface fines (particles size less than 6.3mm). to be 4.4%. This indicates low to moderate impacts to rearing habitat. During a 1994 survey at RM 90.8, the BLM estimated cobble embeddedness to be 39.5% and found spawning gravels to contain 19.5% fines.

Stream channels are highly variable throughout the subbasin. Headwater streams, breakland streams, and smaller tributary streams are predominately steep-gradient, confined channels, with high sediment transport capacity. These steep gradient streams may be subject to frequent scouring events. The larger tributaries are typically moderate gradient and are moderately confined. These channels are also efficient at sediment transport. The upper reaches of some streams flowing through low gradient prairie areas, meadows, or forest stringer meadows generally have Rosgen C and B channel types (Rosgen 1996).

2.1.3.2 Little Salmon River Subbasin

Ths Little Salmon River, approximately 43 miles long, enters the main Salmon at RM 82. The Little Salmon River subbasin includes a total of 372,500 acres, and the PNF manages approximately eleven percent (40,975 acres) of the area. The USFS lands (Payette and Nez Perce National Forests) comprise the majority of the subbasin, followed by Private, BLM, and State

Lands. The upper half of the watershed occurs in a wide valley surrounded by forested mountain slopes. The valley is characterized as pasture and meadowlands, with the Little Salmon River meandering through the valley. Near the mid-section of the watershed, the valley narrows (RM 21.5), and the river flows through a narrow, steep canyon to its mouth. The mean subbasin elevation is 5,430 feet, with elevations ranging from 1,760 to 9,393 feet. Annual precipitation ranges from less than 20 inches at Riggins, Idaho, to over 50 inches near Brundage Mountain.

The riparian vegetation along the Little Salmon River is generally dominated by black cottonwood, willows, red-osier dogwood, syringa, horsetail, black hawthorn, and alder. Along the lower reaches of the Little Salmon River, willow, Douglas hackberry, and poison ivy are common. The meadow riparian areas associated with the upper valley are commonly dominated by willows and sedges. Many of the tributary streams have a narrow riparian vegetation zone confined by steep canyon walls. Common riparian species include red-osier dogwood, syringa, willows, alder, water birch, and blue elderberry. It is often common for conifer species to also occur in the riparian areas. The higher elevation riparian areas may have grand fir, Englemann spruce , subalpine fir, and lodgepole pine, while the lower elevation riparian areas may have Douglas fir and grand fir.

Uplands are characterized by grasslands on dry, south-facing slopes, sometimes with scattered conifers and shrubs. North-facing slopes are vegetated with conifers and tall shrubs. Localized, steep rocky areas exist with low vegetation production, while other areas are heavily forested. Mid to upper elevation areas are dominated by grand fir, Douglas fir, larch, Engelmann spruce, and subalpine fir/lodgepole pine. Whitebark pine is found in localized areas at higher elevations. Common understory shrub species include ninebark, oceanspray, serviceberry, spiraea, snowberry, grouseberry, and big huckleberry.

A large variety of past and present land uses have impacted listed species habitat to varying levels. Human activities in the subbasin include logging, roads, trails, water withdrawal, agriculture, livestock grazing, residences, communities, and recreation. The higher elevation lands administered by the FS have been used for timber harvest, livestock grazing, and recreation. The BLM lands within the subbasin have also been used similar to the FS, primarily for timber harvest and livestock grazing. Major subwatersheds in the Little Salmon River include Rapid River, Elk Creek, Boulder Creek, Hazard Creek, Hard Creek, Round Valley Creek, and Goose Creek. U.S. Highway 95 parallels the Little Salmon River, and encroaches on riparian areas and floodplains in the lower canyon reach. Several small towns occur in the subbasin, ranging in size from a few hundred people to slightly more than one thousand. The predominant uses on PNF lands include roads and timber harvest.

The Little Salmon River drainage (below RM 24.0) provides habitat for listed spring/summer chinook salmon and steelhead, located primarily in the Rapid River, Boulder Creek, Hazard Creek, and Hard Creek watersheds. Rapid River is considered a stronghold for spring/summer chinook salmon and steelhead. The most significant chinook salmon and steelhead spawning and rearing areas are found in Rapid River and Boulder Creek drainages. To a lesser extent, chinook salmon and steelhead spawning and rearing also occurs in Hazard Creek, Hard Creek, and the mainstem Little Salmon River. All tributary streams that are accessible, below RM 24.0, are used for steelhead spawning and rearing. Adult steelhead have been observed in Squaw Creek, Sheep Creek, Denny Creek, Hat Creek, Lockwood Creek, Rattlesnake Creek, Elk Creek, and Trail Creek. These small steep gradient tributaries provide limited production. The mouth areas of these streams or lower reach segments (downstream from barriers) may provide rearing habitat for juvenile chinook salmon, but the value of these small tributaries for rearing is limited.

2.1.3.3 Middle Salmon-Chamberlain Subbasin

The Middle Salmon-Chamberlain Creek subbasin is located in central Idaho and includes the main Salmon River from French Creek at RM 104.8 up to the confluence with the Middle Fork Salmon River RM 191. The Middle Salmon-Chamberlain subbasin includes a total of 1,088,000 acres, and the PNF manages approximately forty percent (435,200 acres) of the area. The subbasin is 98% public lands managed by the USFS (Payette, Nez Perce, Bitterrroot, and Salmon-Challis National Forests).

This subbasin is primarily wilderness. Major portions of the subbasin are in either the Frank Church River of No Return Wilderness or the Gospel Hump Wilderness. Many of the watersheds within the subbasin experienced mining in the past, with some mining activities still in existence today. In particular, larger mining areas include the Marshall Mountain area, Warren Creek, and the vicinity of Dixie. The BLM manages the Marshall Mountain area for mining. There are a number of small private holdings within the subbasin, most less than 500 acres in size. Many of these holdings have, and continue to be, used for mining activities. Timber harvest has occurred on about 3,000 acres of the Cove-Mallard area administered by the Nez Perce National Forest and historically grazing was important within parts of the subbasin.

The hills surrounding the Salmon River canyon are composed of a thick mantle of soil and weathered rock. The thick soil along with sufficient precipitation allows for the near complete forested canopy. The western and southern portions of the subbasin are underlain by batholith, which erodes readily, giving the hills their rounded appearance. The older metamorphic basement rock found in the lower canyon and on the north side above Sabe Creek give the river’s edge its distinctly rugged appearance.

The rolling uplands vary in elevation from greater than 9,000 feet at Cottonwood Butte in the southeast to 6,700 feet at Black Butte on the western edge of the subbasin. Elevations in the canyon at the river’s edge vary from near 3,000 feet at the eastern end of the subbasin to approximately 1,900 feet at the western end. Typically, face drainages and the lower portions of major drainages are higher gradient as water runs off the rolling highlands and then plunges into the deeper canyon to join the Salmon River. Drainages run basically north-south in the subbasin with those on the north side of the canyon draining south and the south side of the canyon draining north. North-south drainages create more east- and west-facing slopes. In general, north-facing slopes are the coolest and south-facing slopes the warmest, since they receive more direct sunlight. East-and west-facing slopes are more intermediate with west-facing slopes slightly warmer as late afternoon sun tends to cause warmer air temperatures.

The USGS gage station for White Bird, which is downstream from the subbasin, and other USGS station data from tributaries in and around the subbasin give some indication of the flows experienced within this subbasin. The drainage area above White Bird is 13,550 square miles and includes most of the Salmon River basin from Stanley to White Bird. This gaging station records the accumulative flow of the Salmon River originating near Galena Summit and includes the North Fork, Middle Fork, South Fork, the Little Salmon River, Lemhi River, Pahsimeroi River and the many tributaries. With an average annual mean flow over 11,000 cfs at White Bird, more than half this flow (6,400 cfs) is added by the Salmon River basin above the Salmon-Chamberlain subbasin, the Middle Fork, and the South Fork. The remaining 5,000 cfs comes from the subbasin and Panther Creek, Little Salmon River, and all the tributaries between the subbasin and White Bird. Five and ten year peak flows approximate 80,000 to 96,000 cfs.

The Salmon River canyon is steep and rocky, with an average gradient of approximately 0.2%, and the channel alternates between large pools and boulder-dominated rapids (IDEQ, 2001). The hydrology of tributaries tends to be dominated by snowmelt runoff from the Sawtooth and Salmon River Mountains in the south and the Clearwater and Bitterroot Mountains in the north. Snowmelt runoff generally produces high gradient, high energy stream systems. Gradients average 7.7% for first and second order streams. Tributaries to the Salmon River tend to be mountainous, high gradient, high energy streams dominated by snowmelt runoff. These streams tend to be in V-shaped valleys, with low sinuosity, in Rosgen (1994) A2 - A3 or B2 - B4 stream types (IDEQ, 2001). Gradients vary from as high as 12% in first order streams to as low as 1% in third order streams.

The subbasin is substantially forested, but the lower elevation canyon walls, along the Salmon River especially, are often in shrublands, sagebrush and/or mountain mahogany. The principle forest types are ponderosa pine within drier elevations at 2,000 to 6,500 feet, especially on the south side of the Salmon River; Douglas fir on more mesic sites; mixed conifers with a predominance of grand fir at mid-elevations between 4,500 to 6,500 feet; and subalpine fir at higher elevations above 6,500 feet (IDEQ, 2001). In addition, a number of other conifers may be present in mixed communities or locally dominant, including western larch, lodgepole pine, whitebark pine, western white pine, and Engelmann spruce.

Chinook salmon spawning or juvenile rearing have been detected in Bargamin, lower Crooked, Sheep, Rhett, Little Mallard and Big Mallard Creeks, and lower Wind River watersheds. Steelhead have been found in Sabe, Bargamin, Big Mallard, lower Sheep, and lower Wind watersheds. Additionally, Chamberlain Creek and West Fork Chamberlain Creek provide spawning and rearing habitat for chinook salmon and steelhead (IDEQ, 2001). Fall chinook salmon have been reported in the Salmon River just downstream of Mackey Bar. Redds were observed that were probably made by fall chinook salmon (IDEQ, 2001).

There are eight stream segments within the subbasin that are listed on the 1998 303(d) list for water quality limited waterbodies in Idaho. The listed segments are located in portions of the subbasin primarily outside of wilderness areas. Six north-side tributaries of the Salmon River are listed for sediment, including: Big Creek, Crooked Creek, Jersey Creek, Big Mallard Creek, Little Mallard Creek, and Rhett Creek. Additionally, Warren Creek, a south-side tributary to the Salmon River, is listed for habitat alteration from its headwaters to the wilderness boundary. The Salmon River is 303(d) listed from Corn Creek to Cherry Creek for unknown pollutants.

2.1.3.4 Upper and Lower Middle Fork Salmon Subbasins

The Middle Fork enters the Salmon River at RM 191 and all 106 miles are included in the national Wild and Scenic Rivers System. The Middle Fork flows through a remote area of central Idaho, which for the most part lies within the Frank Church-River of No Return Wilderness. With an area of 1,876,000 acres, the Middle Fork Salmon subbasin has small inholdings of private and state lands that comprise no more than one percent of the subbasin. The subbasin is managed by three National Forests (Salmon-Challis, Boise, and Payette) with 23% (431,480 acres) of the subbasin administered by the PNF.

The tributary streams in the Middle Fork drainage were subjected to glacial action that formed numerous alpine lakes, hanging valleys, glacial till, and moraines. The Middle Fork flows through the Idaho Batholith where the region's rock consists primarily of granites and volcanic. The topography is rugged and steep. The lower part of the drainage is moderate to steep, while headwater streams become nearly flat and meandering.

The season pattern of water temperatures is typical of Rocky Mountain streams. Approximately 39 inches of precipitation falls primarily as snow each year. Stream discharges peak during a two to six weeks period in May and June as snow melt. The magnitude and timing of spring runoff likely affects steelhead spawning activity. As in other batholith streams, hydrochemical analysis indicates that the Middle Fork and tributaries contain relatively low concentrations of various ions.

Vegetation varies by elevation. Ponderosa and lodgepole pine, Douglas fir, Engelmann spruce, and aspen provide the main tree cover on ridge tops and side slopes. Sagebrush, shrubs, and grasses are common in lower areas, especially on south-facing slopes. Tributaries support riparian growth of alder, water birch, cottonwood, and willows.

Recreational use is an extremely important consideration for this drainage. The lower 97 miles of the Middle Fork is only accessible by air, raft or trail. This river has attained national prominence as a recreational area since it offers outdoor enthusiasts opportunities in whitewater experiences, angling, hunting, or passive enjoyment of scenery. Most of the Middle Fork Salmon has good to excellent quality aquatic habitat. However, some notable exceptions exist. Important as salmon and steelhead habitat, portions of headwaters streams Bear Valley, Marsh, Camas, Big, and Loon creeks lie outside the wilderness area and have been degraded to various degrees by mining, grazing and logging. The entire Middle Fork Salmon River subbasin was designated as a special emphasis watershed for salmon within the NOAA Fisheries biological opinion "Land and Resource Management Plans (LRMPs) for National Forests and Bureau of Land Management Resource Areas in the Upper Columbia River Basin and Snake River Basin Evolutionarily Significant Units" (June 22, 1998). This is because a genetically and ecologically unique sub-population of steelhead has been identified in this subbasin combined with a relatively high density of site-specific Federal actions which are exceptions to programmatic LRMPs as well as a lack of implementing planned restoration actions.

2.1.3.5 Summary of Environmental Baseline

The biological requirements of the listed species are not being met under the environmental baseline. Conditions in the action area would have to improve, and any further degradation of the baseline, or delay in improvement of these conditions would probably further decrease the likelihood of survival and recovery of the listed species under the environmental baseline. Pacific salmon populations also are substantially affected by variation in the freshwater and marine environments. Ocean conditions are a key factor in the productivity of Pacific salmon populations. Stochastic events in freshwater (flooding, drought, snowpack conditions, volcanic eruptions, etc.) can play an important role in a species' survival and recovery, but those effects tend to be localized compared to the effects associated with the ocean. The survival and recovery of these species depends on their ability to persist through periods of low natural survival due to ocean conditions, climatic conditions, and other conditions outside the action area. Freshwater survival is particularly important during these periods because enough smolts must be produced so that a sufficient number of adults can survive to complete their oceanic migration, return to spawn, and perpetuate the species. Therefore it is important to maintain or restore essential features and PFC in order to sustain the ESU through these periods. Additional details about the importance of freshwater survival to Pacific salmon populations can be found in Federal Caucus (2000), NMFS (2000), and Oregon Progress Board (2000).

2.2 Analysis of Effects

Effects of the action are defined as: "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 the action, that will be added to the environmental baseline" (50 CFR 402.02). Direct effects occur at the project site and may extend upstream or downstream based on the potential for impairing essential habitat features of critical habitat. Indirect effects are defined in 50 CFR 402.02 as "those that are caused by the proposed action and are later in time, but still are reasonably certain to occur." They include the effects on listed species or critical habitat of future activities that are induced by the proposed action and that occur after the action is completed (USDI FWS and NMFS 1998). "Interrelated actions are those that are part of a larger action and depend on the larger action for their justification" (50 CFR 403.02). "Interdependent actions are those that have no independent utility apart from the action under consideration" (50 CFR 402.02).

