Appendix HAZ-2

2017 Herbicide Toxicity Information

HAZ-2 HERBICIDES

1.1 INTRODUCTION

This appendix describes the potential human health and ecological effects from the chemicals that were proposed to be used for vegetation treatments under the 2017 VTP PEIR. The information that follows addresses the use and disposal of borax and herbicides and some of their adjuvants, metabolites, and degradates.

The information presented in this appendix describes information pertaining to the direct effects of chemical use to humans and other life forms and the indirect effects associated with impacts on the environment. The information in this appendix was prepared by CAL FIRE in 2010 and was peer-reviewed, updated in 2015 by Bill Williams, Ph.D. (Williams 2015), and included as Appendix D to the 2017 VTP Draft EIR.

Pertinent information to aid in understanding the chemicals that are likely to be used for control of vegetation is included in this appendix. This appendix also contains information pertaining to the herbicide 2,4-D, which has since been removed from the list of herbicides proposed for use under the proposed VTP based on the results of the analysis in the 2017 Draft EIR.

The following outline will aid in reviewing this appendix:

Outline

1.2 REGULATORY AND POLICY RESTRICTIONS ON THE USE OF CHEMICALS

1.2.1 IN ADDITION TO LAWS AND REGULATIONS CONSTRAINING THE USE AND DISPOSAL OF CHEMICALS DISCUSSED IN THE PEIR CONSTRAINTS HAVE ALSO BEEN PLACED ON THE USE OF CHEMICALS IN THE VTP AND ALTERNATIVES BY CAL FIRE POLICY. CHEMICALS ANALYZED

The chemical active ingredients selected for analysis are those that were most often used in forestry and rangeland applications in California from 2001 to 2010, as reported annually in the California Department of Pesticide Regulation (CDPR) Pesticide Use Report Database (CDPR N.D.a). Analyses of these active ingredients cover the range of potential risks, hazards, unknowns, and uncertainties associated with these active ingredients and the product formulations that contain them. Products that are registered and commonly used in California may be mentioned for example purposes, but mention of any trade names is in no way intended by CAL FIRE to be endorsement of or promotion for the use of pesticide products.

Except for borax, which is a fungicide, and NP9E-based surfactants, the chemicals analyzed in this PEIR appendix are herbicides. For the purposes of this analysis, the term herbicide sometimes includes borax. When the term “chemical” is used, it generally refers to herbicides, the fungicide borax, and/or NP9E-based surfactants.

By policy decision of CAL FIRE, after consultation with CDFG and U.S. FWS, atrazine and atrazine related products were removed from the list of potentially funded chemical active ingredients under the VTP and alternatives and are therefore not analyzed in this PEIR. All formulations of chemical containing 2,4-D were also removed from the list of fundable herbicides, due to toxicological concerns. It should be noted that most VTP treatments will

occur on private property not under the control of CAL FIRE so atrazine and/or 2,4-D might be used by landowners outside of the VTP for initial or maintenance treatments.

Table 4.8-1 lists the chemicals being proposed for use under the VTP and alternatives. Due to the uncertainty regarding which herbicides might be used, as well as when and where the chemicals will be applied, the selection of formulations and adjuvants will be made at the activity-specific planning level.

One active ingredient of boron (sodium tetraborate decahydrate, also known as borax), clopyralid (monoethanolamine salt), hexazinone, imazapyr (isopropylamine salt) and sulfometuron methyl are being proposed for use. Four active ingredients of glyphosate (diammonium salt, dimethylamine salt, isopropylamine salt, potassium salt) and two active ingredients of triclopyr (butoxyethyl ester and triethylamine salt) are also being proposed for use under this PEIR.

This risk assessment will not cover in detail the adjuvants or inert ingredients that have the potential to be used when chemicals are applied for vegetation management, apart from surfactants that are of high toxicological concern. Adjuvants, such as surfactants, are additives that improve the effectiveness of a formulation and are added just prior to application of a formulated product. Surfactants are intended to increase the efficacy of the formulation towards eliminating or retarding the target plant (U.S. EPA, 2011b).

Like adjuvants, inert ingredients are not active in directly eliminating or retarding the growth of the targeted species, but instead improve the effectiveness of the active ingredient (FIFRA Sec. 2 [7 U.S.C. 136](m); U.S. EPA, 2011b). Unlike adjuvants, however, inert ingredients are combined with active ingredients to create formulations that are sold as end-use products. Inert ingredient information is considered proprietary (FIFRA Sec 10(f) and 12(a)(2)(D)) and as such is typically only disclosed by formulation registrants to the U.S. EPA. When registering a formulation that contains inert ingredients, toxicity testing is completed on both the technical grade active ingredient (TGAI) and end-use product, which allows for the toxicity of chemicals to be compared.

One surfactant of concern, nonylphenol ethoxylate (NP9E) contains the active ingredient nonylphenol (NP) and its ethoxylates (USDA/FS, 2003b). Another surfactant of concern contains polyethoxylated tallow amine, which is also known as polyoxyethylene amine or POEA, (SERA, 2011d). Each of these surfactants is made up of many related components, making toxicity ambiguous and challenging to classify. Currently, there is concern regarding the toxicity of NPE and POEA compounds to aquatic organisms (SERA 1997a, 2011b and USDA/FS, 2003b). Estrogen mimicry, a potential for NPE, causes concern for both aquatic and terrestrial organisms. Of the active ingredients proposed for use, NP9E is commonly used with clopyralid, glyphosate and/or triclopyr formulations, whereas, POEA is predominately an unspecified inert in glyphosate formulations. NP9E data can be evaluated more easily than

POEA, given that NP9E is a component of surfactants added after purchase, making information less proprietary.

1.2.2 AREA POTENTIALLY TREATED BY CHEMICALS

For analysis purposes, it is assumed that the area potentially treated with chemicals under the VTP and alternatives is the 20.3-million-acre treatable landscape. The VTP and the alternatives propose to treat approximately 6,000 acres with chemical treatments. Based on area treated, if there are no significant effects from chemical treatments in the VTP, there will be no significant effects in the alternatives.

Chemical treatments will potentially occur only on Local Responsibility Area (LRA) or State Responsibility Area (SRA) lands where CAL FIRE has fire suppression responsibility.

There are two basic activity initiators for chemical treatments under the VTP, either state or private land managers. On state lands (State Forests, State Parks, Ecological Reserves, and Wildlife Areas), VTP subsequent activities are initiated by state agencies (CAL FIRE, the CA Department of Parks and Recreation, or CDFG). There are 71,000 acres of State Forests, ~1,500,000 acres of State Parks, 129,000 acres of CDFG Ecological Reserves, and ~563,000 acres of Wildlife Areas, for a total of ~2,263,000 acres. Because these are public lands, inadvertent exposure of the public to chemicals is potentially greater than on private lands.

On private lands, landowners working in partnership with CAL FIRE are the activity initiators. Use of such lands is not considered public use, as people can legally gain access only by invitation of the landowner. Some potential chemical exposure routes to the public, such as eating berries or coming into direct contact with sprayed vegetation, are therefore unlikely.

Herbicide use may occur on any acre available for treatment under the proposed VTP subject to the constraints outlined in the SPRs, MMs, and any additional constraints identified in the Project Specific Analysis (PSA). It is not possible to know exactly where chemical treatments will be located in the State or how many subsequent activities will be in any bioregion in any given year. The percentage of the VTP area potentially treated each year with chemicals is 10%, or 6,000 acres. The area treated in each alternative is like the VTP, therefore each of the alternatives will have a similar impact (see Chapter 3).

1.2.3 TIMING OF CHEMICAL TREATMENTS

Under the VTP and alternatives, herbicides could be used as the initial vegetation treatment or for maintenance of previously treated areas. In shrubland treatments, herbicides are sometimes applied a year prior to prescribed burns to enhance the flammability of shrubs and reduce emissions from burning, by causing shrubs to die and desiccate (“brown”) before ignition. Initial treatments in shrubland are unlikely to use herbicides independent of some type of follow-up treatment to remove the dead fuels. Noxious weeds could also be controlled primarily by herbicide treatments.

Many of the maintenance treatments are expected to utilize herbicides. As discussed in Chapter 2, maintenance treatments are generally related to vegetation habitat, landscape location, and treatment type. For analysis purposes, maintenance with herbicides is assumed to occur at the following time intervals:

- Grasslands – 2-3 years after previous treatments
- Shrublands – 5-10 years after initial treatment
- Forestland – 10-15 years after initial treatment

These maintenance intervals could vary by as much as 2-15 years for specific vegetation types depending on species composition and site quality. Forestland herbicide treatments to establish regeneration following timber harvesting are typically done only once or twice in a 40-70 year rotation. Treatments with borax are likely to occur only once, immediately after trees are cut in thinning operations.

Because the VTP is based on willing landowner participation, not every acre initially treated by whatever method, will receive a maintenance treatment. Some landowners are not receptive to herbicide use as an initial or maintenenace treatment. Alternatives such as manual, herbivory, and mechanical treatments are also likely to be utilized for maintenance treatment under the VTP. The type of follow up treatment and interval between treatments would depend on site conditions and objectives.

1.2.4 APPLICABILITY OF EXISTING RISK ASSESSMENTS

The human and ecological risks associated with all active ingredients being proposed for use in the program and alternatives have been assessed for the USDA/FS vegetation management program by the Forest Service and Syracuse Environmental Research Associates, Inc. (SERA). A review was made of the USDA/FS program for which the risk assessments (RAs) were prepared and in all cases the VTP and alternatives fall well within the parameters of the Forest Service program, so the conclusions of the risk assessments are generally applicable and there is no need to conduct a new and original RA for each chemical (see Title 14 CCR § 15148). Instead, the conclusions of these risk assessments were used as a basis for identifying known hazards for each chemical being proposed for use. Information from U.S. EPA chemical evaluations was used to both supplement and update materials in the SERA and Forest Service risk assessments. Scientifically accurate information from open literature was added, as referenced below, to elaborate on or update any material in U.S. EPA, SERA, and Forest Service assessments.

The full U.S. EPA and USDA/FS risk assessments are available via the Internet. U.S. EPA documents can either be obtained online through the Agency website (http://www.epa.gov/) or the federal regulations database (www.regulations.gov). The most current USDA/FS risk assessments and associated 2012 Excel workbooks with assessment calculations, which are

typically completed by SERA consultants, can be downloaded at http://www.fs.fed.us/foresthealth/pesticide/risk.shtml.

Consultants at SERA developed specific risk assessment methodologies and programs in collaboration with the Forest Service. SERA consultants, for example, currently use a program associated with the Groundwater Loading Effects of Agricultural Management Systems (GLEAMS), which is referred to as Gleams-Driver version 1.9.3 (see user guide, SERA 2007a), to model the fate of chemicals in the environment. Likewise, a downloadable Excel application called WorksheetMaker was created by SERA and the Forest Service to make human and ecological health risk assessment calculations, and this application is regularly updated as new information becomes available. SERA risk assessment methods and user guides, as well as programs and applications, such as Gleams-Driver and WorksheetMaker, are updated frequently. The most current information can be downloaded directly from the SERA website (www.sera-inc.com).

Human and ecological risk values disclosed in Forest Service chemical risk assessments were determined using calculation and scenario methods current at the time that each assessment was completed; however, risk evaluation methods change frequently. As a result, values in risk assessments do not always reflect the current evaluation methods. Thus, for the assessment of chemicals in this PEIR, risk calculations have been updated using the most current version of the Excel application FS WorksheetMaker (version 6.00.10). Generally, for each chemical, separate workbooks were completed for the typical and upper application rates (lbs/acre) for each category of application method applicable (e.g. backpack directed foliar, ground broadcast foliar, or stump application). For example, four workbooks were created for 2,4-D, because it has different typical and upper application rates and is applied using both backpack directed foliar and ground broadcast foliar application methods.

The only chemical risk calculations not updated using the most current version of WorksheetMaker (6.00.10) are the two chemical impurities of concern, hexachlorobenze and NP9E, as these compounds are not included in the current WorksheetMaker application. Hexachlorobenzene calculations were updated as suggested by Patrick Durkin of SERA Inc. using provided workbooks that were created using WorksheetMaker version 6.00.07 (see workbook revisions tab for details). For NP9E, values were taken directly from the USDA/FS (2003b) risk assessment written by David Bakke, as the worksheets have not been updated. Worksheets completed for the chemicals analyzed in this PEIR are in Oliver, 2012.

All of the chemicals proposed for use in the VTP and alternatives have also been extensively evaluated by the U.S. EPA. Conclusions made by the U.S. EPA (also referred to as the Agency) are usually based on findings from a suite of studies completed by the chemical registrants. Although there is disagreement over the validity of such studies, the Agency enforces stringent guidelines for each type of test required during the registration process (see 40 CFR 158.5 for study requirements). If standard protocols are not followed by the

registrant, or test requirements should change over time, studies are considered unacceptable and must be repeated and resubmitted to the Agency for the active ingredient to become or remain registered. If there have not been changes to standard protocols for a given test since initial chemical registration, then the test will continue to be used any time the active ingredient is re-evaluated by the U.S. EPA. A single study completed by the registrant may qualify to fulfill multiple data requirements, which allows registrants to reduce the number of laboratory animals used. For example, a study may be conducted to evaluate dermal irritation and dermal sensitization. The guidelines and standards set by the Agency ensure some level of consistency and allow for comparability of test results for a particular chemical, as well as between multiple chemicals. The Agency evaluates tests for a given active ingredient and summarizes the findings in various Re-registration Eligibility Decision (R.E.D.), or more recently, in Registration Review Decisions and other chemical assessment documents.

Until recently, the U.S. EPA released copies of registrant studies under the Freedom of Information Act, but now most studies are considered proprietary information and are no longer released to the public. Fortunately, most U.S. EPA guidelines have not changed since the initial submission of acceptable studies, so older studies are still acceptable. All Forest Service risk assessments include detailed information regarding U.S. EPA-submitted studies that were acquired prior to 2011 from the U.S. EPA, as well as directly from the chemical registrants. These Forest Service assessments are typically completed under contract by Dr. Patrick Durkin and other consultants from SERA. When contracted to completed Forest Service risk assessments for active ingredients, SERA consultants evaluate these U.S. EPA studies, as well as, toxicology databases, and an enormous amount of open literature, making them particularly valuable resources.

The U.S. EPA and the Forest Service regularly evaluate and re-evaluate new information regarding the human and ecological risks associated with the chemicals proposed for use under the VTP and alternatives. The U.S. EPA reviews the hazards of pesticide active ingredients, as well as surfactants, inerts and/or metabolites of toxicological concern, during the registration, tolerance, and re-registration evaluation process. Similarly, the Forest Service contracts (i.e.,, usually SERA) to have chemical risk assessments created and updated regularly. The U.S. EPA and USDA/FS risk assessment and review history for each chemical proposed under the VTP and alternatives is as follows:

Borax (tetraborate decahydrate) - A R.E.D. was completed by the U.S. EPA (1993b) for boric acid and its salts. Subsequently, certain aspects of toxicity for boric acid and its salts were re-examined in a Tolerance Re-registration Eligibility Decision (T.R.E.D.) and again when scoping in preparation for a R.E.D. that is expected in 2014 (U.S. EPA 2006e and 2009a respectively). The most recent USDA/FS risk assessment for borax, completed by SERA (2006a), specifically assessed the fungicidal product Sporax®, which is 100% sodium tetraborate decahydrate. Note that Cellu-Treat is also a borax

product registered for use in California, which is 98% disodium octaborate tetrahydrate and 2% water. Cellu-Treat, however, is not proposed for use in this document because the SERA risk assessment does not cover the use pattern of this product.

Clopyralid - While extensive toxicity data was submitted to the U.S. EPA by clopyralid registrants, the Agency has yet to complete or propose a R.E.D. for this active ingredient. Despite this, clopyralid tolerance and acute and chronic toxicity information was released by the U.S. EPA after new clopyralid crop uses were evaluated (FR 2002a, 2002b; U.S. EPA 2009b). The initial USDA/FS risk assessment for clopyralid specifically evaluated the product Transline®, which contains the monoethanolamine salt of clopyralid (SERA 1999). Since then, another assessment of clopyralid was completed by SERA (2004a).

Glyphosate - A R.E.D. has been completed for glyphosate by the U.S. EPA (1993c), though toxicity and tolerances have been re-evaluated several times as a result of additional chemical uses, as well as new glyphosate salts being registered (e.g. FR 2007, 2011; U.S. EPA 2006b, 2006c). Glyphosate was also recently evaluated by the U.S. EPA in scoping documents for a proposed R.E.D. expected in 2015 (U.S. EPA 2009c). As for the USDA/FS, specific glyphosate formulations and surfactants were evaluated in the mid-1990s (SERA 1996a & 1997a respectively). Since then, complete glyphosate risk assessments have been done multiple times (e.g. SERA 2003a). The USDA/FS contracted SERA to update a glyphosate program description, as well as a human and ecological health risk assessment (SERA 2010 & 2011b respectively). Rather than simply evaluating the active ingredient, the most recent assessment for glyphosate considered the relative toxicity of technical grade glyphosate, glyphosate formulations, and the POEA surfactant.

Hexazinone - This chemical was first registered in 1975 and several years later a R.E.D. was completed by the U.S. EPA (1994). Later, some tolerance data was revised due to evaluation changes (U.S. EPA 2002a and 2002b). A U.S. EPA (2010c) registration review for hexazinone is expected in 2016. Initially, SERA (1997b) was only contracted by the Forest Service to evaluate selected formulations of hexazinone, though SERA (2005) later fully assessed the active ingredient.

Imazapyr - Technical grade imazapyr was first registered in 1985, though the first grassland uses were not registered until 2003, as discussed in a recent R.E.D. (U.S. EPA 2006d). A subsequent addendum was released in 2008. A USDA/FS human and ecological health risk assessment was completed for imazapyr, which was later updated (SERA 2004b & 2011c, respectively).

Sulfometuron methyl - This chemical was first registered in 1982, but no tolerance studies have been completed since there are no food or feed uses for this herbicide. A R.E.D. (U.S. EPA 2008a) was done in 2008 and a subsequent amendment was completed in 2009. Initially, SERA (1998b) assessed sulfometuron methyl by evaluating the commercial formulation Oust®, as that was the only sulfometuron methyl product

used by the USDA/FS. Subsequently, SERA (2004c) completed a full assessment of sulfometuron methyl.

Triclopyr - This chemical was most recently evaluated by the Agency in a R.E.D. (U.S. EPA 1998). Similarly, during the mid-1990's, SERA (1996b) assessed commercial formulations of triclopyr (Garlon 3A and Garlon 4). Since then, multiple evaluations of triclopyr have been completed (SERA 2003b & 2011d).

NP9E - A hazard characterization of alkylphenols, including *p*-Nonylphenol (NP) compounds, was completed by the U.S. EPA and subsequently an action plan was conducted specifically for NP and NPE compounds (U.S. EPA 2009f & 2010e). David Bakke, USDA/FS Region 5 Pesticide Use Specialist, evaluated the surfactant NPE in 2003 because it is commonly used in forestry and sometimes as an active ingredient (USDA/FS 2003b).

1.2.5 CHEMICAL APPLICATION RATES

Pesticide product labels are regulated by the U.S. EPA and are required to specify maximum product application rates. These rates are based on the specific composition of the product and the labeled product uses. The concentration and form (e.g. salts, esters or amines) of herbicidal ingredients, the presence or absence of "other" or inert ingredients (including water), and the concentration of other ingredients are all factors that influence the composition and potency of a product. Each product is labeled to be used for controlling specific target species on certain types of sites for particular purposes. Each of these factors also influences the application rates specified on the label. Formulation composition and use factors are both considered to determine all application rates on a product label, including those for specific purposes, as well as the maximum rate for each product.

The proportion of a pesticidal ingredient in a formulation directly influences the labeled application rates. It is disclosed differently on labels depending on the composition of the pesticide. Derivatives, such as salts, esters, or amines, are often formulated with the pesticidal/herbicidal compounds to increase the efficacy of pesticide activity (Hager 2009). For example, formulating glyphosate with a salt compound may allow glyphosate to act against the target plant more effectively, because the salt allows for higher absorption of glyphosate through the waxy cuticle of the plant. The presence or absence of derivatives influences how the proportion of pesticidal ingredients is measured and printed on product labels, with the proportion being expressed as either active ingredient (a.i.) or acid equivalent (a.e.) per pound or gallon. "Active ingredient (a.i.)" is commonly used on labels when pesticidal acid compounds are formulated with derivatives, and the derivatives are included in the proportion of the pesticide. The term a.i. is also used on labels of products when there is only one form of a pesticidal compound sold (*ibid*). By contrast, "Acid equivalent (a.e.)" is used when the proportion includes only the amount of pesticidal parent acid that could be theoretically derived from a formulation containing derivatives (*ibid*). Using a.e., rather than

a.i., allows for easy comparison of pesticide concentrations between products that use the same pesticide but different derivatives.

In the Forest Service risk assessments, active ingredients are often evaluated in terms of estimated expected lower, upper, and typical rates of application, which are based on past USDA/FS use of each active ingredient for forestry related applications (see Table D.2-1). In these assessments, application rates were stated as either pounds of a.i. or pounds of a.e. per acre per treatment, as appropriate. For the analysis in this PEIR, the typical and upper application rates were usually used when updating Forest Service risk assessment calculations using FS WorksheetMaker 6.00.10. While application rates determined by the USDA/FS are used in most cases throughout this PEIR, an adjustment for clopyralid was made because California law mandates lower application rates for this active ingredient. The clopyralid product labels registered in California restrict the application rate of clopyralid to a maximum of 0.25 lbs a.e./ac/year, whereas this chemical can be applied at a maximum rate of 0.5 lbs a.e./ac/year in other states.

Though the maximum rates were calculated for comparison and discussion purposes in SERA risk assessments, the USDA/FS actual application rates in the field usually parallel the typical application rates. The application rates potentially used under the VTP and alternatives are expected to be similar to the typical rates projected by the U.S. Forest Service. The SERA risk assessments often use higher maximum application rates (of a.e. or a.i.) for calculations than are actually allowed by product labels. In such cases, the formulation label will always supersede the upper bound specified in SERA assessments, as law prohibits the use of application rates higher than those written on a label. Conversely, for those products that have higher application rates specified on the label than the applicable risk assessment specifies (i.e., glyphosate Accord® products), the U.S. Forest Service maximum rates will not be exceeded under the VTP and alternatives.

Table D.2-1

Chemical Application Rates Proposed for Use Under the Program & Alternatives [1]

Active Ingredient	Ground Application			References
	Typical Applied (lbs /acre)	Lower Range (lbs /acre)	Upper Range (lbs /acre)	
Borax, sodium tetraborate decahydrate	1.0 a.i.	0.10 a.i.	5.00 a.i.	SERA 2006a, p. 2-2
Clopyralid, monoethanolamine salt	0.25 a.e. [2]	0.10 a.e.	0.25 a.e. [2]	SERA 2004a, p. 2-3

Glyphosate	2.0 a.e.	0.29 a.e.	8.00 a.e.	SERA 2011b, Table 7, p. 16-17 & 289
Hexazinone	2.0 a.i.	0.50 a.i.	4.00 a.i.	SERA 2005, p. 2-4
Imazapyr, isopropylamine salt	0.30 a.e.	0.125 a.e.	1.5 a.e.	SERA 2011c, Table 3 & 4, p. 9 & 133
Sulfometuron methyl	0.045 a.i.	0.03 a.i.	0.38 a.i.	SERA 2004c, p. 2-1
Triclopyr	1.0 a.e.	0.10 a.e.	6.60 a.e. [3]	SERA 2011d, p. 10; SERA 2003b, p. 2-5
NP9E	1.67 a.i.	0.167 a.i.	6.68 a.i.	USDA/FS 2003b, p. 4

a.e. = acid equivalent isomer of active ingredient, a.i. = active ingredient. Typical = refers to the average application rate used by the USDA/FS, high/low = refer to upper and lower application rate limits used by the USDA/FS; [1] Application rates are based on those disclosed in Forest Service risk assessments for each chemical, unless otherwise noted. [2] The typical and upper application rates for Clopyralid are 0.35 and 0.50 lbs a.e. per acre respectively for the USDA/FS. In California, however, the maximum application rate is restricted to 0.25 lbs a.e. per acre per year, and as such 0.25 is conservatively used for both the typical and upper application rate throughout this PEIR; [3] A few uses for triclopyr have application rates as high as 10 lbs. a.e./acre, though 6.63 lbs a.e. per acre was the maximum used by the USDA/FS in 2004.

1.2.6 CHEMICAL PROPERTIES AND MOBILITY

Humans could potentially be exposed to herbicides in several ways, such as direct contact, by contact with or inhalation of spray, by ingestion of contaminated materials (such as vegetation, water, fish and game), or by contact with contaminated vegetation. It is therefore imperative to consider the mobility and persistence of proposed herbicides as well as their rates of absorption and degradation in nature.

For herbicides to adversely affect humans offsite from where the chemicals are applied, they must be able to move from the treatment site in sufficient quantities to expose people to harmful doses. Chemicals are mobile to different degrees and for different lengths of time. Pesticide mobility is greatly affected by microsite conditions, such as soil pH, texture, depth, and organic matter content. Climatic conditions, such as a precipitation, temperature, humidity, and wind speed, may also affect how herbicides spread or drift from the area of application.

The ability of chemicals to affect living organisms over time is determined in part by their persistence in the environment. Persistence is determined for both soil and aqueous environments and is measured by the time it takes for one-half of the chemical to become inactive (degraded) in its ability to affect target species. Persistence in soil is primarily affected by soil texture, climate, and microbial action. Persistence in water is primarily affected by temperature, sunlight, flow, and by the type(s) of sediment in the water. Potential modes of transport of chemicals are as follow:

a) Direct spray of waterbodies, special status species, or receptors
b) Off-site drift of spray to waterbodies and terrestrial areas
c) Runoff of surface water from the application area to off-site waterbodies or soils
d) Accidental spills to waterbodies
e) Contamination of water used for irrigation
f) Infiltration into and leaching through soil to groundwater
g) Wind erosion resulting in deposition of contaminated dust
h) On-site volatilization from sprayed surfaces
i) On-site volatilization by burning of sprayed vegetation

Table D.2-2 displays the differences in mobility of the chemicals potentially used in the VTP and alternatives. This table is a synthesis of information from a number of sources, including USDA/FS and SERA risk assessments, the U.S. EPA, the HFQLG FSEIS (in Appendix F-Environmental Fate of Proposed Herbicides) (USDA/FS 2003a), the Diamond Project DEIS (USDA/FS 2006b), the USFWS (USDI U.S. FWS 2007), CDPR Environmental Fate Reviews (http://www.cdpr.ca.gov/docs/emon/pubs/envfate.htm), and other sources. The ratings in this table are not absolutes and should be taken with caution, as mobility of chemicals is variable and highly complex. Substantially different estimates of mobility could be made when different site-specific factors are considered. Estimates of exposure risk based upon movement of chemicals should be considered only as crude approximations of environmentally plausible consequences.

Table D.2-2

Modes of Off-Site Transport for Chemicals Proposed for Use Under the Program & Alternatives

Chemical	Drift	Volatilization	Runoff	Leaching	Wind [3]
Borax, Sodium Tetraborate Decahydrate	L	L	L	L	M
Clopyralid, Monoethanolamine Salt	L	L	H	L [2]	L
Glyphosate, Diammonium Salt	L	VL	L	L	H
Glyphosate, Dimethylamine Salt	L	VL	L	L	H
Glyphosate, Isopropylamine Salt	L	VL	L	L	H
Glyphosate, Potassium Salt	L	VL	L	L	H
Hexazinone	L	L [4]	H	H	L
Imazapyr, Isopropylamine Salt	L	L	H	H	L
Sulfometuron-Methyl	L	VL	M	M	L

Triclopyr, Butoxyethyl Ester (BEE)	M	M	L	L	M
Triclopyr, Triethylamine Salt (TEA)	L	L	L	L	M

H = high mobility, M = moderate mobility, L = low mobility, VL = very low mobility. [1]Also formerly known as isooctyl ester (U.S. EPA 2005d); [2] Field studies indicate minimal leaching due to rapid degradation in soil; [3] Transport of soil particles by wind; [4] Volatilization of the liquid form of Velpar is higher.