In step 3 of the jeopardy and adverse modification analysis, NOAA Fisheries evaluates the effects of proposed actions on listed species and seeks to answer the question of whether the species can be expected to survive with an adequate potential for recovery. In watersheds where critical habitat has been designated, NOAA Fisheries must make a separate determination of whether the action will result in the destruction or adverse modification of critical habitat (ESA, section 3, (3) and section 3(5A)).

2.2.1 Effects of the Proposed Action

NOAA Fisheries will consider any scientifically credible analytical framework for determining an activity's effect. In order to streamline the consultation process and to lead to more consistent effects determinations across agencies, NOAA Fisheries where appropriate recommends that action agencies use the MPI and procedures in NMFS (1996), particularly when their proposed action would take place in forested montane environments. Regardless of the analytical method used, if a proposed action is likely to impair properly functioning habitat, appreciably reduce the functioning of already impaired habitat, or retard the long-term progress of impaired habitat toward PFC, it cannot be found consistent with conserving the species.

For the streams typically considered in salmon habitat-related consultations, a watershed is a logical unit for analysis of potential effects of an action (particularly for actions that are large in scope or scale). Healthy salmonid populations use habitats throughout watersheds (Naiman *et al*. 1992), and riverine conditions reflect biological, geological and hydrological processes operating at the watershed level (Nehlsen *et al*. 1997; Bisson *et al*. 1997; and NMFS 1999).

Although NOAA Fisheries prefers watershed-scale consultations due to greater efficiency in reviewing multiple actions, increased analytic ability, and the potential for more flexibility in management practices, often it must analyze effects in geographic areas smaller than a watershed or basin due to a proposed action's scope or geographic scale. Analyses that are focused at the scale of the site or stream reach may not be able to discern whether the effects of the proposed action will contribute to or be compounded by the aggregate of watershed impacts. This loss of analytic ability typically should be offset by more risk averse proposed action and ESA analysis in order to achieve parity of risk with the watershed approach (NMFS 1999).

The PNF Noxious Weed Control BA provides a detailed analysis of the effects of the proposed action on ESA-listed fish species and their critical habitat in the action area. The analysis uses MPI and procedures in NMFS (1996), the information in the BA, and the best scientific and commercial data available to evaluate elements of the proposed action that have the potential to affect the listed fish or essential features of their critical habitat. The effects analysis in this

Opinion focuses on those elements of the proposed action that have the potential to affect fish, their prey, or riparian functions. The analysis is based primarily on toxic effects of herbicides on ESA-listed fish and their prey, and secondarily on the physical effects of weed removal. Toxic effects may potentially harm listed fish by killing them outright, through sublethal changes in behavior or physiology, or indirectly through a reduction in the availability of prey. Physical effects of weed removal could potentially affect riparian functions such as shade, cover, debris recruitment, and sediment filtering.

2.2.1.1 Activity-Specific Effects

Physical Weed Control. Physical weed removal includes manual or mechanized techniques to remove weeds (hand pulling, grubbing, mowing, tilling, discing, or plowing). The primary effect on aquatic species is exposure of bare topsoil to increased erosion, and subsequent runoff into aquatic systems. In locations where weeds are removed from streambanks, removal of weeds would result in a temporary loss of cover, which would be replaced by new plant growth through natural regeneration, or from re-seeding disturbed sites with desirable vegetation to compete with noxious weeds targeted for control. The amount of area where weeds would be physically removed is a small percentage of the PNF's management area. Soil disturbance and resulting production of sediment from this activity will likely be insignificant.

Regulatory Weed Control Mechanisms. Regulatory control measures would have virtually no effect on listed fish or critical habitat. Proposed regulations would primarily restrict activities that could spread noxious weeds. In situations where vehicle access is restricted to reduce the spread of weeds, there could be a possible reduction in sediment. In situations where livestock grazing is reduced in riparian areas, the condition of riparian vegetation could improve.

Biological Weed Control. Biological weed control may be used in conjunction with other weed control methods, however, biological control is not part of the proposed action. Any introduction of insects or pathogens, that may affect listed species, will be consulted on separately by the United States Department of Agricultural (USDA) Animal Plant Health Inspection Service prior to release. Insects are the primary biological control agent, however, mites, nematodes or pathogens could also be used. The potential effects of biological weed control on listed fish or critical habitat would depend on the specific control agent proposed for use, which are not known at this time.

Chemical Weed Control. In the proposed action, the risks of herbicides effects on salmon and steelhead occur primarily through their toxicological effects on aquatic organisms, rather than physical changes in fish habitat (except for contamination by the herbicides themselves). Herbicides also affect terrestrial vegetation and watershed characteristics by killing or injuring plants, but these terrestrial changes in the proposed action are not expected to noticeably affect the aquatic environment because of the small proportion of land proposed for treatment, restricted use of herbicides in riparian areas, and regrowth of native vegetation in treated areas.

Proposed chemical weed control activities involve the use of five herbicides that include "inert" ingredients that are unknown. No carrier other than water will be used. The ecological risks to aquatic species and toxicological effects are not fully known for the herbicides and formulations in the proposed action. There is ample information available to assess the risk of direct mortality from the active ingredients in the herbicide formulations in the proposed action. There is incomplete information available on ecological effects of the herbicides and their formulations (including effects on the invertebrates on which fish feed), sublethal effects of the active ingredients on listed species, and lethal or sublethal effects of product formulations (mixtures of active ingredients, adjuvents and inert ingredients). Due to concerns about the uncertainty of effects of pesticides on listed salmon and steelhead, Environmental Protection Agency (EPA) has been directed by the U.S. District Court for the Western District of Washington (Washington Toxics Coalition et.al. v. EPA) to consult with NOAA Fisheries on the effects of 55 pesticides, including 2,4-D, which is proposed for use by the PNF.

The effects of the chemicals on ESA-listed fish are dependent on their toxicity to listed fish and other aquatic organisms, and the amount (or likelihood) of exposure to the chemicals. Following this conceptual model, the effects analysis in this Opinion consists of three parts: (1) An evaluation of the likelihood that listed fish and other aquatic organisms will be exposed to the chemicals; (2) an evaluation of the direct effects of herbicide exposure on listed fish; and (3) an evaluation of the indirect effects of the chemicals on the biotic community. The background information for each chemical's toxicity, and information related to the environmental fate and transport is in Appendix A.

2.2.1.2 Likelihood of Exposure to Herbicides

The most effective mechanism to avoid adverse effects of herbicides on ESA-listed fish species is to keep the chemicals out of the waters where listed fish occur. Herbicides can enter water through atmospheric deposition, spray drift, surface water runoff, groundwater contamination and intrusion, and direct application. The proposed action includes numerous BMPs intended to minimize or avoid water contamination from herbicides (See Section I.B.4.a in this Opinion, and the BA). The BMPs include stream and riparian buffers where chemical use is restricted or prohibited, limits on the amount of chemicals carried at a given time or applied to a given area, and rules governing application methods and timing. The BMPs and the likelihood of herbicides entering the water depend on the type of treatment and mode of environmental transport.

Exposure is also affected by the amount of herbicide used (as well as the location and timing of application). The PNF proposes to apply herbicides to no more than 10 - 75 acres per year in any given subbasin (Table 2). Herbicides would be applied primarily through dispersed spot treatments that, in conjunction with BMPs described in Section 1.1.1.1 (above), would limit potential herbicide concentrations in water to very low levels. The actual exposure likely to occur from the proposed action is unknown, since the treatment areas are not precisely known in advance. Consequently, typical exposures reported in published literature are described below, and are used to determine effects on aquatic organisms. These "typical" exposure scenarios are likely to be much higher than actual exposures from the proposed action due to the small amount of chemicals that would be applied.

Water Contamination from Wind Drift. Herbicide spraying can introduce chemicals directly into water through wind drift. Drift may occur during any spraying activity, including aerial applications, boom spraying, and hand spraying. Wind drift is more likely to occur during most aerial applications, and less likely to occur to a significant extent during ground-based spraying, unless sprays are directed into the air, or sprays are delivered in a fine mist. The proposed action does not include aerial applications or boom spraying. Herbicide application will be ground-based and could include: (1) hand-held spray nozzles attached to either backpacks, vehicles or ATVs; and spot gun sprayers mounted on vehicles or ATVs (slip tanks); (2) hand-spreading granular formulations; and (3) wicking, wiping, dripping, painting, or injecting individual target weeds. The amounts of chemicals expected to reach the water from wind drift were not quantified in the BA, and they are not known. However, based on the proposed ground based applications the amount of chemical reaching the water through wind drift is likely to be minimal and below concentrations where lethal or sublethal effects are known to occur in salmon or steelhead.

Water Contamination from Runoff, Leaching, and Percolation. The five herbicides proposed for use can potentially enter streams through water transported by runoff, leaching, or percolation. Rain events could transport chemicals to waterways and result in water contamination, conveying chemicals to chinook salmon or steelhead habitat. The adsorption of herbicides onto soils, stability, solubility, and toxicity of a chemical determine the extent to which it will migrate and adversely affect surface waters and groundwater (Spence et al. 1996). Adsorption of these herbicides to soil varies with the amount of organic matter present in the soil and in with the texture and pH of the soil. Herbicides are susceptible, to a varying degree, to transport by surface runoff, especially if applications are followed immediately by high rainfall events. However, data limitations make it difficult to precisely estimate the degree of ecological risk.

The potential concentrations of chemicals in the water from the proposed action, as a result of runoff, leaching, or percolation, are not known. Off-site movement of herbicides is governed by the binding of herbicides to soil, the persistence of herbicides in soil, as well as site-specific topographic, climatic, and hydrological conditions. The Oregon State University Extension Pesticides Properties Data Base (Vogue et al. 1994) provides a pesticide movement rating, derived from soil half-life, sorption in soil, and water solubility (Table 6). The pesticide movement rating indicates the propensity for a pesticide to move toward groundwater. There are five nominal ratings in Table 6, ranging from very low to very high. As indicated by the movement ratings, glyphosate is least likely to reach groundwater or move from the site, while chemicals such as picloram and dicamba are highly mobile and are likely to be transported by runoff or percolation. Rain fall rates, soil properties, topography, vegetation, and other parameters are factors that influence actual pesticide movement at any given location.

Table 6. Herbicide Movement Rating[†]

Herbicide	Herbicide Movement Rating	Soil Half-Life (days)	Water Solubility (mg/l)	Sorption Coefficient (soil Koc)
Clopyralid	Very High	40	300,000	6
Glyphosate	Very Low	47	900,000	24,000
Picloram	Very High	90	200,000	16
2,4-D	Moderate	10	100	100
Metsulfuron-Methyl	High	30	9500	35

[†] From Vogue et al. (1994); This database relies heavily on the SCS/ARS/CES Pesticide Properties Database for Environmental Decision Making (Wauchope et al., 1992).

Exposure to Escort® (Metsulfuron methyl) - Metsulfuron methyl is generally active in the soil. It is usually absorbed from the soil by plants. The adsorption of metsulfuron methyl to soil varies with the amount of organic matter present in the soil, and with soil texture and pH (USDA Forest Service 1995c). Adsorption to clay is low. Metsulfuron methyl remains unchanged in the soil for varying lengths of time, depending on soil texture, pH and organic matter content. The half-life of metsulfuron methyl can range from 120 to 180 days (in silt loam soil). Soil microorganisms break down metsulfuron methyl to lower molecular weight compounds under anaerobic conditions. Metsulfuron methyl in the soil is broken down to nontoxic and non-herbicidal products by soil microorganisms and chemical hydrolysis. 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.

The breakdown of metsulfuron-methyl in soils is largely dependant on soil temperature, moisture content, and pH. The chemical will degrade faster under acidic conditions, and in soils with higher moisture content and higher temperature (Extoxnet, 1996). The chemical has a higher mobility potential in alkaline soils than in acidic soils, as it is more soluble under alkaline conditions. Metsulfuron-methyl is stable to photolysis, but will break down in ultraviolet light. Half-life estimates for metsulfuron-methyl in soil are wide ranging from 14 - 180 days, with an overall average of reported values of 30 days (Extoxnet, 1996). Reported half-life values (in days) for soil include: clay - 178; sandy loam - 102; clay loam - 70, 14-28, 14-105; silty loam - 120-180. The dissipation time for metsulfuron-methyl was investigated in a mixed wood/boreal forest lake. The DT50 or length of time required for half of the material to dissipate in water was >84 days when high concentrations of metsulfuron-methyl were applied, and 29.1 days at concentrations that might be expected if the chemical is applied for forestry uses (Extoxnet, 1996). The chemical is stable to hydrolysis at neutral and alkaline pHs, and has a half-life of three weeks at pH 5.0, 25 degrees C and >30 days at 15 degrees C.

No field studies have been encountered on the fate of metsulfuron methyl in ambient water. Thompson et al. (1992) measured the persistence of metsulfuron methyl in experimental enclosures in a forest lake with pH ranging from 6.7 to 7.3 and a water temperatures of 22°C (71.6°F). At water concentrations of 0.01 mg/L, the halftime for metsulfuron methyl was 29 days. At 1.0 mg/L, however, the half-life was 84 days. SERA (2000) estimated that peak water level of about 0.044 mg/L metsulfuron methyl (adjusted to the maximum application rate of 2.0 oz/acre as proposed by the PNF) can be anticipated in a 65 x 15 x 1 meter deep pond, under worst case conditions. Below annual rainfall rates of 10 inches per year, no substantial off site movement by percolation or runoff is anticipated. At annual rainfall rates of 25 to 250 inches per year, peak water concentrations vary from about 0.021 mg/L for clay and 0.038 mg/L for sand.

Exposure to Rodeo® (Glyphosate) - Glyphosate is strongly adsorbed to most soils, and 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 130 days (USDA Forest Service 1995b). 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 (U.S. EPA 1993).

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 1996).

There are several relevant monitoring studies that are useful for estimating exposure to glyphosate in water. After an aerial application of Roundup at a rate of 2 kg a.i./ha [about 1.8 lb a.i./acre] over a 10 km^2 area in Vancouver Island, British Columbia, maximum concentrations in streams that were intentionally oversprayed reached about 0.16 mg a.e./L and rapidly dissipated to <0.04 mg a.e./L after 10 minutes. After a storm event, peak concentrations in stream water were <0.15 mg a.e./L, and rapidly dissipating to <=0.02 mg a.e./L before the end of the storm event (SERA 1996). At the same application rate, another Canadian study noted maximum stream concentrations of 0.109–0.144 mg a.e./L, occurring 7–28 hours after aerial application. Similar results were noted in a study conducted in Oregon, in which forest streams were oversprayed at a rate of 3.3 kg a.i./ha [2.9 lb a.i./acre]. Maximum water levels in streams reached 0.27 mg a.e./L (SERA 1996). When normalized for application rates, the maximum levels in stream water from these three studies range from 0.088 to 0.093 mg a.e./L per lb a.i. applied.

Concentrations of glyphosate in ponds that are oversprayed appear to be somewhat less than those found in streams (SERA 1996). For example, in three forest ponds oversprayed at 2.1 kg a.i./ha [1.9 lb a.i./acre], maximum initial glyphosate concentrations were <0.1 mg a.e./L or 0.05 mg a.e./L@lb a.i. applied (Goldsborough and Brown 1993). These findings indicate that modeling the fate of herbicides and the potential concentration within a pond could reasonably be an underestimate of the potential concentrations in streams.

Exposure to Tordon 22K® (Picloram) - Picloram is highly soluble in water, readily leaches through soil, and is resistant to biotic and abiotic degradation processes with a field half-life of 20 to 300 days. Ismail and Kalihasan (1997) found that picloram moves rapidly out of the top 5 cm 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 most relevant monitoring studies are those in which picloram has been detected in ambient water after the application of picloram at a known amount. Two such studies have been encountered involving the contamination of streams (Davis and Ingebo 1973; Michael and Neary 1993). Michael and Neary (1993) summarize monitoring data on the concentrations of picloram in surface water after the application of picloram by injection, broadcast ground, and broadcast aerial applications. Normalized for application rate, the reported peak concentrations of picloram in water are 7.4 to 37 mg/L per lb/acre for injection, 2.2 mg/L per lb/acre for broadcast ground applications, and 48 to 78 mg/L per lb/acre for broadcast aerial applications (Michael and Neary 1993, Table 3, p. 407). The injection data is also summarized in Michael et al. (1994) and the ground and aerial application data are detailed further in Neary et al. (1993). Both the ground and aerial broadcast application involved pellet formulations and the ground application involved a 140 meter buffer. In another study using an application rate of 10.4 kg/ha (9.3 lb/acre), the maximum concentration noted in stream water draining from the watershed was 370 µg/L, which occurred after a 6.4 cm rainfall (Davis and Ingebo 1973). This is equivalent to 40 µg/L per lb/acre [370 µg/L÷9.3 lb/acre], much lower than the concentration rates reported in Michael and Neary (1993). In a study by Watson et al. (1989), no picloram was detected in streams, at a limit of detection of 0.0005 mg/L, after the application of picloram at rates of 0.28 kg a.e./ha or 1.12 kg a.e./ha in areas with loam or sandy loam soil. As discussed further below, these apparent discrepancies are most probably due to rainfall patterns as well as local conditions which impact runoff potential.

The estimated rate of contamination of ambient water, using GLEAMS, that could result from the normal application of picloram is 0.025 (range 0.01 to 0.06) mg a.e./L at an application rate of 1 lb a.e./acre (SERA, 1999a). For acute exposure scenarios, the highest estimated concentration of picloram in water after an accidental spill is about 0.8 mg a.e./L with a range of about 0.27 to 27 mg a.e./L (SERA, 1999a). Based on the results of GLEAMS modeling in SERA (1999a), contamination of ground or surface water from clay or sand is not likely in areas with annual rainfall of less than 50 inches. These calculations were made based on an application of picloram along a 10 acre right-of-way adjacent to a 10 acre pond with an average depth of 1 meter. The assumptions of the model are that runoff from the treatment site is distributed evenly over the 1 acre pond and that the herbicide mixes evenly within the pond to yield the rate of contamination.

Because of the general rather than site-specific nature of the GLEAMS modeling, some loss could occur in arid areas during unusually severe rainfalls, at least at sites with high runoff or leaching potential. There is a high potential for runoff where herbicides are used to treat weeds in roadside ditches because roadside ditches channel storm water and often drain directly into stream channels. Such a scenario would likely result in short term concentrations greater than those predicted by SERA (1999a) due to the fact that the herbicide would be delivered to a point along the stream channel and not distributed evenly across a ten acre pond. Spraying of herbicides in roadside ditches that drain directly into stream channels increases the risk of exposure.

Exposure to Transline® (Clopyralid) - Clopyralid's half-life in the environment averages one to two months and ranges up to one year. It is degraded almost entirely by microbial metabolism in soils and aquatic sediments. Clopyralid is not degraded by sunlight or hydrolysis. Similar to picloram, 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 one year period (Hawes and Erhardt-Zabik 1995). Because clopyralid does not bind with sediments readily, it can be persistent in an aquatic environment. Clopyralid is stable in water over a pH range of five to nine (Woodburn 1987) and the rate of hydrolysis in water is extremely slow, with a half-life of 261 days (Concha and Shepler 1994).

In SERA (1999b) the monitoring data from Leitch and Fagg (1985) was used to estimate the concentrations in ambient water that could be associated with the application of clopyralid. For the characterization of risk, an exposure level of 0.0089 mg a.e./L per lb a.e. clopyralid is used to characterize the risks associated with chronic exposure scenarios, and 4.54 mg a.e./L per lb a.e. clopyralid is used to characterize the risks associated with acute exposure scenarios (SERA 1999b). In the study by Leitch and Fagg (1985), rainfall rates averaged about 0.4 in/hour. The amount of rain predicted for a storm within the action area that occurs on average every ten years and has a duration of one hour is between 0.6 and 0.8 inches (Dunne and Leopold 1978). For a storm that occurs on average every one hundred years and has a duration of one hour the predicted rainfall is at least 1.0 inch. Heavy rainfall events frequently occur within the action area and the potential for localized high intensity storms is great. Runoff and contamination of waterways could occur during unusually severe rainfalls, at least at sites with high runoff or leaching potential. There is a high potential for runoff where herbicides are used to treat weeds in road-side ditches. Road-side ditches channel storm water and often drain directly into stream channels.

Exposure to Weedar® 64 (2,4-D) - Weedar® 64 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. However, it 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 (USDA Forest Service 1995a). 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 one to six weeks in the soil and will degrade to half of its original concentration in several days

(USDA Forest Service 1995a). 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.

2,4-D is likely to contaminate surface waters if rains occur shortly after application, and unlikely to enter surface waters if rain is unlikely. The Washington Department of Ecology collected 32 stream samples downstream from a helicopter application of 2,4-D conducted according to Washington's BMPs. The 2,4-D was found in all samples collected, in highest concentrations following a rainstorm the day after the spraying (Rashin and Graber 1993). In another study, 2,4-D was found in 19 of 20 basins sampled throughout the U.S. (USGS 1998). In the USGS (1998) study, 2,4-D was found in 12% of agricultural stream samples; 13.5% of urban stream samples; and in 9.5% of the samples from rivers draining a variety of land uses. The study by Rashin and Graber (1993) demonstrates a greater likelihood of 2,4-D contamination in an environment with frequent precipitation, while the broader USGS (1998) study shows lower rates of contamination when averaged across a range of climatic conditions.

In SERA (1998) a contamination rate of 0.002 mg/L per lb a.e. applied is used when assessing chronic exposure to water contaminated with 2,4-D. This rate is based on the average level in groundwater of 0.19 μg/L during 1985 in the study by Waite et al. (1992). Over the 3-year period, the average amount of 2,4-D applied to the 6,900 acre watershed was 322 kg or 708 lbs. Thus, the average application rate over the entire area was about 0.1 lb/acre (708 lbs/6,900 acres). Thus, the monitored level of 0.19 μg/L is rounded to 0.0002 mg/L and divided by 0.1 lb/acre to yield a rate of 0.002 mg/L per 1 lb applied/acre. The PNF proposes to use a maximum of 2 lb a.e./acre which would yield a conservative level of water contamination of 0.004 mg/L for 2,4-D in treated watersheds. However, the actual level of water contamination could be much less because proposed treatments do not include broadcast spraying and would not cover entire watersheds.

2.2.1.3 Likelihood of Direct Effects

Most direct effects of herbicides on listed salmon and steelhead are likely to be from sublethal effects, rather than outright mortality from herbicide exposure. Sublethal effects are considered under the ESA to constitute "take," if the sublethal effects "harm" listed fish. NOAA Fisheries defines harm as "an act which actually kills or injures fish or wildlife. Such an act may include significant habitat modification or degradation which 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). These behavioral patterns, and their underlying physiological processes are typically reported for individual test animals. However, the ecological significance of sub-lethal toxicological effects depends on the degree to which they influence behavior that is essential to the viability and genetic integrity of wild populations. It is important to note that many sublethal toxicological endpoints or biomarkers may harm fish in ways that are not readily apparent. When small changes in the health or performance of individual fish are observed (e.g. a small percentage change in the activity of a certain enzyme, an increase in oxygen consumption, the formation of pre-neoplastic hepatic lesions, etc.), it may not be possible to infer a significant loss of essential behavior patterns of fish in the wild, even in circumstances where a significant loss could occur.

An analysis of the direct impacts of herbicides on salmonids should relate the site-specific exposure conditions (i.e., expected environmental concentration, bioavailability, and exposure duration) to the known or suspected impacts of the chemical on the health of exposed fish. Where possible, such analyses should consider: (1) The life history stage (and any associated vulnerabilities) of the exposed salmonid; (2) the known or suspected mechanism of toxicity for the active ingredient (or adjuvant) in question; (3) local environmental conditions that may modify the relative toxicity of the contaminant; and (4) the possibility of additive or synergistic interactions with other chemicals that may enter surface waters as a result of parallel or upstream land use activities.

A probabilistic risk assessment (PRA), based on the relationship between the likelihood of exposure and the magnitude of effect is used to evaluate the proposed action. Traditionally, a PRA incorporates data from a standard lethal concentration (LC_{50}) exposure study as well as chronic exposure data to predict the sensitivity of an organism to the pesticide or chemical. The lethality endpoint has little predictive value for assessing whether pesticide exposure will cause sublethal neurological and behavioral disorders in wild salmon (Scholz et. al 2000), but in most cases, the LC_{50} is the only toxicity data available. Although little information is available on the sublethal effects of the herbicides on listed fish, there can be subtle sublethal effects that can potentially affect the survival or reproduction of large population segments. For example, Scholz et al. (2000), and Moore and Waring (1996) indicate 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 similar effects with the chemicals proposed for use is unknown.

Based on the analysis provided in the BA, and available literature, it appears unlikely that the proposed herbicide use would cause outright fish kills at concentrations of the active ingredients likely to occur in water from the proposed action, except for circumstances where high herbicide concentrations result from heavy rainfall shortly after herbicides are applied, or as a result of an accidental spill. All LC_{50} concentrations for salmonids, for the active ingredients in the herbicides proposed for use, and reported in the literature cited in the BA and in this Opinion, are above 1 mg/L (see Appendix A), while environmental concentrations would typically be at least 1 to 2 orders of magnitude lower. While the active ingredients appear to pose a low risk of mortality, the likelihood of outright mortality of listed fish from exposure to product formulations that include unknown adjuvants is virtually unknown due to the paucity of information available.

Although lethal effects are not expected to occur under most circumstances, listed fish are likely to be exposed to herbicide concentrations where sublethal effects could occur. Potential sublethal effects, such as those leading to a shortened lifespan, reduced reproductive output, other types of “ecological death” (e.g. Kruzynski et al. 1994; Kruzynski and Birtwell 1994) or other deleterious biological outcomes is a threat to listed species from the proposed action. The toxicological endpoints identified below are possible for a variety of pesticides and 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, eggs fertilized, or eggs 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.
- Changes in migratory behavior.

The consequences of these sublethal effects are uncertain, but the loss of physiological or behavioral functions can adversely affect the survival, reproductive success, or migratory behavior of individual fish. Such effects, in turn, can be expected to reduce the viability of wild populations. Additional endpoints could also be significant if a clear relationship is established between the observed impairment and the “essential biological requirements” of salmonids (i.e. the likelihood that the exposed animal will survive the various phases of its life cycle and return to its natal river system to spawn.). Most of these endpoints (above) have not been investigated for the herbicides in the proposed action, however some limited data are available.

Direct effects of Escort® (Metsulfuron methyl). The lowest reported concentration at which mortality has been observed is 100 mg/L (SERA 2000). However, the investigators do not consider this effect significant because there was no mortality at the 1000 mg/L concentration. SERA (2000) reported that metsulfuron-methyl had no effect on rainbow trout hatching, larval survival, or larval growth over a 90-day exposure period at a concentration of up to 4.7 mg/L. Concentrations of metsulfuron-methyl greater than 8 mg/L resulted in small but significant decreases in hatching and survival of fry. For a 96-hour exposure test, SERA (2000) reported that although there was no mortality in adult fish at a concentration of 150 mg/L metsulfuron-methyl, some of the fish exhibited erratic swimming, rapid breathing and were lying on the bottom of the test container.

Direct Effects of Rodeo® (Glyphosate). The toxicity of glyphosate to aquatic species depends on the acidity (pH) of the water (SERA 1996). Glyphosate is more toxic in relatively highly acidic water (pH.6) by up to a factor of about 10, compared with alkaline water (pH.10). Generally, the reported LC_{50} values (concentration which will kill approximately half of the subjects) for aquatic animals range from approximately 10 to 400 mg/L, depending on the species and pH of the water. Information on sublethal effects of glyphosate is available for some of the above endpoints, and of those reported, Rodeo® appears to have a low risk for sublethal effects. Technical grade glyphosate (purity 62%) is of moderate toxicity to aquatic species, and the surfactant used in Roundup® is more toxic than glyphosate alone. Rodeo® contains 46.8% dimethylamine salt of 2,4-D. Carp exposed to 5 mg/L of technical grade glyphosate for two weeks were found to have gill damage and liver damage at concentrations of 10 mg/L (Neskovic et al., 1996). Neskovic et al. (1996) found that subacute toxic effects of glyphosate to fish correlated well with findings of other researchers and that pesticides cause changes in enzyme activity (in the first place transaminases) as well as biochemical alterations of some organ and tissue constituents (total lipids, glucose, glycogen, etc.).

***Direct Effects of Tordon 22K® (Picloram*).** Acute (96-hour) LC_{50} (concentration which will kill approximately half of the subjects) values for trout range from about 5 mg/L to about 20 mg/L (SERA 1999a). 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). Maximum exposure concentrations not affecting survival and growth of cutthroat trout ranged from 29 to 86 mg/L in Woodward's (1979) study. Tests with the early life-stages of rainbow trout showed that picloram concentrations of 0.9 mg/L reduced the length and weight of rainbow trout larvae, and concentrations of 2 mg/L reduced survival of the larval fish (Mayes et al. 1987). Woodward (1976), in a study of lake trout, found that picloram reduced fry survival, weight, and length at concentrations of 0.04 mg/L, and that the rate of yolk sac absorption and growth of lake trout fry was reduced in flow-through tests at concentrations as low as 0.35 mg/l. Yearling coho salmon exposed to 5 mg/L of picloram for six days suffered "extensive degenerative changes" in the liver and wrinkling of cells in the gills (U.S. EPA 1979).

Direct Effects of Transline® (Clopyralid). No information was found for the potential sublethal effects of clopyralid on fishes. The LC_{50} (concentration which will kill approximately half of the subjects) for rainbow trout exposed to the monoethanolamine salt of clopyralid is 700 mg a.e./L (SERA 1999b).