Two models are used to evaluate chemical mobility and fate in Forest Service risk assessments: AgDRIFT® and GLEAMS-Driver (SERA, 2012). AgDRIFT® is a cooperative development effort between the U.S. EPA-ORD, USDA Agriculture Research Service, USDA Forest Service, and the Spray Drift Task Force, a consortium of approximately 42 agricultural chemical registrants. AgDRIFT® was developed to provide the U.S. EPA with an evaluation tool to estimate the environmental exposure from spray drift at the time chemicals are applied. GLEAMS (Groundwater Loading Effects of Agriculture Management Systems), by contrast, is a root zone model developed by the USDA Agricultural Research Service to assess the fate of chemicals applied to a variety of soils under varying hydrogeological and meteorological conditions. Gleams-Driver was developed by the USDA Forest Service in Region 8 as a "*user-friendly Windows program that serves as a pre-processor and post-processor for GLEAMS. It prepares input files for GLEAMS, runs the GLEAMS program, and then reads and processes the output from GLEAMS to make estimates of concentrations of pesticides in soil (target and nontarget fields) as well as surface water (streams and ponds)*" (SERA 2006b). Metabolite information is also sometimes modeled when using Gleams-Driver. Information from AgDRIFT® and GLEAMS-Driver modeling is important to assess exposures relevant to both human and ecological risk assessment

In Forest Service risk assessments, GLEAMS models are used to evaluate how the properties of a chemical influence their spread through the environment. Chemical properties required include foliar, aquatic sediment, soil and water halftimes. Additionally, chemical solubility in water and the fraction of a chemical that washes off of foliage were used. Coefficients relating to chemical concentrations in water and sediment, as well as soil absorption, were also established for USDA/FS models. Since chemical binding to soil is influenced by the specific characteristics of different soils, Forest Service risk assessments usually modeled three soil textures: clay, loam, and sand (SERA, 2006b). Table D.2-3 shows the chemical and site parameters used in the modeling for loam (a combination of clay and sand) in USDA Forest Service risk assessments evaluated for this PEIR. For further details regarding the GLEAMS models refer to the Gleams-Driver User Guide (SERA 2007a) and Modifications to Gleams-Driver Version 1 (SERA, 2006b) documents.

Very small amounts of chemicals are likely to be used under the VTP and alternatives relative to agricultural, urban, and other uses of pesticide. A review of scientific literature on drinking water from forests and grasslands in North America did not identify the chemicals analyzed

in this PEIR “in surface or ground water at sufficiently high concentrations as to cause drinking water problems. Their rapid break down by physical, chemical, and biological routes coupled with use patterns precludes the development of water contamination problems unless they are applied directly to water” (USDA Forest Service 2002a). Chemicals will not be applied directly to water under the VTP and alternatives.

Surface water monitoring conducted in 1999-2002 to measure off-site transport of atrazine, 2,4-D, glyphosate, and triclopyr in the lower Klamath River watershed found that there was no detectable off-site movement of atrazine or triclopyr following the first rainfall event after ground applications. Glyphosate and 2,4-D were not applied by ground application, so those results are not reported here (CDPR, 2003, Table 18, p. 40).

Ground water monitoring conducted in the late 1990s to measure off-site transport of ground applications of hexazinone applied in pellet form at rates of 34.7 and 41.4 kg/ha (31 and 37 lbs/acre) on the Stanislaus National Forest found no detectable amounts in monitoring wells in the first year of application. In the following six years of monitoring in one well, detectable amounts (0.44-3.1 µg/L) were found until the last year of monitoring. In the following four years of monitoring in the other well, detectable amounts (0.16-2.2 µg/L) were found until the last year of monitoring. For both wells, the detectable amount of hexazinone was far less than the California Department of Water Resources’ water quality standard of 400 µg/L. (DeGraff et. al., 2007, p. 359)

Monitoring of a ground application of liquid hexazinone on the Sierra National Forest demonstrated that hexazinone penetrated a significant distance into a 25-foot buffer zone on either side of a Class 4 (CA Forest Practice Rules Class III) channel centerline. It penetrated at least 15 feet into the buffer zone in surface water, at least 10 feet in surface soil, and leached to a depth of 6 feet at 20 feet into the buffer zone. However, the detectable concentrations were a full magnitude lower than the California Department of Water Resources’ water quality goal. “The pattern of mobility at these sensitive sites clearly shows peak concentrations of hexazinone in surface water following the first storm event and a gradual rise to peak concentrations of hexazinone in the vadose zone water after several storm events.” (DeGraff et. al., 2007)

Table D.2-3

Chemical & Site Parameters Used in GLEAMS Modeling

Active Ingredient	Foliar wash-off fraction [1]	Soil Adsorption Coefficient ($K_{o/c}$) [2]	Sediment-Water Distribution Coefficient (K_d) [3]	Water Solubility (mg/L)	Persistence (Half-Life in Days) [4]				Primary Degradation Processes	Reference
					Foliar	Soil	Water	Aquatic Sediment		
Borax, sodium tetraborate decahydrate	1.0	0.11	0.0165	42,700	10,000	10,000	infinity	infinity	NA (not microbial)	SERA 2006a Table 3-1
Clopyralid	0.95	3.15	0.02	1,000	2	25	261	1,000	slow microbial	SERA 2004a, Table 3-1
Glyphosate	0.6	3,100 (2,000-24,000)	420 (18-1,000)	12,000	10	5.4	21	208	slow microbial	SERA 2011b Table 15
Hexazinone	0.9	54	0.59	33,000	30	120	730	230	slow microbial, photolysis	SERA 2005 Table 3-4
Imazapyr, isopropylamine salt	0.9	53 (8-110)	0.64 (0.07-3.4)	11,100	30 (15-37)	2,150 (313-2,972)	19.9-199	5,000	slow microbial	SERA 2011c Table 10
Sulfometuron methyl	0.65	78	0.6	300	10	30	113	60	hydrolysis, microbial,	SERA 2004c Table 3-1

Triclopyr, butoxyethyl ester (BEE)	0.7	1,233 (640-1,650)	NA	7.4	26.9 (16.5-73)	0.2	0.5	4.1 (1.1-15)	hydrolysis, photolysis, microbial	SERA 2011d Table 22
Triclopyr, triethylamine salt (TEA)	0.95	59 (25 to 134)	NA	440	26.9 (16.5-73)	14 (8 - 28.4)	426	6.2 (2.6 – 15)	hydrolysis, photolysis, microbial	SERA 2011d Table 22

[1] Fraction of a chemical on the foliage of plants available for washoff by rainfall; [2] organic carbon partition coefficient; [3] skin permeability coefficient; [4] Time for 1/2 of total chemical applied to be dissipated; NA = Not Available. NP9E has not been analyzed using GLEAMS modeling, hence its absence from this table.

The mobility of chemicals is of particular concern to the Native American community, including the California Indian Basketweavers Association, due to the potential for contamination of plants traditionally used in their culture. Such plants are still used and are gathered by hand, in the traditional manner, primarily on public wildlands and tribal reservations. Plants that are used for weaving baskets are handled with bare hands and are often placed in the mouth at some time during the weaving process. Other plants, or plant parts, are used as food, or have additional uses.

There have been a number of studies in the field to assess the effects of herbicides on plants important to Native Americans. A four-year study from 1997-2001 by the CDPR monitored residues of glyphosate (Accord®), triclopyr (Garlon® 4), and hexazinone (Velpar® L – liquid form & Pronone® 10G – granular form) on bracken fern, buckbrush, golden fleece, and manzanita on three national forests (CDPR 2002). The study also determined herbicide dissipation rates and estimated the potential for off-site movement (Table D.2-4). The half-lives of these chemicals are also well documented (Table D.2-5).

Table D.2-4

The Mean Number of Weeks Observed from the Maximum Herbicide Concentration to the Non-Detectable Level

Herbicide	Plant Part Sampled			
	Bracken Fern Roots	**Buckbrush Shoots**	**Golden Fleece Foliage**	**Manzanita Berries**
Glyphosate	6 weeks	NA[1]	42 weeks	NA
Triclopyr	11 weeks	NA	56 weeks	NA
Hexazinone Velpar® L	4 weeks	130 weeks	20 weeks	6 weeks
Hexazinone Pronone® 10G	29 weeks	4 weeks	15 weeks	8 weeks

Source: CDPR 2002, Table 7. [1] NA means a non-detectable level was not recorded, either because there was no vegetation left to sample, or the non-detect level was never reached.

Table D.2-5

Mean Half-Life of Four Forestry Herbicides In Plant Parts Used by California Indians

	Mean Half-Life for Plant Media Sampled (weeks)			
Herbicide	**Bracken Fern Roots**	**Buckbrush Shoots**	**Golden Fleece Foliage**	**Manzanita Berries**
Glyphosate	11.5 (1) [1]	9.8 (3)	8.2 (2)	NA [2]
Triclopyr	6.1 (2)	2.4 (3)	5.1 (3)	NA
Hexazinone - Velpar® L	18.5 (1)	17.6 (2)	0.6 (2)	NA
Hexazinone - Pronone® 10G	NA	NA	NA	1.7 (1)

Source: CDPR 2002, Table 8. [1] The number in parentheses indicates the sample size used for the calculation of the mean. [2] NA denotes that no meaningful regression could be obtained and, therefore, no mean half-life was calculated.

Herbicide half-lives were variable, ranging from 1 to 19 weeks. The CDPR (2002) study found, "[i]n decreasing order, half-lives were longest for liquid hexazinone, glyphosate, triclopyr, and then granular hexazinone treated plant materials."

It can be seen from these tables that there is considerable variation in the dissipation rates between the herbicides themselves and between the various plant parts that were tested. For example, hexazinone in the granular form had the slowest dissipation rate by far in bracken fern roots and the fastest in golden fleece foliage, while the liquid form had the fastest rate in bracken fern roots and the next to the fastest rate in golden fleece foliage. Residues dissipated most slowly in buckbrush shoots.

The highest residue levels on the day of application were with glyphosate treated plants, followed by those treated with liquid hexazinone, triclopyr, and then granular hexazinone. Although granular hexazinone had the lowest residue level, by the 28th week following application, both liquid and granular hexazinone had similar residue levels in roots, shoots, and foliage.

A residue study in redbud, used for making baskets, following application of hexazinone around the base of plants showed no hexazinone in plant shoots after 0, 4, 8, and 12 weeks. The maximum detectable level of herbicide for redbud is 0.05 ppm (CDPR, 2002). Native Americans are also concerned about herbicide residues in oak acorns, which are used for food. Several studies of residues in acorns have been done by CDPR. Acorns were collected from under trees 28-36 days after spraying with liquid and granular

hexazinone and glyphosate. No herbicide residues were found (at a 0.1 ppm maximum detectable level) for either of the chemicals (*ibid*).

Monitoring conducted in 1999-2002 to measure impacts to culturally significant plants (i.e., beargrass: stems and leaves, huckleberry: berries, yarrow: stems and leaves, manzanita: berries, Oregon grape: roots, willow: shoots, and tanoak: acorns) from off-site transport of 2,4-D and triclopyr in the lower Klamath River watershed found that drift from aerial applications at 10-50 feet above the ground (no ground applications were monitored) of the herbicides was detectable at two of four application sites. The farthest distance that residues were detected on plants was 30 to 41 feet outside the application area, where plant samples averaged 0.14 ppb and 0.10 ppb for triclopyr and 2,4-D, respectively. Dissipation of herbicides after application was monitored over time at six sites in five treatment areas. Plants in four of the sites contained no detectable herbicide residues by approximately 150 days. The other two sites had measurable amounts of herbicide at approximately day 60, but contained no residues at the next sampling date of 370 days (53 weeks). Samples of new growth on plants collected more than a year after application contained no detectable amount of triclopyr or 2,4-D (CDPR 2003).

1.3 DIRECT EFFECTS FROM IMPLEMENTING THE VTP AND ALTERNATIVES

The U.S. EPA and States register and license pesticides in the United States under the authority of the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA). The U.S. EPA is also responsible for issuing “Experimental Use Permits”, required to test an unregistered product. Additionally, the Agency continually reviews new information available on each active ingredient in an attempt to keep pace with new scientific findings and changes in policy and practices. This information is made available to the public in a Re-registration Eligibility Decision (RED) or in Registration Review Decisions. Before a new chemical can be registered, or an existing chemical registered for a new use, the U.S. EPA requires a minimum of 120 different scientific studies and tests from the applicants (usually registrant chemical companies), which can take up to 15 years to complete. These studies are reviewed by the U.S. EPA to determine, with reasonable certainty, that the use of the chemical will not pose a risk to human health or the environment.

State agencies further regulate pesticides per state laws. California State laws that regulate pesticide use, which are enforced by the CDPR, are more restrictive than regulations of the U.S. EPA and most other states. Therefore, pre-registration and registration requirements are more stringent than in other parts of the United States. CDPR reviews the studies submitted to the U.S. EPA and evaluates its findings, as well as State laws, to determine if additional label requirements or studies are needed.

1.3.1 HUMAN HEALTH RISK ASSESSMENT

1.3.1.1 Introduction

There is considerable concern among some members of the public over the long-term health risks of chemicals used in forest and rangeland management. Concern comes from the belief that exposure to even small amounts of these chemicals will result in cancer or other debilitating or life-threatening diseases. It is generally thought that the level of agricultural pesticide use in California is excessive and that any use in forested areas, which are generally the headwaters of much of California's water supply, is increasing the risks to public health.

When considering risk, it should be recognized that nothing we do is risk free. Driving a car, swimming, climbing a ladder, or having a medical X-ray all have risks. Calculations by the U.S. Forest Service of cancer risk to the public from forestry herbicides used on National Forests in the Southeast showed a 1 in 10 million risk (the risk of getting cancer following an X-ray treatment is 7 in 1 million). This estimate is:

> *[B]ased on an extremely conservative approach, which assumed that the herbicides were carcinogenic (cancer causing) and exposure levels were high over long periods of time (70 years). The fundamental assumption of carcinogenicity is subject to much debate and to date no forestry herbicide has been conclusively shown to be carcinogenic (McNabb 1997).*

Although there are risks associated with the use of the chemicals likely to be used in the VTP and alternatives, not using these chemicals will not necessarily result in a higher margin of public safety. Other methods for treating vegetation have their own unique risks to workers and the public. For example, manual methods can lead to worker injuries, manual (chainsaw) and mechanical methods produce greenhouse gases and other pollutants, prescribed fire produces greenhouse gases, smoke pollution, and escaped fires, and prescribed herbivory can increase water pollution. These treatments are generally more expensive than herbicide treatments, and will thus deplete public and private funds more rapidly, resulting in fewer acres treated under the VTP and alternatives. There are risks associated with treating fewer acres due to fiscal constraints, such as less protection from wildfire and fewer acres of noxious invasive plants treated.

Herbicides are designed to kill or retard plants by disrupting or altering one or more of their metabolic processes or by disrupting some physical structure, such as cell membranes. Borax can be used as an insecticide, but under the VTP and alternatives will only be used as a fungicide to prevent infection of heterobasidion root disease in conifers. While few adverse effects to humans or animals are likely, as herbicides primarily affect processes exclusive to plants and borax is a common natural compound found in soil, any chemical in great enough quantities can have adverse effects. Therefore, risk

analyses conducted by the Forest Service and others relating to forestry and rangeland management, were used and referenced for this PEIR.

There are many important factors that must be considered when evaluating the potential risks of chemical use to human health. The level of risk depends on the inherent toxicity of end-use products, additives, and chemical mixes being used in the field. Risk is also dependent on the chemical concentrations, route of exposure, and the duration of chemical exposure. When humans are briefly exposed to pesticides, they may experience acute (short-term) toxicity symptoms, such as irritation of the eyes, skin or throat (causing coughing), as well as headaches and/or dizziness. When humans are exposed to chemicals over longer time periods (sub-chronically or chronically) adverse signs or symptoms of toxicity, such as cancer, the heritable mutations, reproductive issues, and/or neurotoxicity, may be observed. Individuals often respond to chemicals differently, with some being more sensitive than others. Additionally, most conclusions relating human health effects, including chemical toxicity, exposure, and risk characterization, are derived from studies using surrogate mammals, such as rodents, rabbits and dogs. These factors and others add different levels of variability and uncertainty.

With this in mind, USDA/FS risk assessments take a conservative approach when assessing acute and chronic exposure for the public and workers, by using worst case scenarios for each type of exposure (e.g. dermal, consumption of contaminated water). However, it should be kept in mind whenever conclusions of acceptable or minimal risk are presented in this document that **the use of chemicals is never without risk and that precautions should be taken to minimize human exposure to chemicals**. Adequate warning signage, for example, must be posted to lessen exposure to members of the public, while workers applying chemicals must wear personal protection equipment. Mitigation measures (e.g. streamside buffer zones) and additional measures created at the subsequent activity level must be followed to further protect humans.

This PEIR does not specify which herbicide will or should be used in what bioregion. This would be unrealistic, given the immense scope of the VTP and the tremendous variation in vegetation management needs, ecosystems, environmental fate conditions, land use, etc. across the program area. Such decisions will be made on the subsequent activity level. Proposed chemicals are assessed for human health risks based on the assumption that chemicals are used. In this PEIR, herbicides and one fungicide will potentially be used in the VTP and alternatives. Thus, all sections and appendices relating to the use of chemicals are relevant to the VTP and alternatives.

Following the U.S. EPA and the Forest Service protocol, the human health risk assessment in this appendix will follow the four steps established by the National Research Council of the National Academy of Sciences to evaluate both human health risks and ecological effects associated with herbicide use. The steps include 1) hazard

identification, 2) exposure assessment, 3) dose-response assessment, and 4) risk characterization (NRC 1983, as ordered by section in SERA 2012). Hazard identification assesses the toxicity of a given chemical agent to different organisms through different routes, doses and durations of exposure. Exposure assessment evaluates potential routes of exposure to workers and the public and to other organisms. Dose-response assessment evaluates the magnitude of exposure and the likelihood that adverse effects occur due to exposure. The risk characterization sections indicate the magnitude of risk once uncertainty factors are incorporated.

1.3.1.2 Hazard Identification

A suite of studies is commonly completed and/or evaluated by pesticide companies, regulatory agencies, and independent institutions to determine the risks of adverse human health effects related to the use of pesticides. Studies are often conducted to understand the effects of exposure duration (i.e., acute, sub-chronic, and chronic) and dose-response relationships. Other studies are conducted to specifically test for developmental toxicity and reproductive issues or test for mutagenicity and carcinogenicity. Additionally, specific studies are sometimes conducted to consider immunotoxicity, neurotoxicity, and endocrine disruption. Conclusions made by the U.S. EPA and SERA for each of these areas of toxicological concern are summarized throughout this section. Chemical properties, such as how chemical agents are metabolized and moved through the body (pharmacokinetics), are also important to hazard identification. Through these studies, the overall toxicity assessment of active ingredients and new formulations can be completed.

The U.S. EPA requires registrants of any new active ingredient or product to submit human health data from the studies discussed above, for technical grade active ingredient (TGAI), end-use product, and/or manufacturing-use product, depending on what is being registered (see 40 CFR 158.5 for study requirements). In particular circumstances, however, the U.S. EPA waives acute toxicity data requirements or allows registrants to fulfill these requirements with substitution of data from another product (U.S. EPA, 2011a). Waivers may be granted, for example, if an acute oral toxicity study is inappropriate because the chemical exists in only a gas form (*ibid*). By contrast, data substitution (referred to as *data bridging*) may only occur when identical products are registered and re-packaged, or a new formulation is sufficiently similar to an existing formulation (*ibid*).

Given ethical constraints for chemical toxicity testing on human subjects, extensive toxicity tests are conducted on other physiologically similar mammals, primarily rodents, rabbits and dogs (see 40 CFR 158.5). This data on surrogate mammal species then provides a pesticide toxicology profile for each active ingredient. Judgments are made by the scientific community and regulatory agencies regarding the equivalency of the results

to evaluate the potential adverse effects of chemicals towards humans. When available, documented incidents of human poison and human population effects are evaluated in conjunction with mammalian toxicity data.

The Office of Prevention, Pesticides and Toxic Substances (OPPTS), the U.S. EPA department responsible for developing test guidelines relating to pesticides, is now named the Office of Chemical Safety and Pollution Prevention (OCSPP), though guidelines are still often labeled using the acronym OPPTS. The OCSPP recently "harmonized" multiple test guidelines, which are listed and linked to pdf documents at http://www.epa.gov/ocspp/pubs/frs/home/ guidelin.htm. Harmonized human health effects test guidelines are as linked in Series 870. Similarly, U.S. EPA Endocrine Disruptor Screening Program (EDSP) guidelines are in series 890, with current EDSP information found at: http://www.epa.gov/scipoly/oscpendo/index.htm. Alternatively, guidelines can be located directly at www.regulations.gov by OPPTS number.

1.3.1.2.1 Form Equivalency

When initial studies were conducted for the registration of a new active ingredient, chemical and toxicology properties were compared to any similar active ingredients already registered, in order to assess for chemical equivalency. Currently, for example, nine active ingredients exist in the 2,4-D case file, including an acid, salts, and esters, and these forms were all found to have equivalent properties, with only a few exceptions (WHO 1996, 1997, 1998 as referenced in USDA/FS 2006a). Generally speaking, when multiple active ingredients are found to be equivalent in chemical properties and toxicity, the group was discussed generically, such as 2,4-D, without specifying form information. Moreover, in these cases data from one active ingredient form is chosen to represent the group of active ingredients, with any equivalency exceptions being clearly disclosed in risk assessment documents.

Like 2,4-D, the acid forms of borax, clopyralid, glyphosate and imazapyr are generally representative toxicologically to salt forms proposed for use in this PEIR. On the other hand, the BEE and TEA forms of triclopyr are not always toxicologically equivalent to the acid form, so each of these two active ingredients are usually considered separately in USDA/FS and U.S. EPA documents. Hexazinone and sulfometuron methyl are each only used as a single active ingredient. Any important exceptions to these generalizations are clarified as needed.

1.3.1.2.2 Acute Toxicity

Acute toxicity is determined for oral, dermal, inhalation, and ocular routes of exposure. In general, exposure during these studies is by a single dose of the chemical agent. For

background information regarding acute toxicity testing, refer to the U.S. EPA's guidelines (OPPTS 870.1000).

Acute oral and dermal toxicity studies assess systemic effects of exposure to the chemical agent. Results are quantified using "Lethal Dose 50" (LD_{50}), which estimates the amount of a pesticide per test animal bodyweight (usually displayed as mg/kg) required to kill 50% of a test animal population over a specific period of time (WHO 2009; Marer 1999). For acute oral testing, rodents (preferably rats) are usually fed a single dose of the chemical agent by gavage (OPPTS 870.1100) and observations are made to document any signs of systemic toxicity. For the acute dermal toxicity test, chemicals are applied to the skin in graduated doses to several groups of experimental animals (usually albino rabbits), with one dose being used per group and the study is typically conducted for 14 days (OPPTS 870.1200). Results establish a baseline systemic toxicity (via LD_{50}) and effects resulting from exposure. Sometimes detailed information on absorption is also obtained from this study.

Skin is also tested for irritation and sensitization effects. When assessing dermal irritation the test animal (usually a rabbit) has the chemical applied directly to only one patch of shaven skin and an area of the skin without chemical treatment serves as the control (OPPTS 870.2500). This study determines if the chemical causes irritation and/or corrosion to the skin, as well as irreversibility/reversibility of the effects, for no more than 14 days. Dermal sensitizations studies typically use one of three methods, with the most common being the Guinea-Pig Maximization Test (GPMT), which is the test used for active ingredients assessed in this PEIR (OPPTS 870.2600). The GPMT is intended to test for whether the test agent is likely to cause or elicit skin sensitization reactions (allergic contact dermatitis). This study may also indicate systemic toxicity symptoms associated with repeated exposures to the chemical agent. Dermal sensitization is categorized as either being present or absent.

Like oral and dermal tests, the acute inhalation and eye irritation tests follow standard protocols. Unlike acute oral tests, however, inhalation is measured by "Lethal Concentration 50" (LC_{50}), which is typically measured by the concentration of a chemical in the air (mg of chemical per liter of air) that it takes to kill 50% of the test animals over a set time (WHO 2009; Marer 1999). Acute inhalation studies are intended to determine the effects and mortality from inhaling pesticide vapor using graduated dosing with rodents (OPPTS 870.1300). In general, chemicals have greater toxicity via inhalation relative to ingestion (oral) routes of exposure, due to factors such as more rapid absorption and distribution of the chemical through the body. The eye irritation studies measure whether the test chemical has irritating or corrosive effects on the eye and if effects are reversible, usually by adding test material to one eye of a rabbit, while the other eye serves as a control (OPPTS 870.2400). Observations of the eyes are taken anywhere from 72 hours to 21 days after application.

Results from each study discussed above, with the exception of dermal sensitization, are categorized into one of four toxicity categories, in order to easily compare relative acute toxicity from each potential exposure route (Table 5.17.12). During studies any behavioral or physiological changes (e.g. gross lesions, body weight changes) are evaluated, as well as the reversibility of observed anomalies, animal mortality, and any other toxic effects. The U.S. EPA uses initial acute toxicity categories to establish dosing information for chronic and subchronic testing, as well as to establish an overall hazard potential of the chemical agent and to determine label requirements.

Acute toxicity category information, as well as inert ingredient information, is also used to determine product labeling requirements. The U.S. EPA requires that all chemicals that are considered to have toxic properties have a precautionary statement on the label. This statement is determined by the acute test with the most severe toxicity category, or the presence of a special inert at a concentration of 4% or more (U.S. EPA no date). Labels for each category are as follows:

Toxicity Category I or special inert	*DANGER*
Toxicity Category II	*WARNING*
Toxicity Category III	*CAUTION*
Toxicity Category IV	*None Required*

Though a signal word is not required if a chemical meets Category IV criteria, when a signal word is used the word must be CAUTION (U.S. EPA N.D.). In addition to this labeling, the term POISON, as well as a skull and crossbones symbol, are required by the U.S. EPA if either a) any of the acute dermal, oral, or inhalation tests result in a Toxicity Category I classification, or b) there is 4% or more of a known toxic inert, particularly methanol, in any formulation (see Table D.3-1 for labeling examples). This additional labeling must be in red on a contrasting background. All pesticide labels must have a “Keep out Of Reach of Children” warning (U.S. EPA N.D.).

Table D.3-1

Acute Toxicity Criteria Used by the U.S. EPA for Pesticide Classification & Labeling

Study	Category I DANGER	Category II WARNING	Category III CAUTION	Category IV Not Required
Acute Oral Toxicity OPPTS 870.1100	≤ 50 mg/kg body weight	> 50 - 500 mg/kg body weight	> 500 - 5,000 mg/kg body weight	> 5,000 mg/kg body weight
Acute Dermal Toxicity OPPTS 870.1200	≤ 200 mg/kg body weight	> 200 - 2,000 mg/kg body weight	> 2,000 - 5,000 mg/kg body weight	> 5,000 mg/kg body weight
Acute Inhalation* Toxicity OPPTS 870.1300	≤ 0.05 mg/liter	> 0.05 - 0.5 mg/liter	> 0.5 - 2 mg/liter	> 2 mg/liter
Acute Eye Irritation OPPTS 870.2400	Corrosive (irreversible destruction of ocular involvement or irritation persisting for more than 21 days	Corneal involvement or other eye irritation clearing in 8-21 days	Corneal involvement or other eye irritation clearing in 7 days or less	Minimal effects Clearing in Less than 24 hours
Acute Skin Irritation OPPTS 870.2500	Corrosive (tissue destruction into the dermis and/or scarring)	Severe irritation at 72 hours (severe erythema or edema)	Moderate irritation at 72 hours (moderate erythema)	Mild or slight irritation at 72 hours (no irritation or slight erythema)

*4 hr exposure; Adapted from U.S. EPA N.D., Table 1, p. 7-2 to 7-3. The dermal sensitization results are not used for labeling information

Table D.3-2

Examples of U.S. EPA Signal Word Determination

Type of Study	Product A	Product B	Product C*	Product D	Product E*
Acute Oral Toxicity	III	IV	I*	III	II
Acute Dermal Toxicity	IV	III	III	IV	II
Acute Inhalation Toxicity	III	IV	III	III	II
Acute Eye Irritation	III	II	I	I	II
Acute Skin Irritation	IV	IV	II	IV	II
Special Inert, e.g., methanol	No	No	No	No	Yes*
SIGNAL WORD	**CAUTION**	**WARNING**	**DANGER & POISON**	**DANGER**	**DANGER & POISON**

Source: U.S. EPA N.D., Table 2, p 7-4. *Product C and Product E must have additional labeling of a skull & crossbones symbol in close proximity to the word "POISON". This is as a result of Product C having a Category I classification for one of the first three acute toxicity studies (oral in this case) and Product E being made of at least 4% of a special inert.

During U.S. EPA pesticide evaluation processes, most relevant registrant-submitted studies and any new information are continuously reviewed. The most current findings for each active ingredient proposed for use are in Table D.3-3. All chemicals potentially used under the VTP and alternatives have low (Categories III or IV) acute oral, dermal and inhalation toxicity and also low (all Category IV) acute dermal irritation. Acute eye irritation is minimal (Category III) for monoethanolamine salt of clopyralid, glyphosate, sulfometuron methyl, and triclopyr BEE. Acute eye irritation is moderate (Category II) for NP9E, and thus products with this active ingredient must have a WARNING on the label. However, acute eye irritation is high (Category I) for borax, clopyralid acid, hexazinone, and triclopyr TEA, and thus products with these active ingredients must have DANGER on the label. Imazapyr is listed as Category I or III, depending on the percent of technical grade active ingredient used in the test study. Proposed chemicals are not dermal sensitizers, with the exceptions of triclopyr BEE and TEA. Nonylphenol and its ethoxylates

(on average 9 ethoxylates, so abbreviated NP9E) are severe eye and skin irritants, but this chemical mixture is not a skin sensitizer. There is currently no inhalation study for NP9E. Given the low acute oral, dermal, and inhalation toxicity, none of the proposed chemicals are required to be labeled with the word POISON and a skull and crossbones, according to U.S. EPA regulations, unless a particular formulation has a special inert that warrants additional labeling.