Direct effects of Weedar® (2,4-D). In general, fish are less sensitive to the dimethylamine formulation than the acid. However, SERA (1998) recommends using the LC_{50} (concentration which will kill approximately half of the subjects) for the acid as a conservative estimate. The LC_{50} for lake trout is 45 mg/L and for cutthroat trout it is 64 mg/L. Little et al. (1990) examined behavior of rainbow trout exposed for 96 hours to sublethal concentrations of 2,4-D amine, and observed inhibited spontaneous swimming activity and swimming stamina. The ability of rainbow trout to capture food was reduced when exposed to 5 mg/L of 2,4-D amine (Little 1990). Changes in schooling behavior and red blood cells, reduced growth, impaired ability to capture prey, and physiological stress were reported for 2-4,D (NIH 2002; Gomez et. al. 1998; Cox 1999). The 2,4-D can also combine with other pesticides and have a synergistic effect, resulting in increased toxicity. Combining 2,4-D with picloram damages the cells of catfish (*Ictalurus spp.*) gills, although neither individual pesticide has been found to cause this damage (Cox 1999). Adverse sublethal effects on fish behavior [i.e., no changes in the reproductive behavior (nest guarding) in red ear or bluegill sunfish] were not observed by Bettoli and Clark (1992) at concentrations of 11 mg/L of 2,4-D dimethylamine. In characterizing the likelihood of fish being exposed to relatively high concentrations of 2,4-D after a spill, it is worth noting that fish will avoid concentrations of 2,4-D in water at levels as low as 1 mg/L (Sassaman et al. 1984).

2.2.1.4 Likelihood of Indirect Effects

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 pesticides, including the chemicals 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; however, 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., 2nd Sess. 12 (1979)].

It is becoming increasingly evident that indirect effects of contaminants on ecosystem structure and function are a key factor in determining a toxicant's cumulative risk to aquatic organisms (Preston 2002). Moreover, aquatic plants and macroinvertebrates are generally more sensitive than fish to the acutely toxic effects of herbicides. Therefore, chemicals can potentially impact the structure of aquatic communities at concentrations that fall below the threshold for direct impairment in salmonids. The integrity of the aquatic food chain is an "essential biological requirement" for salmonids, and the possibility that herbicide applications will limit the productivity of streams and rivers should be considered in an adverse effects analysis.

The potential effects of herbicides on prey species for salmonids are also an important concern. Juvenile Pacific salmon feed on a diverse array of aquatic macroinvertebrates (i.e. larger than 595 microns in their later instars or mature forms; Cederholm et al. 2000). Terrestrial insects, aquatic insects, and crustaceans comprise the large majority of the diets of fry and parr in all salmon species (Higgs et al. 1995). Prominent taxonomic groups 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. daphniids), the impacts of pesticides on salmonid prey taxa have not been widely investigated. Where acute toxicity for salmonid prey species are available, however, they should be used to estimate the potential impacts of herbicide applications on the aquatic food chain.

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). In the case of herbicides, a primary 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 environmentally relevant concentrations (DeLorenzo et al. 2001). In many cases, however, the acute sensitivities of algal species to herbicides are not known. In addition, Hoagland et al. (1996) identify key uncertainties in the following areas: (1) The importance of environmental modifying factors such as light, temperature, pH, and nutrients; (2) interactive effects of herbicides where they occur as mixtures, (3) indirect community-level effects, (4) specific modes of action, (5) mechanisms of community and species recovery, and (6) mechanisms of tolerance by some taxa to some chemicals. Herbicide applications have the potential to impair autochthonous production and, by extension, undermine the trophic support for stream ecosystems. However, existing data gaps make it difficult to precisely estimate the degree of ecological risk, and limited information is available on the ecological effects of the chemicals in the proposed action.

The growth of salmonids in freshwater systems is largely determined by the availability of prey (Chapman 1966; Mundie 1974). For example, supplementation studies (e.g. Mason 1976) have shown a clear relationship between food abundance and the growth rate and biomass yield or productivity 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, that is, competition among foragers can be expected to increase 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). For example, a recent 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; Healy 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).

Indirect effects of Escort® (Metsulfuron methyl). Aquatic plants are far more sensitive than aquatic animals to the effects of metsulfuron methyl, although there appear to be substantial differences in sensitivity among species of macrophytes and unicellular algae. *Lemna gibba* has a EC_{50} value of 0.00036 mg/L and a "No Observable Effect Concentration" (NOEC) value of approximately 0.00016 mg/L (SERA 2000). There appears to be substantial variation in the toxicity of metsulfuron methyl to algal species with reported EC_{50} values above 0.01 mg/L (SERA 2000). Metsulfuron methyl appears to be relatively non-toxic to aquatic invertebrates, based on acute bioassays in daphnia, with an acute LC_{50} value for immobility of 720 mg/L and an NOEC for reproduction of 150 mg/L. The only effect seen in a 21-day daphnia study was a decrease in growth (Hutton 1989), which was observed at concentrations as low as 5.1 mg/L.

SERA (2000) estimated that peak water level of about 0.044 mg/L (adjusted to the maximum application rate of 2.0 oz/acre as proposed by the PNF) can be anticipated in a 65 x 15 x 1 meter deep pond, under worst case conditions. Below annual rainfall rates of 10 inches per year, no substantial off site movement by percolation or runoff is anticipated. At annual rainfall rates of 25 to 250 inches per year, peak water concentrations vary from about 0.021 mg/L for clay and 0.038 mg/L for sand. Concentrations lethal to aquatic macrophytes and algae are unlikely to occur unless metsulfuron methyl is directly added to water or if a rainfall washes the chemical into a stream shortly after it is applied.

Indirect Effects of Rodeo® (Glyphosate). Glyphosate is highly toxic to all types of terrestrial plants and is used to kill floating and emergent aquatic vegetation. Glyphosate does not appear to have similar toxicity to algae. Glyphosate is considered by EPA to be "slightly toxic" to aquatic invertebrates (SERA 1996). LC_{50} values of 780 and 930 mg/L have been reported for daphnia. Hildebrand et al. (1980) found that Roundup treatments at concentrations up to 220 kg/ha did not significantly affect the survival of daphnia or its food base of diatoms under laboratory conditions. In addition, Simenstad et al. (1996) found no significant differences between benthic communities of algae and invertebrates on untreated mudflats and mudflats treated with Rodeo. It appears that under most conditions, rapid dissipation from aquatic environments of even the most toxic glyphosate formulations prevents build-up of herbicide concentrations that would be lethal to most aquatic species.

Indirect Effects of Transline® (Clopyralid). From information reported in SERA (1999b) it appears that there could be potential losses in primary productivity from algae killed by clopyralid, based on an EC_{50} (concentration causing 50% inhibition of a process) for algae of 6.9 mg/L. However, concentrations lethal and sublethal 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. 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 232 mg a.e./L for the acid LC_{50}. If other invertebrates respond similarly to daphnia, then lethal effects on aquatic invertebrates are unlikely. However, both sublethal effects and the concentration at which sublethal effects occur within aquatic invertebrates and plants are unknown.

Indirect Effects of Tordon 22K® (Picloram). While most grasses are resistant to picloram, it is highly toxic to many broad-leafed plants. 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, nontarget plants may be exposed to chemical concentrations many times the levels that have been associated with toxic effects. The lowest reported adverse effect (the EC_{25} for the inhibition of seed emergence in soybeans) for the potassium salt of picloram is 0.000014 kg or about 0.000012 lb a.e./acre (SERA, 1999a). Picloram's mobility allows it to pass from the soil to nearby, nontarget plants. It can also move from target plants, through roots, down into the soil, and into nearby nontarget plants. Given this capability, an applicator does not have to spray within the riparian buffer zone in order to affect the riparian vegetation. Spray drift may kill plants some distance away from the area being treated. Additionally, crop damage from irrigation water contaminated by picloram has been documented by the U.S. EPA (U.S. EPA 1995, USDA Forest Service 1995d).

Although picloram is toxic to salmonids, it is not as toxic to daphnia or algae at the same concentrations. In *Daphnia*, the reported acute (48 hours) LC_{50} value is 68.3 (63 to 75) mg/L (SERA 1999a). Chronic studies using reproductive or developmental parameters in daphnia report a no-effect level of 11.8 mg/L and an adverse effect level of 18.1 mg/L. Based on standard bioassays in aquatic algae, the lowest effect level for the potassium salt of picloram (EC_{25} for growth inhibition in *S. capricornutum*) is 52.6 mg/L with a corresponding “no observable adverse effects level “(NOAEL) of 13.1 mg/L. Concentrations lethal to algae and daphnia are unlikely to occur unless picloram is directly added to water or if a rainfall washes the chemical into a stream shortly after it is applied.

Indirect effects of Weedar® (2,4-D). Most of the toxicity studies have focused on the ester and acid formulations of 2,4-D, which are generally more toxic than the amine salt. The SERA (1998b) report suggests that amine and acid formulations have relatively low toxicity to aquatic invertebrates and aquatic plants, although the effects are highly variable. Insect larvae are most susceptible to adverse effects, while zooplankton are the least susceptible (Sarkar 1991). Acute toxicity tests exposing the cladoceran, *Simocephalus vetulus*, to the sodium salt of 2,4-D show complete mortality of test subjects following 96 hours of exposure to concentrations ranging from 0.5 to 5.0 mm (110-550 mg/L) (Kaniewska-Prus 1975). The EPA (1988) classifies 2,4-D as slightly toxic to aquatic invertebrates, but found that the dimethylamine was highly toxic to grass shrimp with a LC_{50} of 0.15 mg/L.

SERA (1998b) concluded that some species of aquatic algae are sensitive to concentrations of approximately 1 mg/L 2,4-D, however, low levels of the compound may stimulate algal growth in some species. Aquatic macrophytes are more sensitive to 2,4-D than other aquatic organisms. 2,4-D affected the concentration and ratio of chlorophyll a and chlorophyll b in a freshwater fern, *Salvinia natans*, at concentrations as low as 0.3 mg/L. Concentrations lethal to aquatic invertebrates and aquatic plants are unlikely to occur unless 2,4-D is directly added to water or if a rainfall washes the chemical into a stream shortly after it is applied. However, sublethal concentrations can be exceeded in the event of an accidental spill. SERA (1998b) found that after an accidental spill, maximum initial concentrations of 2,4-D in water are estimated at 6 mg/L per lb applied. After the application of 2,4-D butoxyethyl ester, Aqua-Kleen, to ambient waters a maximum concentration of 0.02 mg/L was observed (SERA 1998b). Ester formulations have much greater toxicity, but are not proposed for use by the PNF.

2.2.1.5 Physical Effects of Herbicides on Watershed and Stream Functions

The use of herbicides can affect watershed or stream functions through the removal of vegetation and exposing bare soil. For hand and spot applications, the potential for significant increases in erosion or water yield is limited because treatments would consist of small, scattered areas, and vegetation would typically be reestablished within a few months to a year. Additionally, the proposed BMPs should minimize the effects of drift, chemical leaching, or other effects of weed spraying on riparian vegetation.

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 desired vegetation re-occupies the treated site. Only ground based spot/selective spraying will be authorized within riparian areas or within 100 feet of live water (whichever is greater). This will significantly reduce risks associated with spraying of non-target riparian vegetation. Noxious weed control measures will reduce weed competition with native riparian species and other upland species. Herbicide spraying in riparian areas will be minimal and will primarily be associated with spot spraying along road right-of-ways, and spot spraying of small patches of noxious weeds or individual plants. No aerial applications of herbicides are authorized by the proposed action.

2.2.2 Cumulative Effects

Cumulative effects are defined in 50 CFR 402.02 as “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.” These activities within the action area also have the potential to adversely affect the listed species and critical habitat. Future Federal actions, including the ongoing operation of hydropower systems, hatcheries, fisheries, and land management activities are being reviewed through separate section 7 consultation processes. Federal actions that have already undergone section 7 consultations have been added to the description of the environmental baseline in the action area.

The action area contains federal lands administered by the USFS, which comprise the majority of the watershed acreage, interspersed with state and privately-owned lands. The BA identified a risk from activities occurring on private, state, and other federal lands within the action area. The BA revealed that activities occurring on private, state, and other federal lands within the action area may contribute to the need to maintain or increase the proposed level of noxious weed treatment for many years into the future. Land use within the analysis area includes agricultural, timber harvest, roads, development, recreation, mining, and livestock grazing. Current levels of these uses are likely to continue, but detailed information on other federal and non-Federal activities in the action area are not available.

Livestock grazing may partially thwart weed control efforts. Cattle can spread weeds through their droppings, and create conditions that increase the likelihood that invasive weeds will out-compete native plants. Riparian cattle grazing on non-Federal lands is likely to cumulatively affect water temperature and water quality in portions of the action area.

Streamflows in the action area are not appreciably affected by water diversions, but a small number of stream diversions, in mostly headwater streams, exist on private lands. Reduced streamflows in smaller streams could appreciably increase the likelihood of reaching herbicide concentrations where adverse effects would occur.

Impaired water quality from on-going agricultural activities is likely to be one of the largest cumulative effects present in the action area. Cultivated croplands are likely to produce large amounts of sediment and increase water yield, and relatively large amounts of pesticides are also likely to be applied to croplands in the action area. City, state, and county governments also have on-going weed spraying programs with less-stringent measures to prevent water contamination. Weeds are sprayed along road right-of-ways annually by city, state, and county transportation departments, sometimes several times a year. NOAA Fisheries staff have observed county road crews spraying herbicides on streambank vegetation and directly into the water in Idaho County, and it is probable that similar practices will continue.

Any herbicide contamination that occurs from the proposed PNF action could potentially combine with contaminants from other Federal and non-Federal activities, and contribute to formation of chemical mixtures or concentrations that could kill or harm listed steelhead or salmon. In addition, fish stressed by elevated sediment and temperatures are more likely to susceptible to toxic effects of herbicides. While the mechanisms for cumulative effects are clear, the actual effects cannot be quantified.

2.2.3 Consistency with Listed Species ESA Recovery Strategies

Recovery is defined by NOAA Fisheries regulations (50 CFR 402) as an "improvement in the status of listed species to the point at which listing is no longer appropriate under the criteria set out in section 4 (a)(1) of the Act." Recovery planning is underway for listed Pacific salmon in the Northwest with technical recovery teams identified for each domain. Recovery planning will help identify measures to conserve listed species and increase the survival of each life stage. NOAA Fisheries also intends that recovery planning identify the areas/stocks most critical to species conservation and recovery and thereby evaluate proposed actions on the basis of their effects on those areas/stocks.

The USFS has specific commitments to uphold under the Basinwide Salmon Recovery Strategy (Federal Caucus 2000). Commitments made pertinent to the proposed action include the following:

1. Consult with NOAA Fisheries and FWS on land management plans and actions that may affect listed fish species following the Streamlined Consultation Procedures for Section 7 of the ESA, July 1999;

2. Collaborate early and frequently with states, tribes, local governments and advisory councils in land management analyses and decisions;

3. Share information, technology and expertise, and pool resources, in order to make and implement better-informed decisions related to ecosystems and adaptive management across jurisdictional boundaries; and

4. Require that land management decisions be made as part of an ongoing process of planning, implementation, monitoring and evaluations. Incorporate new knowledge into management through adaptive management.

The proposed action is consistent with the specific commitments and primary objectives of the Basinwide Salmon Recovery Strategy.

2.2.4 Summary of Effects

The fourth step in NOAA Fisheries' approach to determine jeopardy and adverse modification of critical habitat is to determine whether the proposed action, in light of the above factors, is likely to appreciably reduce the likelihood of species survival and recovery in the wild or adversely modify or destroy critical habitat. For the jeopardy determination, NOAA Fisheries uses the consultation regulations and, where appropriate, the Habitat Approach (NMFS 1999) to determine whether actions would further degrade the environmental baseline or hinder attainment of PFC at a spatial scale relevant to the listed ESU.

The proposed action is likely to adversely affect listed salmon and steelhead through lethal or sub-lethal chemical effects on listed fish, through alteration of the food web from toxic chemical effects, loss of desired riparian vegetation from contact with herbicides, or through restoration of native vegetation or more naturally-functioning watershed processes that are impaired by infestations of invasive weeds.

The risk of harm to listed salmon and steelhead from contact with herbicides is a function of chemical concentration to which listed fish are exposed, and the toxicity of the chemical. Available literature cited above indicates that "typical" exposure to the herbicides proposed for use by the PNF is well-below concentrations where the herbicides kill outright listed salmon or steelhead once they matured beyond the fry stage. Concentrations where outright mortality is reported for steelhead that have matured beyond the fry stage are at least one to two orders of magnitude higher than the "typical" exposure. "Typical" concentrations of herbicides reported in the literature approach the range where mortality has been observed in salmon eggs and fry, however, the levels of exposure expected under the proposed action are likely to be far less than the typical exposure scenarios described above, due to the small amounts of herbicide that would be applied in a given watershed and additional protection through the BMPs. Consequently, outright mortality from herbicide exposure is unlikely to occur under the proposed action. Herbicide spraying in the vicinity of steelhead or salmon eggs or fry, could result in direct mortality if chemicals are sprayed into the water, or if rainfall occurs shortly after application. The relatively small amount of area treated within a given watershed, use of BMPs to reduce the likelihood of exposure, and the dilute concentrations proposed for use reduce the probability that direct mortality would occur from chemical exposure.

Table 7 contains a summary of the effects pathways and relevant properties of each herbicide proposed for use by the PNF. The longevity of each herbicide is expressed as the half-life (the time required for half the amount of a the substance in or introduced into the ecosystem to be eliminated or disintegrated by natural processes). The half-life can be rapid especially if the herbicide is actively degraded in soil and/or water by microorganisms. The solubility (in water) and adsorption (to soil particles) characterize the potential for an herbicide to leach through the soil and also for the herbicides to be transported offsite in runoff. All of the herbicides are soluble in water and only one of them binds well to soil particles under most environmental conditions. The direct and indirect effects include sublethal effects to fish and to other aquatic species which includes prey species and algae. For the most part, direct and indirect effects are poorly documented and the information that exists largely represents the results of lethal assays. Sublethal effects of the herbicides are unknown for many of the effects pathways identified within Section II.A.2.c of this Opinion.

Table 7. Summary of the Effects Pathways and Relevant Properties of Each Herbicide Proposed for Use by the Payette National Forest.

	Longevity (half-life)	Solubility (in water)	Adsorption (to soil particles)	Direct Effects (toxicity to fish)	Indirect Effects (toxicity to other aquatic species)
Tordon 22K® (Picloram)	Half-life of less than 300 days and resistant to degradation.	High	Low	Very toxic with effects noted at levels as low as 0.04 mg/L.	Slightly toxic
Weedar 64® (2,4-D)	Half-life of several days; remains active in soil for up to six weeks.	High	Low to Moderate depending on organic content	Caused behavioral changes at 5 mg/L; minimal information found for sublethal effects on salmonids.	Aquatic algae are sensitive to 1 mg/L and most invertebrates have a high tolerance but effects have been noted at 0.15 mg/L.
Rodeo® (Glyphosate)	Half-life of less than 130 days; rapid microbial degradation in soil and water.	High	High	Caused deformities in carp at 5 mg/L; found minimal information for sublethal effects.	Low toxicity
Transline® (Clopyralid)	Half-life of less than one year; microbial degradation in soils and water.	High	Low	No sublethal effects analysis found. Results of lethal assays report an LC_{50} of 700 mg/L for fish.	LC_{50} of 350 mg/L for aquatic invertebrates; EC_{50} of 6.9 mg/L for algae.
Escort® (Metsulfuron methyl)	Half-life of less than 180 days; rate of microbial degradation varies with soil characteristics.	High	Varies depending on organic content, texture and pH	No sublethal effects analysis found. Results of lethal assays report no lethal effects observed at 4.7 mg/L.	Aquatic plants are very sensitive; the NOEC is 0.00016 mg/L.

Although outright mortality from herbicide exposure is not expected to occur, adverse effects reported in sub-lethal assays may occur, and include reductions in reproductive success, weight loss, physiological effects (endocrine system, blood chemistry, liver function, etc.), and reductions in growth, prey capture ability, and swimming ability, all of which are associated with reduced survival. Information available on sub-lethal effects is incomplete, and few herbicide formulations have been thoroughly tested for sublethal effects on salmon or steelhead. Consequently, the extent of sub-lethal effects could be much greater than indicated by available information. Harm to listed fish from effects of chemicals on food webs are also possible, but difficult to quantify due to the paucity of information.

Given the presence of listed fish in the action area, the range of soil properties in the action area, chemicals proposed for use, rainfall patterns, and proposed spray activities, it is likely that circumstances will arise where herbicide concentrations in water will reach levels where delayed mortality or reduced reproductive success would occur. Such circumstances would arise in isolated instances when various combinations of factors occur, such as: use of chemicals that persist in the environment for several months or longer; conditions that allow chemicals to move rapidly through soils; when precipitation occurs before the chemicals break down, bind to soil particles, or get taken up by plants; where ESA-listed fish or redds are in the vicinity of a spray site; or where the amount of chemical applied to an area is great enough to reach concentrations that could harm listed fish. Heavy rainfall events frequently occur within the action area and the potential for localized high intensity storms is great. Runoff and contamination of waterways could occur as a result of heavy rainfalls, at least at sites with high runoff or leaching potential. Treatment in road-side ditches presents a unique problem because they channel overland flow and storm waters, and often drain directly into stream channels. The majority of the herbicide treatments proposed by the PNF will be within roadside ditches (P. Grinde, Level 1 meeting 5/01/02). Spraying of roadside ditches that drain directly into stream channels increases the risk of effects to steelhead and chinook salmon. Specific locations where harm is likely to occur from the proposed action cannot be identified at this time, since most of the above factors will not be known until spray sites are selected.

Changes in vegetation from weed spraying or other control methods can beneficially or adversely affect riparian and watershed functions. Adverse effects have been reported in instances where herbicides killed non-target plants, particularly riparian trees killed as a result of spray drift or uptake by roots. Beneficial effects to aquatic systems from noxious weed control are not well-documented, but could conceivably occur in circumstances where weed treatments kill exotic plants that would otherwise create a disclimax riparian plant community or displace native plants that provide shade, cover, habitat complexity, streambank stability, or recruitment of terrestrial invertebrate prey.

In summary, NOAA Fisheries' analysis of effects finds the following:

1. The proposed action is not likely to impair physical habitat conditions or processes, since the majority of weed treatment sites are dispersed areas that would not be large enough to have any discernable effect on stream functions, and in instances where weed control activities occur in riparian areas or over large contiguous blocks of land, the activities are restricted by BMPs that prevent or minimize adverse effects.

2. The proposed action is likely to impair water quality where herbicides enter the stream, however, such impairments are expected to occur in isolated cases, and be of short duration (*e.g*. spikes in concentration following a rainfall).

3. Although toxicity of the herbicides may be underestimated due to gaps in the information available on toxic effects, gross errors in the effects analysis are not anticipated because the area where adverse effects could occur is less than 0.15% of any given subbasin.

4. The proposed action will not appreciably reduce the survival of listed Snake River salmon or steelhead because instances where listed fish are likely to be killed or harmed are

expected to be uncommon because, under most circumstances (with the exceptions noted above), the BMPs in the proposed action are expected to largely prevent herbicides from reaching water in concentrations where listed fish would be killed or harmed by the chemicals.

2.3. Conclusion

The final step in NOAA Fisheries' approach to determine jeopardy/adverse modification is to determine whether the proposed action, in light of the above factors, is likely to appreciably reduce the likelihood of species survival in the wild or adversely modify critical habitat. NOAA Fisheries has determined that, when the effects of the proposed action are added to the environmental baseline and cumulative effects occurring in the action area given the status of the stocks and condition of critical habitat, the action is not likely to jeopardize the continued existence of the listed Snake River steelhead and chinook salmon and ESUs considered in this Opinion. Further, NOAA Fisheries concludes that the subject action is not likely to destroy or adversely modify designated critical habitat for the Snake River chinook salmon ESUs considered in this Opinion.

In reaching these determinations, NOAA Fisheries used the best scientific and commercial data available.

2.4 Conservation Recommendations

Conservation recommendations are defined as "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). 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. The conservation recommendations listed below are consistent with these obligations, and therefore should be implemented by the PNF.

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

2. The PNF should investigate the utility of alternative forms of weed control that do not involve the use of chemicals toxic to aquatic organisms. Examples of alternatives include substitution of vinegar or acetic acid formulations for spot-spraying weeds, and use of steam or other heat-killing methods.

In order for NOAA Fisheries to be kept informed of actions minimizing or avoiding adverse effects, or those that benefit listed species or critical habitat, NOAA Fisheries requests notification of the achievement of any conservation recommendations when the action agency submits its monitoring report describing action under this Opinion or when the project is completed.

2.5 Reinitiation of Consultation

As provided in 50 CFR 402.16, reinitiation of formal consultation is required if: (1) The amount or extent of taking specified in the Incidental Take Statement is exceeded, or is expected to be exceeded; (2) new information reveals effects of the action may affect listed species in a way not previously considered; (3) the action is modified in a way that causes an effect on listed species that was not previously considered; or (4) a new species is listed or critical habitat is designated that may be affected by the action. In instances where the amount or extent of incidental take is exceeded, any operations causing such take must cease, pending conclusion of the reinitiated consultation.

2.6 Incidental Take Statement

The ESA at section 9 [16 USC 1538] prohibits take of endangered species. The prohibition of take is extended to threatened anadromous salmonids by section 4(d) rule [50 CFR 223.203]. Take is defined by the statute as "to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, or collect, or to attempt to engage in any such conduct" [16 USC 1532(19)]. Harm is defined by regulation as "an act which actually kills or injures fish or wildlife. Such an act may include significant habitat modification or degradation which actually kills or injures fish or wildlife by significantly impairing essential behavior patterns, including, breeding, spawning, rearing, migrating, feeding or sheltering" [50 CFR 222.102]. Harass is defined as "an intentional or negligent act or omission which creates the likelihood of injury to wildlife by annoying it to such an extent as to significantly disrupt normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering" [50 CFR 17.3].

Incidental take is defined as "any taking otherwise prohibited, if such taking is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity" [50 CFR 17.3]. The ESA at section 7(o)(2) removes the prohibition from incidental taking that is in compliance with the terms and conditions specified in a section 7(b)(4) incidental take statement.

An incidental take statement specifies the impact of any incidental taking of endangered or threatened species. It also provides reasonable and prudent measures that are necessary to minimize impacts and sets forth terms and conditions with which the action agency must comply in order to implement the reasonable and prudent measures.

2.6.1 Amount of Extent of Take

The proposed action is reasonably certain to result in incidental take of the listed species. NOAA Fisheries is reasonably certain the incidental take described here will occur because: (1) the listed species are known to occur in the action area; (2) the proposed action would kill or harm individual listed salmon and steelhead through lethal or sub-lethal exposure to herbicides, respectively, as a result of accidental spills, failure of BMPs to keep chemical concentrations below expected levels, unexpected toxic effects that have not been reported in the scientific literature, or additive or synergistic effects of herbicides from multiple sources in the action area; and (3) the proposed action would adversely affect availability of invertebrate prey through toxic effects of herbicides on primary productivity and invertebrate prey.

Despite the use of best scientific and commercial data available, NOAA Fisheries cannot quantify a specific amount of incidental take of individual fish or incubating eggs for this action. Instead, the quantity of take depends on the circumstances at the specific locations where treatments will occur (which are not known at this time). In circumstances where the amount of take cannot be quantified, the extent of incidental take is described (50 CFR 402.14 [I]). The extent of take in the action area is anticipated to be no more than 626 acres (acreage proposed for treatment, see Table 2), and NOAA Fisheries anticipates that take will not occur in all of streams within the treatment areas.

2.6.2 Reasonable and Prudent Measures

Reasonable and Prudent Measures (RPMs) are non-discretionary measures to minimize take, that may or may not already be part of the description of the proposed action. They must be implemented as binding conditions for the exemption in section 7(o)(2) to apply. The PNF has the continuing duty to regulate the activities covered in this incidental take statement. If the PNF fails to adhere to the terms and conditions of the incidental take statement, or fails to retain the oversight to ensure compliance with these terms and conditions, the protective coverage of section 7(o)(2) may lapse. NOAA Fisheries believes that activities carried out in a manner consistent with these RPMs, except those otherwise identified, will not necessitate further site-specific consultation. Activities which do not comply with all relevant RPMs will require further consultation.

NOAA Fisheries believes that the following RPMs are necessary and appropriate to minimize take of listed fish resulting from implementation of the action. These RPMs will also minimize adverse effects on designated critical habitat.

1. The PNF shall minimize the amount and extent of incidental take from use of herbicides by implementing precautionary measures that keep chemicals out of water.

2. The PNF shall monitor and report on the effectiveness of the proposed conservation measures in minimizing incidental take, and report this information to NOAA Fisheries.

3. The PNF shall report to NOAA Fisheries the activities actually completed during the 2003 treatment season.

2.6.3 Terms and Conditions

To be exempt from the prohibitions of section 9 of the ESA, the action must be implemented in compliance with the following terms and conditions, which implement the RPMs described above for each category of activity. These terms and conditions are non-discretionary.

1. To Implement RPM #1, above, the PNF shall:

    a. Implement all BMPs described in the Proposed Action section of this Opinion.

    b. Use only Rodeo® (glyphosate) within roadside ditches that drain directly into Critical Habitat and/or stream channels inhabited by steelhead or chinook salmon.

    c. Avoid the use Tordon 22K® (Picloram), Transline® (Clorpyralid), and Escort® (Metsulfuron Methyl) within annual floodplains where the water table is within six feet of the surface and soil permeability is high (silt loam and sand soils).

    d. Ensure all chemical storage, chemical mixing, 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, unprotected ephemeral waterway, or wetland.

2. To implement RPM #2, above, the PNF shall:

    a. Implement a monitoring strategy that includes:

        (1) Drift monitoring with use of spray cards on a representative sample of streams.

        (2) Monitoring of non-target plant mortality in riparian areas to determine if mortality of non-target plants is affecting riparian functions in NOAA Fisheries' matrix (NMFS 1996).

    b. Report monitoring results to NOAA Fisheries after the 2003 field season, and prior to 2004 weed control activities, if a similar action is proposed in the following season.