Table D.3-3

Acute Toxicity of Proposed Chemicals According to the U.S. EPA [*, 1]

Acute Oral Toxicity OPPTS 870.1100	MRID	Results	Toxicity Category	Reference
Boric acid	00006719	rat LD_{50} males = 3,450 mg/kg rat LD_{50} females = 4,080 mg/kg	III	U.S. EPA 2006e, Table 1, p. 3
	00064208	beagle dog LD_{50} > 631 mg/kg		
Borax, sodium tetraborate decahydrate	40692303	rat LD_{50} males = 4,550 mg/kg rat LD_{50} females = 4,980 mg/kg	III	U.S. EPA 2006e, Table 2, p. 3
	40692304	dog LD_{50} > 974 mg/kg		
Clopyralid, acid	41641301	rat LD_{50} > 5,000 mg/kg (M+F)	IV	U.S. EPA 2009b, Table A2.a, p. 27
Clopyralid, monoethanolamine salt	00147690	rat LD_{50} > 5,000 mg/kg	IV	SERA 2004a, Table Appendix 1, Appendix 1-1
Glyphosate [2]	41400601	LD_{50} > 5,000 mg/kg	IV	U.S. EPA 2006b, Table 4.1a, p. 9
Hexazinone	41235004	rat LD_{50} = 1,200 mg/kg	III	U.S. EPA 2010d, Table 4, p. 16
Imazapyr	41551002 93048016	rat LD_{50} > 5,000 mg/kg	IV	U.S. EPA 2005c, Table 4.1a, p. 15
Sulfometuron methyl	43089201	rat LD_{50} > 5,000 mg/kg (M+F)	IV	U.S. EPA 2008a, Table 2, p. 8
Triclopyr, butoxyethyl ester (BEE)	40557004	rat LD_{50} = 578 mg/kg (F)	III	SERA 2011, Appendix 2, Table 1, p. 3
Triclopyr, triethylamine salt (TEA)	41443301	rat LD_{50} = 1,847 mg/kg (F)	III	U.S. EPA 1998, Table 3, p. 7; SERA 2011d, p. 21

NP9E	none	rat LD_{50} = 1,410-5,600 mg/kg rabbits, mice LD_{50} = 620 – 4,400 mg/kg	III	USDA/FS 2003b, Appendix 3-Table 1, p. A-12
Acute Dermal Toxicity **OPPTS 870.1200**	**MRID**	**Results**	**Toxicity Category**	**Reference**
Boric acid	00106011	rabbit LD_{50} > 2,000 mg/kg	III	U.S. EPA 2006e, Table 1, p. 3
Borax, sodium tetraborate decahydrate	43553201	rabbit LD_{50} > 2,000 mg/kg	III	U.S. EPA 2006e, Table 2, p. 3
Clopyralid, acid	41641302	rat LD_{50} males > 5,000 mg/kg rat LD_{50} females > 5,000 mg/kg	IV	U.S. EPA 2009b, Table A2.a, p. 27
Clopyralid, monoethanolamine salt	none	None	IV	U.S. EPA 2009b, p. 8
Glyphosate	41400602	LD_{50} > 5,000 mg/kg	IV	U.S. EPA 2006b, Table 4.1a, p. 9
Hexazinone	00104974	rabbit LD_{50} > 5,278 mg/kg	IV	U.S. EPA 2010d, Table 4, p. 16
Imazapyr	41551003 93048017	rabbit LD_{50} > 2,000 mg/kg	III	U.S. EPA 2005c, Table 4.1a, p. 15
Sulfometuron methyl [3]	43089202	rabbit LD_{50} > 2,000 mg/kg	III	U.S. EPA 2008a, Table 2, p. 8
Triclopyr, butoxyethyl ester (BEE)	40557005	rabbit LD_{50} > 2,000 mg/kg	III	U.S. EPA 1998, Table 4, p. 7, 187 & 199
Triclopyr, triethylamine salt (TEA)	41443302	rabbit LD_{50} > 2,000 mg/kg	III	U.S. EPA 1998, Table 3, p. 7, 180 & 201
NP9E	none	rabbit LD_{50} > 2,830 mg/kg	III	USDA/FS 2003b, Appendix 3, Table 1, p. A-12
Acute Inhalation Toxicity **OPPTS 870.1300**	**MRID**	**Results**	**Toxicity Category**	**Reference**
Boric acid	00005592	rat LC_{50} > 0.16 mg/L (no deaths)	II [4]	U.S. EPA 2006e, Table 1, p. 3
Borax, sodium tetraborate decahydrate	43500801	rat LC_{50} > 2.03 mg/L	IV	SERA 2006a, p. Appendix 1-17

Clopyralid, acid	41848300	rat LC_{50} males > 5.0 mg/L (M+F)	IV	U.S. EPA 2009b, Table A2.a, p. 27
Clopyralid, monoethanolamine salt	none	none	IV	U.S. EPA 2009b, p. 8
Glyphosate [5]	none	LC_{50} requirement waived	none	U.S. EPA 2006b, Table 4.1a, p. 9
Hexazinone [6]	41756701	rat LC_{50} > 3.94 mg/L (4 hr)	III	U.S. EPA 2010d, Table 4, p. 16
Imazapyr	00132032 93048018	rat LC_{50} > 1.3 mg/L (gravimetric) rat LC_{50} > 5.1 mg/L (nominal)	III	U.S. EPA 2005c, Table 4.1a, p. 15
Sulfometuron methyl	43089203	rat LC_{50} > 5.0 mg/L	IV	U.S. EPA 2008a, Table 2, p. 8
Triclopyr, butoxyethyl ester (BEE)	40557006	rat LC_{50} > 4.8 mg/L	IV	U.S. EPA 1998, Table 4, p. 7 & 187
Triclopyr, triethylamine salt (TEA)	41443303	rat LC_{50} > 2.6 mg/L	IV	U.S. EPA 1998, Table 3, p. 7 & 181
NP9E	none	none	NA	no data
Acute Eye Irritation OPPTS 870.2400	**MRID**	**Results**	**Toxicity Category**	**Reference**
Boric acid	00064209	rabbit - conjunctiva irritation clearing by Day 4	III	U.S. EPA 2006e, Table 1, p. 3
Borax, sodium tetraborate decahydrate	43553203	rabbit - corrosive	I	U.S. EPA 2006e, Table 2, p. 3
Clopyralid, acid	41641304	rabbit - severe irritation at 7 days (corrosive)	I	U.S. EPA 2009b, Table A2.a, p. 27
Clopyralid, monoethanolamine salt	none	slight eye irritant or not irritant	none	U.S. EPA 2009b, p. 8
Glyphosate	41400603	corneal opacity or irritation clearing in 7 days or less	III	U.S. EPA 2006b, Table 4.1a, p. 9
Hexazinone	00106003	rabbit - severe irreversible corneal opacity	I	U.S. EPA 2010d, Table 4, p. 16

Imazapyr	41551001 93048019	rabbit - 2/6 with corneal opacity at 21 days; discharge in 1/6 at 21 days; vascularization of cornea in 1/6 at 21 days; irreversible eye damage	I Tested with 99.3% technical fine powder	U.S. EPA 2005c, Table 4.1a, p. 15
	Accession # 252004	rabbit - corneal opacity cleared within 72 hrs; conjunctivitis reversible by day 7	III Tested with 93% technical	
Sulfometuron methyl	00071412	rabbit - minimally irritating	III	U.S. EPA 2008a, Table 2, p. 8
Triclopyr, butoxyethyl ester (BEE)	40557007	rabbit - minimally irritating	III	U.S. EPA 1998, Table 4, p. 7 & 187
Triclopyr, triethylamine salt (TEA)	41443304	rabbit - corrosive	I	U.S. EPA 1998, Table 3, p. 7 & 181
NP9E	none	rabbit - moderate to highly irritating	II	USFS/FS 2003b, Appendix 3, Table 1, p. A-12
Acute Dermal Irritation **OPPTS 870.2500**	**MRID**	**Results**	**Toxicity Category**	**Reference**
Boric acid	00106011	rabbit - skin irritant	III	U.S. EPA 2006e, Table 1, p. 3
Borax, sodium tetraborate decahydrate	43553202	rabbit - not a skin irritant	IV	U.S. EPA 2006e, Table 2, p. 3
Clopyralid, acid	41641305	rabbit - not a skin irritant	IV	U.S. EPA 2009b, Table A2.a, p. 27
Clopyralid, monoethanolamine salt	none	not a skin irritant	IV	U.S. EPA 2009b, p. 8
Glyphosate	41400604	mild or slight skin irritant	IV	U.S. EPA 2006b, Table 4.1a, p. 9
Hexazinone	00106004	rabbit - mild skin irritant	IV	U.S. EPA 2010d, Table 4, p. 16
Imazapyr	41551004 93048020	rabbit - non-irritating to slight erythema and edema	IV	U.S. EPA 2005c, Table 4.1a, p. 15
Sulfometuron methyl	41672808	rabbit - not a skin irritant [3]	IV	U.S. EPA 2008a, Table 2, p. 8
Triclopyr, butoxyethyl ester (BEE)	40557008	rabbit - not a skin irritant	IV	U.S. EPA 1998, Table 4, p. 7 & 187

Triclopyr, triethylamine salt (TEA)	41443305	rabbit - not a skin irritant	IV	U.S. EPA 1998, Table 3, p. 7 & 181
NP9E	none	rabbit - minimally to severely irritating	II	USFS/FS 2003b, Appendix 3, Table 1, p. A-12
Skin Sensitization **OPPTS 870.2600**	**MRID**	**Results**	**Toxicity Category**	**Reference**
Boric acid / Sodium borate salts	none	no evidence of absorption across intact skin	N/A	U.S. EPA 2006e, Table 3, p. 6
Clopyralid, acid	41641306	guinea pig - not a skin sensitizer	N/A	U.S. EPA 2009b, Table A2.a, p. 27
Clopyralid, monoethanolamine salt	none	not a skin sensitizer	N/A	U.S. EPA 2009b, p. 8
Glyphosate	41642307	guinea pig - not a skin sensitizer	N/A	U.S. EPA 2006b, Table 4.1a, p. 9
Hexazinone	4123005	guinea pig - not a skin sensitizer	N/A	U.S. EPA 2010d, Table 4, p. 16
Imazapyr	00131607 93048021	guinea pig - not a skin sensitizer	N/A	U.S. EPA 2005c, Table 4.1a, p. 15
Sulfometuron methyl [7]	43089204	[guinea pig] - not a dermal sensitizer	N/A	U.S. EPA 2008a, Table 2, p. 8
Triclopyr, butoxyethyl ester (BEE)	40557009	guinea pig - sensitizer	N/A	U.S. EPA 1998, Table 4, p. 7 & 187
Triclopyr, triethylamine salt (TEA)	41443306	guinea pig - sensitizer	N/A	U.S. EPA 1998, Table 3, p. 7 & 181
NP9E	none	guinea pig - not a skin sensitizer	N/A	U.S. EPA 2009f, Table 4, p. 33 & 38

*2,4-D acid, boric acid, and clopyralid acid were shown for comparison purposes and are not proposed for use in this program. [1] Technical grade active ingredient (TGAI) was specified as used for all acute toxicity tests of imazapyr and sulfometuron methyl (SMM: at least 98.8%, purity), triclopyr (BEE: 97.1% a.i. & TEA: 44.4% a.i.) and, though not specified in all U.S. EPA documents, use of TGAI is likely for other ingredients as well. [2] All glyphosate salts disassociate to the acid and associated ions (FR 2007), and thus independent hazard characterization and toxicology studies are not required for each salt active ingredient. [3] From the sulfometuron methyl R.E.D. (U.S. EPA 2008a): "Minimal skin irritation was [also] noted in the acute dermal toxicity study [using rats] (MRID No. 43089202) and an older dermal irritation study [using rabbits] of a 75% formulation (MRID No. 00071411)". [4] The TRED report (U.S. EPA 2006e, p. 3, Table 1) expressed values show, though the U.S. EPA Health Effects Division stated in the earlier preparation documents "[b]oric acid is classified as Toxicity Category II by the inhalation route but only a single dose was tested and an LC50 was not determined", and the subsequent R.E.D. scoping document (U.S. EPA 2009a, p. 2)

listed inhalation as Category III for both acid and borax inhalation. [5] Technical grade glyphosate was used as used for acute tests listed in the R.E.D. and it was specified the "[a]cute inhalation study was waived by the Agency since glyphosate technical is a nonvolatile solid and adequate inhalation studies were conducted on the end-use product formulations (U.S. EPA 1993c)." [6] Given the test result, it is unclear why the acute inhalation was not listed as category III and not IV. [7] Incorrectly labeled as rabbit in original Table given, the Append. D citation specifies guinea pig. [4] The TRED report (U.S. EPA 2006e, p. 3, Table 1) expressed values show, though the U.S. EPA Health Effects Division stated in the earlier preparation documents "[b]oric acid is classified as Toxicity Category II by the inhalation route but only a single dose was tested and an LC50 was not determined", and the subsequent R.E.D. scoping document (U.S. EPA 2009a, p. 2) listed inhalation as Category III for both acid and borax inhalation.

The WHO, like the U.S. EPA, places pesticides in categories based on hazard potential and promotes the use of statements on labels that reflect chemical hazards (Table D.3-4; also see WHO 2009). Since 1975 the WHO classification system has used five categories, rather than the U.S. EPA's four, and precautionary language is required for all chemical products, even if found to be virtually non-toxic. Also different from the U.S. EPA classification system, the WHO primarily uses only oral and dermal acute toxicity test results to determine classification. The WHO (2009) did not find any chemicals potentially used in the VTP and alternatives to be extremely or highly hazardous (Table D.3-6). Hexazinone, and triclopyr are categorized as moderately hazardous and borax, clopyralid and glyphosate are only slightly hazardous. Imazapyr and sulfometuron methyl were found to be unlikely to present acute hazard in normal use.

In December of 2002 the WHO refined its classification system (see Table D.3-5) when:

> *. . the United Nations Committee of Experts on the Transport of Dangerous Goods and on the Globally Harmonized System of Classification and Labeling of Chemicals (UNCETDG/GHS) approved a document called "The Globally Harmonized System of Classification and Labeling of Chemicals" with the intent to provide a globally-harmonized system1 (GHS) to address classification of chemicals, labels, and safety data sheets. The GHS (with subsequent revisions) is now being widely used for the classification and labeling of chemicals worldwide . For this revision of the Classification the WHO Hazard Classes have been aligned in an appropriate way with the GHS Acute Toxicity Hazard Categories for acute oral or dermal toxicity as the starting point for allocating pesticides to a WHO Hazard Class (with adjustments for individual pesticides where required) . It is anticipated that few of the more toxic pesticides will change WHO Hazard Class as a result of this change. (WHO 2009)*

The WHO classifications are for the active ingredients only and are not for any specific formulation. The final classification of these chemicals might be different, depending upon their formulation. However, evidence suggests that overall, whether assessed by the U.S. EPA or the WHO, chemicals potentially used in the VTP and alternatives do not pose a high acute toxicity hazard except for those few that are severely or moderately irritating to the eye.

Table D.3-4

Acute Toxicity Criteria Used by the WHO for Pesticide Hazard Classification

Study	Class Ia Extremely Hazardous	Class Ib Highly Hazardous	Class II Moderately Hazardous	Class III Slightly Hazardous	Class U Unlikely to Present acute Hazard
Acute Oral Toxicity (rat LD_{50})	< 5 mg/kg body weight	5 - 50 mg/kg body weight	50 - 2,000 mg/kg body weight	Over 2,000 mg/kg body weight	5,000 mg/kg body weight or higher
Acute Dermal Toxicity (rat LD_{50})	< 50 mg/kg body weight	50 - 200 mg/kg body weight	200 - 2,000 mg/kg body weight	Over 2,000 mg/kg body weight	5,000 mg/kg body weight or higher

WHO = World Health Organization; Adapted from WHO 2009, p. 10

Table D.3-5

Acute Toxicity Criteria Used by the WHO for the Globally Harmonized System (GHS) for Pesticide Hazard Classification

Study	Category 1 Fatal if Swallowed or in Contact with Skin	Category 2 Fatal if Swallowed or in Contact with Skin	Category 3 Toxic if Swallowed or in Contact with Skin	Category 4 Harmful if Swallowed or in Contact with Skin	Category 5 May Be Harmful if Swallowed or in Contact with Skin
Acute Oral Toxicity (rat LD_{50})	< 5 mg/kg body weight	5 - 50 mg/kg body weight	50 - 300 mg/kg body weight	Over 300 - 2,000 mg/kg body weight	2,000 - 5,000 mg/kg body weight
Acute Dermal Toxicity (rat & rabbit LD_{50})	< 50 mg/kg body weight	50 - 200 mg/kg body weight	200 - 1,000 mg/kg body weight	Over 1,000 - 2,000 mg/kg body weight	2,000 - 5,000 mg/kg body weight

WHO = World Health Organization; Adapted from WHO 2009, p. 10

Table D.3-6

Acute Toxicity of Chemicals Potentially Used Under the VTP & Alternatives, as Reported by the WHO [1/]

Common Name as Listed by WHO	Equiv. Names Used by U.S. EPA	CAS no	Classification		LD_{50} mg/kg	WHO Remarks	Reference
			WHO	GHS			
Borax [ISO]	Borax, sodium tetraborate decahydrate	1303-96-4	III	5	4,000	ICSC 567	WHO 2009, Table 4, p. 34
Clopyralid	Clopyralid, monoethanolamine salt	57754-85-5	III	5	4,300	Severe irritant to eyes; ICSC 443	WHO 2009, Table 4, p. 35
Glyphosate [ISO]	Glyphosate	1071-83-6	III	5	4,230	EHC 159, DS 91; ICSC 160; JMPR 1987a	WHO 2009, Table 4, p. 36
Hexazinone [ISO]	Hexazinone	51235-04-2	II	4	1,690		WHO 2009, Table 3, p. 28
Imazapyr	Imazapyr (CAS # Arsenal)	81334-34-1	U	5	> 5,000	Irritant to eyes	WHO 2009, Table 5, p. 42
Sulfometuron	Sulfometuron methyl	74223-56-6	U	5	> 5,000		WHO 2009, Table 5, p. 45
Triclopyr [ISO]	Triclopyr (salts and esters)	55335-06-3	II	4	710		WHO 2009, Table 3, p. 32

WHO = World Health Organization; Information adapted from WHO 2009; See Table 5.17.8 for WHO Classification definitions. Sulfometuron methyl (CAS no. 74222-97-2) not listed, though Sulfometuron (CAS no. 74222-97-2) was listed as a Classification U - Unlikely to present acute hazard in normal use. * 2,4- D is a Phenoxyacetic acid derivative. TERMS: DS denotes a WHO/FAO Data Sheet on Pesticides, EHC an Environmental Health Criteria monograph, HSG = Health and Safety Guide, IARC Monographs on the Evaluation of Carcinogenic Risks to Humans, ICSC an International Chemical Safety Card, JMPR an evaluation by the Joint FAO/WHO Meeting on Pesticide Residues. [ISO] denotes common name of the a.i. approved by the International Organization for Standardization.

1.3.1.2.3 Subchronic and Chronic Toxicity

Subchronic and chronic toxicity studies form the basis of most quantitative values used in risk assessments. In contrast to acute testing, subchronic and chronic testing involves laboratory animals being given repeated doses. At least two different chemical doses are tested on separate, but otherwise identical, same-sexed groups of animals for both subchronic and chronic tests.

Subchronic and chronic toxicity are typically measured by determining the *No-observed-adverse-effect-level* (NOAEL) or *No-observed-adverse-effect-concentration* (NOAEC), which is defined as “effects that are attributable to treatment but do not appear to impair the organism's ability to function and clearly do not lead to such an impairment” (SERA 2012). The measure of *lowest-observed-adverse-effect-level* or *concentration* (LOAEL or LOAEC) is also often used, and is defined as the lowest exposure level or concentration associated with an adverse effect (SERA 2012). NOAELs/NOAECs and LOAELs/LOAECs are usually expressed as milligrams of chemical per kilogram of test animal body weight per day and notated as mg/kg bw/day (or just mg/kg bw). This section summarizes general signs of systemic toxicity and quantifies no-observable-adverse-effect levels (NOAELs) for the identified endpoints, as well as levels associated with adverse effects such as LOAELs.

Subchronic tests may include repeated doses via consumption in 28-day (OPPTS 870.3050) or 90-day studies, using rodents - preferably rats - (OPPTS 870.3100), as well as a 90-day study using a non-rodent species, which is typically dog (OPPTS 870.3150). Other subchronic studies include 21/28-day and 90-day skin exposure tests using rats, rabbits, or guinea pigs (OPPTS 870.3200 and 870.3250 respectively). Along with these tests, a 90-day inhalation study (OPPTS 870.3465) using a rodent species (preferably rats) may be conducted. Additionally, reproduction (including fertility) and development toxicity screening tests with repeat dosing are also completed using rats as part of the subchronic process (OPPTS 870.3550, 870.3800, 870.3700 and 870.3650). Symptoms of neurotoxicity, immune toxicity, and endocrine disruption are also evaluated as part of the subchronic and chronic testing suite.

Chronic toxicity evaluates the effects of repeated daily exposure of experimental animals to a chemical by the oral, dermal, or inhalation routes of exposure for a minimum of 12 months (OPPTS 870.4100). Chronic toxicity and carcinogenicity studies (OPPTS 870.4200) should be completed using two mammal species. Alternatively, registrants often examine both chronic toxicity and carcinogenicity of a chemical using a single combined study (OPPTS 870.4300). Chronic toxicity studies typically use rat and dog species, while rat and mice species are preferred in carcinogen studies. When the combined chronic toxicity and carcinogenicity alternative is used, rats are the preferred species for oral and inhalation routes of exposure and mice are preferred for dermal exposure (OPPTS 870.4300).

No attempt is made in this document to display all completed subchronic and chronic toxicity-associated studies, or all associated endpoints, as this is beyond the scope of this assessment. Instead, the most significant findings that resulted from subchronic and chronic dosing are summarized below (Table D.3-7). For further details regarding endpoints for specific tests, refer to the U.S. EPA and SERA risk assessments referenced throughout this subchronic and chronic section and other sections below, which evaluate more specifically effects associated with reproduction and development, carcinogenicity and mutagenicity, or effects on nervous, immune, and endocrine systems. Since effects are only summarized, refer to the sources for information by the author(s) of the original study.

Borax (Source: SERA 2006a) - The developing fetus and the male reproductive system are the primary targets for borate-inducted toxicity during developmental, subchronic and chronic toxicity studies. Gestational exposure of rodents and rabbits to boric acid resulted in increased fetal deaths, decreased fetal weight, and increased fetal malformations (e.g. abnormalities of the eyes, skeleton, central nervous and cardiovascular system) in one or both species. Testicular atrophy, degeneration of the spermatogenic epithelium and spermatic arrest were observed during subchronic exposure of rats and dogs via food and water.

Other considerations regarding repeated doses of 2,4-D, include systemic effects and inhalation. The acute dermal exposure for borax is rated as Category 3, as no significant signs of toxicity developed. Single dose inhalation exposure of borax resulted in ocular and nasal discharge, hunched posture, and hypoactivity. This limited data suggests that borax has the potential to cause irritant and systemic toxic effects following inhalation by laboratory mammals.

Clopyralid (Source: SERA 2004a) - While several studies have been submitted to the U.S. EPA during registration, no studies are currently published as open literature. Some information is available from the U.S. EPA as a result of reviews conducted for registration of new uses for clopyralid (e.g. U.S. EPA 2009b) since the 2004 Forest Service risk assessment was completed.

Decreased body weight as well as increases in relative kidney and liver weights consistently result from dietary exposures to clopyralid, though when looking at the indicators for liver damage, histopathologic damage was not apparent. The U.S. EPA determined the chronic NOAEL to be 15 mg/kg/day based on gastric epithelial hyperplasma at the LOAEL of 150 mg/kg/day.

Significance of effects during skin, eye, and inhalation studies varied for clopyralid. Persistent eye damage, characterized by redness, conjunctiva swelling and discharge, is known to result from directly applying clopyralid to the eye. While redness to the skin may occur just after application of clopyralid, there are no symptoms that indicate this chemical is a potent skin irritant for either the penta process clopyralid or electrochemical process

clopyralid. The only effects noted during acute inhalation studies for registration were labored breathing and red stains around nares, as well as lung discoloration.

Glyphosate (Source: SERA 2011b) - The U.S. EPA evaluated subchronic and chronic exposure during registration processes, using studies that tested with technical grade glyphosate. These studies are summarized in the SERA (2011b) risk assessment and associated appendix. Decreased body weight gain is the most consistent signs of subchronic, chronic, and reproductive exposure for test mammals (i.e., rats, mice and rabbits) using technical grade glyphosate. Decreases in body weight may be attributed to glyphosate possibly being an uncoupler of oxidative phosphorylation and/or may be secondary to decreased consumption of food. Other signs of toxicity resulting from technical grade glyphosate seem inconsistent, general and non-specific. Changes in liver weight, kidneys, and blood chemistry have been reported in some studies.

Separate, more specific subchronic and chronic toxicity studies for each glyphosate formulation are not required for pesticide registration by the U.S. EPA, and thus no such studies have been identified in U.S. EPA reports. Only one study evaluating subchronic toxicity was discussed in the SERA (2011b) report as being found in open literature, though the study was on a Brazilian formulation and the study was ambiguous in several regards, including test doses used. Nevertheless, results of the study did not substantially differ from those in the studies submitted to the U.S. EPA, with liver pathology being observed only at the highest dose. No overt toxic effects were noted at dose up to 360 mg a.e./kg bw/day, which is consistent with the NOAEL of 500 mg a.e./kg bw/day from a 90-day study in mice submitted to the U.S. EPA.

The primary signs of subchronic and chronic toxicity to the POEA surfactant included gastrointestinal irritation in rats and dogs. This effect was also noted and attributed to the POEA surfactant for humans in cases of suicidal ingestion of glyphosate formulations. The NOAEL of POEA in rats appears to be about 36 mg/kg bw. The studies inconsistently indicated that POEA by itself appears to be more toxic than technical grade glyphosate. Specific effects in dogs are well characterized, with the toxicity of POEA higher than technical grade glyphosate by a factor of 10, though results with other mammals are less clear.

Hexazinone (Source: SERA 2005) - No studies indicate a specific target organ or mode of action. Decreases in body weight, increases in liver weight, and changes in blood enzyme levels associated with liver toxicity are the effects most commonly observed during long-term exposure. While the decrease in body weight often appears to be a secondary effect related to a decrease in food consumption in dogs and rodents, evidence for female rats in one study suggests instead that decrease in body weight sometimes relates to food conversion efficiency (i.e., in female rats). Thus, the U.S. EPA used such weight-related evidence to establish a chronic RfD.

Imazapyr (Source: SERA 2011c) - The commonality between all studies was the lack of any adverse effects noted at doses at about 2,000 mg/kg bw/day in rodents and about 250 mg/kg bw/day in dogs. Increased food consumption for rats and mice was sometimes observed, but there was no significant corresponding weight gain. The reasons behind these observations remain unclear. The NOAEL of 1,700 mg/kg bw/day was established based on the highest dose tested in rats. While the dog NOAEL is much lower, this is because of study design and doses used, rather than an indication of imazapyr being more toxic to dogs than other mammals. Nevertheless, the current chronic RfD of 2.5 mg/kg bw/day is from the study using dogs.

NP9E (Source: USDA/FS 2003b) - Target organs for both NP9E and NP appear to be the liver and kidneys, based on subchronic and chronic exposure studies. The mitochondria of cells appeared to be affected by long-term NP exposure, though they were not affected with NP9E exposure. Subchronic and chronic studies of NP9E most commonly revealed changes to liver, kidney and sometimes spleen (e.g. increased weight), as well as weight loss in dogs and/or rats. As with NP9E, liver and kidney weights, as well as decreases in body weight and food consumption, appear to most commonly characterize subchronic and chronic exposure effects of NP.

Sulfometuron methyl (Source: SERA 2004c) - Sulfometuron methyl toxicity often involves changes in blood and decreased body weight, though some other more general signs also occurred inconsistently. Changes in blood appear to be consistent with hemolytic anemia. Inconsistent symptoms of sulfometuron methyl include reduced testicular size in a rat, mild testicular lesions in another rat, increased alkaline phosphatase activity and increased serum cholesterol (in females), as well as decreased serum albumin and creatinine. Likewise, increased liver weights and thymus were also observed in particular sexes.

Triclopyr (Sources: SERA 2011d, g) - When mammals are exposed to triclopyr, the kidneys appear to be the most targeted organ and dogs are more sensitive than other lab mammals tested. The LOEL in dogs is 2.5 mg/kg/day and is associated with phenolsulfonphthalein (PSP) urinary excretion, as well as reduced absolute and relative kidney weights. This value was initially used by the U.S. EPA to establish a provisional RfD of 0.025 mg/kg/day for humans. In a subsequent study, the same dose was associated with increases in serum urea nitrogen and creatinine in male dogs. This study resulted in the U.S. EPA lowering the provisional RfD to 0.005 mg/kg/day. Kidney effects were observed in rodents (i.e., hematological and histopathological changes and increased kidney weight) in a 90-day subchronic study at doses as low as 70 mg/kg/day. The other general systemic toxic effects of triclopyr are unremarkable. At high doses, signs of liver damage may be apparent, as well as decreases in food consumption, growth rate, and gross body weight.