3. To implement RPM #3, above, the PNF shall:

    a. Report to NOAA Fisheries the actual number of acres treated, the chemicals used, application method, and location of treatment sites.

3. MAGNUSON-STEVENS FISHERY CONSERVATION AND MANAGEMENT ACT

3.1 Background

The objective of EFH consultation is to determine whether the proposed action may adversely affect designated EFH for relevant species, and to recommend conservation measures to avoid, minimize, or otherwise offset potential adverse effects to EFH resulting from the proposed action.

3.2 Statutory Requirements

The MSA, as amended by the Sustainable Fisheries Act of 1996 (Public Law 104-267), established procedures designed to identify, conserve, and enhance EFH for those species regulated under a Federal fisheries management plan.

Pursuant to the MSA:

- Federal agencies must consult with NOAA Fisheries on all actions, or proposed actions, authorized, funded, or undertaken by the agency, that may adversely affect EFH (section 305(b)(2)).

- NOAA Fisheries must provide conservation recommendations for any Federal or state action that may adversely affect EFH (section 305(b)(4)(A));

- Federal agencies must provide a detailed response in writing to NOAA Fisheries within 30 days after receiving EFH conservation recommendations. The response must include a description of measures proposed by the agency for avoiding, mitigating, or offsetting the impact of the activity on EFH. In the case of a response that is inconsistent with NOAA Fisheries EFH conservation recommendations, the Federal agency must explain its reasons for not following the recommendations (section 305(b)(4)(B)).

The EFH means those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity (MSA section 3). For the purpose of interpreting this definition of EFH: waters include aquatic areas and their associated physical, chemical, and biological properties that are used by fish and may include aquatic areas historically used by fish where appropriate; substrate includes sediment, hard bottom, structures underlying the waters, and associated biological communities; necessary means the habitat required to support a sustainable fishery and the managed species' contribution to a healthy ecosystem; and "spawning, breeding, feeding, or growth to maturity" covers a species' full life cycle (50 CFR 600.10). Adverse effect means any impact which reduces quality and/or quantity of EFH, and may include direct (*e.g.*, contamination or physical disruption), indirect (*e.g*., loss of prey or reduction in species fecundity), site-specific or habitat-wide impacts, including individual, cumulative, or synergistic consequences of actions (50 CFR 600.810).

The EFH consultation with NOAA Fisheries is required for any Federal agency action that may adversely affect EFH, including actions that occur outside EFH, such as certain upstream and upslope activities.

The objectives of this EFH consultation are to determine whether the proposed action may adversely affect designated EFH and to recommend conservation measures to avoid, minimize, or otherwise offset potential adverse effects on EFH.

3.3 Identification of EFH

Pursuant to the MSA the Pacific Fishery Management Council (PFMC) has designated EFH for three species of Federally-managed Pacific salmon: chinook (*O. tshawytscha*); coho (*O. kisutch*); and Puget Sound pink salmon (*O. gorbuscha*)(PFMC 1999). Freshwater EFH for Pacific salmon includes all those streams, lakes, ponds, wetlands, and other waterbodies currently, or historically accessible to salmon in Washington, Oregon, Idaho, and California, except areas upstream of certain impassable man-made barriers (as identified by the PFMC 1999), and longstanding, naturally-impassable barriers (i.e., natural waterfalls in existence for several hundred years). Detailed descriptions and identifications of EFH for salmon are found in Appendix A to Amendment 14 to the Pacific Coast Salmon Plan (PFMC 1999). Assessment of potential adverse effects to these species' EFH from the proposed action is based, in part, on this information.

3.4 Proposed Action

The proposed action is detailed above in Section I.B. of this Opinion. The action area consists of all rivers and streams on PNF lands in the following 4th code HUCs: Lower Salmon (17060209), Little Salmon (17060210), Middle Salmon-Chamberlain (17060207), Lower Middle Fork Salmon (17060206), and Upper Middle Fork Salmon (17060205); and rivers and stream downstream from PNF lands that potentially receive herbicide inputs through direct contamination, runoff, or percolation. The action area contains designated EFH for chinook salmon.

3.5 Effects of the Proposed Action on EFH

The effects on EFH-listed fish species are the same as previously described in Section II.B.1 of this Opinion for ESA-listed fish species. The proposed activities may result in detrimental effects on water quality (chemical contamination). Herbicide concentrations are expected on occasion to reach concentrations where salmon would be harmed by exposure to toxic chemicals, or through effects of toxic chemicals on the species' prey.

3.6 Conclusion

NOAA Fisheries concludes that the proposed action may adversely affect EFH for Pacific salmon.

3.7 EFH Conservation Recommendations

Pursuant to section 305(b)(4)(A) of the MSA, NOAA Fisheries is required to provide EFH conservation recommendations to Federal agencies regarding actions that would adversely affect EFH. The conservation measures proposed for the project by the PNF, all Conservation Recommendations outlined above in section II.D. and all of the RPMs and the Terms and Conditions contained in sections IV.B and IV.C. are applicable to EFH. Therefore, NOAA Fisheries incorporates each of those measures here as EFH recommendations.

3.8 Statutory Response Requirement

Pursuant to the MSA (section 305(b)(4)(B)) and 50 CFR 600.920(j), Federal agencies are required to provide a detailed written response to NOAA Fisheries' EFH conservation recommendations within 30 days of receipt of these recommendations. The response must include a description of measures proposed to avoid, mitigate, or offset the adverse impacts of the activity on EFH. In the case of a response that is inconsistent with the EFH conservation recommendations, the response must explain the reasons for not following the recommendations, including the scientific justification for any disagreements over the anticipated effects of the proposed action and the measures needed to avoid, minimize, mitigate, or offset such effects.

3.9 Consultation Renewal

The PNF must reinitiate EFH consultation with NOAA Fisheries if the proposed action is substantially revised in a manner that may adversely affect EFH, or if new information becomes available that affects the basis for NOAA Fisheries' EFH conservation recommendations (50 CFR 600.920(l)).

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APPENDIX A - Description of Herbicide Properties

Transline® (clopyralid) - 3,6-dichloro-2-pyridinecarboxylic acid, Monoethanolamine salt, is an auxin growth regulator that acts as a synthetic auxin or hormone, altering a plant's metabolism and growth characteristics. Clopyralid is in the pyridine carboxylic acid family which includes herbicides such as picloram. The formulation of Transline® is manufactured by Dow Agro and contains 40.9% clopyralid as the monoethanolamine salt and 59.1% inert ingredients. The inert ingredients in Transline® are water, isopropyl alcohol, and polyglycol 26-2 (USFS 1995a). Technical grade clopyralid contains hexachlorobenzene and pentachlorobenzene as contaminants. Hexachlorobenzene and pentachlorobenzene contaminate clopyralid during its manufacture. Hexachlorobenzene is a compound that has shown carcinogenic activity in three mammalian species and has been classified as a potential human carcinogen by the United States (U.S.) Environmental Protection Agency (EPA) (SERA 1999a; Cox 1998a). Nominal or average concentrations of hexachlorobenzene are less than 2.5 mg/L. Nominal or average concentrations of pentachlorobenzene are less than 0.3 mg/L (Lade 1998).

Clopyralid is registered by the EPA as a general use pesticide (GUP) in the U.S. and is used to control many types of broadleaf weeds. The registered use rate is 0.0625 to 4.0 lbs. of acid equivalent (a.e.)/acre (USFS 1995a).

Clopyralid may be persistent in soils under anaerobic (no oxygen) conditions and in soils with a low microorganism content. The half-life in soil can range from 15 to 287 days (USFS 1995a). Soil microorganisms break down clopyralid. The only degradation product that has been identified is carbon dioxide. Clopyralid is highly soluble in water, does not adsorb to soil particles, and is not readily decomposed in some soils, and may leach into ground water. Ground water may be contaminated if clopyralid is applied to areas where soils are very permeable and the water table is shallow (USFS 1995a). There is a potential for clopyralid to contaminate ground water if it is applied to soils containing sinkholes or severely fractured surfaces. Clopyralid is of low toxicity to fish and aquatic invertebrate animals.

Clopyralid does not bind tightly to soil and thus would seem to have a high potential for leaching. While there is little doubt that clopyralid will leach under conditions that favor leaching - i.e., sandy soil, a sparse microbial population, and high rainfall - the potential for leaching or runoff is functionally reduced by the relatively rapid degradation of clopyralid in soil (SERA 1999). A number of field lysimeter studies and one long-term field study indicate that leaching and subsequent contamination of ground water is not likely to be substantial. This conclusion is also consistent with a short-term monitoring study of clopyralid in surface water after aerial application.

The most relevant monitoring data for this exposure scenario is the study by Leitch and Fagg (1985) in which clopyralid (LONTREL L) was aerially applied at a rate of about 2.5 lb a.i./acre over 56 hectares - i.e., about 140 acres [56 ha × 2.471 acres/ha = 138.376 acres] As detailed in worksheet B07, this application rate is equivalent to an application rate of about 1.90 lb a.e./acre.

Clopyralid was monitored in stream water during application and subsequently for 72 hours after application at a site 0.5 kilometers downstream from the application site (see Leitch and Fagg 1985, Figure 2, p. 203). The limit of detection in this study was 0.001 mg/L. During and immediately after application, only trace levels of clopyralid were detected in the stream water, suggesting that direct spray of the stream was negligible. The highest levels of clopyralid occurred during or shortly after storm events - i.e., rainfall at hourly rates of about 1 to 20 mm/hour. The maximum level in the stream water was 0.017 mg/L. This occurred shortly after the initial rainfall event during which the highest rainfall rate was about 4 mm/hour. Heavy rainfalls during the following 24 hours resulted in much lower levels of clopyralid in the water. Generally, the monitored concentrations of clopyralid were near the limit of detection of 0.001 mg/L. While Leitch and Fagg (1985) do not provide a tabular summary of the data, visual inspection of Figure 2 (p. 203) in the publication suggests that 0.004 mg/L, the geometric mean of the range from 0.001 mg/L to 0.017 mg/L [$(0.017 \times 0.001)_{0.5}$], is a reasonable estimate of a central value for the concentration of clopyralid in stream water.

For this risk assessment, the monitoring data from Leitch and Fagg (1985) are used to estimate the concentrations in ambient water that could be associated with the application of clopyralid. As detailed in worksheet B07, the central estimate is 0.0021 mg/L per lb a.e. clopyralid that is applied per acre - i.e., units of (mg/L) ÷ (lb a.e./acre). The range for this estimate is from 0.00053 to 0.0089 (mg/L) ÷ (lb a.e./acre).

This approach is clearly extremely conservative. The study by Leitch and Fagg (1985) involved the application of about 266 lbs a.e.. This is about the total amount of clopyralid used by the Forest Service during all of 1997 when clopyralid was applied as the sole herbicide - i.e., 303.43 lbs from Table 2-2. In addition, the monitoring data are from only a relatively brief period after application but are used to estimate longer term exposures for humans. For the characterization of potential human health effects (section 3.4), nonetheless, this extremely conservative approach makes no difference because the exposure levels are far below those of toxicological concern.

Clopyralid also has a low level of toxic risk to aquatic species based on field studies. At application rates of 1 lb. per acre, the observed contamination in water was about 50 times lower than the lowest LC_{50} (concentration where 50% of test organisms are killed) for aquatic animals (.0021 mg active ingredient per liter, or a.i./L). For fish, only 96-hour toxicity bioassays are available with the lowest reported LC_{50} for clopyralid being 103 mg a.i./L. Macro-invertebrates may be less sensitive with LC_{50} s of 232 mg a.i./L (SERA 1999). The effects of clopyralid seem to be the greatest on aquatic plants. The lowest reported effective concentration expected to cause a biological effect on 50% of a group of test animals (EC_{50}) for growth inhibition of green algae is 6.9 mg/L (SERA 1999). The EC_{50} for growth inhibition in duckweed was measured at 89 mg/L (DOW AgroSciences 1998). At lower concentrations growth of other aquatic macrophytes is stimulated (Forsyth et al. 1997). Clopyralid does not build up (bioaccumulate) in fish tissues. The acute toxic level for daphnia is an LC_{50} of 232 mg/L.

The chemical formulation of Transline® includes isopropyl alcohol which is cited on EPA inert ingredient list three (Inerts of unknown toxicity for which there is little concern about toxicity) (USFS 1995a). Excessive exposure to isopropyl alcohol may cause eye, nose and throat irritation and at prolonged (hours) and high exposures may cause a lack of coordination, confusion, low blood pressure, low body temperature, circulatory collapse, respiratory arrest and even death. The manufacturer has not revealed the identity of the surfactants used in the formulated products (USFS 1995a).

Weedar® 64 (2,4-D) - Acetic acid, (2,4-dichlorophenoxy)-, dimethylamine salt, is a registered product of Nufarm, Inc. Weedar® 64 is registered by the EPA as a GUP in the U.S. and is used to control many types of broadleaf weeds. Weedar® is toxic to most broad leaf crops, especially cotton, tomatoes, beets, and fruit trees. The registered use rate is 0.475 to 3.8 pounds a.i./acre, and the method of application may be aerial and ground spraying, lawn spreaders, cut surface treatments, foliar spray, basal bark spray, or injection (Extoxnet 1996a).

2,4-D is a member of the chlorinated phenoxy family and interferes with normal plant growth processes by stimulating nucleic acid protein synthesis, and by affecting enzyme activity, respiration, and cell division. Uptake of the compound occurs through leaves, stems, and roots (Extoxnet 1996a).

There are many forms or derivatives of 2,4-D. Herbicides containing 2,4-D use the amine salt or ester forms of the compound. The amine and ester forms may differ in health-related activity and environmental fate and effects from the parent 2,4-D acid. Unless otherwise noted below, "2,4-D" refers collectively to the acid, amine salt, and ester forms.

Commercially produced 2,4-D contains one or more inert ingredients. The percentage and type depends upon the company creating the product, and whether the compound is an amine salt, ester, or the rarely used pure parent acid form. For example, HiDepr (liquid)® contains dimethylamine salt of 2,4-D (33.2%) and diethanol-amine salt of 2,4-D (16.3%), with ethylene glycol (10%) and other inerts (40.3%).

Depending upon the formulation used, the aquatic ecotoxicity rating can range from Very Highly Toxic to Practically Nontoxic to aquatic organisms (USFS 1995b). For cutthroat trout (*Salmo clarki*), LC_{50} values range between 1.0 and 100 mg/L. The Handbook of Acute Toxicity of Chemicals to Fish and Aquatic Vertebrates (1980) reports a LC_{50} of 64 mg/L for 96 hours for cutthroat trout (95% confidence limit 57-72 mg/L) using 2,4-D acid, granular 100%/ wt 0.3 grams and pH at 7.2-7.5. Channel catfish (*Ictalurus punctatus*) had less than 10% mortality when exposed to 10 mg/L for 48 hours. Green sunfish (*Lepomis cyanellus*), when exposed to 110 mg/L for 41 hours, showed no effect on swimming response. Limited studies indicate a half-life of less than two days in fish and oysters (Extoxnet 1996a).