Table D.3-7

Human Health Toxicological Reference Doses, Target Organs, and Endpoints of Chemicals Proposed for Use Under the VTP & Alternatives

Active Ingredient	Exposure Scenario	NOAEL Dose	UF	RfD Dose	Study and Toxicological Effects Used For RfD	Target Organs and Most Sensitive Endpoints	References
Borax	**Acute**	chronic used[1]			**Two developmental toxicity studies in rats** - LOAEL for each study ~13.6 and ~13.3 mg B/kg/day based on decreased fetal weight. One study lacked a defined NOAEL, while the other had one of 9.6 mg B/kg/day.	The male reproductive system and the developing fetus appear to be the most sensitive endpoints, with the developing fetus more sensitive than the male reproductive system. Toxicity effects related to fetal development include fetal deaths, decreased in fetal weight, and increased fetal malformations. The testis is a primary target organ for borates based on atrophy, degeneration of the seminiferous epithelium, and sterility (i.e., NOAEC = 25 with an LOAEC of ~50 mg B/kg/day).	SERA 2006i, p. 3-8 & 3-21
	Chronic	NOAEL= 10.6 mg/kg/day	UF = 66	RfD = 0.2 mg/kg/day			
Clopyralid	**Acute**	NOAEL= 75 mg/kg/day	UF = 100	RfD = 0.75 mg/kg/day	**Developmental toxicity studies in rats (gavage)** - decreased maternal body-weight gain and reduced food consumption at the LOAEL of 250 mg/kg/day.	Only non-specific toxicity effects observed. Thus, no primary target organ is indicated during subchronic and chronic toxicity testing; anticipated exposures do not	SERA 2004a, p. 3-27

	Chronic	NOAEL= 15 mg/kg/day	UF = 100	RfD = 0.15 mg/kg/day	**2-year combined chronic/carcinogenicity rat feeding study** - histopathology in stomach at the LOAEL of 150 mg/kg/day.	exceed the RfD values. Contamination of hexachlorobenzene and pentachlorobenzene is not significant in terms of potential systemic-toxic effects.	U.S. EPA 2009b, p. 13
Glyphosate	**Acute**	chronic used			**Developmental toxicity study in rabbits** - LOAEL of 350 mg/kg/day based on diarrhea, nasal discharge and death in maternal animals. Both rabbit and rat dams appear more sensitive than offspring. Represents all populations.	Chronic feeding/carcinogenicity studies in rats revealed systemic effects only at the highest test dose and LOAEL of 940 mg/kg/day, based on decreased body-weight gain in females and increased cataracts and lens abnormalities, decreased urinary pH, increased absolute liver weight, and increased relative liver weight/brain weight in males. Suggestions that glyphosate targets testes are not substantiated using U.S. formulations at doses below or equal to the NOAEL.	SERA 2011b, p. 52, 61 & 102 U.S. EPA 2009c, p. 5 & 22
	Chronic	NOAEL= 175 mg/kg/day	UF = 100	RfD = 2.0 mg/kg/day [2]			
Hexazinone	**Acute**	NOAEL= 400 mg/kg/day	UF = 10	RfD = 4.0 mg/kg/day	**Developmental toxicity study in rats** - LOAEL of 900 mg/kg/day based on decreased male and female fetal weight, kidneys with no papilla and misaligned sternebrae. Protective of females 13-50 years of age	No effects were observed in reproductive tissues (i.e., testes and ovaries) that indicated direct toxicological effects of hexazinone exposure. Decrease weights of testes and other organs during a chronic feeding study with dogs, and multigenerational study with rats, appear to be incidental and not associated with organ specific toxicity.	SERA 2005, p. 3-10 & 3-35 U.S. EPA 2010d, p. 16
	Chronic	NOAEL= 5.0	UF = 10	RfD = 0.05 mg/kg/day	**Chronic dog feeding study** - LOAEL of 41.24/37.57 (m/f) mg/kg/day based on severe body weight		

		mg/kg/day			decrements and clinical chemistry changes.		
Imazapyr	**Acute**	chronic used			**1-year dog feeding study** - due to an absence of an appropriate endpoint attributable to a single dose, the USDA/FS used this study to establish both acute and chronic RfD values. No LOAEL was demonstrated with imazapyr at doses up to 250 mg/kg/day [the highest dose of the study].	The most remarkable aspect of all of the subchronic and chronic studies is the failure to note any adverse effects at doses of up to about 2000 mg/kg /day in rats and mice and about 250 mg/kg /day in dogs.	SERA 2011c, p. 20 & 47 U.S. EPA 2006d, p. 7
	Chronic	NOAEL= 250 mg/kg/day	UF = 100	RfD = 2.5 mg/kg/day			
NP9E	**Acute**	chronic used			**2-generation rat reproduction study (nonylphenol)** - LOAEL of 50 mg/kg/day, based on increases in pituitary weight (F0 males), decreased ovary weight (F0 females), accelerated vaginal opening (F1 females), decreases in # of implanted and live F2 pups (NOAEL 10 mg/kg/day). **3-generation rat reproduction study (nonylphenol)** - LOAEL of 30 mg/kg/day based on acceleration of vaginal opening by ~2 days and ~6 days in F1, F2 and F3 generations following dietary exposure at 30 and 100 mg/kg/day respectively (NOAEL ~9 mg/kg/day).	In studies of nonylphenol, the kidney has been identified as a target organ based on increased kidney weight, tubular dilatation, and cyst formation. Evidence further suggests the liver is a target organ, which is indicated by effects such as decrease in liver polysaccharides at a dose of 50 mg/kg/day (the LOAEL) in one study.	USDA/FS 2003b, p. 29 U.S. EPA 2010f, p. 20
	Chronic	NOAEL= 10 mg/kg/day	UF = 100	RfD = 0.10 mg/kg/day			

Sulfometuron methyl [3]	**Acute**	NOAEL= 86.6 mg/kg/day	UF = 100	RfD = 0.870 mg/kg/day	**Acute teratology study in rats** - NOAEL is based on decreased maternal and fetal body weights in rats after 10-day gestational exposure of dams.	No specific organs appear to be targeted by sulfometuron methyl, though hemolytic anemia and decreased body-weight gain were found. These effects are the basis of the past acute and chronic RfD of 0.27 mg/kg/day, which were derived from a study with a NOAEL of 27.5 mg/kg/day and LOAEL OF 148.5 mg/kg/day in both sexes. It is plausible that effects on blood are likely, at least in part, to be attributable to sulfonamide and saccharin.	SERA 2004c, p. 3-23 U.S. EPA 2008a, p. 8
	Chronic	NOAEL= 2.0 mg/kg/day	UF = 100	RfD = 0.02 mg/kg/day	**2-year rat feeding study** - NOAEL is based on hematological effects in male rats at higher doses, with a NOAEL of 3 mg/kg/day for comparable hematological effects in females.		
TCP	**Acute**	NOAEL= 25 mg/kg/day	UF = 1000	RfD = 0.025 mg/kg/day	**Developmental toxicity study in female rabbits** - a LOAEL of 100 mg/kg/day based birth defects including hydrocephaly and dilated ventricles. No dietary RfD is derived for members of the general population.	3,5,6-trichloro-2-pyridinol (TCP) is a major metabolite of triclopyr in both mammals and the environment. This compound does not have the phytotoxic potency of triclopyr; however, according to the RfD values used by the U.S. EPA, TCP is more toxic than triclopyr to mammals and other aquatic animals.	SERA 2011d, p. 16 & 71
	Chronic	NOAEL= 12 mg/kg/day	UF = 1,000	RfD = 0.012 mg/kg/day	**Chronic toxicity study in dogs** - an LOAEL of 48 mg/kg/day is based on clinical chemistry.		
Triclopyr	**Acute**	NOAEL= 100 mg/kg/day	UF = 100	RfD = 1.0 mg/kg/day	**Developmental study in female rats with triclopyr BEE** - NOT APPLICABLE TO FEMALES OF CHILDBEARING AGE. The more protective chronic RfD is used as the	The liver and kidney are suggested to be primary target organs. Like most weak acids, triclopyr is excreted primarily in the kidney by an active transport process. At very	SERA 2011d, p. 71 & 232

					acute RfD for such females. The LOAEL is based on severe maternal toxicity.	high doses, this process may become saturated causing triclopyr to reach toxic levels. At sufficiently high doses, triclopyr may cause toxic effects, including death. Nonetheless, triclopyr has a low order of acute lethal potency. The dog appears to be the most sensitive test species.	
	Chronic	NOAEL= 5.0 mg/kg/day	UF = 100	RfD = 0.05 mg/kg/day	**Two generation dietary reproduction study with triclopyr acid** - this RfD is used for all occupational exposures, acute exposure for women of childbearing age and chronic exposure of individuals. The LOAEL is based on kidney toxicity.		

NOAEL = no observed adverse effect level; LOAEL - lowest observed adverse effect level; UF = uncertainty factor; RfD = reference dose; [1] Typically, the chronic NOAEL is used for the acute RfD calculation in USDA/FS risk assessments when a dose in a single day did not result in toxic effects. [2] The chronic RfD used by the U.S. EPA is 1.75 mg/kg/day, and this value was rounded to 2.0 in the SERA risk assessment. [3] The U.S. EPA (2008a) R.E.D. for sulfometuron methyl listed equal acute and chronic RfD values (0.275 mg/kg/day) for drinking water exposure and dietary RfD values were not calculated since this chemical is not used on food commodities; in lieu of this, the more detailed RfD values from the SERA (2004c) risk assessment used throughout this PEIR.

1.3.1.2.4 Reproductive and Developmental Effects

The analysis in this PEIR distinguishes between *reproductive* and *developmental toxicity*, as defined by the U.S. EPA (1991, 1996). The U.S. EPA human health effects test guideline for reproduction and development include OPPTS 870.3550, 870.3650, 870.3700, and 870.3800.

In the *U.S. EPA Guidelines for Reproductive Toxicity Risk Assessment*, reproductive toxicology is defined as (U.S. EPA 1996):

> *The occurrence of biologically adverse effects on the reproductive systems of females or males that may result from exposure to environmental agents. The toxicity may be expressed as alterations to the female or male reproductive organs, the related endocrine system, or pregnancy outcomes. The manifestation of such toxicity may include, but not be limited to, adverse effects on onset of puberty, gamete production and transport, reproductive cycle normality, sexual behavior, fertility, gestation, parturition, lactation, developmental toxicity, premature reproductive senescence, or modifications in other functions that are dependent on the integrity of the reproductive systems.*

Multigenerational reproduction studies with rats are conducted as outlined in guidelines (OPPTS 870.3800) using standardized protocols as part of the reproduction test suite. In general for these studies, males and females are dosed equally via oral route with the chemical agent at 5 to 9 weeks old. These males and females serve as the parental (P) animals and are mated. Chemical doses are often given continuously through weaning of offspring (F1). If a second-generation study is conducted, these steps are repeated with F1 male and female offspring to produce a second generation of offspring (F2). During experiments, animals are observed for gross signs of toxicity and other effects, such as length of the estrous cycle, assays on sperm and other reproductive tissue, and the number, viability, and growth of offspring.

In the *U.S. EPA Guidelines for Developmental Toxicity Risk Assessment*, developmental toxicology is defined as (U.S. EPA 1991):

> *The study of adverse effects on the developing organism that may result from exposure prior to conception (either parent), during prenatal development, or postnatally to the time of sexual maturation. Adverse developmental effects may be detected at any point in the lifespan of the organism. The major manifestations of developmental toxicity include: (1) death of the developing organism, (2) structural abnormality, (3) altered growth, and (4) functional deficiency.*

In summary, developmental studies are designed to exam whether a compound has the potential to cause birth defects. Chemicals in these studies are typically administered to

rats or rabbits using gavage or dermal application methods. The U.S. EPA generally is not concerned with reproductive and developmental effects that are experienced at dosages that cause toxicological maternal or parental effects. Per U.S. EPA chemical assessments, toxicity symptoms only occurred at chemical dosages that were *above/at the threshold of parental toxicity* (ATPT) for chemicals potentially used in the VTP and alternatives, with the exception of borax (Table D.3-8). None of the chemicals potentially used are listed on the California U.S. EPA's Safe Drinking Water and Toxic Enforcement Act of 1986 (Proposition 65) as chemicals known to cause reproductive toxicity (OEHHA 2011).

Table D.3-8

Reproductive and Developmental Toxicity of Chemicals Proposed for Use Under the VTP & Alternatives

Active Ingredient	Reproductive Toxicity	Developmental Toxicity	Reference
Boric acid/ borate salts	at LOAEL testicular atrophy and reduced sperm production leading to reduced male fertility; reduced survival when doses are ATPT	decreased fetal weight and skeletal abnormalities sometimes when doses are BTPT; visceral, heart/vessel, and brain abnormalities when doses are ATPT	U.S. EPA 2006a, p. 1 U.S. EPA 2006e, p. 4 U.S. EPA 2009a, p. 3
Clopyralid	no effects when doses are BTPT; effects sometimes when doses are ATPT (e.g. changes in pup body and liver weights)	no effects when doses are BTPT; sometimes decreased fetal body weight and hydrocephalus when doses are ATPT	U.S. EPA 2009b, p. 8
Glyphosate	no significant effects when doses are BTPT; effects sometimes when doses are ATPT include decrease in implantation	no significant effects when doses are BTPT; sometimes symptoms when doses are ATPT (e.g. decrease in mean fetal body weight and increase in fetuses with unossified sternebrae)	U.S. EPA 2010b, p. 4 & 11
Hexazinone	no significant effects, with both fetal and maternal endpoints based on decreased body weights	no significant effects, with both fetal and maternal endpoints based on decreased body weights	U.S. EPA 2010d, p. 5
Imazapyr			U.S. EPA 2006d, p. 7

	no reproductive effects up to highest dose tested	no developmental effects up to highest dose tested	FR 2003, Table 2, p. 55478
Sulfometuron methyl	no effects on fetal or maternal endpoints at the highest tested dose; abortions when doses were ATPT; note that some studies had deficiencies	no effects on fetal or maternal endpoints at the highest tested dose; abortions when doses were ATPT; note that some studies had deficiencies	U.S. EPA 2008a, p. 8,9 & 18
Triclopyr	no effects when doses are BTPT for BEE or TEA; systemic effects occur when doses are ATPT (e.g. decreased litter size, # of litters, and mean pup weight, decreased parent body weight and weight gain, and increased pup death and proximal tubular degeneration)	no effects when doses are BTPT for BEE or TEA; effects occur when doses are ATPT (e.g. decreased # live fetuses and mean fetal weight gain, increase in fetal death and post-implantation loss, increased incidence bone abnormalities)	U.S. EPA 1998, p. 11-14 & 29 SERA 2011d, p. 25
NP9E	no significant effects when doses BTPT; when doses ATPT effects on adults included less food consumption and decreased weight gain, as well as a decrease in sperm for males, and for females increased estrous cycle length and decreased ovarian weights and decrease in number of implants	when doses BTPT acceleration in the vaginal opening in pups; no evidence when doses are ATPT though kidneys, liver and spleen thought to target organs from general toxicity; weak estrogenic effects at high doses that decrease with increased ethoxylate numbers	USDA/FS 2003b, p. 6, 8 & 11 U.S. EPA 2009f, p. 23, 24 & 28

ATRC = at/above threshold of renal clearance, ATPT = at/above threshold of parental toxicity, BTPT = below threshold of parental toxicity. [1] Only 2,4-D acid and DEA forms have any effects when ATPT

1.3.1.2.5 Carcinogenic and Mutagenic Effects

CAL FIRE defers to the U.S. EPA and CDPR on issues relating to quantitative risk assessment for potential carcinogenic and mutagenic effects in humans. Carcinogenicity refers to the ability of an agent, in this case a pesticide, to cause cancer. Generally, results from chemical effects studies, such as mammal acute, subchronic, and chronic toxicity studies, as well as genetic toxicity (including mutagenicity) studies are used to assess the likelihood a chemical may be a carcinogen. Carcinogenicity is also evaluated by examining chemical profile studies (e.g. metabolism, environmental fate) for indications of whether cancer is a feasible hazard. Some studies are designed to evaluate carcinogenicity of a chemical directly as well (OPPTS 870.4200 and 870.4300). Each

chemical is categorized based on carcinogenic likelihood. Since 1999 five carcinogenicity standard hazard descriptors have been recommended for use by the U.S. EPA: *"Carcinogenic to Humans," "Likely to Be Carcinogenic to Humans," "Suggestive Evidence of Carcinogenic Potential," "Inadequate Information to Assess Carcinogenic Potential," and "Not Likely to Be Carcinogenic to Humans"* (U.S. EPA 2005b).

However, many existing U.S. EPA and USDA/FS risk assessments use the earlier (1986) classification system, which has the following six general categories (often with slight variation): *"A – human carcinogen," "B1 – probably carcinogen, limited human evidence," "B2 - probable carcinogen, sufficient evidence in animals," "C - possible human carcinogen," "D – not classifiable," and "E – evidence of noncarcinogencity."*

In the context of evaluating the effects of pesticides, mutagenicity is defined as the capacity of a chemical to induce transmitted genetic changes or increase their frequency. The mutagenic effects of a pesticide on humans are associated with changes in gamete (germ cell) and/or somatic (tissue/organ) cells (U.S. EPA 1986). Mutations that occur in gamete cells, such as eggs and sperm, have the potential to be inherited by the next generation. Somatic cell mutations, by contrast, effect tissues and organs of the affected individual, and are thought to subsequently cause several disease states (e.g. cancer). Point mutations (i.e., changes in DNA sequence) and structural or numerical chromosome aberration, for example, are mutations that have the potential to cause adverse effects in humans (U.S. EPA 1986). Mutations, however, may not alter DNA directly, but instead interfere with mechanisms essential to cells, such as DNA synthesis or nuclear division processes (ibid). When such mutations occur in gamete cells, offspring may develop skeletal abnormalities, cataracts, or other morphological anomalies. Background, risk assessment, and toxicity study information for various mutation types can be found in *Guidelines for Mutagenicity Risk Assessment* (U.S. EPA 1986) and test guidelines 870.51 through 870.59. Additionally, information relating to hazard identification and toxicity tests for cancer and mutations thought to cause cancer may be found in *Guidelines for Carcinogen Risk Assessment* (U.S. EPA 2005b).

Per the U.S. EPA, none of the active ingredients proposed for use in the VTP and alternatives are known carcinogens or mutagens (Table D.3-9). Similarly, none of the chemicals proposed for use are on the California EPA's Safe Drinking Water and Toxic Enforcement Act of 1986 (Proposition 65) list of chemicals that are known to cause cancer (Cal EPA 2011). While clopyralid is not thought to be a carcinogen, hexachlorobenzene is a carcinogenic impurity of particular concern. Thus, the carcinogenicity of this impurity is considered in this risk assessment.

Table D.3-9

Carcinogenicity and Mutagenicity of Chemicals Proposed for Use Under the Program & Alternatives

Active Ingredient	Carcinogen Class	Mutagenicity	Reference
Boric acid/ borate salts	"not likely to be carcinogenic to humans"	Negative for mutagenic effects	U.S. EPA 2009a, p. 9
Clopyralid	"not likely to be carcinogenic to humans"	Negative for mutagenic effects	U.S. EPA 2009b, p. 8-9
Glyphosate	Group E [evidence of non-carcinogenicity for humans]	Negative for mutagenic effects	U.S. EPA 2010b, p. 27
Hexazinone	Group D [not classifiable as to human carcinogenicity]	Negative for mutagenic effects usually, though structural chromosomal aberrations occurred during one study.	U.S. EPA 2010d, Table 5, p. 17 U.S. EPA 2002b, Table 2, p. 10-12
Imazapyr	Group E [no evidence of carcinogenicity in at least 2 adequate animal tests in different species]	Negative for mutagenic effects	U.S. EPA 2006d, p. 7 FR 2003, Table 2 & 3, p. 55475-55479
Sulfometuron methyl	no evidence of carcinogenicity reported from current toxicity studies	Negative for mutagenic effects	U.S. EPA 2008a, p. 8 SERA 2004c, p. 3-1
Triclopyr	Group D [not classifiable as to human carcinogenicity]	Negative for mutagenic effects	U.S. EPA 1998, p. 14 & 18 SERA 2011d, p. 27
NP9E	no evidence of carcinogenicity reported from current toxicity studies	Negative for mutagenic effects	U.S. EPA 2010e, p. 4 U.S. EPA 2009f, p. 30 & 32 USDA/FS 2003b, p. 4

1.3.1.2.6 Effects on Nervous System

Neurotoxicants are chemical agents that disrupt the function of neurons, either by interacting with neurons specifically, or with supporting cells in the nervous system (e.g., neuroglia, Schwann cells, sensory receptors). The above definition is central

to this discussion because it distinguishes agents that act directly on the nervous system (direct neurotoxicants), from those agents that might produce neurologic effects that are secondary to other forms of toxicity (indirect neurotoxicants) (O'Donoghue, 1994). SERA (2002)

While specific neurotoxicity studies are now required as a part of new data requirements in the 40 CFR §158 (OPPTS 870.6100, 870.6200, 870.6300, 870.6500, 870.6850, and 870.6855), these tests have not yet been completed for all chemicals proposed for use under this PEIR. Nevertheless, it is likely that any effects to the nervous system after exposure to a chemical would be observed during other toxicology studies for chemicals that are neurotoxic. While only direct effects are relevant to evaluating neurotoxicity, in some cases, it can be difficult to determine if the observed effects are a result of direct or indirect neurotoxicity. Currently, most conclusions regarding neurotoxicity of chemicals are usually based on observations from toxicological studies not specific to evaluating the nervous system (see Table D.3-10). Of chemicals potentially used in the VTP and alternatives, direct effects to the nervous system were only found for boric acid/ borate salts at high dosages.

1.3.1.2.7 Effects on Immune System

Immunotoxicants are chemical agents that disrupt the function of immune system. These agents can impair immune responses (immune suppression) or produce inappropriate stimulation of immune responses (hyperreactivity). Suppression of immune responses to microbes or abnormal cells can enhance susceptibility to infectious diseases or cancer. Hyperreactivity can give rise to allergy or hypersensitivity, in which the immune system or genetically predisposed individuals inappropriately responds to chemical agents (e.g., plant pollen, cat dander, flour gluten) that pose no threat to other individuals or autoimmunity, in which the immune system produces antibodies to self components leading to destruction of the organ or tissue involved. SERA (2002)

While immunotoxicity studies are now required as a part of new data requirements in the 40 CFR §158 (OPPTS 870.7800), these tests have not yet been completed for all chemicals proposed for use in the program. Nevertheless, it is likely that any effects to the immune system after exposure to a chemical would be observed during other toxicology studies for chemicals that are immunotoxic. While only direct effects are relevant to evaluating immunotoxicity, it can be difficult in some cases to determine if the effects observed are a result of direct or indirect immunotoxicity. Currently, most conclusions regarding immunotoxicity of chemicals are usually based on observations from toxicological studies not specific to evaluating the immune system (see Table D.3-10). Direct immunotoxicity effects were not observed for any herbicides proposed for use under the VTP.

1.3.1.2.8 Effects on Endocrine System

An endocrine disruptor is an exogenous agent (from outside of the body) that produces adverse effects on an organism or population of organisms by interfering with endocrine function (Kavlock et al., 1996). The endocrine system is highly regulated to achieve hormone activities in amounts needed to respond to physiological demands. Endocrine disruption is a state of uncontrolled hormone action, in which hormone responses are absent or insufficient when needed, or occur inappropriately when they are not needed. These can result in abnormalities in growth and development, reproduction, body composition, homeostasis, and behavior. (SERA 2002)

At the time this appendix was prepared, the U.S. EPA had recently developed an Endocrine Disruptor Screening Program (EDSP), the guidelines for which are in series 890. Current information regarding the program and which herbicides are to be assessed can be found at: http://www.epa.gov/scipoly/oscpendo/index.htm. In short, Tier 1 consists of several assays to identify the potential of a chemical substance to interact with the estrogen, androgen, or thyroid hormonal systems. If it is found that there are direct effects on these systems resulting from chemical exposure, a second group of tests will be chosen as appropriate, given initial results. This second group of studies, referred to as "Tier 2", is used to identify any adverse endocrine related effects caused by the substance, as well as to establish a dose-response relationship between the dose and any effects found on the estrogen, androgen, and/or thyroid hormonal systems.

While all chemicals may be subject to additional screening and/or testing to specifically assess endocrine disruption potential in the future, evaluation of chemicals for endocrine disruption has been prioritized based on the potential for human exposure (e.g. via food and water, residential activity) and effects observed during previous studies evaluating all aspects of chemical toxicity. Currently, information regarding endocrine disruption is vague, though per U.S. EPA and USDA/FS risk assessments, glyphosate, hexazinone, imazapyr and sulfometuron methyl are thought to have the potential to cause effects on the endocrine system with exposure, though it remains unclear if the effects are direct or indirect (see Table D.3-10).

Table D.3-10

Neurotoxicity, Immunotoxicity, and Endocrine Disruption of Chemicals Proposed for Use Under the VTP & Alternatives

Active Ingredient	Neurotoxicity	Immunotoxicity	Endocrine Disruption	Reference
Boric acid/ borate salts	evidence of neurotoxicity from toxicity studies at high dose levels (e.g. depression, ataxia and convulsion)	no conclusive evidence of direct immunotoxicity from any toxicity studies	no evidence of direct endocrine disruption; changes in hormones thought indirect resulting from testicular toxicity	U.S. EPA 2006a, p. 17 & 42 U.S. EPA 2006e, p. 4 & 13 SERA 2006a, p. 3-1, 3-6 to 3-8
Clopyralid	no conclusive evidence of direct neurotoxicity from any toxicity studies	no conclusive evidence of direct immunotoxicity from any toxicity studies	no conclusive evidence of direct endocrine disruption from any toxicity studies	U.S. EPA 2009b, p. 4, 10 & 18 SERA 2004a, p. 3-5 & 3-6
Glyphosate	no conclusive evidence of direct neurotoxicity from any toxicity studies	no conclusive evidence of direct immunotoxicity from any toxicity studies	potential evidence of direct endocrine disruption; effects observed may be indirect	U.S. EPA 2010b, p. 4, 11 to 15 SERA 2011b, p. 40 to 51
Hexazinone	no conclusive evidence of direct neurotoxicity from any toxicity studies	no conclusive evidence of direct immunotoxicity from any toxicity studies	potential evidence of direct endocrine disruption; effects observed may be indirect	U.S. EPA 2010d, p. 5 U.S. EPA 2002b, p. 3 SERA 2005, p. 3-7 to 3-9
Imazapyr			no conclusive evidence of direct endocrine	U.S. EPA 2006d, p. 7 & 27 FR 2003, p. 55481

	no conclusive evidence of direct neurotoxicity from any toxicity studies	no conclusive evidence of direct immunotoxicity from any toxicity studies	disruption from any toxicity studies	SERA 2011c, p. 23
Sulfometuron methyl	no conclusive evidence of direct neurotoxicity from any toxicity studies	no conclusive evidence of direct immunotoxicity from any toxicity studies	potential evidence of direct endocrine disruption; effects observed may be indirect	U.S. EPA 2008a, p. 8 & 14 SERA 2004c, p. 3-6 to 3-7
Triclopyr	no conclusive evidence of direct neurotoxicity from any toxicity studies	no conclusive evidence of direct immunotoxicity from any toxicity studies	no conclusive evidence of direct endocrine disruption from any toxicity studies	U.S. EPA 1998, p. 14 & 18 SERA 2011d, p. 22 to 25
NP9E	no conclusive evidence of direct neurotoxicity from any toxicity studies	no conclusive evidence of direct immunotoxicity from any toxicity studies	no conclusive evidence of direct endocrine disruption from any toxicity studies	U.S. EPA 2009f USDA/FS 2003b, p. 4

1.3.1.2.9 Metabolites and Impurities

No chemical exists without some metabolites and impurities. When evaluating human health effects related to chemical use, it is important to consider how a chemical is metabolized, what byproducts result, what other impurities exist, and the toxicity of any unintended compounds. As chemicals are broken down, either through energy production by a living organism (aka metabolism) or environmental degradation processes (aka environmental fate), metabolites are created. During the synthesis of technical grade product, there may be unintended impurities including un-reacted starting material, side reaction products, contaminants, and degraded products (as listed in 40 CFR 158.153(d)). There is concern regarding the toxicity of metabolites and impurities, but this is lessened by the fact that tests are completed using the technical grade product of each active ingredient that includes metabolite production and contains impurities. Thus, any toxicity effects of metabolites and impurities would be encompassed in the technical grade of the active ingredient (TGAI) toxicity evaluation.

All known metabolites and impurities in chemicals proposed for use under this PEIR were identified and examined for toxicity concerns (see Table D.3-11). Of the chemicals potentially used in the VTP and alternatives, only triclopyr produces a metabolite [i.e., 3,5,6-trichloro-2-pyridinol (3,5,6-TCP)] that is toxic beyond the level of concern in some scenarios. Clopyralid contains the impurities hexachlorobenzene and pentachlorobenzene, which are known carcinogens. Similarly, some formulations of glyphosate that contain POEA surfactants contain the known carcinogenic contaminant 1,4-dioxane. These three carcinogens, however, are at concentrations well below the cancer risk level used by the USDA/FS and U.S. EPA when assessing carcinogenicity. Nicotinic acid, which is also known as Vitamin B3, is a metabolite of imazapyr and is a known neurotoxin; however, the minute amount in imazapyr poses no toxicity concern.

Table D.3-11

Metabolism, Metabolites and Impurities from Chemicals Proposed For Use Under the VTP & Alternatives

Active Ingredient	Metabolism	Metabolites/Degradates	Metabolite Concern	Impurities/Contaminants	Impurities Concern	Reference
Boric acid/ borate salts	in mammals, not metabolized, so is eliminated in urine unchanged; in the environment, at physiological pH borate salts convert to boric acid	boric acid	no concern; assessed as active ingredient	none identified	NA	U.S. EPA 2009a, p. 8 & 20 SERA 2006a, p. 3-11
Clopyralid	in mammals, rapidly absorbed and then excreted in urine, primarily unchanged or	parent clopyralid	assessed as active ingredient	4,5,6-trichloro-2-pyridinecarboxylic acid (<0.1%)	no concern	U.S. EPA 2009b, p. 4, 7 & 19
		3,6-DCPA-glycine		hexachlorobenzene [*, 1]	no concern given cancer risk level for these two impurities of	SERA 2004a, p. 3-2, 3-9, 3-28 to 3-31, 3-33 & 3-38

	as parent compound			pentachlorobenzene [*, 1]	3 in 100,000,000 is well below trigger level of 1 in 1,000,000 used by USDA/FS and U.S. EPA; cancer risk factor=1.6 (mg/kg/day)-1	
Glyphosate	in mammals, primarily excreted in the feces and urine unchanged	aminomethyl phosphonate (AMPA)	no concern	N-nitrosoglyphosate (NNG) *	no concern	U.S. EPA 2009c, p. 2, 6 & 7
		N-acetyl-AMPA		1,4-dioxane [*, 1]	no concern given cancer risk level of 1	U.S. EPA 2010b, p. 4, 12

		N-acetyl-glyphosate	no concern; equivalent to glyphosate	(contaminant in POEA)	in 1,500,000 below trigger level of 1 in 1,000,000 used by the USDA/FS and U.S. EPA; cancer potency factor=0.011 $(mg/kg/day)^{-1}$	SERA 2011b, p. 83-86
Hexazinone	In mammals, rapidly metabolized by hydroxylation and demethylation, and eliminated in urine and feces; in the environment, the data indicate that hexazinone is metabolized by hydroxylation to metabolite	3-(4-hydroxycyclohexyl)-6-(dimethylamino)-1-methyl-1,3,5-triazine-2,4(1H,3H)-dione)	no concern; tolerance expressions include hexazinone (parent) and metabolites; hexazinone and its metabolites do not exceed level of concern	names not released by the U.S. EPA	no concern	U.S. EPA 2010d, Tables 1+7, p. 5-7
		3-cyclohexyl-6-(methylamino)-1-methyl-1,3,5-triazine-2,4-(1H,3H)-dione				U.S. EPA 2002b, p. 5
		3-(4-hydroxycyclohexyl)-6-(methylamino)-1-methyl-1,3,5-triazine-2,4-(1H,3H)-dione				U.S. EPA 1994, p. 14-16
		3-cyclohexyl-1-methyl-1,3,5-triazine-2,4,6-(1H,3H,5H)-trione				SERA 2005, p. 3-16 & 3-17

	A which is then metabolized to metabolite C by demethylation and to metabolite E after oxidation.	3-(4-hydroxycyclohexyl)-1-methyl-1,3,5-triazine-2,4,6-(1H,3H,5H)-trione				
		3-cyclohexyl-6-amino-1-methyl-1,3,5-triazine-2,4-(1H,3H)-dione				
Imazapyr	in mammals, rapidly absorbed when administered orally and then excreted in urine and feces, primarily unchanged; in the environment, photolysis is the only identified mechanism for imazapyr degradation	pyridine hydroxy-dicarboxylic acid	no concern; no more toxic than parent	names not released by the U.S. EPA	no concern given TGAI mammal toxicity tests	U.S. EPA 2006d, p. 17
		pyridine dicarboxylic acid				
		nicotinic acid (aka Vitamin B3)*[2]	no concern for low exposures			SERA 2011c, p. 31

Sulfometuron methyl	in mammal, nearly all is excreted in urine; in both mammals and bacteria, sulfometuron methyl is degraded similarly in multiple stages	sulfometuron pyrimidine amine	no concern given TGAI mammal toxicity tests	no information	no concern given TGAI mammal toxicity tests	U.S. EPA 2008a, p. 10
		sulfometuron sulfonamide				
		saccharin				SERA 2004c, p. 3-11
Triclopyr	in mammals, excretion is rapid though urine typically unchanged at low doses; in the environment, it degrades slowly under aerobic aquatic conditions by aqueous photolysis, in soil it is degraded by biotic mechanisms	3,5,6-trichloro-2-pyridinol (3,5,6-TCP)*	more toxic than a.i. to mammals and aquatic organisms; exceeds level of concern for contaminated vegetation and fruit at upper bounds of analysis	none identified	NA	U.S. EPA 1998, p. 16, 30, 34 & 51
		glucuronide	no concern			SERA 2011d, p. 4, 15, 80-81
		sulfate conjugates of 3,5,6-TCP				

NP9E	appears to be rapidly metabolized and excreted primarily through feces and secondarily in urine.	nonylphenol (conjugates/ neutral and acidic species)	act as estrogen mimics; also concern for aquatic spp.	ethylene oxide [*,1, 3]	carcinogen risk is an acceptable level for USDA/FS (both carcinogens well below the 1 per 1 million cancer risk potential)	USDA/FS 2003b, p. 5 & 18
		sulphate conjugates	no concern	1,4-dioxane [*, 1]		U.S. EPA 2010e, p. 4
		glucuronide				

* Potentially toxic if in high enough quantities; [1] Probable human carcinogen according to U.S. EPA; [2] Possible neurotoxin; [3] Possible mutagen

1.3.1.3 Exposure Assessment

1.3.1.3.1 Chemical Exposure

In Forest Service risk assessments, chemical exposure of workers and members of the public are considered. Each of these groups is assessed in terms of general exposure and accidental/incidental exposure (SERA 2012). General exposure refers to exposure that is expected to occur from normal chemical use, whereas accidental/incidental exposure results from unforeseeable events and improper handling of chemicals. There are innumerable potential circumstances that lead to chemical exposure, though it is most important in all cases to assess the level of exposure (i.e., percentage of body exposed), the chemical concentration, and the duration of the exposure (*ibid*). To assess potential chemical exposure, several scenarios were created for the USDA Forest Service risk assessments (Table D.3-12). These standard sets of scenarios were designed with the intention of being conservative (in the sense of over-estimating risks) and applicable to a wide range of circumstances.

Exposure scenarios are only summarized in the worker and public exposure subsections below. For further details regarding scenarios, including calculation methods and values, refer to SERA 2012, specific chemical risk assessments, and associated Excel workbooks. In depth calculations for each scenario are on worksheets within workbooks generated by FS WorksheetMaker for each chemical (Table D.3-12). The worker and public exposure results are also summarized on worksheets E01 and E03 respectively in each workbook. Once the levels of exposure are determined for each scenario, the dose responses of the chemicals are assessed and the risk of exposure is then characterized.

As discussed previously, methodologies and information regarding chemical exposure continuously changes. Empirical evidence, such as study information, from SERA, USDA/FS, and U.S. EPA reports was used extensively to complete human health risk summaries for each chemical. Calculations from the latest FS WorksheetMaker, however, were used to update values in previous USDA/FS risk assessments using revised methods. Calculations changed for several of the chemicals, though sometimes only to a minor extent. In all cases, the newest calculations and methodologies have been used throughout this appendix, replacing those disclosed in original USDA/FS risk assessments and workbooks.

Different scenarios were designed for occupational (worker) exposure to chemicals than for public exposure, which is discussed in more detail below. There are, however, commonalities among all scenarios used in Forest Service risk assessments. While humans may be exposed through oral, inhalation and ocular routes, clear empirical evidence is limited, with studies having inconsistent findings (SERA 2012). Dermal absorption information, however, is relatively well characterized and understood for most chemicals. Thus, this data is often used directly, or models are created to approximate how dermal absorption relates to other routes

of exposure, such as inhalation, when risk assessments are conducted for the Forest Service (*ibid*).

Table D.3-12

Standard Scenarios Used in USDA/FS Risk Assessments

Scenario	Receptor	Worksheet
OCCUPATIONAL EXPOSURE		
Accidental/Incidental Acute Exposures (dose in mg/kg bw/event)		
Contaminated gloves, 1 minute	Worker	C02a
Contaminated gloves, 1 hour	Worker	C02b
Spill on hands, 1 hour	Worker	C03a
Spill on lower legs, 1 hour	Worker	C03b
General Chronic Exposures (doses in mg/kg bw/day)		
Directed foliar ground applications	Worker	C01
Broadcast ground applications	Worker	C01
Other ground applications (e.g. directed soil and stump)	Worker	C01
PUBLIC EXPOSURE		
Accidental/Incidental Acute Exposures (dose in mg/kg bw/event)		
Direct spray of child, whole body	Child	D01a
Direct spray of woman, feet and lower legs	Adult female	D01b
Water consumption (spill)	Child	D05
Fish consumption (spill)	Adult male	D08a
Fish consumption (spill)	Subsistence populations	D08b
Non-Accidental Acute Exposures (dose in mg/kg bw/event)		
Vegetation contact, shorts and t-shirt	Adult female	D02
Contaminated fruit consumption	Adult female	D03a
Contaminated vegetation consumption	Adult female	D03b

Swimming, one hour	Adult female	D11
Water consumption	Child	D06
Fish consumption	Adult male	D09c
Fish consumption	Subsistence populations	D09d
Chronic/Longer Term Exposures (dose in mg/kg bw/day)		
Contaminated fruit consumption	Adult female	D04a
Contaminated vegetation consumption	Adult female	D04b
Water consumption	Adult male	D07
Fish consumption	Adult male	D09a
Fish consumption	Subsistence populations	D09b

1.3.1.3.1.1 Workers

General occupational exposure scenarios relate to exposure while handling chemicals during normal use, whereas accidental/incidental exposure scenarios account for occurrences of misuse, mishandling and unexpected events that result in exposure higher than expected during typical chemical application. For USDA/FS risk assessments, dermal exposure is assessed in terms of absorption-based modeling, where the amount of chemical handled is used to estimate the amount of chemical absorbed through the skin (SERA 2012). While such estimates are often considered crude, additional information is incorporated into risk assessments for each chemical, as available (*ibid*).

While aerial application is assessed in USDA/FS risk assessments, it is not under consideration in the VTP and alternatives and is thus not assessed. Per SERA (2012), ground application methods are grouped into two predominate categories in USDA/FS risk assessments:

(1) directed foliar applications (i.e., cut surface, backpacks), and

(2) broadcast foliar applications

At first glance these grouping may seem unexpectedly broad, however current empirical evidence does not indicate that more detailed application categories are usually necessary, as the degree of chemical exposure does not significantly vary between specific methods within each application category (*ibid*). A standard set of dermal absorption rates was established using nine commonly used chemicals (SERA 2012 and Table D.3-13). Generally, these estimated dermal absorption rates, which are in terms of the amount of chemical

handled, are used to calculate estimates of worker and public chemical exposure. The one exception for chemicals assessed in this PEIR is triclopyr BEE, which has been found to have higher dermal absorption rate than the other chemicals considered (SERA 2011d). There are different exposures rates for each category of application due to the different amounts of dermal exposure. Worker exposure estimates are a product of the exposure rate (in mg/kg bw/lb of chemical handled) and the pounds of chemical handled per day or event (SERA 2012). The resulting chemical exposures are expressed as milligrams of chemical per kilograms of body weight per day or event (mg/kg bw/day or mg/kg bw/event).

Table D.3-13

Occupational Exposure Rates Used In Forest Service Risk Assessments

Worker Application Group	Rate (mg/kg bw/lb chemical handled)			References
	Central	Lower	Upper	
Directed foliar	0.003	0.0003	0.01	SERA 2012, Table 6
Broadcast foliar	0.0002	0.00001	0.0009	SERA 2012, Table 6
Triclopyr BEE directed foliar *	0.0058	0.00086	0.039	Middendorf 1992b as referenced in SERA 2011d, Table 18
Triclopyr BEE broadcast foliar *	0.00038	0.00003	0.0035	Adjusted [1]

*Evidence suggests triclopyr BEE has higher rates of exposure than triclopyr TEA and other chemicals. [1] Adjusted as defined in SERA 2011d: "The ratio of rates from Middendorf (1992b) to standard Forest Service rates for directed foliar spray are approximately 1.9, 2.9, and 3.9 based on the central estimate, lower bound, and upper bound. These ratios are used to adjust rates for ground boom…applications of triclopyr BEE based on the standard rates for these application methods."

Accidental/incidental exposures to workers are most likely related to accidental spills or splashing the chemical agent on skin or in eyes. Information on ocular exposure primarily refers to effects, so qualitative discussion is reserved for the risk characterization. Dermal exposure is the predominant exposure route and is studied in depth, so it is assessed quantitatively in USDA/FS risk assessments (SERA 2012). Some standard scenarios involve the amount of dermal absorption associated with direct contact, by wearing contaminated gloves or by full immersion of the hands in a field solution over specific time intervals (i.e., usually one minute and one hour). Other scenarios involve spilling the chemical agent directly onto hands or lower legs (Table D.3-12). For these scenarios, the exposure rate is measured as milligrams of chemical per kilogram of body weight per event (mg/kg bw/event) in USDA/FS risk assessments.

1.3.1.3.1.2 Public

The public may be exposed to chemicals acutely or chronically through several routes. Chemical exposure has the potential to occur to members of the public via direct spray or indirect contact by wind-drifted spray. Exposure may also occur by consumption of, or contact with, contaminated surface or ground water. Consumption and/or contact with contaminated fish, game or plants may also be routes of undesired chemical exposure.

Potential exposure to humans in part depends on the ownership of land being treated with herbicides. VTP treatments may occur on public lands such as State Parks, State Recreation Areas, and lands owned by the Department of Fish and Wildlife. Chemical treatments on these lands have a greater possibility of directly impacting members of the public, at least in part because more people are likely to be exposed on public lands relative to private lands. Under the VTP and alternatives, private lands make up the bulk of the landscape available for treatment. Given that members of the public have limited access to private lands (i.e., by invitation only) the risk of direct chemical exposure is minimal.

While relatively few public lands are proposed for treatment, developed recreation areas, which include trailheads, campgrounds, picnic areas, recreation sites, boat ramps, ski areas, and work centers, have the potential to be chemically treated, especially on State Park lands. Treatments in or near these areas would have the greatest potential for exposing the public to chemicals. Under normal (non-accidental) application conditions, there is no expectation that the public will be exposed to chemicals above acceptable risk levels, given protections required by law and the mitigation measures outlined in Section 4.8.3. Decisions to treat vegetation with chemicals under this program will ultimately be made by landowners and CAL FIRE project coordinators.

Similar to workers in Forest Service risk assessments, exposure to members of the public is grouped into *general exposure* from normal use of chemicals and more severe *accidental/incidental exposure* resulting from misuse or unusual circumstances (SERA 2012). In Forest Service risk assessments, a number of specific scenarios are consistently used to characterize exposure of the general public (*ibid* and Table D.13-12).

The exposure assessments developed in Forest Service risk assessments are based on Extreme Values rather than a single value. Extreme value exposure assessments, as the name implies, bracket the most plausible estimate of exposure (referred to statistically as the central or maximum likelihood estimate) with lower and upper bounds of credible exposure levels. This Extreme Value approach is essentially an elaboration on the concept of the Most Exposed Individual (MEI), sometime referred to as the Maximum Exposed Individual. As this name implies, exposure assessments that use the MEI approach attempt to characterize the extreme but still plausible upper limit on exposure. This common approach to exposure assessment is used by the U. S. EPA, other government agencies, and the International

Commission on Radiological Protection. In most Forest Service risk assessments, upper bounds on exposure are intended to encompass exposures to the MEI.

As with workers, exposure to the public is assessed in USDA/FS risk assessments using acute and chronic exposure scenarios (Table D.3-12). Some scenarios involve direct sprays and are modeled for ground application in a similar way to accidental spills for workers. For such scenarios, it is assumed that some of the chemical remains on the skin and is absorbed by first-order kinetics (SERA 2012). Another scenario involves dermal exposure, which assumes that an adult woman is wearing shorts and a t-shirt when coming into contact with contaminated vegetation. The outcome of this scenario depends on estimates of dislodgeable reside and dermal transfer rates (*ibid*). There are multiple scenarios involving contaminated water, which are broken into categories involving accidental spill as well as accidental direct spray of or drift to a pond or stream (*ibid*). Several scenarios also evaluate the acute and chronic consumption of contaminated fish, broadleaf vegetation, and fruit. One scenario also involves the dermal exposure from swimming in contaminated surface water, which is calculated essentially identically to the contaminated glove scenario for worker exposure (*ibid*). Short- term peak and long-term average water contamination rates (WCRs) are determined for the scenarios involving water as shown in Table D.3-14. Together, these scenarios assess a wide range of potential chemical exposure outcomes.

Table D.3-14

Water Concentration Rates of Chemicals Proposed for Use*

	Short-term peak concentrations (mg/L)			Longer-term average concentrations (mg/L)		
	Peak			**Average**		
Chemical	*Central*	*Lower*	*Upper*	*Central*	*Lower*	*Upper*
Borax	0.03	0.006	0.1	0.014	0.002	0.07
Clopyralid	0.02	0.005	0.07	0.007	0.001	0.013
Glyphosate	0.011	0.0013	0.083	0.00019	0.000088	0.0058
Hexachlorobenzene	N/A	N/A	N/A	0.00039	0.00004	0.005
Hexazinone	0.1	0.0005	0.4	0.02	0.00001	0.07
Imazapyr	0.02	0.000009	0.26	0.007	0.000003	0.12
NP9E	6.1	3.0	15.1	0.007	0.0	0.014

Sulfometuron methyl	0.001	0.00006	0.02	0.00004	0.00001	0.00007
Triclopyr BEE	0.0004	0.00000015	0.03	0.000002	2.0×10^{-10}	0.00007
Triclopyr TCP	0.0009	0.00000001	0.028	0.00005	3.0×10^{-12}	0.002
Triclopyr TEA	0.003	0.000001	0.24	0.001	2.0×10^{-10}	0.06

*All values calculated using FS WorksheetMaker workbooks (worksheets B04Rt and B04a), except those for NP9E that come from USDA/FS 2003b

An important consideration for scenarios involving consumption of fish is the propensity of a chemical to accumulate in fish tissues. The ratio of chemical concentration in fish tissue relative to the chemical concentration in water is referred to as the bioconcentration factor (BCF). If, for example, the concentration in an organism is 5 mg/kg and the concentration in the water is 1 mg/L, the bioconcentration factor (BCF) is 5 L/kg [5 mg/kg ÷ 1 mg/L] (SERA 2012). BCF values ≤1 indicate that chemicals are not expected to bioconcentrate in fish (USDA/FS 2006a). Generally speaking, the amount of chemical accumulation depends on the concentration of the chemical agent in the water and the maximum concentration that can occur in the tissue of the organism (*ibid*; see OPPTS 850.1730 for U.S. EPA test protocols). As with most absorption processes, bioconcentration depends initially on the duration of exposure, but eventually reaches a steady state (SERA 2012). Separate BCF values are calculated for acute (24 hour) and long-term (steady state) exposures and are used in respective scenarios to determine plausible exposure through consumption of contaminated fish (Table D.3-15).

Table D.3-15

Bioconcentration Factors of Chemicals Proposed for Use As L/Kg Fish*

Chemicals	Edible portion, acute	Edible portion, chronic	Whole fish, acute	Whole fish, chronic
Borax	1.0	1.0	1.0	1.0
Clopyralid	1.0	1.0	1.0	1.0
Glyphosate	0.38	0.38	0.52	0.52
Hexachlorobenzene	2,000	20,000	2,000	20,000
Hexazinone	1.0	2.1	2.0	5.5
Imazapyr	0.5	0.5	0.5	0.5
NP9E	1.0	1.0	1.0	1.0

Sulfometuron methyl	3.0	3.5	7.0	6.0
Triclopyr BEE	0.06	0.06	0.83	0.83
Triclopyr TCP	0.06	0.06	0.83	0.83
Triclopyr TEA	0.06	0.06	0.83	0.83

*All values calculated using FS WorksheetMaker except those for NP9E, which are disclosed in USDA/FS 2003b

1.3.1.3.2 Chemical Dose Assessments

The most recent SERA and USDA/FS risk assessments for each chemical were used to summarize the exposure assessment in this PEIR. Values disclosed in this section, however, have been updated using the most current version of FS WorksheetMaker for each chemical. As done for Forest Service risk assessments, exposure is summarized in terms of the typical application rate and discussions regarding the potential impacts of higher application rates are restricted to the risk characterization section for each chemical.

Borax (Sources: FS WSM ver. 6.00.10; SERA 2006a)

The chemical sodium tetraborate decahydrate, alternatively called borax, is a fungicide used to treat heterobasidion root disease. As well as being a fungicide, the application methods of this chemical are different than any other chemicals proposed in the PEIR, because the chemical is only applied directly to freshly cut tree stumps. Thus, many of the scenarios are not appropriate for the application of borax. The per acre application rate is approximate, based on the cumulative area of freshly cut stump surface. One product registered in California for forestry use is Sporax, which is a granular product composed only of sodium tetraborate decahydrate. The USDA/FS risk assessment is only written in terms of Sporax, and not other products. Thus, for the purposes of this document, references to borax are specifically referring to sodium tetraborate decahydrate and the associated product Sporax, and not other boron derived products. Since the chemical component of concern is boron, toxicity information above and all exposure information is expressed in boron equivalents (B).

Boron is a naturally occurring element that is ubiquitous in nature. The use of borax by the Forest Service is not thought to substantially contribute to human exposure through soil and water, except perhaps in extreme cases. Given that Sporax is only applied in a granular form in a specialized way, several of the standard exposure scenarios are not applicable. Other scenarios were adapted in the USDA/FS risk assessment to more accurately reflect potential exposures. Inapplicable scenarios relating to general worker exposure, direct spray, oral exposure by ingestion of contaminated vegetation, fruit, or fish, and direct exposure from contaminated vegetation were omitted from the Forest Service risk assessments. The scenario involving a child being directly sprayed with a chemical was

adapted to a child ingesting borax directly from a freshly treated stump. Scenarios considered in the human health risk assessment also include contact with contaminated gloves for workers and exposure via consumption of water contaminated by an accidental spill or by run-off.

Only the most extreme scenarios related to borax applications by the Forest Service are likely to substantially contribute to levels of boron exposure in humans. The modeled exposures for workers relate to wearing contaminated gloves for 1 minute or 1 hour, with upper bounds at an application rate of 1 lbs a.i. per acre being 2.88×10^{-5} and 2.30×10^{-4} mg/kg bw/event respectively. The scenario of a child consuming Sporax directly from a tree stump resulted in the greatest exposure, with an upper bound of 3.24 mg B/kg bw/day. This estimate was calculated for the Forest Service using the average daily soil consumption by a child. All other public exposures were substantially lower, with remaining upper bounds ranging from 0.0024 to 0.14 mg B/kg bw/event, relating to chronic ingestion of contaminated water by an adult male and acute ingestion of contaminated pond water by a child after a spill respectively.

Clopyralid (Sources: FS WSM ver. 6.00.07 & 6.00.10; SERA 2004a)

The typical rate of application for clopyralid in the USDA/FS programs is 0.35 lb/acre and this was the rate used to calculate exposure values in the SERA 2004a risk assessment. In California, however, the maximum application rate for clopyralid is restricted to 0.25 lbs/acre, and thus clopyralid exposure is anticipated to be lower under the VTP and alternatives than predicted for Forest Service projects. Given the clopyralid restrictions in California, the application rate of 0.25 lb/acre was used as both a typical and upper application rate for calculations.

For acute or chronic exposure scenarios of the public, the scenario relating to a child consuming water after the contamination of a small pond had the highest exposure estimate (e.g. an upper bound of 1.28 mg/kg bw). All other occupational and public scenarios result in often substantially lower exposures. General occupational exposures for terrestrial applications, for example, range from the lowest bound of 1.13×10^{-4} mg/kg bw/day for direct foliar spray, to the upper bound of 0.038 mg/kg bw/day for broadcast spray at an application rate of 0.25 lb a.e./acre. All occupational exposures associated with accidental/incidental events lead to exposures below the broadcast spray upper bound for general occupational exposures. This is in large part because all incidental exposure scenarios involve dermal absorption, and clopyralid is not readily absorbed through the skin. With public exposure scenarios, the upper bounds for non-accidental public exposure range from 3.0×10^{-8} mg/kg bw to 0.338 mg/kg bw, which resulted from the scenarios involving an adult female swimming in contaminated water for one hour, and one consuming contaminated vegetation, respectively. All chronic exposures for the public result in doses lower than the upper bound for contact with contaminated vegetation.

Important impurities of technical grade clopyralid are hexachlorobenzene and pentachlorobenzene, which are found at average concentrations of about 2.5 ppm and 0.3 ppm respectively. Hexachlorobenzene is a common contaminate found in industrial emissions, at hazardous waste sites and on contaminated foods. This impurity is thus found in detectable concentrations in most individuals, and background levels of exposure are thought to be around 1.0×10^{-6} mg/kg/day. The use of clopyralid in the VTP and alternatives are not thought to contribute substantially to ambient levels of the impurity.

Local exposure to hexachlorobenzene, however, for workers and the public from the use of clopyralid was empirically evaluated and discussed in the SERA 2004a risk assessment for clopyralid. Calculations were updated in 2006 using version 4.04 of WorksheetMaker. These calculations were outdated, however, so Patrick Durkin of SERA Inc. graciously provided a workbook completed using WorksheetMaker 6.00.07 that evaluated hexachlorobenzene in picloram, and suggested changing the application rate to that applicable to clopyralid (i.e., 8.75×10^{-7} lb/acre). For workers, the highest dose is associated with the upper bound of broadcast spray (1.32×10^{-7}), which is well below the background level of hexachlorobenzene ($>1 \times 10^{-6}$). In the new version 6.00.7 workbooks, there are no exposure values or assessments for either accidental exposure of workers, or acute exposure of the public. All chronic exposures to the public lead to exposures less than the background levels of the compound.

Glyphosate (Sources: FS WSM v. 6.00.10; SERA 2011b)

Workbooks were created for each applicable application method (broadcast and direct foliar), as well as for more and less toxic formulations for glyphosate using WorksheetMaker. The level of exposure did not vary between the more and less toxic formulations of glyphosate. When considering general occupational exposure, the central estimate for directed foliar spray (0.026 mg/kg bw/day) is lower than the broadcast foliar spray estimate (0.045 mg/kg bw/day) at 2 lb a.e./acre. The upper bounds of exposure are 0.16 mg/kg bw/day for directed foliar exposure, whereas the upper bound for broadcast exposure was 0.30 mg/kg bw/day. All accidental worker exposure scenarios resulted in estimates that were lower than those associated with general worker exposure of the equivalent bound, in part because this chemical is not readily absorbed through the skin.

When considering exposure of the public, there is a wide range of estimated exposures, ranging from the lower bound of 2.54×10^{-10} mg/kg bw for the scenario of a woman swimming for one hour, to the highest upper bound of 4.10 mg/kg bw for exposure resulting from the scenario of a child consuming contaminated water after a spill in a small pond. The second highest estimated exposure for the public, at an application rate of 2 lb a.e./acre, is 2.70 mg/kg bw for an adult woman who consumes contaminated vegetation. All other acute scenarios for accidental and incidental events led to exposure estimates lower than 2.70 mg/kg bw, and corresponding estimates for chronic exposure were smaller still.

Hexazinone (Sources: FS WSM v. 6.00.10; SERA 2005)

The USDA/FS uses both liquid and granular formulations of hexazinone for vegetation management. Both formulations will be potentially used under the VTP and alternatives. It should be noted that some granular formulations, such as Velpar DF, are mixed with water prior to application, and such formulations are evaluated as equivalent to liquid formulations in terms of exposure in USDA/FS risk assessments, as the foliage application is the same. Only formulations such as Velpar ULW, which are applied in the granular form directly to soil, are considered using granular workbooks. The typical application rate of 2 lbs a.i./acre has been used both liquid and granular formulations.

Evidence shows that general worker exposure rates do not differ whether the formulation is liquid or granular, whereas accidental exposures do vary between liquid and granular formulations. For general worker exposure, broadcast foliar spray has the highest upper bound (0.30 mg/kg bw/day) relative to exposure during direct soil or foliar application (0.16 mg/kg bw/day). When considering accidental exposures to workers, scenarios involving spills are not applicable for granular formulations, while scenarios of wearing of contaminated gloves are relevant.