2,4-D amine salt forms are generally non-toxic to fish. However, studies have also shown that toxicities of two amine salts to fathead minnows (*Pimephales promelas*) did not change after aging test solutions 21 days. Also, fry and fingerlings are considerably more sensitive than eggs to two amine salts of 2,4-D. In fathead minnows, tests with the dimethyl amine of 2,4-D yielded 96-hour LC_{50} values ranging from 320-6300 mg/L for fingerlings and swim-up fry, compared with over 1,400 mg/L for the egg stage. In rainbow trout, tests with dodecyl/tetradodecyl amine against several life stages yielded LC_{50} values (mg/L) of 3.2 for fingerlings, 1.4 for swim-up fry, 7.7 for yolk-sac fry, and 47 for eggs (USFWS 1980). Research has shown bioconcentration in fish tissue. (Cox 1999b, NIH 2002a, Walters 1999).

The 2,4-D compound that is most toxic to fish, particularly juvenile salmonids, is the butoxyethanol ester formulation. Acute LC_{50} values for this particular formulation have been found for chinook salmon fry and smolts of < 0.4 mg/L; juvenile chum salmon (*Oncorhynchus keta*), < 0.8 mg/L; and juvenile pink salmon (*Oncorhynchus gorbuscha*), <1.0 mg/L. Sublethal effects on the growth of juvenile chinook salmon (*Oncorhynchus tshawytscha*) have been investigated. Growth was reduced by using 0.6 mg/L of the butoxyethanol ester formulation. Using the same formulation, physiological stress responses in sockeye salmon occurred at 0.3 mg/L. 2,4-D acid in its pure form at 100 mg/L caused slight mortality in fingerling bream and largemouth bass (NIH 2002a). The 96-hour LC_{50} is reported for the granular form of 2,4-D, acid, for lake trout (*Salvelinus namaycush*) as being 45 mg/L (95% confidence limit 35-56 mg/L), 100%/ wt 0.3 g, pH 7.2-7.5. (USFWS 1980). The 48-hour LC_{50} for rainbow trout is reported at 1.1 mg/L.

Sublethal effects for the amine salt form include the reduction in the ability of rainbow trout to capture food at 5 mg/L (Cox 1999b). Sublethal effects studies showed that on the growth of juvenile chinook salmon was reduced with a concentration of 0.6 mg/L of the butoxyethanol ester formulation. Using the same formulation, physiological stress responses in sockeye salmon (*Oncorhynchus nerka*) occurred at 0.3 mg/L (NIH 2002a). One experimental model studied acute lesions in the area of the kidney that produces red blood cells in tench (*Tinca tinca*) caused by continuous exposure to 2,4-D acid dissolved in water at 400 mg/L. Fifty fish were used; 15 for calculating the LC_{50} and 35 were euthanized in five treatment and two control groups. Tissue samples revealed marked alteration of red blood cells, characterized by progressive swelling and tissue death, and activation of white blood cells. The LC_{50} at 96 hours demonstrated the importance of the species and chemical form used as factors in calculating a product's toxicity (Gomez et. al.1998).

A relationship exists between toxicity and pH level in a waterbody. In one study, the percent of fathead minnows surviving a particular concentration of 2,4-D increased as the pH increased in the water. At a concentration of 7.43 mg/L, 60% of the fish survived in 192 hours at pH 7.6, whereas 100% survived at pH 9.8. At the former concentration, normal schooling behavior was completely disrupted and equilibrium lost after 24 hour exposure. At the latter concentration, neither effect was noted, with pH measured at 8.68 and 9.08. A relationship between pH and the degradation of 2,4-D is present in soil medium, as well (NIH 2002a).

It should be noted that degradation of 2,4-D with varying pH levels varies between forms of 2,4-D. Research has shown the dodecyl/tetradodecyl amine form to be nearly four times more toxic to fathead minnows at certain aquatic pH levels (8.5), while the acid form, a butyl ester form, and a dimethyl ester form were about half as toxic to fish at this same pH level (USFWS 1980).

The fate of 2,4-D may also be affected by several processes including runoff, adsorption, chemical and microbial degradation, photodecomposition, and leaching. In general, 2,4-D has a moderate persistence in soil with a field dissipation half-life of 59.3 days, aerobic half-life of 66 days, and a hydrolysis half-life of 39 days. For some chemicals, such as 2,4-D, the influence of soil pH is mainly responsible for transformation from anionic[6] to nonionic[7] forms with decreasing pH. This can, in turn, affect adsorption. At pH levels below 6.0, 2,4-D is in nonionic form. Increasing the pH above 6.0 turns 2,4-D anionic. In slightly acidic soils, 2,4-D will be adsorbed at a pH level of less than 6.0 but will not be readily adsorbed at a pH level of 7.0 if in the anionic form, because the negative charges of the soil and of the chemical repel each other (Welp and Brommer1999; Walters 1999).

Overall, the persistence of 2,4-D depends upon formulation, pH, soil moisture, soil type, temperature, microbes, and the status of pre-exposure to 2,4-D or its salts or esters (which alter concentrations of 2,4-D applications in the soil). Once in soil, 2,4-D esters and salts are first converted to the parent acid prior to degradation (Walters 1999).

The rate of microbial degradation is dependent upon the water potential, depth, and temperature of the soil. Han and New (1994) found that sandy loam soil containing 2,4-D degrading single-celled filamentous bacteria (actinomycetes) and fungi had the lowest degradation rates at a low water potential, and an increase in water potential resulted in increased rates of breakdown. Dry soil conditions inhibit 2,4-D mineralization by restricting mobility, reducing the degrading activity of organisms, and suppressing the 2,4-D degrading microorganism populations. The rate of microbial degradation decreases with increased soil depths and lower temperatures (Walters 1999).

In coarse-grained sandy soils where both biodegradation and adsorption will be low, or with very basic soils, leaching to groundwater may occur (NIH 2002a). Because of the different formulations, 2,4-D ranges from being mobile to highly mobile in sand, silt, loam, clay loam, and sandy loam. Grover (1977) found that higher volumes of water were required to leach 2,4-D from soils with a high organic content. Leaching was correlated with the pH of soils, with 2,4-D leaching more readily in soils with a pH of 7.5 and higher reflecting higher adsorption to organic matter in more acidic soils.

[6] Negatively charged ion

[7] No charge on the ion

Despite its potential mobility, 2,4-D generally persists within the top few inches of the soil. Walters (1999) applied 2,4-D at the rate of 4.49 kg/ha in the ester form to nursery plots with varying crop covers. The 2,4-D remained in the top 20 cm of the soil.

Timing and intensity of rainfall are important factors in determining the movement and extent of 2,4-D leaching in soil. It was found that 2,4-D is susceptible to runoff if rain events occur shortly after application, with runoff concentrations decreasing over time (Walters 1999). Also, the amount of litter and debris on the soil surface will provide infiltration, as 2,4-D adsorbs to the surfaces of a litter and humus layer.

Norris (1981) states that entry into waterbodies via leaching is not a significant transport method for significant quantities of 2,4-D, since most of it is adsorbed onto organic material and later readily degraded by microbial organisms. Despite assurances such as these, 2,4-D has been detected in groundwater supplies in at least five U.S. states and Canada, and very low concentrations have been detected in surface waters throughout the United States (Extoxnet 1996a).

Persistence of 2,4-D in water is dependent upon the formulation, volatilization, level of nutrients present, pH level, temperature, oxygen content, and previous contamination with 2,4-D or other phenoxyacetic acids. Microbial degradation is a possible route for the breakdown of 2,4-D, but it is very dependent on the characteristics of the water. In the lab, studies have shown that in warm, nutrient rich water previously treated with 2,4-D microbial degradation can be a major factor for dissipation. However, natural surface waters are generally cool with nutrient concentrations less than those needed to maintain 2,4-D degrading microorganism populations. These conditions would not promote the growth of microorganisms needed to achieve microbial degradation (Walters 1999). Microbial activity will play a important role in waters with bottom mud sediments and sludge. Degradation increases with sediment load (Extoxnet 1996a, NIH 2002a).

2,4-D should not be applied directly to water or wetlands, such as swamps, bogs, marshes, and potholes, and the issue of contamination by drift into such areas should be addressed (Extoxnet 1996a).

2,4-D can combine with other pesticides and have a synergistic effect, resulting in increased toxicity. Combining 2,4-D with picloram damages the cells of catfish (*Ictalurus spp*) gills, although neither individual pesticide has been found to cause this damage. Application of the insecticide carbaryl in the same area as 2,4-D ester can result in rainbow trout mortality, as carbaryl increases uptake of 2,4-D (Cox 1999b).

Tordon® 22k (picloram) - 4-Amino-3,5,6-trichloropicolinic acid, potassium salt, is registered to Dow AgroSciences and the EPA lists it as a "Restricted Use" pesticide. Sale and use of these pesticides are limited to licensed pesticide applicators or their employees, only for uses covered by certification. Picloram was placed in this category due to its mobility in water, combined with the extreme sensitivity of many important crop plants (Extoxnet 1996b).

Picloram is registered for control of woody plants and a wide range of broad-leaved weeds. Most grasses are resistant to picloram, so it can be used in range management programs to control bitterweed, knapweed, leafy spurge, locoweed, larkspur, mesquite, prickly pear, and snakeweed on rangeland in the western states. Picloram is formulated either as an acid (technical product), a potassium salt, a triisopropanolamine (TIPA) salt, or an isooctyl ester, and is available as either soluble concentrates, pellets, or granular formulations (Extoxnet 1996b). The registered use rate depends upon the plant(s) and formulation:

1. Picloram, TIPA salt: 0.27 to 2.16 pounds a.e per acre (lb a.e./A).

2. Picloram, isooctyl ester: used for basal bark treatment only.

3. Picloram, potassium salt: 1.0 to 8.5 lb a.e./A.

Picloram is a pyridine carboxylic acid herbicide. Other herbicides in this class include clopyralid, quinclorac and thiazopyrs. It is absorbed by the plant roots, leaves, and bark. It moves both up and down within the plant, and accumulates in new growth, interfering with the plant's ability to make proteins and nucleic acids (Extoxnet 1996b).

Both Grazon® PC and Tordon® K contain essentially the same amount of picloram (potassium salt) at 24.4%. “Inert ingredients”, include water and dispersing agents, including surfactants, at 75.6% (Extoxnet 1996b). Additionally, picloram contains hexachlorobenzene a compound that has shown carcinogenic activity in three mammalian species and has been classified as a potential human carcinogen by the U.S. EPA (SERA 1999a; Cox 1998a). Hexachlorobenzene contaminates picloram during its manufacture; as part of picloram’s reregistration, concentrations of hexachlorobenzene were certified by its manufacturer to be no more than 100 mg/L.

The parent acid is characterized as moderately toxic to freshwater fish, with a LC_{50} of 5.5 mg/L and slightly toxic to freshwater invertebrates (LC_{50} of 34.4 mg/L). The parent material has been tested on rainbow trout in various life stages, yielding a 96-hour LC_{50} of 8.0 mg/L for the yolk sac stage, 8.0 mg/L for the swim-up stage, and 11.0 mg/L for the fingerling stage (Extoxnet 1996b; USGS 2001). Field runoff studies conducted with cutthroat trout conclude that concentrations as low as 290 µg/l and 610 µg/l of the parent acid will affect survival & growth of cutthroat trout. Examining the toxicity of the individual picloram formulations, the EPA characterizes picloram TIPA salt as slightly toxic to freshwater fish, with a LC_{50} of 25 mg/L. A test with coho salmon yielded a LC_{50} of 20 mg/L (EPA 1995). The reported 96-hour LC_{50} for the isooctyl ester in rainbow trout is 4 mg/L, and in channel catfish is 1.4 mg/L, giving it a “moderate toxicity” rating. Other LC_{50} values in aquatic invertebrates ranged from 10 to 68 mg/L (Extonet website). The picloram potassium salt is characterized by the

EPA as “slightly toxic” to freshwater fish, with a LC_{50} of 13 mg/L and “slightly toxic” to freshwater invertebrates (LC_{50} of 68.3 mg/L). Fish early-life stage and Life-Cycle Aquatic Invertebrate Studies provided Lowest Observed Effect Concentrations (LOECs) of 0.88 mg/L and 18.1 mg/L, respectively (EPA 1995). In static tests of the toxicity of picloram acid to cutthroat and lake trout, the 96 hr LC_{50}'s ranged from 25 to 86 mg/L for picloram (Woodward 1976).

In a simulated field study, Mayes (1984) found that concentrations greater than 13 mg/L following rainfall increased fry mortality in cutthroat trout and concentrations greater than 0.61 mg/L decreased growth. In a study with bull trout, no adverse affect was noted from less than 0.29 mg/L (Woodward 1979).

The toxicity of technical picloram, picloram potassium salt, and picloram TIPA salt to aquatic organisms was evaluated in static acute toxicity tests. Species tested were the fathead minnow, rainbow trout, bluegill, and the daphnia (*Daphnia magna*). Rainbow trout was the most sensitive species tested with LC_{50} 96 hour median lethal concentrations of 19.3, 48, and 51 mg/L for the three picloram forms, respectively (all “slightly toxic” ratings). These LC_{50} values are 36-fold greater than picloram concentrations detected in freshwater following application to experimental watersheds (NIH 2002b).

Woodward (1976) found that the rate of yolk sac absorption and growth of lake trout fry was reduced in flow-through tests at concentrations as low as 0.35 mg/L of picloram. His research also indicated that chronic toxicity on early life stages of lake trout is more significant than might be anticipated on the basis of only acute tests with fingerlings (Woodward 1976).

Picloram is not expected to accumulate appreciably in aquatic organisms; the measured bioconcentration factor in bluegill sunfish was less than 0.54 (Extoxnet 1996b).

It should be noted that although most grasses are resistant, picloram is highly toxic to many non-target plants, and there is potential for damaging riparian habitat by spraying too close to a riparian buffer. Picloram is persistent in the environment, and may exist at levels toxic to plants for more than a year after application at normal rates.

One study examined persistence, rainfall induced migration, potential contamination of surface and groundwater, and losses by photodegradation by monitoring treatment sites for 445 days. Picloram was applied to control spotted knapweed on two sites in the Northern Rockies to represent best case and worst case conditions for on site retention of picloram. A valley bottom was treated with 0.28 kg/ha in the spring of 1985 and sampled over 445 days. In the spring of 1986, picloram was applied to both sides of a minimal construction logging road extending 4 km along a stream draining a granitic upper mountain watershed. Of the 17.1 sq km watershed, 0.15% was sprayed. Vegetation, soils, surface water, and groundwater near the road were sampled during the 90 days following application. After 90 days, 78% of the picloram remained in the mountain watershed. It was not detected in the surface water or the groundwater during the 90 days after application. At the valley bottom site, 36, 13, and 10.5% of the picloram persisted after 90, 365, and 445 days. It was concluded that loss by photodegradation was an important factor at both sites during the first seven days (NIH 2002b).