While most applicable exposure scenarios were below the levels of the general worker exposure, this was not the case for all central, upper and lower bounds with the scenario involving a contaminated glove being worn for 1 hour. The upper bound for this scenario for liquid and granular formulations was 0.33 and 0.23 mg/kg bw/event, respectively. The point was made in the SERA assessment that the:

> *. . relatively minor difference* [between upper bounds of granule and liquid formulations] *is due to the fact that the upper range of exposure to liquid formulation exceeds the solubility of hexazinone in water, a limiting factor in exposures for the granular formulation. The high exposure to the liquid formulation appears to be associated with the presence of adjuvants in the liquid formulation (probably ethanol) that functionally increases the solubility of hexazinone in the field solution.* (SERA 2005, p. 3-18)

For the public, most accidental and non-accidental exposure scenarios pertain to both granular and liquid formulations, though direct spray scenarios were not applicable to granular application. Doses from acute accidents were lowest for the scenario of a male consuming fish after a spill, with granular and liquid lower bounds at 0.016 and 0.0016 mg/kg bw/event respectively. By contrast, the highest dose from acute accidents was for the scenario that a child consumed water after a spill into a small pond, with both granular and liquid upper bounds being about to 4.1 mg/kg bw/event. The acute non-accident scenario that indicates the lowest dose relates to a female swimming for one hour in contaminated water, with a lower bound of 6.3×10^{-8} mg/kg bw/event for both granular and liquid formulations. The highest dose for non-accident scenarios, by contrast, relates to an

adult female consuming contaminated vegetation, with upper bounds of 2.7 and 1.1 mg/kg/event for liquid and granular formulations respectively.

Overall, chronic exposure scenarios resulted in estimates much lower than acute scenarios respectively for both liquid and granule products. The most substantial chronic exposure difference between liquid and granular formulations involved chronic exposure to contaminated vegetation. The granule formulation ranged from 0.001 to 0.045 mg/kg bw/day, whereas the liquid formulation ranged from 0.0095 to 1.14 mg/kg bw/day for the contaminated vegetation scenario. The difference between the upper bounds of the two formulations is a factor of 25, which likely results from the propensity of the liquid to deposit onto vegetation more readily than granules of hexazinone.

For hexazinone the assumption is made that there is no dissipation in plants over the course of the chronic contaminated vegetation scenario. This is due to the soil-active nature of hexazinone and its continual uptake into plants through the root system (SERA 1997b). This assumption is consistent with a study conducted by the California Department of Pesticide Regulation in which low but persistent levels of hexazinone were found in four species of plants of interest to Native Americans (CDPR 2002).

Imazapyr (Sources: FS WSM v. 6.00.10; SERA 2011c)

While both direct foliar and broadcast application methods are assessed for worker exposure in this document, it is acknowledged that broadcast application is not likely with imazapyr. When examining general worker exposure, the upper bound of direct foliar application is 0.02 mg/kg/day, whereas broadcast application leads to an upper bound of 0.045 mg/kg/day at the typical USDA/FS application rate of 0.3 lb a.e./acre. Occupational exposure estimates for accidental or incidental exposure scenarios were lower than estimates for general daily occupational exposure. The estimate for wearing contaminated gloves for 1 hour, for instance, has the highest upper limit for the accidental/incidental exposure scenarios, at only 0.009 mg/kg bw/event.

When considering the public, the highest upper limit estimate for the acute accident scenario of a child consuming contaminated water just after a spill is 0.6 mg/kg bw/event at a 0.3 lb a.e./acre application rate. As with other chemicals, the parameters for this scenario are considered highly arbitrary. The non-accidental acute exposure levels are highest with the consumption of contaminated vegetation scenario (i.e., upper bound of 0.41 mg/kg bw/day event at 0.3 lb a.e./acre), though most are considerably lower. The lowest estimate results from the scenario of an adult female swimming in contaminated water (2.0×10^{-11} mg/kg bw/event). Chronic exposure estimates are much lower than for the corresponding acute exposure scenarios.

NP9E (Sources: FS WS ver. 2.02; USDA/FS 2003b)

Central and upper estimates for general worker exposure are higher for broadcast spray application (0.037 and 1.01 mg/kg bw/day respectively) than for direct foliar applications (0.53 and 0.01 mg/kg bw/day). The highest accidental/incidental exposure

estimate for workers relates to individuals wearing a contaminated glove for one hour, and resulted in a central estimate of 0.01 mg/kg bw/event, with a range of 0.0019 to 0.066 mg/kg bw/event.

For the public, most exposure estimates were lower than the general worker exposures, except for accidental exposures involving the public. The accidental scenario that lead to the highest exposure involved a child consuming contaminated water from a small pond, which had a typical exposure of 0.46 mg/kg bw/event, with exposures ranging from 0.14 to 1.71 mg/kg bw/event Beyond the contaminated water scenario, other accidental event estimates ranged from 1.25 mg/kg bw/day for short-term consumption of contaminated fruit, to 3.6 x 10^{-6} mg/kg bw/day for a woman making dermal contact with contaminated vegetation. As with other chemicals, accidental exposure scenarios should be regarded as extreme, but to some extent plausible. Chronic exposure scenarios for the public led to a wide range of upper limits, from 2.0 x 10^{-6} to 0.02 mg/kg bw/day.

Sulfometuron methyl (Sources: FS WSM v. 6.00.10; SERA 2004c)

While both direct foliar and broadcast application methods are assessed for worker exposure in this document, it is acknowledged that broadcast application is not likely with sulfometuron methyl. Exposure estimates for workers are highest for broadcast application, with central and upper bounds of 0.001 and 0.007 mg/kg bw/day at the typical Forest Service rate of application of 0.045 lb/acre. Directed foliar application, by contrast, leads to central and upper exposure estimates of 0.0006 and 0.004 mg/kg bw/day. Exposure estimates for accidental exposures related to workers fell within the ranges of the general exposures for workers.

There is variation as to whether exposure estimates for the public were higher or lower than those for general worker exposures. The highest short-term accidental exposure involves a small child consuming water from a small pond that has been contaminated (upper bound of 0.094 mg/kg bw/day). As with other chemicals, this scenario is particularly implausible and arbitrary. The highest estimates for acute and chronic non-accidental exposure to members of the public were substantially lower and related to the consumption of contaminated broadleaf vegetation (upper bounds of 0.06 mg/kg bw/event and 0.0097 mg/kg bw/day, respectively). By contrast, the lowest estimates for acute and chronic non-accidental exposure involved an adult female swimming in contaminated water for 1 hour (1.4 x 10^{-12} mg/kg bw/event) and an adult male consuming contaminated fish (2.3 x 10^{-10} mg/kg bw/day).

Triclopyr (Sources: FS WSM v. 6.00.10; SERA 2011d)

As discussed in the USDA/FS risk assessment, the standard worker exposure rates (mg/kg bw/lb/acre) that are typically used to evaluate general occupational exposure are not applicable to all forms of triclopyr. Current evidence regarding dermal absorption suggest that no exposure rate adjustments are needed for the TEA form of triclopyr, though

the BEE form of triclopyr was found to have a much higher exposure rate than the standard (Table 5.17.25). Thus, the USDA/FS adopted rates established in a study for backpack spraying (Middendorf 1992b as referenced in SERA 2011d) and to use this information to adjust the rates for broadcast foliar application methods. SERA (2011d) contains details of studies and the rationale used by the Forest Service to adapt the exposure rates of BEE. Substantial differences were found in the risk characterization of TEA and BEE for workers.

For worker exposure, BEE form had a higher dose rate than the TEA form regardless of application method. That said, broadcast application led to higher exposure estimates than direct foliar application for both TEA and BEE. For example, the upper bound for broadcast application of BEE was 0.588 mg/kg/day, whereas the same exposure for TEA was 0.15 mg/kg/day, at an application rate of 1 lb. a.e./acre. The upper bound accidental/incidental exposure estimates for TEA involving workers were below the upper bound for general exposures (i.e., <0.15 mg/kg/day) likely during broadcast application of TEA. This was also true for BEE applications, except for BEE exposure from wearing contaminated gloves, which led to an exposure of 7.49 mg/kg/event at an application rate of 1 lb a.e./acre.

When considering public exposure scenarios, the consumption of water by a child shortly after a spill led to the greatest exposure rate for both BEE and TEA (upper bound of 2.05 mg/kg/day). Consumption of broadleaf vegetation shortly after spraying led to the next highest exposure rate for both forms of triclopyr (upper bound of 1.35 mg/kg/day). Other scenarios involving skin contact and consumption of contaminated water, fish, vegetation and fruit resulted in substantially lower exposures, with upper bounds ranging from 6.0×10^{-10} to 0.07 mg/kg/day. Whether considering occupational or public exposure, triclopyr TEA may cause moderate to severe ocular damage if splashed into the eye, though this potential effect is only qualitatively considered in the most recent USDA/FS risk assessment.

The metabolite 3,5,6-trichloro-2-pyridinol (TCP) is known to be more toxic than triclopyr, particularly to some aquatic organisms, and thus the potential exposure was quantitatively assessed for the USDA/FS risk assessment using all available information. The accidental spill scenario led to a peak concentration of triclopyr in water of about 1.8 (0.23 to 18.2) mg a.e./L. While no such direct comparative data exists for TCP, the concentrations after aquatic triclopyr application have been determined in several studies, and this information has was used to approximate spill information as discussed in SERA (2011d). After aquatic applications, triclopyr was several magnitudes higher than TCP in concentration. In the Forest Service risk assessments, studies evaluating concentrations of triclopyr and TCP were used to approximate “the concentrations of TCP in a pond following an accidental spill are estimated at about 0.0077 (0.0004 to 0.13) mg/L” (see SERA 2011d). Scenarios involving direct spraying or drift of triclopyr into ponds and streams would lead to exposure levels much lower than those for similar direct spill scenarios, and thus TCP levels would

also be much less. Calculations of pond and stream contamination vary depending on several environmental and application factors, as modeled in Gleams-Drivers (SERA 2007a).

Given the toxicity of TCP, the Forest Service risk assessment evaluated the contamination of fruits, vegetable, and water using models with what limited information was available. TCP was found to be "somewhat more persistent in soil when compared to triclopyr, but less persistent than triclopyr in water." Acute and chronic exposure to triclopyr is greater through consumption of vegetation than compared to fruit (e.g. acute upper bounds: 1.35 and 0.19 mg/kg/event respectively). Exposure to TCP through consumption of vegetation and fruit also follows this pattern (e.g. acute upper bounds 0.38 and 0.053 mg/kg/event respectively), though overall exposure to TCP is less than for triclopyr.

1.3.1.4 Dose-Response Assessment

In addition to understanding the likelihood of human exposure from chemical applications, it is important to consider how the amount, or dose, of a chemical affects the degree or severity of risk (SERA 2012). In USDA/FS assessments, this is quantified in terms of Reference Doses (RfD) or Reference Concentrations (RfC) for each chemical. The units for oral doses (RfD values) are mg/kg/day, whereas inhalation doses are measured as RfC values, in mg/m^3. These values are most often taken directly or derived from U.S. EPA studies, as the U.S. EPA is better equipped to provide analysis and review that is outside of the scope of USDA/FS risk assessments. Beyond clear budgetary benefits, this approach promotes information sharing between federal and state agencies and other organizations, rather than a duplication of efforts. In the SERA (2012) report reference doses are described as "point estimates (single numbers rather than ranges) of doses that are not believed to be associated with any adverse effect and that are not directly related to a dose-response model." Using a reference dose methodology ensures a conservative approach to dose-response assessment.

Both chronic and acute RfDs are used to characterize risk in USDA/FS risk assessments. Per SERA (2012) "[c]hronic RfD values are intended to estimate dose levels associated with a negligible or at least defined level of risk over a lifetime of exposure." Chronic No-Observed-Adverse-Effect-Level (NOAEL) values used are typically based on long-term (chronic or subchronic) toxicity studies, or multigenerational studies (SERA 2012). When there is no NOAEL available, a Lowest-Observed-Adverse-Effect-Level (LOAEL) may be used in conjunction with an uncertainty factor (UF). RfD values result from experimental toxicity values (NOAEL or LOAEL) divided by uncertainty factors. Uncertainty factors are typically established in factors of 10. If several factors are applicable to the data of a particular NOAEL used for establishing a chemical RfD, the factors are multiplied to determine an overall uncertainty value. For example, several of the chemicals under consideration were assigned an uncertainty factor of 100, which in some cases represents a factor of 10 for differences between species multiplied by a factor of 10 for within species uncertainty.

While comparable to chronic RfDs conceptually, acute RfD values are intended to only assess risks associated with one day or less of exposure to a chemical (SERA 2012). Acute RfDs have only recently been determined for U.S. EPA risk assessments, and are determined differently, depending on the chemical, for Forest Service assessments (*ibid*). There seems to be little difference, however, between acute and chronic toxicity of chemical agents that appear to have weak dose-duration relationships, and in such cases the chronic RfDs are used (*ibid*). When risks are apparent, further attempts should be made to categorize these risks. Table D.3-16 displays RfD values used in the most current USDA/FS risk assessments for chemicals that will potentially be used in the VTP and alternatives.

Table D.3-16

Reference Doses (RfD) Values Used By the USDA/FS for Proposed Chemicals

Active Ingredient	ACUTE mg/kg bw/event	CHRONIC mg/kg bw/day	References
Borax	chronic used	0.200	SERA 2006a, p. 3-21
Clopyralid	0.750	0.150	SERA 2004a, p. 3-27
Glyphosate	chronic used	2.000	SERA 2011b, p. 102
Hexazinone	4.000	0.050	SERA 2005, p. 3-35
Imazapyr	chronic used	2.500	SERA 2011c, p. 47
Sulfometuron methyl	0.870 [1]	0.020	SERA 2004c, p. 3-23
Triclopyr	1.000	0.050 [2]	SERA 2011d, p. 71
TCP - Triclopyr metabolite	0.025	0.012	SERA 2011d, p. 71
NP9E	0.1	0.100 [1]	USDA/FS 2003b, p. 29

[1] While the USDA/FS usually uses the RfD determined by the U.S. EPA, additional data was used to establish this value. [2] Also the acute RfD value for women of childbearing age.

Dose-severity relationships are important to consider only when plausible exposures are above a level of concern (LOC). Given the conservative nature of exposure and dose-response assessments done by the USDA/FS, no elaboration was needed in cases where upper ranges of plausible exposure are below the LOC. However, when risks were apparent, the Forest Service would compare any, often sparse data, such as LOAELs and NOAELs, though explicit dose-response models were not used. The intention for doing this type of dose-

response assessment allowed for estimates when explicit data is lacking, which can then be discussed in the risk characterization section for each chemical.

Chemicals potentially used under the VTP and alternatives are not classified as carcinogens, although some impurities and/or metabolites in technical grade active ingredients or surfactants have the potential to be carcinogens. Hexachlorobenzene, for example, is a manufacturing by-product of clopyralid that is a known carcinogen. The U.S. EPA determines values, known as the cancer potency factors, to approximate the cancer risk of chemicals. These values are adopted from the U.S. EPA for use in Forest Service risk assessments.

Borax (Sources: FS WSM ver. 6.00.10; SERA 2006a)

The U.S. EPA used two developmental studies on boric acid and borates to establish a chronic RfD of 0.2 mg B/kg/day for boron. Decreased fetal weights observed during these studies on rats served as the most sensitive endpoints. This was calculated using a benchmark response (BMR) level, divided by an uncertainty factor of 66, which considers both interspecies and sensitive individual variability. No acute RfD has been established for boron at the time the Forest Service risk assessment was written, and thus, the chronic RfD was also used for one-day exposures.

Clopyralid (Sources: FS WSM ver. 6.00.07 & 6.00.10; SERA 2004a)

The Forest Service used acute and chronic RfD values of 0.75 and 0.15 mg/kg bw/day for clopyralid, as derived by the U.S. EPA. An acute NOAEL of 75 mg/kg bw/day was the basis for the short-term RfD. A NOAEL of 15 mg/kg bw/day from a 2-year dietary study was used to establish the chronic RfD. An uncertainty factor of 100 was used to obtain both acute and chronic RfD values. As is commonly observed in chronic toxicity studies, changes in body, liver and kidney weights were noted in several additional studies with clopyralid. It was also indicated that some mammals developed thickening in some epithelial tissue. The importance of this less common effect is not well understood. Most the anticipated exposures were below the RfD and those that were above the RfD only marginally exceeded this dose. Thus, there was no need for further modeling to complete the risk characterization.

Technical grade clopyralid is contaminated with hexachlorobenzene and pentachlorobenzene. The presence of these contaminants was quantitatively evaluated in the Forest Service risk assessment, to a limited extent. Due to the low abundance of these contaminants in technical grade clopyralid and the low potency of each contaminant relative to clopyralid, these contaminants were not anticipated to substantially influence any systemic-toxic effects associated with clopyralid. The carcinogenicity of hexachlorobenzene, however, was considered separately using the U.S. EPA's cancer potency parameter.

Glyphosate (Sources: FS WSM v. 6.00.10; SERA 2011b)

The chronic RfD of 2 mg/kg bw currently used in Forest Service risk assessments was derived by the U.S. EPA Office of Pesticide Programs, based on a chronic developmental study using rabbits that defined both an NOAEL of 175 mg/kg bw/day and definitive LOAEL of 350 mg/kg bw/day. Two uncertainty factors of 10 (one for sensitive individuals and one for species extrapolation) were multiplied, for a total uncertainty factor of 100. There is no acute RfD defined by the U.S. EPA, so the chronic RfD of 2 mg/kg bw/day was used for both acute and chronic exposure characterizations in the USDA/FS assessment.

Some reservations regarding the use of this RfD are discussed in detail in the Forest Service assessment. Moreover, this RfD was established using technical grade glyphosate, though some surfactants, such as POEA, are known to have comparable or greater toxicity than glyphosate. Thus, the RfD equivalency of technical grade glyphosate and mixtures containing POEA surfactants may be questioned. The NOAEL was then divided by the UF, and in the case of glyphosate, the result was rounded. As discussed in the USDA/FS risk assessment, surfactants in glyphosate formulations have the potential to be more toxic in some circumstances, however, currently there is not compelling evidence that would suggest an alternative RfD is necessary for formulations used in the U.S. The margin between the NOAEL and LOAEL is narrow when considering that some dam mortality was observed at the LOAEL, which indicates that the NOAEL may be viewed as a frank effect level. Concern should be given for any doses that exceed the RfD of 2 mg/kg bw/day, especially in terms of sensitive individuals, though defining a clear threshold for adverse effects is difficult for glyphosate.

Hexazinone (Sources: FS WSM v. 6.00.10; SERA 2005)

The USDA/FS adopted the acute and chronic RfD values of 4 mg/kg bw/event and 0.05 mg/kg bw/day, as derived by the U.S. EPA. The acute RfD was based on reproductive/ developmental studies using rabbits and rats that resulted in NOAELs of 400 mg/kg bw/day. This dosage was then divided by an uncertainty factor of 100. The chronic RfD, by contrast, was developed from a study that resulted in a NOAEL of 5 mg/kg bw/day using dogs. Again, an uncertainty factor of 100 was used, which in this case consisted of two factors of 10 to account for species-to-species extrapolation and sensitive subgroups.

Imazapyr (Sources: FS WSM v. 6.00.10; SERA 2011c)

A chronic RfD of 2.5 mg/kg bw/day was established by the U.S. EPA and used in the USDA/FS risk assessment, based primarily on a dog study with a NOAEL of 250 mg/kg bw/day, which is reinforced by additional rat and mice studies. Uncertainty factors of 10 for sensitive individuals in the human population and 10 for species extrapolation were multiplied, for an overall uncertainty factor of 100. There is no acute RfD defined by the U.S. EPA, so the chronic RfD of 2.5 mg/kg bw/day was used for both acute and chronic exposure characterizations in the USDA/FS assessment. Dose-severity relationships could not be made, in part because doses could not be associated with any adverse effects and

none of the HQs exceed the LOC. Thus, data does not show that young animals are more susceptible to adverse effects from imazapyr exposure.

NP9E (Sources: FS WS ver. 2.02; USDA/FS 2003b)

The U.S. EPA has not derived an RfD for this surfactant active ingredient. A NOEL of 10 mg/kg bw/day for NP, however, was used by the USDA/FS to establish a chronic RfD, by dividing by an uncertainty factor of 100 to account for interspecies and intraspecies differences. Using an RfD based on NP is protective of both NP and the less toxic NP9E, and is specifically protective of estrogenic or reproductive effects. Acute exposures of NP9E are not anticipated to be associated with any adverse health effects at doses of 0.1 and 0.4 mg/kg bw/day. These RfD values are based on NP, but in reality, only a portion of NP9E would degrade into the more toxic NP compound.

Sulfometuron methyl (Sources: FS WSM v. 6.00.10; SERA 2004c)

Contrary to the approach taken in most Forests Service risk assessments, acute and chronic RfD values were not adopted from the U.S. EPA. No acute RfD has been established by the U.S. EPA for sulfometuron methyl. One developmental study using rats, however, established a NOAEL of 86.6 mg/kg bw/day based on observed decreases in maternal and fetal body weights after 10 days of gestational exposure. The Forest Service uses this study to establish a provisional acute RfD of 0.87 mg/kg/day that was calculated using the NOAEL of 86.6 mg/kg/day and an uncertainty factor of 100. Although the U.S. EPA uses a chronic RfD of 0.24 mg/kg/day, the more conservative provisional RfD of 0.02 mg/kg bw/day was derived by the Forest Service from a chronic feeding study using rats. This study had a NOAEL of 2 mg/kg bw/day as a result of hematological effects in male rats. An uncertainty factor of 100 was used, which represents two factors of 10 to account for species to species extrapolation and sensitive subgroups.

Triclopyr (Sources: FS WSM v. 6.00.10; SERA 2011d)

The U.S. EPA established acute and chronic RfDs for triclopyr, and separate RfD values for the metabolite 3,5,6-4 trichloro-2-pyridinol (TCP), which are used in USDA/FS risk assessments without adaptations. The RfD values for triclopyr are 1.0 and 0.05 mg/kg bw/day for acute and chronic exposure respectively. Each of these RfD values was derived from NOAEL findings from studies using rats. The UF used to calculate both RfD values was 100.

The acute RfD of 1 mg/kg bw/day was intended to be used for the general population. This RfD was established because marked maternal toxicity in rats was not seen until a dose of 300 mg/kg bw/day was administered, although fetal toxicity was observed with a dose of 100 mg/kg bw/day. However, the RfD of 1 mg/kg bw/day was not acceptable for human females of reproductive age (13 to 50 years) due to maternal toxicity being observed at 30 mg/kg bw/day with a NOAEL of 5 mg/kg bw/day for the developmental study. Thus,

the more conservative RfD of 0.05 mg/kg bw/day for both acute and chronic exposure is most appropriate for women in this age group (SERA 2011d, p. 72).

Triclopyr contains the metabolite/degradate 3,5,6-trichloro-2-pyridinol (TCP), which has the potential to be toxic, so this compound is quantitatively assessed. Acute and chronic RfDs, of 0.025 and 0.012 mg/kg bw/day respectively, were derived by the U.S. EPA and adopted by the Forest Service. Both RfDs were derived using a UF of 1000, because, as with triclopyr, there were uncertainties relating to species to species extrapolations and sensitive individuals. In addition to these a third factor was added to account for the potential for children having a higher sensitivity to TCP than adults.

The acute RfD originated from a developmental study of triclopyr resulting in a NOAEL of 25 mg/kg bw/day that was then divided by a UF of 1000 for TCP. This resulted in an RfD of 0.025 mg/kg bw/day. By contrast, the data that was used to establish the chronic RfD for TCP was derived from a chronic study on dogs. A NOAEL of 12 mg/kg/day resulted from this study as well as a LOAEL of 48 mg/kg/day. Once divided by 1000, as done for the acute RfD, the resultant RfD that remains for chronic exposure is 0.012 mg/TCP/kg bw/day.

1.3.1.5 Risk Characterization

In Forest Service risk assessments, the exposure and the dose-response assessments are used to quantitatively characterize risks. Hazard quotients (HQ) are values used to categorize risk for systemic toxicity effects (SERA 2012). All HQ values are directly proportional to the chemical application rate (i.e., an HQ value of 2 at an application rate of 1 lb a.e./acre would be 6 at an application rate of 3 lb a.e./acre). For acute exposures, HQs are in units of mg/kg bw/event whereas chronic exposures are in units of mg/kg bw/day. The HQ is usually calculated by dividing a projected level of exposure by an acceptable level of exposure, such as an RfD (*ibid*). Generally, an HQ greater than 1 indicates that risk is above the Level of Concern (LOC), or unacceptably high for the situation being considered, and that adverse health outcomes may be plausible. By contrast, an HQ less than or equal to 1 indicates that exposures are below the LOC and adverse effects are not expected. Still, when HQ values are 1 or greater, the plausibility of scenarios and assumptions made for each scenario should be considered before conclusions regarding risk levels are drawn. For example, the parameters set for the scenario relating consumption of contaminated water after a pond spill is designed to show varying consequences of spilling different amounts of the chemical under consideration (USDA/FS 2006a). The amounts of a chemical spilled are set at the amounts needed to treat from 1 to 100 acres. Such assumptions in this scenario are arbitrary and may be unrealistic. Given its arbitrary nature, this scenario can usually be used only to quantitatively assess risk to a limited extent.

When characterizing risk, it is important to consider the severity of the toxicological effects used to establish effect levels. Distinctions between adverse effect levels (AELs) and frank effect levels (FEL, defined as “gross and immediately observable signs of toxicity”) are

important. These levels are subject to misinterpretation, so judgments should be made with caution (SERA 2012). When no FELs are found, this implies that no overt effects are anticipated, though this does not mean that all HQs are acceptable or comparably acceptable. In some cases, hazard levels of exposure may be greatly exceeded and humans may be asymptomatic. This does not mean, however, that subclinical changes have not occurred that should justify rational people to minimize exposure to chemicals. It needs to be emphasized that for the risk characterizations that follow, regardless of studies and findings, "***[a]bsolute safety cannot be proven and the absence of risk can never be demonstrated***" (*ibid*). There are always uncertainties, such as those associated with using data from surrogate mammals to represent human health risk. Thus, individuals should remain prudent and minimize chemical exposure when possible.

Biologically sensitive individuals also need consideration as part of chemical risk characterization. Certain individuals have severe sensitivities when exposed to chemicals, often even when the chemical is below levels of concern (*ibid*). Individuals who are biologically sensitive to chemicals are those who are significantly more sensitive than the general population. Factors such as age (young or old), lifestyle and behavior, as well as the presence of genetic conditions or pre-existing disease states, may increase susceptibility to chemicals (*ibid*). Individuals who are at a high risk due to a high level of exposure, however, are not included in this group. There is also a condition referred to as multiple chemical sensitivities (MCS), which is where individuals report having multiple sensitivities to different types of chemicals, including pesticides (SERA 2011b). These individuals notice effects at very low doses relative to folks without MCS. To date, there is debate about whether this condition is psychosomatic, but regardless, the condition exists (*ibid*). This condition has been particularly noted in the case of glyphosate.

In the risk characterization section of each USDA/FS risk assessment, "connected actions" are also evaluated in terms of adverse effect risks. The Council on Environmental Quality (CEQ) defines connected actions as actions that are closely related, and they are connected if they:

> *(i) Automatically trigger other actions which may require environmental impact statements. (ii) Cannot or will not proceed unless other actions are taken previously or simultaneously. (iii) Are interdependent parts of a larger action and depend on the larger action for their justification (40 CFR 1508.5).*

In terms of USDA/FS risk assessments and pesticide use, connected actions most commonly refer to adverse effects associated with inert ingredients, metabolites, impurities, and synergism. As applicable, these actions are summarized below for each chemical being proposed for use.

In Forest Service risk assessments on specific chemicals, risk is characterized in terms of cumulative effects, when appropriate. The USDA/FS described the cumulative effects section of a chemical-specific risk assessment as considering "known chemical interactions or actions, which taken in consideration with the proposed pesticide use, would affect the quality of human health and the environment (i.e., modify risks to human health and ecological receptors within the context of the risk assessment)" (USDA/FS 2006a). Given the scope of the chemical risk assessment, the Forest Service makes no attempt to identify and consider all agents that could potentially interact with a specific chemical. When applicable, the USDA/FS and the risk assessment in this PEIR try to discuss interactions and associated effects in terms of the most current information.

Borax (Sources: FS WSM ver. 6.00.10; SERA 2006a)

Only some of the standard worker and public exposure scenarios usually used by the USDA/FS are applicable to the use of borax, as it is only applied directly as a dry substance to freshly cut stumps of trees. Of general and accidental worker exposure scenarios, only the ones that involve wearing contaminated gloves for a minute or one hour were applicable, and even at the upper application rate (5 lbs a.i./acre) none of the HQ values indicated that toxic effects were plausible. When considering scenarios pertaining to public exposure, the standard direct spray scenario was adapted to assess the hazards of a child consuming dry borax from a stump. The HQ values for this scenario indicated that adverse effects are plausible at typical and upper application rates. At the typical rate (1 lb a.i./acre) the central, lower and upper HQ values were 4.2, 2.1, and 16.2 respectively, whereas HQ values were 21.2, 10.6 and 80.9 at the upper application rate (5 lbs a.i./acre) for the direct consumption scenario. Per SERA 2006a, such "estimated levels of exposure are below levels of exposure associated with nonlethal effects such as diarrhea and vomiting…". Thus, if a child consumes borax from a stump, the child would likely experience vomiting and diarrhea as symptoms of toxicity. The only other applicable standard scenarios included acute and chronic consumption of borax contaminated water. Of these scenarios, HQ values are only above levels of concern for central and upper bounds at an application rate of 5 lbs a.i./acre for a child consuming water contaminated by borax shortly after a spill (HQ values = 1.2 and 3.6 respectively).

Certain precautions should be used when handling boron products. Borax is known to be an eye irritant (sometimes severe), and be absorbed more rapidly through damaged skin compared to intact skin. While no scenarios specifically evaluate these factors, borax usually only comes in contact with eyes and damaged skin when the chemical is mishandled. Individuals with large areas of damaged skin should avoid using boron products such as Sporax®. Moreover, prudence should be taken to ensure that proper pesticide application procedures be followed, such as wearing appropriate personal protection equipment, implementing sound hygiene practices and using proper pesticide handling procedures.

Other factors important to risk characterization of borax include sensitive subgroups, connected actions, and cumulative effects. Developing fetuses are a primary target of boron exposure. Since the RfD is based on the adverse fetal effect of weight loss, the reproduction related subgroups are accounted for throughout the entire Forest Service risk assessment. Testes are also targeted in male mammals and thus, while data is currently lacking, males with underlying testicular dysfunction may be at an increased risk of testicular issues induced by boron exposure. Connected action consideration is not a concern since borax is not mixed with other chemicals. In terms of cumulative effects, multiple exposures are not concerns given that the chronic RfD was used to calculate risk through the entire boron assessment. The concern is also lessened by the fact that boron is ubiquitous in nature. Exposures occur naturally at rates of 0.14 to 0.36 mg/kg/day and the Forest Service application rates do not substantially contribute to the already existent background levels.

Clopyralid (Sources: FS WSM ver. 6.00.07 & 6.00.10; SERA 2004a)

The application rate of clopyralid is restricted to a maximum of 0.25 lb a.e./acre in California, and this rate is used as the typical and central rate of application for evaluation in this PEIR, given that it is lower than the typical rate used by the USDA/FS.

Empirical evidence does not indicate that use of clopyralid poses unreasonable risk to workers and member of the public. At an application rate of 0.25 lb a.e./acre, none of the general or incidental exposures to workers lead to HQ values above the level of concern. Similarly, none of the short or long-term exposure scenarios relating to the public approach a level of concern based on central estimates. Only the upper bounds of scenarios involving a child drinking water after a spill, and chronic consumption of contaminated vegetation, resulted HQ values just over the level of concern (1.7 and 1.2 respectively). The exposure scenarios for the consumption of contaminated water and vegetation are arbitrary scenarios: scenarios that are more or less severe, all of which may be equally probable or improbable, easily could be constructed. Nonetheless, these acute scenarios help to identify the types of scenarios that are of greatest concern and may warrant the greatest steps to mitigate. For clopyralid, as with most other chemicals, spills of relatively large amounts into a small body of standing water and clopyralid applications on or near vegetation that might be collected for food would require remedial action to limit public exposure.

Though not assessed quantitatively, evidence suggests that dermal and ocular damage may occur when in direct contact with high levels of clopyralid acid, so precautions should be taken, such as wearing personal protection equipment to avoid direct contact while handling clopyralid.

Current evidence does not clearly indicate that there are subgroups sensitive to or connected actions affiliated with clopyralid exposure. In toxicity studies clopyralid has been implicated in

causing decreased body weight, increased kidney and liver weight, deceased red blood cell counts, as well as hyperplasia in gastric epithelial tissue. However, the likely critical effect in humans cannot be identified and effects are not consistent among test species or even between different studies on the same species Thus, it is unclear if individuals with pre-existing kidney, liver, or blood diseases would be particularly sensitive to clopyralid exposures (SERA 2004a, p. 3-35). Regarding potential connected actions, although clopyralid may be applied in combination with 2,4-D or other herbicides, “there are no data in the literature suggesting that clopyralid will interact, either synergistically or antagonistically with these or other compounds” (SERA 2004a, p. 3-36).

Using the assumptions and methods typically applied in Forest Service risk assessments, there is no plausible basis for asserting that the contamination of clopyralid with hexachlorobenzene or pentachlorobenzene will result in any substantial risk of cancer in workers applying clopyralid under normal circumstances. Per the clopyralid risk assessment, the Forest Service has adopted a cancer risk level of one in one-million (1÷1,000,000) as a trigger that would require special steps to mitigate exposure or restrict and possibly eliminate use. In the case of hexachlorobenzene that contaminates clopyralid, the highest risk level is at about 3 in 100,000,000. The scenario that leads to this highest estimate involved a subsistence population consuming contaminated fish. This was the primary scenario for exposure to hexachlorobenzene because of the tendency for the chemical to bioconcentrate from water into fish tissue. The prolonged use of clopyralid at the highest plausible application rate, 0.25 lb. a.e./acre, could approach a level of concern in areas with small ponds or lakes used for fishing and in areas with local conditions that favor runoff. In such cases, site-specific exposure assessments and/or monitoring of hexachlorobenzene concentrations in water could be considered.

Glyphosate (Sources: FS WSM v. 6.00.10; SERA 2011b)

When using the HQ approach to assessing risk from exposure to glyphosate, values indicate that concern for workers is minimal. The highest HQ for worker exposure is the upper bound for general broadcast spraying (HQ of 0.2 at typical application rate of 2 lb a.e./acre). Similarly, at the highest rate of application used by the USDA/FS of 8 lbs a.e./acre, the highest upper bound associated with workers participating in broadcast foliar application (HQ of 0.6).

In terms of public exposure, only two of the public exposure scenarios indicate the potential for adverse effects related to glyphosate exposure (HQ values greater than 1). The accidental acute exposure scenario involving contaminated water after a spill, for example, has an upper bound HQ of 2.1 at the typical application rate (2 lbs a.e./acre), and 8.1 at the upper application rate (8 lbs a.e./acre). The only non-accidental exposure of potential concern involves consumption of contaminated vegetation shortly after application, with an upper bound HQ of 1.4 for the typical application rate (2 lb a.e./acre) and 5.4 at the upper application rate (8 lb a.e./acre). An HQ of 5 may raise concerns regarding adverse health effects to

pregnant women and fetotoxicity. Chronic exposure scenarios never resulted in levels of concern, even when the maximum application of 8 lbs a.e./acre was used, as 0.9 was the highest HQ, which was for the chronic scenario involving consumption of contaminated vegetation. South American formulations that contain surfactants have been associated with genotoxicity, though it is currently unclear if this finding is applicable to the U.S. formulations.

There are some glyphosate specific issues, such as sensitive subgroups, connected actions, and cumulative effects, which can only be qualitatively discussed. Sensitive subgroups include women and fetuses, but these are accounted for since a developmental study was used to establish the NOAEL and subsequent RfD. While not well understood, MCS may be a potential concern for glyphosate, as with other chemicals. For glyphosate use, the most important connected action is associated with surfactants. Given that glyphosate functions to inhibit some mixed-function oxidases, this is a plausible mechanism of interaction for other chemicals that function similarly. There has been no evidence of such effects, however, and this is only likely to be a potential when glyphosate is applied at much higher rates than done by the Forest Service or likely under the VTP and alternatives. Individuals may be exposed to glyphosate applied by the USDA/FS though several routes (e.g. contaminated water and fruit), though this is thought to be inconsequential, particularly since the consumption of contaminated vegetation is the only substantial exposure scenario. The Food Quality Protection Act requires chemicals that have the same mode of action relating to toxicity be assessed, but currently the U.S. EPA has not determined if glyphosate shares toxicity mechanisms with other chemicals.

Some glyphosate formulations may pose the risk of skin and eye irritation. As stated in SERA 2011b, the original Roundup formulation is about as irritating to the skin as standard dish washing detergents, all-purpose cleaners, and baby shampoos. This risk characterization, however, may not be applicable to all formulations of glyphosate that contain a surfactant. Some surfactant containing formulations of glyphosate appear to be greater irritants to the skin and eyes compared with other nominally similar formulations. Because formulations may change over time, care should be taken to read and understand the MSDS for any formulation of glyphosate which may contain a surfactant.

Hexazinone (Sources: FS WSM v. 6.00.10; SERA 2005)

Risks to workers are the dominant element in the risk characterization for potential effects in humans. The highest HQ associated with accidental/incidental exposure of worker is well below the LOC (HQ values ≤ 0.2) for all scenarios at the upper application rate of 4 lbs/acre, regardless of application method. The upper bounds of general exposure for workers is above the LOC at a typical rate of 2 lbs/acre, regardless of whether liquid and granular formulations of hexazinone are applied by broadcast (HQ of 6) or directed foliar (HQ of 3) methods. Since HQ values are proportional to the application rate, HQ values double when considered at the upper application rate of 4 lbs/acre. It should be noted, however, the lower bounds of hazard

quotients for general worker exposure do not exceed a level of concern at typical or upper application rates. The simple interpretation of these hazard quotients is that worker exposures to hexazinone during application are likely to exceed exposures that would generally be regarded as acceptable unless workers follow prudent handling practices that minimize exposure.

In addition to hazards associated with systemic toxicity, hexazinone can cause eye irritation. Quantitative risk assessments for irritation are not derived; however, from a practical perspective, eye irritation is probably the overt effect that is most likely to be observed as a consequence of mishandling hexazinone. This effect can be minimized or avoided by using sound industrial hygiene practices during handling of the chemical.

For the public, few of the scenarios led to HQ values above the LOC. One such scenario of acute accidental exposure involves consumption of contaminated water after a spill into a small pond, which results in an upper bound HQ of 2, for the highest application rate (4 lbs a.e./acre). While no acute non-accidental scenarios resulted in HQ values that substantially exceed the level of concern at the upper application rate, the highest value is associated with consumption of contaminated vegetation (i.e., upper bound HQ of 1.4 for liquid formulations). Chronic scenarios with the highest upper HQ values are those associated with consumption of contaminated vegetation (HQ of 45 at 4 lbs/acre) and fruit (HQ of 6 at 4 lbs/acre) after the application of liquid formulations. Remaining chronic scenarios, other than those relating to vegetation and fruit consumption, resulted in upper bound HQ values ≤ 0.2 for liquid formulations at the upper application rate. The risk of exposure is much lower for granular formulations of hexazinone. Upper HQ values, for example, associated with consumption of contaminated broadleaf vegetation and fruit are 1.8 and 0.3 respectively for granular formulations.

As discussed in SERA 2005, the chronic RfD is based on a NOAEL of 5 mg/kg/day. The corresponding LOAEL was about 40 mg/kg/day based on minor body weight changes and changes in blood chemistry indicative of liver toxicity. This LOAEL is a factor of 8 above the NOAEL. At the highest dose tested, about 160 mg/kg/day and a factor of 32 above the NOAEL, effects included decreased body weight gain, more pronounced changes in blood chemistry indicative of liver damage, and some changes in the liver. The relationship of the experimental NOAEL to the LOAEL or higher doses cannot be used as a direct measure of plausible effects in humans at doses above the chronic RfD. Nonetheless, the hazard quotient of 6 at the lowest application rate (0.5 lb a.i./acre) is a concern. The hazard quotient of 23 at the application rate of 2 lbs a.i./acre and the hazard quotient of 45 at an application rate of 4 lbs a.i./acre are clearly a serious concern. Given that granular application methods result in less residue on plants, particularly on the leaves of broadleaf vegetation and other plant parts that might collect similar levels of residue, this method should be favored over liquid hexazinone applications where public consumption of contaminated vegetation is probable.

Other factors that should be considered include sensitive subgroups, connected actions and cumulative effects. Hexazinone can induce fetal resorptions and other adverse developmental effects, so pregnant women and developing offspring may be sensitive subgroups particularly vulnerable to adverse effects of hexazinone. This potential has been explicitly accounted for given that the developmental endpoint was used in the risk assessment. The literature does not report any other subgroups that may be sensitive to hexazinone and there is no indication that it causes allergic responses or sensitization. In terms of connected actions, while there is almost no information available on the interaction of hexazinone with other compounds, there is no indication that the inerts and adjuvants in its formulations will increase the toxicity of hexazinone in humans or mammals. It is not unreasonable, however, to suspect hexazinone would interact additively, synergistically or antagonistically with chemicals that share similar metabolic pathways. Such potential connected actions are beyond the scope of the risk assessment in this PEIR and are not evaluated by the Forest Service or the U.S. EPA. Cumulative effects may result from repeated exposures, multiple routes of exposure (i.e., oral and dermal), or exposures to chemicals that have connected modes of action. Forest Service risk assessments consider the effects of multiple, long-term exposures, evaluating risk in terms of both acute and chronic exposures to workers and the public.

Imazapyr (Sources: FS WSM v. 6.00.10; SERA 2011c)

No hazards have been identified for this chemical other than the potential for eye irritation. None of the scenarios result in an HQ that exceeds 1 when calculated at an application rate of 1 lb a.e./acre. When using the maximum application rate of 1.5 lb a.e./acre the only exposure scenario that exceeded an HQ of 1 was from the upper exposure limit on drinking water from a pond immediately after a spill (HQ 1.2). Given the lack of adverse effects detected, HQ values that do exceed 1 are difficult to interpret. Currently, no evidence suggests that systemic effects are likely to occur among workers and the public because of imazapyr exposure. Eye irritation is the only clear risk to humans and is most pertinent to workers. Injury to the eye is most likely to occur with occupational mishandling of imazapyr, and thus workers would be prudent to follow personal protection measures, such as wearing goggles.

Given the low toxicity of imazapyr, effects on sensitive subpopulations, the occurrence of connected actions, and cumulative effects are thought to be minimal. Because imazapyr is a weak acid it would most likely be affected by other weak acids that are similarly excreted by the kidneys, though only at unrealistically high doses that nearly saturate kidneys. In terms of connected actions, both the low HQ values and conservative assumptions support that impacts of inerts, impurities and metabolites are minimal to imazapyr risk characterization. Potential adjuvant interactions, however, are a potential but were beyond the scope of the USDA/FS risk assessment for imazapyr (as with other chemicals). When characterizing risk of chemical use, cumulative effects may result if humans experience multiple exposures to imazapyr via multiple routes and/or events, or if humans are exposed to additional chemicals with the same toxicity mechanisms at the same time as exposure to imazapyr. At present,

common mechanisms of toxicity have not been found between imazapyr and any other chemicals (similar or otherwise). Given this, the USDA/FS found no evidence to suspect cumulative effects should occur with the use of imazapyr, particularly in lieu of the low chemical toxicity to humans.

NP9E (Sources: FS WS ver. 2.02; USDA/FS 2003b)

No evidence indicates that typical acute and chronic exposures for workers would lead to doses that exceed the level of concern, though some of the upper bounds did exceed it. Accidental exposure is not anticipated to cause adverse health effects, with the highest HQ of 0.7 from wearing a contaminated glove for one hour. The upper bounds of general worker exposure resulted in levels above concern, with the level of concern being double for broadcast application (HQ of 10) than directed ground spray (HQ of 5). Despite the high levels of concern at the upper bounds, there is not a high likelihood that workers will use such high levels (the upper application rate of 6.68 lb a.i./acre or 40 gallons per acre of a 2.5% solution) of surfactants containing NP9E on a chronic basis. Additionally, workers are expected to use industrial hygiene practices while handling chemicals, which are not accounted for in worker exposures.

For members of the public, chronic exposure leads to levels below concern, though some accidental exposure scenarios lead to exposures of concern. Per the USDA/FS risk assessment, there should not be any substantial risk of long-term exposure to NP9E-based surfactants to the public. Only the scenarios for consumption of contaminated water (spill or ambient/drift) and/or fish (the latter for subsistence populations), as well as contaminated fruit exposures lead to acute or accidental exposures with unacceptable risk. The scenario relating to consumption of water by a child after a spill leads to the highest risk at typical, lower and upper exposures levels (HQ values of 5, 1.4 and 17 respectively). Beyond water consumption after a spill, only the upper bounds of other scenarios were above the level of concern. As discussed in USDA/FS 2003b, an HQ of 5 represents a risk of subclinical effects to the liver and kidney. The upper HQ of 17 represents an increasing risk of clinical effects to the kidney, liver, and other organ systems. These finding indicate that oral, rather than dermal, exposures are of the greatest concern for NP9E, and help determine where the greatest mitigations may be necessary to minimize exposures to the public.

NP9E exposures directly to the eye may lead to irritation and damage when at relatively high levels, and undiluted NP9E may lead to skin sensitization. Such exposures, however, are only likely to occur in cases where the chemical is mishandled, and thus the use of personal protective equipment and industrial hygiene procedures are imperative.

There are several groups of people that have the potential to be part of sensitive subgroups. There is some indication that some sensitive individuals are prone to develop contact allergies related to NP9E exposures. In addition, there is evidence that NP9E targets the kidneys and

liver in mammals, so sensitive subgroups may consist of those individuals that have pre-existing impairment of the liver or kidneys. Per the Forest Service risk assessment, the likelihood of NP9E inducing reproductive effects should be low, though acute exposures may occur that are within the range where fetal effects may occur, therefore pregnant women could be considered a sensitive subgroup.

Potential connected actions and cumulative effects of NP9E are important to consider. NP9E has not been connected to any antagonistic or synergistic interactions relating to human health effects when mixed with other chemicals. This group of surfactants is not known to increase dermal absorption of herbicides and synergistic effects are not expected with repeated exposures of NP related compounds. Toxicological response appears to be dependent on daily doses rather than the duration of exposures. Additionally, any repeated-exposure effects should have been counted for through use of the chronic RfD. That said, there is the potential for additive estrogenic effects to arise if NP related compounds or chemicals that act via similar estrogen-like (xenoestrogen) pathways cumulatively reach a high enough concentration. NP9E exposure may result from a number of non-forestry related sources (e.g. personal care products, industrial and institutional detergents and cleaners, and the environment), and the amount of human exposure to NP9E as a result of forestry use may be negligible in comparison.

Sulfometuron methyl (Sources: FS WSM v. 6.00.10; SERA 2004c)

At the typical application rate used by the Forest Service (0.045 lb a.e./acre), none of the upper limit HQ values for workers or the public are at or above levels of concern. The highest general worker exposure is the upper bound for broadcast application, with an HQ of 0.34 for at the typical application rate. At the higher application rate of 0.38 lb a.i./acre, however, the upper bounds for both broadcast and direct foliar application are above the level of concern (HQ values of 2.9 and 1.5 respectively). None of the scenarios for the public resulted in levels of concern at the typical application rate (0.045 lb a.i./acre) At the highest application rate, however, the upper bounds for the scenario involving chronic consumption of contaminated broadleaf vegetation indicated that adverse effects are plausible (HQ of 4.1).

The interpretation by the Forest Service is that an unacceptable level of risk could be expected for workers if the maximum application rates are used, the maximum acreage is treated per day, and the workers are not prudent in using sound hygiene practices and personal protection equipment. Given the low likelihood that all these factors would occur, and the conservative provisionary RfDs used by the Forest Service, it is unlikely that workers or the public alike would experience observable adverse effects. Proper chemical handling and hygiene practices should minimize potential irritation or damage to eyes and skin. Similarly, the risk of adverse effects to the public would be reduced or eliminated if lower application rates and fewer acres were treated.

No adverse effects associated sensitive subgroups, connected actions, or repeated exposures, were identified in the 2004 risk assessment for sulfometuron methyl conducted for the Forest Service. Given hematology and thyroid effects observed in mammalian studies, it was suggested that individuals with pre-existing anemia or thyroid function issues may be more susceptible to adverse effects. Per the Forest Service risk assessment, sulfometuron methyl formulations have not been connected to synergistic or antagonistic effects related to the mixing of sulfometuron methyl with other active ingredients and surfactants. Cumulative effects are not anticipated given that repeated exposures were explicitly considered through using a chronic RfD to evaluate the level of concern with repeated exposure.

Triclopyr (Sources: FS WSM v. 6.00.10; SERA 2011d)

The acute RfD for general worker exposures is 1) less conservative than using the chronic RfD, 2) only applicable to sporadic applications of triclopyr, and is 2) only applicable to men, so these results will not be summarized here (see SERA 2011d for acute details). Overall, triclopyr TEA had a higher HQ values than BEE for ground application methods. Based on the chronic RfD of 1.0 mg/kg bw, central HQ values for workers applying the typical application arte of 1 lb. a.e./acre are below the level of concern for both triclopyr TEA and triclopyr BEE, for all ground application methods. The upper bound HQ values for all ground application methods at this rate, however, were above the level of concern for both TEA and BEE forms of triclopyr. When considering these upper bounds, HQ values of TEA range from 1.6 to 3, and BEE values 6 to 12 with the typical application rate (1 lb/acre). At the expected upper application rate (6.6 lbs/acre), upper HQ values for all ground application methods range from 11 to 20 for TEA; whereas equivalent values range from 41 to 78 for BEE.

Whether the HQ values exceed for public exposure scenarios depends on if the acute or chronic RfD is used, the application rate and the form of triclopyr being evaluated. The chronic RfD used for females (0.05 mg a.e./kg bw/day) results in HQ values 20 times higher than those for males calculated using the acute RfD value (1 mg a.e./kg bw/day). When based on the acute RfD of 1 mg/kg/day, accidental exposures of workers to formulations containing triclopyr TEA do not lead to HQs that exceed a level of concern. When using the chronic RfD of 0.05 mg/kg bw for women, none of the HQs for accidental scenarios for triclopyr TEA formulations exceed a level of concern at an application rate of 1 lb a.e./acre either. The highest HQ at 1 lb a.e./acre is 0.02 for male and 0.3 for female workers, which is associated with wearing contaminated gloves for 1 hour.

When the maximum application rate of 6.6 lbs a.e./acre is used, none of the accidental HQs reach a level of concern for male workers. The accidental scenarios for wearing contaminated gloves for 1 hour as well as 1-hour exposures resulting from spills onto the lower legs reach upper bound HQs of 0.1 for both scenarios, using the acute RfD of 1 mg a.e./kg bw/day. Using the RfD of 0.05 mg/kg bw/day for female workers results in an HQ of about 3 for both scenarios. For triclopyr BEE, the accidental exposure from wearing a contaminated glove for

an hour results levels above concern when considered for male workers (acute RfD of 1 mg a.e./kg bw/day), with an upper HQ of about 8 at the typical rate, and an upper HQ of 50 at the 6.6 lbs a.e./acre. Based on triclopyr toxicology, HQs that approach or exceed a factor of 5 could be regarded as clearly unacceptable and possibly hazardous. The development of subclinical adverse effects cannot be ruled out.

Beyond quantitative levels of concern, one of the most likely exposures and risks for workers is from chemicals being splashed into eyes, as the chemical is moderately to severely damaging. This is an avoidable hazard, as long as workers wear eye protection while handling triclopyr.

Risks to the public associated with terrestrial applications of triclopyr TEA and triclopyr BEE are identical for many exposure scenarios. For exposure scenarios involving dermal absorption, the risks associated with triclopyr BEE formulations are only modestly greater than those for triclopyr TEA formulations. The only exposure scenarios of substantial concern involve the consumption of contaminated vegetation, and these risks do not differ between TEA and BEE formulations of triclopyr. Scenarios of concern involving exposures to TCP are also limited to the consumption of contaminated vegetation. The upper bound of the acute exposure scenario for the consumption of contaminated vegetation by a young woman is 27, exceeding the corresponding upper bounds for general exposures in workers applying triclopyr BEE based on the chronic RfD - i.e., HQs of 11 to 22.

Potential exposures to the TCP metabolite of triclopyr also exceed the level of concern at the upper bound of the HQs for both the acute and longer-term consumption of contaminated vegetation and fruit. For TCP, the upper bound of HQs for acute exposures is less than the upper bound of the HQs for longer-term exposures. For the central estimates and the lower bounds, the opposite pattern is apparent. While this may seem incongruous, the calculations are correct and reflect the interplay of the lower chronic RfD and the different half-lives used to estimate the longer-term time-weighted average doses.

The qualitative interpretation of the HQs for TCP is similar to that of the HQs for triclopyr. For TCP, the LOAEL associated with the acute RfD is a factor of 4 higher than the NOAEL on which the RfD is based. As with the discussion of the reproductive NOAELs and LOAELs for triclopyr, this ratio does not indicate that adverse reproductive effects would be predicted in humans at an acute HQ of 4; however, the relationship of the NOAELs to LOAELs in the animal studies does enhance concern for HQs in the range of 4. For TCP, the upper bound acute HQs range from 2 to 15.

As discussed above, exposure to triclopyr has resulted in adverse developmental effects in female mammals, which leads to concerns regarding reproduction and development in female humans. Such effects were only found with doses that also caused frank maternal toxicity in

mammals. Concern is lessened because evidence of frank maternal toxicity or reproductive effects in humans was not found associated with the use of triclopyr.

The primary sensitive subgroups thought to be most susceptible to adverse effects from exposure to triclopyr include women of childbearing age and individuals with kidney disease. Women of child bearing age are thought to be of concern due to reproductive and developmental effects found in exposure studies using mammals. Despite the lack of epidemiological evidence, there is a certain level of uncertainty, regarding the possibility of triclopyr causing adverse reproductive effects. One Forest Service study demonstrated a marginal relationship between herbicide use and miscarriages in woman, which creates a level of uncertainty even though triclopyr was not specifically named as one of the herbicides. Current evidence suggests, however, that toxicity to a fetus would only occur at doses that also caused frank signs of maternal toxicity. Despite the years triclopyr has been used, this chemical has never been implicated in causing frank signals of toxicity in male or female humans. Regardless, the current Forest Service risk assessment interprets findings to mean that some woman may be exposed to triclopyr at levels that are of concern. Individuals with kidney disease may also be at greater susceptibility to adverse effects, since the kidneys are the target organ for triclopyr. Despite this concern, however, no evidence associates adverse effects towards people with kidney disease from exposure to triclopyr.

Connected actions of triclopyr are associated with exposure to the triclopyr metabolite 3,5,6-trichloro-2-pyridinol (TCP). Exposure to TCP is quantitatively considered throughout the human and ecological health sections of the Forest Service risk assessment. The U.S. EPA assessments consider all exposures to this compound as below the level of concern, although the Agency does not consider all oral exposures assessed in the Forest Service risk assessments, as discussed previously. Like many herbicides, adjuvants are commonly used with triclopyr and some may be hazardous, however, evaluation of each surfactant is beyond the scope of Forest Service risk assessments.

The cumulative effects associated with triclopyr may include those associated with any additive effects that could potentially result from mixing of triclopyr with other chemicals, as well as effects resulting from repeated exposures. The additive effects associated with mixing adjuvants with triclopyr are beyond the scope of the USDA/FS risk assessments. It should be noted, however, that triclopyr is a weak-acid auxin herbicide, and thus, when mixed with other similar weak acids that function by the same mechanisms, such as clopyralid, additive risks would result. Repeated exposure is a cumulative effect accounted for by the use of chronic exposure information in each Forest Service risk assessment.

1.3.2 ECOLOGICAL EFFECTS

1.3.2.1 Introduction

This ecological effects analysis mirrors the protocol used in SERA Risk Assessments (RAs) (SERA 1996a & b, 1997a & b, 1998a & b, 1999, 2003a & b, 2004a, b, & c, 2005, 2006i, 2010c, 2011d, e, & f) and is adapted primarily from those RAs. Information from SERA RAs is supplemented by other sources, including a U.S. Forest Service RA for NPE (USDA/FS 2003b) and for 2,4-D (USDA/FS 2006a), U.S. EPA RAs for NPE (U.S. EPA. 2010e & f), the U.S. EPA risk assessments for the California red-legged frog (U.S. EPA 2007b, 2008b & c, 2009d & e), the Alameda whipsnake (U.S. EPA 2009d), and endangered and threatened salmon and steelhead (U.S. EPA 2004).

As discussed above, the chemical active ingredients and formulations and surfactants likely to be used in the VTP and alternatives and the parameters under which they will be used are well within the USFS programs for which the RAs were developed. To reiterate, chemicals will not be applied directly to water or riparian areas under the VTP and alternatives and they will not be applied aerially.

As in the human health assessment, the SERA RAs assess ecological effects in four parts, as follow:

1. First, the hazards of specific chemical active ingredient formulations to terrestrial organisms (mammals, birds, invertebrates, microorganisms, and plants) and aquatic organisms (fish, amphibians and reptiles, invertebrates, and plants) are identified. Hazards are based on toxicities to surrogate species tested under controlled conditions. Testing on certain species groups, notably amphibians and reptiles, is generally inadequate or non-existent. For these species groups, tests are done on surrogate species, namely freshwater fish as a substitute for amphibians and birds as a substitute for reptiles.
2. Next, the potential for exposure to chemicals by terrestrial organisms (from direct spray, indirect contact, ingestion of contaminated vegetation or prey, and ingestion of contaminated water) and by aquatic organisms (from direct spray, off-site drift, runoff, contaminated irrigation water, and wind erosion) are assessed.
3. Then the effects (responses) on terrestrial and aquatic organisms (those tested for hazard identification) from potential doses of chemicals are assessed.
4. Finally, the risk of adverse effects is determined for the terrestrial and aquatic organisms tested for hazard identification.