Environmental fate data indicate that picloram is mobile and persistent in laboratory and field studies (EPA 1995). Picloram is classified as moderately to highly persistent in the soil environment, with field half-lives generally from 20 to 300 days. However, some experiments show persistence exceeding five years. The estimated average is 90 days. Photodegradation is significant only on the soil surface and volatilization is insignificant. Degradation by microorganisms is mainly aerobic, and dependent upon application rates. Increasing soil organic matter increases the sorption of picloram and increases the soil residence time. Picloram adsorbs to clay and organic matter and is highly soluble in water. Picloram is poorly bound to soils lacking clay or organic matter, and can be leached out of the soil. These properties, combined with its persistence, mean it may pose a risk of groundwater contamination. Picloram has been detected in the groundwater of 11 states at concentrations ranging from 0.01 ug/l to 49 ug/l (Extoxnet 1996b).

Picloram is water soluble and can be carried by surface run-off. If released in water, it will not appreciably adsorb to sediments, evaporate, or readily hydrolyze. It is subject to photolysis[8], if it is near the water's surface, with reported half-lives ranging from 2.3 to 41.3 days. In laboratory studies, sunlight readily broke down picloram in water, with a half-life of 2.6 days. In the field, herbicide levels in farm ponds were 1 mg/L following spraying and decreased to 0.01 mg/L within 100 days, primarily due to dilution and sunlight (Extoxnet 1996b, NIH 2002b).

Picloram may used alone or mixed into formulations with 2,4-D and applied on deep-rooted perennials on non-cropland, or as pellets or in combination with 2,4-D or 2,4,5-T for brush control. In one study, coho salmon smolts exposed to Tordon 101 (Picloram and 2,4-D) at 0.6 - 1.8 mg/L for 96 hours prevented successful migration upon their release (Wedemeyer 1980).

Rodeo® (glyphosate) - Isopropylamine salt of N-(phosphonomethyl)glycine, isopropylamine salt of glyphosate, is registered to Monsanto Company by the EPA as a GUP. There are several formulations of glyphosate, but the Payette National Forest (PNF) will only use the commercial formulation of glyphosate known as Rodeo®. Glyphosate is a broad-spectrum, non-selective systemic herbicide used to control grasses, herbaceous plants including deep rooted perennial weeds, brush, some broadleaf trees and shrubs, and some conifers. The registered use rate is 0.3 to 4.0 pounds of a.i/acre and may be applied by aerial spraying; spraying from a truck, backpack or hand-held sprayer; wipe application; frill treatment; or cut stump treatment. It is absorbed by leaves, moves rapidly through the plant, acting to prevent production of an essential amino acid that inhibits plant growth. In some plants, glyphosate is metabolized or broken down while other plants do not break it down (Extoxnet 1996c).

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[8] chemical decomposition by the action of radiant energy

Glyphosate itself is an acid, but it is commonly used in salt form (isopropylamine salt). It may also be available in acidic or trimethylsulfonium salt forms. It is generally distributed as water-soluble concentrates and powders (Extoxnet 1996c). Most commercially produced glyphosate, such as Accord® and Rodeo®, contain glyphosate (41.5%) and water (58.5%) , although some brands, such as Roundup,® include a surfactant (polyethoxylated tallowamine surfactant, 15%) (Extoxnet 1996c). The PNF will only use the commercial formula of glyphosate known as Rodeo®.

Glyphosate acid and its salts are classified as "moderately toxic" compounds by the EPA. Technical glyphosate acid (parent compound) is "practically nontoxic" to fish and may be "slightly toxic" to aquatic invertebrates. The 96-hour LC_{50} is 86-140 mg/L in rainbow trout and 120 mg/L in bluegill sunfish. LC_{50} values for chinook salmon and rainbow trout range from 10 mg/L to 220 mg/L within a pH range of 6.3-8.2 (lower LC_{50}'s at higher pH). However, there is a lot of variability in the results of toxicity analysis. The results of a rainbow trout yolk-sac 96-hour LC_{50} static bioassay ranged from 3.4-5.3 mg/L (USGS 2002). In SERA (1996) a reference concentration of 1 mg/L is recommended for assessing the potential for toxic effects in fish. The 48-hour LC_{50} for glyphosate in daphnia (water flea), an important food source for freshwater fish, is 780 mg/L.

There is a very low potential for the compound to build up in the tissues of aquatic invertebrates or other aquatic organisms (Extoxnet 1996c). In one study of bioaccumulation and persistence, glyphosate was applied to two hardwood communities in Oregon coastal forest and none of the 10 coho salmon fingerlings analyzed had detectable levels of the herbicide or its metabolite aminomethylphosphonic acid, although levels were detectable in stream water for three days and in sediment throughout the 55-day monitoring period (NIH 2002c).

Looking at the different formulations, the Accord® and Rodeo® formulations are practically nontoxic to freshwater fish (LC_{50} = >1,000 mg/L) and aquatic invertebrate animals (LC_{50} = 930 mg/L for *Daphnia*). The Roundup® formulation, which contains the surfactant, is moderately to slightly toxic to freshwater fish (LC_{50} = 5-26 mg/L) and aquatic invertebrate animals (LC_{50} = 4-37 mg/L for *Daphnia*). Glyphosate and its formulations have not been tested for chronic effects in aquatic animals (Extoxnet 1996c). The EPA conducted surfactant testing for both cold-water and warm-water fish for glyphosate (EPA 1993). The application rate used was lower than for technical glyphosate. A formulation of 41.2% isopropylamine salt and 15.3 "AA" surfactant provided a rainbow trout LC_{50} of 120 mg/L, which EPA considers to be "practically nontoxic." Bluegill sunfish experienced similar results, with a LC_{50} of greater than 180 mg/L. Bluegill and rainbow trout were found to be similar in sensitivity to the glyphosate formulation containing the "W" surfactant, with LC_{50} values of 150 and >100 mg/L, respectively. Neither rainbow trout (LC_{50} of 240 mg/L) nor bluebill (LC_{50} of 830 mg/L) were very sensitive to the x-77 (.5) surfactant and glyphosate (7.03%) (EPA 1993).

The surfactant MON0818 was tested separately, producing an LC_{50} of 13 mg/L for channel catfish, indicating that it is slightly toxic for catfish, who appear to be the most tolerant to this surfactant. Rainbow trout are the most sensitive, with a LC_{50} of 0.65 mg/L, classifying this as highly toxic. Based upon the available data, products containing MON0818 must include the statement: "This pesticide is toxic to fish." (EPA 1993).

In the aquatic environment with freshwater fish, toxicity appears to increase with increasing temperature and pH. As reported in the Handbook of Acute Toxicity of Chemicals to Fish and Aquatic Invertebrates (USFWS 1980), glyphosate was twice as toxic to rainbow trout at 17°C than at 7°C. With bluegills, toxicity was twice as toxic at 27°C compared to 17°C. Toxicity was also two to four times greater to bluegills and rainbow trout at a pH level of 7.5 to 9.5 than at pH 6.5 (pH of 7.0 is considered "neutral water").

Glyphosate is classified as moderately persistent in soil, with an estimated average half-life of 47 days. Field half-lives range from 1 to 74 days. It is strongly adsorbed to most soil types, including types with low organic and clay content. Therefore, even though it is also highly soluble in water, it has a low potential for runoff (except as adsorbed to colloidal matter) and leaching. One study estimated that two percent of the applied chemical was lost to runoff.

Microbes appear to be the primary pathway for degradation of phyphsate (biodegradation), while volatilization or photodegradation (photolysis) losses are negligible (Extoxnet 1996c). Under laboratory conditions, glyphosate has been rapidly and completely biodegraded by soil microorganisms under both aerobic and anaerobic conditions. In one study, after 28 days under aerobic conditions, 45% to 55% of the glyphosate was mineralized using Ray silt loam soil, Lintonia sandy loam soil, and Drummer silty clay loam soil. Norfolk sandy loam mineralized glyphosate at a much slower, but still significant, rate. Data indicate half-life values of 1.85 and 2.06 days in Kickapoo sandy loam and Dupo silt loam, respectively (EPA 1993).

Although glyphosate has a low propensity for leaching, it can enter water bodies by other means, such as overspray, drift, and erosion of contaminated soil. Once in water, glyphosate is strongly adsorbed to any suspended organic or mineral matter and is then broken down primarily by microbes. Sediment adsorption and/or biodegradation represents the major dissipation process in aquatic systems. Half-lives in pond water range from 12 days to 10 weeks (Extoxnet 1996c).

Evidence from studies suggest that glyphosate levels first rise and then fall to a very low, or even undetectable level, in aquatic systems. After glyphosate was sprayed over two streams in rainy British Columbia, levels in the streams rose dramatically after the first rain event, 27 hour post-application, and fell to undetectable levels 96 hours post-application. The highest glyphosate residues were found in sediments, indicating strong adsorption characteristics of this herbicide. Residues persisted for the entire 171-day monitoring period. It was found that suspended sediment is not a major mechanism for glyphosate transport in rivers (NIH 2002c).

Questions have been raised about the role photodegradation plays once glyphosate is in a waterbody, particularly when laboratory versus field conditions are involved. The EPA states in the Registration Eligibility Document (1993) that glyphosate is stable to photodegradation in pH 5, 7, and 9 buffered solutions under natural sunlight.

Escort® (Metsulfuron methyl) - methyl 2-[[[[(4-methoxy-6-methyl-1,3,5-triazin-2-yl) -amino]carbonyl] -amino]-sulfonyl] benzoate, commonly known as Escort. Escort contains metsulfuron methyl (60%) and inert ingredients (40%). The registered use rate is 0.33 - 4.0 ounces of active ingredient per acre for non-cropland uses. Metsulfuron methyl is absorbed through the roots and foliage and moves rapidly through the plants. It inhibits cell division in the roots and shoots, which stops growth.

Metsulfuron methyl is generally active in the soil. It is usually absorbed from the soil by plants. The adsorption of metsulfuron methyl to soil varies with the amount of organic matter present in the soil, and with soil texture and pH. Adsorption to clay is low. Metsulfuron methyl remains unchanged in the soil for varying lengths of time, depending on soil texture, pH and organic matter content. The half-life of metsulfuron methyl can range from 120 to 180 days (in silt loam soil). Soil microorganisms break down metsulfuron methyl to lower molecular weight compounds under anaerobic (no oxygen) condition. Metsulfuron methyl in the soil is broken down to nontoxic and nonherbicidal products by soil microorganisms and chemical hydrolysis. Metsulfuron methyl dissolves easily in water. Metsulfuron methyl has the potential to contaminate ground water at very low concentrations. Metsulfuron methyl leaches through silt loam and sand soils. Because metsulfuron methyl is soluble in water, there is a potential for surface waters to be contaminated if metsulfuron methyl is applied directly to bodies of water or wetlands. Tests show that the half-life for metsulfuron methyl in water, when exposed to artificial sunlight, ranges from one to eight days.

There are major areas of uncertainty and variability in assessing potential levels of exposure in soil. In general, metsulfuron methyl absorption to a variety of different soil types will increase as the pH decreases (i.e., the soil becomes more acidic). The persistence of metsulfuron methyl in soil is highly variable, and reported soil half-times range from a few days to several months, depending on factors like temperature, rainfall, pH, organic matter, and soil depth.

In order to encompass a wide range of field conditions, Groundwater Loading Effects of Agricultural Management Systems simulations were conducted for clay and sand at annual rainfall rates ranging from 5 to 250 inches and the typical application rate of 0.02lbs a.i./acre. In sand or clay under arid conditions (i.e., annual rainfall of about 10 inches or less) there is no percolation or runoff and the rate of decrease of metsulfuron methyl concentrations in soil is attributable solely to degradation rather than dispersion. At higher rainfall rates, plausible concentrations in soil range as high as 0.007 mg/L, and under a variety of conditions, concentrations of 0.0005 mg/L and greater may be anticipated in the root zone for appreciable periods of time. Metsufuron methyl exposure to aquatic species is affected by the same factors that influence terrestrial plants, except the directions of the impact are reversed. In other words, in very arid environments (i.e., where the greatest persistence in soil is expected) substantial contamination of water is unlikely. In areas with increasing levels of rainfall, toxicologically significant exposures to aquatic plants are more likely to occur.

These estimates of persistence in soil and transport to water should be considered only as crude approximations of plausible levels of exposure. A substantial impact on these assessments could result from a variety of site-specific factors, particularly, application rate, microbial activity, soil binding of metsulfuron methyl, depth of water table, proximity to open water, and rates of flow in and volumes of groundwater, streams, ponds, or lakes, and specific patterns of rainfall. These site-specific considerations could lead to substantial variations from the modeled values upward or downward.

Analysis using lethal assays indicates that metsulfuron methyl tends to be much more toxic to aquatic plants than to aquatic animals. Mortality in adult fish is not likely to be observed at concentrations less than or equal to 1000mg/L. For longer-term effects (e.g., hatching, larval survival, or larval growth over 90-day exposure period) mortality is observed at much lower levels with a No Observable Effect Concentration (NOEC) of 4.7 mg/L for a corresponding effect level at 8 mg/L. In the USDA risk assessment for metsulfuron methyl the following clinical observations from one study of the lethal and sublethal effects of metsulfuron methyl on rainbow trout are noted: at 24 hours, three fish exposed to 150 mg/L exhibited erratic swimming, rapid breathing and were lying on the bottom of the test container; two out of three fish recovered by 48 hours; the third fish remained affected throughout the entire study.

An acute LC_{50} value of 720 mg/L for immobility and NOEC of 150 mg/L for reproduction was observed for aquatic invertebrates. Aquatic plants are far more sensitive than aquatic animals to the effects of metsulfuron methyl, although there appear to be substantial differences in sensitivity among species of macrophytes and unicellular algae. For macrophytes, the most sensitive species appears to be *Lemna gibba* with a reported EC_{50} value of 0.00036 mg/L and a NOEC value of approximately 0.00016 mg/L. There appears to be substantial variation in the toxicity of metsulfuron methyl to algal species with reported EC_{50} values ranging from about 0.01 to about 1 mg/L.

There is no published information regarding the impurities in technical grade metsulfuron methyl or any of its commercial formulations. Information on all of the impurities in technical grade metsulfuron methyl was disclosed to the U.S. EPA (Brennan 1995), and the information was obtained and reviewed as part of the risk assessment. Because this information is classified as confidential business information, details about the impurities cannot be disclosed. Nonetheless, all of the toxicology studies on metsulfuron methyl involve technical metsulfuron methyl, which is presumed to be the same as or comparable to the active ingredient in the formulation used by the Forest Service. Thus, if toxic impurities are present in technical metsulfuron methyl, they are likely to be encompassed by the available toxicity studies using technical grade metsulfuron methyl.

Escort, the commercial formulation of metsulfuron methyl used by the Forest Service, contains materials other than metsulfron methyl that are included as adjuvants to improve either efficacy or ease of handling and storage. The identity of these materials is confidential. The additives were disclosed to the U.S. EPA (DuPont) 1985b,c) and were reviewed in the preparation of this risk assessment. All that can be disclosed explicitly is that none of the additives are classified by the U.S. EPA as toxic.

As reviewed by Levine (1996), testing requirements for pesticide inerts that have been used as additives or adjuvants for many years are minimal, and this is a general problem in many pesticide risk assessments. For new inerts, the U.S. EPA does require more extensive testing (Levine 1996).

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