For an in-depth discussion of how Syracuse Environmental Research Associates, Inc. (SERA) conducts ecological risk assessments, refer to “Preparation of Environmental Documentation and Risk Assessments for the USDA/Forest Service” (SERA 2012). The exposure assessments for ecological effects are conceptually like those conducted in the

human health risk assessment, and for many terrestrial organisms the exposure assessments are parallel to those used in the human health risk assessment. Similarly, exposures of aquatic species are typically based on the same estimates of concentrations of the chemical in water that are used in the human health risk assessment.

The following from the SERA RA for hexazinone (SERA 2005, p. xviii) illustrates the uncertainty of ecological risk assessments in general, including the one in this PEIR:

> *As with most ecological risk assessments, the characterization of risk for hexazinone is limited by the comparison of the available data to the number of species that might be exposed and the interactions that could occur among these species. Hexazinone has been tested in only a limited number of species and under conditions that may not well-represent natural populations of nontarget organisms. This leads to uncertainties that may result in underestimates or overestimates of risk. The methods and assumptions used in both the exposure and dose-response assessments are intended to consider these uncertainties by using protective assumptions in developing both the exposure and dose-response assessments which form the basis of the risk characterization.*

As is true for the human health risk assessment, it needs to be reiterated that absence of risk can never be demonstrated and absolute safety cannot be proven. Available data does not, however, indicate that significant adverse effects to populations of terrestrial and most aquatic sentient organisms are likely from most of the chemicals potentially used under the VTP and alternatives.

1.3.2.2 Hazard (Toxicity) Identification

1.3.2.2.1 Introduction

As in the human health risk assessment, the results of various types of acute toxicity bioassays may be used to classify chemicals into various levels of toxicity, namely highly toxic to virtually nontoxic. As with the corresponding classification scheme for human health effects, acute toxicity is only used in the hazard identification to categorize chemicals and is not directly used in the risk characterization. To support pesticide registration, longer-term studies in most organisms are also required, typically for the active ingredient but not for chemical formulations.

1.3.2.2.2 Terrestrial Organisms

Toxicity data for terrestrial species from the most recent SERA RAs (SERA 2003a & b, 2004a, b, & c, 2005, 2006a, 2011b, c, d), U.S. Forest Service RA for NPE (USDA/FS 2003b) and for 2,4-D (USDA/FS 2006a), U.S. EPA RAs for NPE (U.S. EPA. 2010e & f), and/or U.S. EPA risk assessments for the Alameda whipsnake (U.S. EPA 2009d) is summarized in Table D.3-17.

Detailed toxicity data for each terrestrial species group is included below in *Chemical-Specific Hazard (Toxicity) Identification* for each chemical analyzed in this PEIR.

Mammals - As stated in the "Hazard Identification Overview" in SERA 2012, p. 76):

> *The hazard identification for wildlife mammals is usually based on the same information considered in the human health risk assessment, and this information is typically much more detailed than the information available on other groups because studies are often available on both lethal and sublethal effects. Data on the other groups is typically much less detailed. While information on sublethal effects is often available for some groups, much of the information consists of acute bioassays for lethality. This reflects a major conceptual difference between human health and ecological risk assessment. Human health risk assessment focuses on preventing the occurrence of any effect in any individual. Ecological risk assessment tends to focus on preventing adverse effects at the population level.*

Many of the pesticides used by the Forest Service, particularly the herbicides, are weak acids. Weak acids are often removed from the blood by the kidney, with eventual secretion in the urine. Part of this process involves active transport from the blood into kidney cells. This active transport process in dogs is much less active than the active transport process in primates and other mammals. Consequently, dogs are less able to eliminate weak acids and may be substantially more sensitive to weak acids than other mammals. Thus, in risk assessments on weak acids, any available information on the pharmacokinetics or toxicity of the compound in dogs relative to other mammalian species will be emphasized. If dogs appear to be more sensitive than other mammals, this may be considered further in the dose-response assessment and separate NOAEL or NOEC values may be derived for dogs and other canids. These values may then be used to characterize risks for other canid species that may be covered in the risk assessment – e.g., the consumption of a small mammal by a predator such as a coyote or wolf.

Birds - Information on the toxicity of pesticides to birds is typically much more limited than that for mammals. While some toxicity studies on birds may be available in the open literature, most of the information is usually from studies required specifically by the U.S. EPA for the registration of pesticides.

The acute studies, both oral and dietary, most commonly involve tests on mallard ducks and northern bobwhite quail. The acute oral study involves administration of a single dose and is observed for 14 days, although this period can be extended to 21days if mortality is seen. As with the mammalian oral study a limit test may be conducted at a single dose of 2,000 mg/kg. If no mortality occurs, the LD_{50} value may be expressed as >2,000 mg/kg and no additional testing is required. As with the mammalian studies, the risk assessment will distinguish this

type of information from studies in which some, but less than 50%, mortality occurred at the maximum dose.

The avian acute dietary toxicity study is similar to the acute oral study in general design and test species. Occasionally, however, other species may be used such as pigeon, Japanese quail, ring-necked pheasant, and red-legged partridge. The chemical is administered in the standard diet for a period of 5-days, and is sometimes referred to as a 5-day dietary or 8-day dietary study, which can lead to some confusion if the duration of exposure is not clearly distinguished from the duration of observation. As with the acute oral study, the duration of observation can be increased up to 21 days if signs of toxicity are noted during the standard 3-day post-exposure observation period. Either the acute oral study or acute dietary study will often serve as the basis for an acute NOAEL or NOEC that is used in the dose-response assessment for birds.

Chronic studies in birds analogous to those conducted in mammals – i.e., studies that span a full or significant fraction of the life span of the animal – are almost never available. Typically, the consequences of longer-term exposure scenarios for birds are evaluated using the avian reproductive toxicity study. These studies are generally conducted on mallard ducks or bobwhite quail. After egg laying begins, the study is continued for an additional 8 to 10 weeks. During all three periods, dietary exposure is maintained and thus the total period of exposure is 16to 21 weeks.

Reptiles and Amphibians (Terrestrial-Phase) - Data on terrestrial phase amphibians and reptiles are typically sparse to non-existent. When data are available, the studies are assessed in a manner similar to that used for mammals and birds. Typically, avian toxicity studies are substituted for those on amphibians. As stated in the U.S. EPA “Risks of 2,4-D Use to the Federally Threatened California Red-legged Frog (*Rana aurora draytonii*) and Alameda Whipsnake (*Masticophis lateralis euryxanthus*)” (U.S. EPA 2009d, p. 110): “[a]*s specified in the Overview Document, the Agency uses birds as a surrogate for reptiles and terrestrial-phase amphibians when toxicity data for each specific taxon are not available (U.S. EPA, 2004).*”

Terrestrial Invertebrates - There is substantial variability in the types of information that are available on terrestrial invertebrates. The U.S. EPA assumes that herbicides are generally not directly toxic to insects, so only requires relatively simple and standard bioassays: the honeybee acute contact toxicity, the honeybee toxicity of residues on foliage, and the earthworm subchronic toxicity test. Earthworms and honeybees comprise only a very small fraction of the terrestrial invertebrates. The acute contact toxicity study in honeybees is often the only kind of invertebrate toxicity study available on herbicides. This acute study is similar in design to acute toxicity studies conducted on mammals and birds, but involves direct application.

The earthworm toxicity test (OPPTS 850.6200) involves exposing a species of earthworm, typically *Eiseniafetida*, to various concentrations of the test compound in soil for a period of 28-days. The use of limit tests is not discussed in the OPPTS protocol. Range-finding studies are conducted as 0.1, 1.0, 10, 100, 1,000 mg/kg dry weight artificial soil.

Terrestrial Plants (Macrophytes) - The testing requirements for the effects of herbicides on terrestrial plants are relatively rigorous, since terrestrial vegetation is the usual target of herbicides. Studies on seedling emergence and vegetative vigor are the two basic types of bioassays that are covered and used in Forest Service risk assessments. Seedling emergence studies typically involve soil exposure and vegetative vigor studies typically involve direct spray. The former is used to characterize risk associated with soil contamination by runoff, and the latter are used to characterize risks associated with direct spray or spray drift.

Terrestrial Microorganisms - Studies on terrestrial microorganisms are not required for pesticide registration in the United States. Nevertheless, assays on microbial toxicity submitted directly to U.S. EPA for registration involve soil exposures, as these are directly relevant to the risk assessment. Many microbial toxicity studies in the open literature involve pure cultures of microorganisms in artificial media, such as agar or liquid culture. These types of assays are less directly relevant and are clearly distinguished from soil assays in the risk assessment.

Table D.3-17

Terrestrial Wildlife Acute Toxicity Summary

Herbicide	Sources	Mammals	Birds	Invertebrates
Boric Acid	SERA 2006a	moderately toxic	practically nontoxic	practically nontoxic
Borax (STD)	SERA 2006a	moderately toxic	practically nontoxic	practically nontoxic
Clopyralid	SERA 2004a; U.S. U.S. EPA 2009b	relatively nontoxic	slightly toxic	particularly nontoxic
Glyphosate	SERA 2011b	slightly toxic	practically nontoxic	practically nontoxic
Diammonium Salt	SERA 2011b	slightly toxic	practically nontoxic	practically nontoxic
Isoproplyamine Salt	SERA 2011b	slightly toxic	practically nontoxic	practically nontoxic
Potassium Salt	SERA 2011b	slightly toxic	practically nontoxic	practically nontoxic
Hexazinone	SERA 2005	slightly toxic	practically nontoxic	slightly to practically nontoxic (bees)
Imazapyr	SERA 2011c.	slightly toxic	slightly toxic	practically nontoxic (bees)
Sulfometuron-Methyl	SERA 2004c, U.S. EPA 2008a	slightly to practically nontoxic	slightly to practically nontoxic	practically nontoxic (bees)
Triclopyr Acid	SERA 2011d	slightly toxic	slightly to practically nontoxic	practically nontoxic (bees)
BEE	SERA 2011d	slightly toxic	practically nontoxic	practically nontoxic (bees)
TEA	SERA 2011d	slightly toxic	practically nontoxic	practically nontoxic (bees)

NP9E	USDA/FS 2003b; U.S. EPA 2010e & f	slightly to practically nontoxic	NA	NA

[1/] Toxicity ranges (from the most recent SERA, USDA Forest Service, and U.S. EPA RAs) are due to variable toxicities to different species in the same class. NA = no published data and/or no reliable data

1.3.2.2.3 Aquatic Organisms

Acute toxicity data for aquatic species from the most recent SERA RAs (SERA 2003a & b, 2004a, b, & c, 2005, 2006a, 2011b, c, d), U.S. Forest Service RA for NPE (USDA/FS 2003b) and for U.S. EPA RAs for NPE (U.S. EPA. 2010 e & f), and/or U.S. EPA risk assessments for the California red-legged frog (U.S. EPA 2007, 2008b & c, 2009d & e) and endangered and threatened salmon and steelhead (U.S. EPA 2004) is summarized in Table D.3-18. Detailed toxicity data for each aquatic species group is included below in *Chemical-Specific Hazard (Toxicity) Identification* for each chemical analyzed in this PEIR.

Fish - Three general types of relatively standardized studies may be available on fish: acute toxicity studies; egg-and-fry studies, also referred to as early life-stage studies and full life cycle studies. To support pesticide registration, longer-term studies in fish and most other organisms are typically required for the active ingredient but are not required on pesticide formulations.

Freshwater species that are commonly used in acute assays preferred by the U.S. EPA include rainbow trout and bluegill sunfish. A large number of other freshwater and saltwater species may be used. The design of the acute toxicity bioassays is similar to the design of other acute toxicity bioassays. Range-finding studies as well as limit assays may be used. The common limit concentration is 1000 mg/L – if less than half of the fish die at a concentration of 1000 mg/L, further testing may not be required and the LC_{50} value may be reported as >1000 mg/L. In Forest Service risk assessments, NOEC and LOEC values are reported if available. The U.S. EPA will typically use an LC_{50} value for risk characterization while the Forest Service prefers to use an NOEC for sublethal effects.

Early life-stage studies in fish are analogous to mammalian teratology studies. The test involves exposing fertilized eggs to various concentrations of the chemical and maintaining the exposure until the fish are free-feeding. Freshwater species commonly used in this assay include rainbow trout, fathead minnow, zebra fish, and rice fish. The sheepshead minnow is the only saltwater species that is typically used. Results are typically reported as NOEC and LOEC values. While these studies are not true chronic studies, they are often the only longer-term study available on a presumably sensitive life-stage, and these studies often serve as the basis for the longer-term dose response assessment in fish.

Fish life cycle toxicity studies involve essentially egg-to-egg exposures. As with the early life-stage study, the life cycle study starts with fertilized eggs and continues throughout the life of the initial generation and continues until this generation produces eggs. This type of test is almost always conducted on either the fathead minnow (freshwater) or the sheepshead minnow (estuarine). When available, these tests are used for assessing the consequences of longer-term exposures unless egg-and-fry studies on other species appear to be more sensitive indicators of risk –i.e., have lower NOEC values.

Field studies that include observations on fish are occasionally available as well as mesocosm (e.g., littoral enclosure) studies. These studies are used to the extent possible as a check on the available laboratory toxicity studies. The general limitations on field studies apply to observations from field studies that involve fish. Better controlled mesocosm studies are generally more useful in assessing the relevance of standard laboratory studies to potential hazards in the field.

Amphibians (Aquatic-Phase) – The documented decline of amphibian populations worldwide has raised concerns that these species are being impacted by pesticides. Californians for Alternatives to Toxics has published a database ("Reptile, Amphibian and Pesticides", aka RAP) (CATS 2006) of the most recent international research on the effects of pesticide use on amphibians and reptiles. The list includes over 320 scientific papers published since 1999 on the effects of pesticides on amphibians, as well as almost 130 research papers on the impacts of pesticides on reptiles. This list was reviewed and 11 citations were found specifically addressing three of the chemicals analyzed in this PEIR (nine on glyphosate, one on sulfometuron methyl, and one on triclopyr). Some findings from these studies follow.

Amphibians appear to be especially vulnerable to pesticides as they readily absorb chemicals and are cutaneous breathers, breathing through their skin, as well as through a developed pair of lungs. It has been found that low levels of pesticides can cause fatal immune system suppression in amphibians (Davidson 2002). Field studies show that there are toxicological effects at much lower doses than in laboratory studies (Davidson 2004).

The "Complaint for Declaratory and Injunctive Relief" filed against the U.S. EPA and the U.S. FWS by the Center for Biological Diversity on October 19, 2011 in the U.S. District Court, Northern District of California, San Francisco Division (CBD v. U.S. EPA & U.S. FWS. 2011), asserts that the California red-legged frog (*Rana aurora draytonii*), California's largest native frog, has lost more than 70 percent of its historic range. It is believed that the use of pesticides has significantly contributed to the decline of this federally threatened subspecies and continues to pose a hazard to it:

> *Because amphibians like the California red-legged frog respire through their permeable skin, [so] they are especially vulnerable to chemical contamination. Additionally, the California red-legged frog's eggs float exposed on the water surface, where pesticides tend to concentrate. Once hatched, larvae live solely in aquatic environments for five to seven months before they metamorphose, making agricultural pesticides introduced into wetlands, ponds, and streams particularly harmful. (CBD v. U.S. EPA & U.S. FWS. 2011, p. 9)*

The "Complaint for Declaratory and Injunctive Relief" further states that

> *Pesticide contamination may cause deformities, depressed immune system functions, endocrine disruption, and death to the California red-legged frog, as well as impairment to the frog's swimming, predator avoidance, reproduction, or other key behaviors. Pesticides can also adversely affect the frog by impacting its food supplies and habitat. (ibid, p. 10)*
>
> *Due to their sensitivity to chemical contaminants, California red-legged frogs are a strong barometer for the health of California's human residents. Ultimately, the pesticides found in the frogs' habitat also migrate into Californians' drinking water, food, homes, and schools, posing a disturbing health risk. (ibid)*
>
> *The "Complaint for Declaratory and Injunctive Relief" requests the court to order the completion of interagency consultations between the U.S. EPA and the U.S. FWS on the effects of 64 pesticides on the federally listed California red-legged frog, including five of the herbicides proposed for use in the VTP and alternatives (2,4-D, glyphosate, hexazinone, imazapyr, and triclopyr) and one (atrazine) that might be used off-Program. Until the consultation process has been completed, it requests the Court to order restrictions on, or prohibit use of, the 64 identified pesticides where they may affect the California red-legged frog or its habitats.*

Battaglin and Fairchild (2002) found that:

> *There has been relatively little research directed at determining the risk of environmental mixtures of pesticides to non-target aquatic organisms. This research gap is due to several factors: (1) the difficulties arising from weather and the timing and rate of application in estimating exposures of organisms to various chemicals; and (2) the expense of conducting toxicity tests on the myriad of potential pesticides (and nutrient) mixtures found in the environment.*

"Environmental mixtures", as used in the above quote, are the combinations of pesticide(s) and chemicals in the environment, including nutrients.

Amphibians (e.g., frogs, salamanders, and toads) are cold-blooded animals that spend time both on land and in water, but breeding and development typically occur in water. Although the amount of information on the toxicity of pesticides to amphibians is increasing, very little toxicity data are generally available on amphibians compared to other aquatic species. The most commonly available study is the Frog Embryo Teratogenesis Assay – Xenopus (FETAX) bioassay. This study typically involves exposing frog embryos to the test chemical for a 96-hour period. The study is similar in design to acute toxicity study in fish in terms of the number of concentrations and reporting of results. The endpoints include observations of mortality as well as malformations.

Testing of certain species groups, notably amphibians (especially terrestrial adults) and reptiles, is inadequate or non-existent for most chemicals. As stated in the U.S. EPA "Risks of 2,4-D Use to the Federally Threatened California Red-legged Frog (*Rana aurora draytonii*) and Alameda Whipsnake (*Masticophis lateralis euryxanthus*)" (U.S. EPA 2009d, p. 104):

> *Although several registrant-submitted and ECOTOX studies evaluating the acute toxicity to aquatic-phase amphibians were reviewed, EFED [Environmental Fate and Effects Division] determined that the use of freshwater fish data is preferable to the use of aquatic-phase amphibian data because it is unknown where the CRLF would fall on a species sensitivity distribution. Because amphibian data is not required from the registrant, it is EFED's standard approach to use freshwater fish as a surrogate for aquatic-phase amphibians. In addition, because acute amphibian data were less sensitive than acute freshwater fish data, the use of freshwater fish as a surrogate provides a more conservative estimation of risk to the aquatic-phase CRLF. Chronic aquatic-phase amphibian toxicity data were not available.*

Because of the relative scarcity of data available on toxic effects to amphibians and the high level of concern with effects on amphibians, any available information on effects to amphibians are typically reviewed in some detail. If the data are sufficient, these data are used in the dose-response assessment.

Aquatic Invertebrates - Many aquatic invertebrates are relatively simple organisms to culture and test in aquatic toxicity studies, and standard acute toxicity protocols from U.S. EPA are available on a number of invertebrate species. These tests are similar in design to acute toxicity studies in fish, although some may involve somewhat shorter periods of exposure – e.g., the daphnid study typically only lasts for 48 hours. Acute toxicity studies will often be available in the open literature as well and may be conducted on a large number of different species, although the overall designs of most studies are similar to those (and often follow) standard protocols from either the U.S. EPA. Chronic studies on invertebrates are generally limited to daphnids or mysid shrimp. These are true chronic studies. The chronic daphnid study is typically the only study available on the chronic toxicity of a pesticide to freshwater invertebrates.

Aquatic Plants - Aquatic plants comprise both macrophytes (large multicelluar plants) and algae (microscopic plants). Bioassays in aquatic algae typically involve freshwater green alga, a freshwater diatom, amarine diatom, and a blue-green alga or cyanobacterium. The duration of exposure for algae is typically 48-hours. Bioassays on macrophytes typically use a species of duck weed and the duration for duckweed assays is typically 7-days to 14-days. Both types of studies measure growth (either as cell count, gross weight or length, or frond count) and express results as effective concentrations (e.g., EC50) rather than lethal concentrations. As with most other types of bioassays, the studies often report NOEC and LOEC values, and NOEC values are typically used in the dose-response assessment. Field studies may be

relatively abundant for some herbicides, particularly for those that are intended for aquatic weed control. These studies may be directly useful in the dose-response assessment as long as concentrations in water are reported and can be associated with NOAECs or LOAECs.

Table D.3-18

Aquatic Organism Acute Toxicity Summary

Herbicide	Sources	Aquatic Invertebrates	Fish	Amphibians
Boric Acid	SERA 2006a	practically nontoxic	practically nontoxic	practically nontoxic
Borax (STD)	SERA 2006a	practically nontoxic	practically nontoxic	practically nontoxic
Clopyralid	SERA 2004a	practically nontoxic	practically nontoxic	no data
Glyphosate [2/]	SERA 2011b; U.S. EPA 2004, 2008b	practically nontoxic	slightly to practically nontoxic	(slightly to practically nontoxic) [1/] [3/]
Diammonium Salt	SERA 2011b; U.S. EPA 2004, 2008b	practically nontoxic	slightly to practically nontoxic	(slightly to practically nontoxic)
Isoproplyamine Salt	SERA 2011b; U.S. EPA 2004, 2008b	practically nontoxic	slightly to practically nontoxic	(practically nontoxic)
Potassium Salt	SERA 2011b; U.S. EPA 2004, 2008b	practically nontoxic	slightly to practically nontoxic	(slightly to practically nontoxic)
Hexazinone	SERA 2005; U.S. EPA 2008c	practically nontoxic	practically nontoxic	no data
Imazapyr	SERA 2011c; U.S. EPA 2007b	practically nontoxic	practically nontoxic	no data
Sulfometuron-Methyl	SERA 2004c; U.S. EPA 2008a	slightly to practically nontoxic[1/]	slightly to practically nontoxic [1/]	(toxic) [3/]
Triclopyr Acid	SERA 2011d; U.S. EPA 2009e	(toxic) (~90X less than BEE)	(toxic) (~250X less than BEE)	(toxic) (~30X less sensitive than BEE)
BEE	SERA 2011d; U.S. EPA 2009e	highly toxic	highly toxic	(highly toxic)

		(~140X more than TEA)	(~240X more than TEA)	(~4X less sensitive than for fish) [6/]
TEA	SERA 2011d; U.S. EPA 2009e	toxic	(toxic)	(toxic)
NP9E [4/]	USDA/FS 2003b; U.S. EPA 2010e & f	(slightly toxic) [5/]	(slightly toxic) [5/]	(slightly toxic) [5/]

[1/] Toxicity ranges are due to variable toxicities for different species in the same class. [2/] Some formulations contain surfactants that have been shown to be moderately toxic to fish and other aquatic organisms; [3/](toxic) Toxicity characterizations in parentheses are based upon limited data; [4/] Toxicity is variable, depending on species; [5/] Toxicity is 100X less than for NP, one of the "highly toxic" parent compounds. [6/]This is for triclopyr BEE formulations. No data are available on the toxicity of unformulated triclopyr BEE in amphibians.

1.3.2.2.4 Chemical-Specific Hazard (Toxicity) Identification

1.3.2.2.4.1 Borax (Sources: FS WSM ver. 6.00.10; SERA 2006a)

Under conditions typically found in the environment, borate salts are rapidly converted to boric acid. Since organisms are primarily exposed to boric acid in most surface waters and at physiologic pHs, information on boric acid is used as surrogate data in this risk assessment and data are expressed in terms of the dose or concentration of borate compound (borax or boric acid) and in terms of boron equivalents (B), to facilitate comparisons between borax and boric acid.

Mammals - Although the mode of action of borax and other borate salts in mammals is not well understood, based on the results of acute exposure studies, borax is classified as *moderately toxic* to mammals. However, the Sporax® form of borax can cause severe, irreversible eye damage to the eyes of terrestrial organisms.

Developmental studies show that the developing fetus is the primary target for borate-induced toxicity. Gestational exposure of rats, mice, and rabbits to boric acid resulted in increased fetal deaths and malformations and decreased fetal weight.

Subchronic and chronic dietary exposure studies in adult rats and dogs show that at higher exposure levels adverse testicular effects and infertility can persist for at least 8 months, although at lower exposure levels testicular effects and infertility may be reversed.

Birds - Although acute single and dietary exposure studies have been conducted on borax and boric acid in standard avian test species, only limited information is available on either acute or chronic effects.

Acute exposure studies of borax show that it is *practically non-toxic* to birds, with no significant clinical signs of toxicity at dietary concentrations up to 5000 ppm borax (567 ppm B equivalent to 567 mg B/kg diet). No chronic exposure studies (21-week studies) on borax or boric acid using standard test avian species were identified. It appears that longer-term dietary exposure to boron compounds results in adverse reproductive effects in avian species.

Reptiles and Amphibians (Terrestrial-Phase) - No acute or chronic toxicity studies were found for reptiles or terrestrial-phase amphibians.

Terrestrial Invertebrates - No studies on the acute or chronic effects of borax in terrestrial invertebrates were identified in the available literature. A single study on the effects of acute topical exposure of honeybees to boric acid showed that boric acid is *practically non-toxic* to honey bees. However, borax is used in the control of termites, ants and house flies, so toxic effects may occur in other insects.

Terrestrial Plants (Macrophytes) - Although boron is an essential trace element for terrestrial plants, the amount of boron required for optimal growth and development varies widely between species and even between strains of the same species. Excess boron can lead to the development of phytotoxicity and the amount of boron required for optimal growth and the amount that is phytotoxic can be within a narrow range for some species.

There are many studies evaluating the phytotoxicity of boron compounds, but few provide data that are useful in a quantitative assessment of the risk of boron toxicity. Data are available for only a limited number of domestic plants. Per the product label for

Sporax (Wilbur-Ellis Company, no date), borax spilled or applied to crops may retard plant growth or kill plants. The label does not specify which plants species are at greatest risk for borax-induced phytotoxicity.

Terrestrial Microorganisms - Boron is apparently not an essential nutrient for soil microorganisms. A study of soil treated with borax showed no effect on total soil counts of actinomyces, fungi, protozoa and bacteria involved in nitrification. Although data needed to provide an adequate assessment of the effects of borax in nontarget microorganisms is unavailable, given the effectiveness of borax in the control of annosum root disease, it is likely that borax will have effects on nontarget microorganisms.

Fish - There is limited information available on the effects of acute borax exposure in fish. However, since borax is converted to boric acid in water, studies on boric acid can be used. Based on these studies, the U.S. EPA classifies borax as *practically nontoxic* to fish.

Acute exposure studies on borax using rainbow trout and western mosquito fish resulted in 48-hour LC_{50} values for rainbow trout of LC_{50} = 387 mg B/L and for mosquito fish LC_{50} = 930 mg B/L. Data is also available for acute exposure to boric acid in bluegill sunfish, rainbow trout, Colorado squawfish, razorback sucker, bonytail, and young salmon fry. Razorback sucker fry appear to be the most sensitive to acute boron exposure (96-hour LC_{50} of 233 mg B/L) and rainbow trout appear to be the most tolerant species (96-hour LC_{50} >1100 mg B/L).

A single open literature publication reported longer-term toxicity studies on borax that were conducted using rainbow trout, channel catfish, and goldfish. The studies show a similar degree of sensitivity for the three-species tested, with the lowest estimated NOAEC (for mortality) of 0.5 ppm B for goldfish and the highest estimated NOAEC (for mortality) of 1.0 ppm B for rainbow trout and channel catfish. The relative tolerance to borax of the different species cannot be determined, as different exposure times were used for each of the three-species tested (up to 28 days for trout, 9 days for catfish, and 7 days for goldfish).

Amphibians (Aquatic Phase) - Although very little information is available on the effects of borax to amphibians, boric acid and borax appear to be *practically nontoxic* to amphibians.

As stated in SERA 2006a (p. 4-6):

> *A single study in larval leopard frogs exposed to borax for 7.5 days reports an LC_{50} of 47 ppm B, with an estimated NOAEC (for mortality) of 1.0 ppm B and an estimated LOAEC (for mortality) of 5.0 ppm B (Birge and Black 1977). Thus, toxicity of borax to leopard frogs appears to be relatively low. Results of a study in wood frog, Jefferson salamander, spotted salamander, and American toad show that boron concentrations of 50 and 100 mg B/L caused a dose-related decrease in proportion of eggs hatching in American toad, while hatching was unaffected in the other three species (Laposata and Dunson 1998). In this same study, a dose-dependent increase in proportion of deformed larvae was observed in wood frog, Jefferson salamander, and spotted salamander (not assessed in American toad)*

Aquatic Invertebrates - Although the "confidential business information" literature did not include standard bioassays of the acute or chronic toxicity of borax or boric acid to aquatic invertebrates, some studies are available in the open literature.

Results of acute toxicity studies in *Daphnia magna* to borax and boric acid show similar LC_{50} values for borax (48-hour LC_{50} = 141 mg B/L) and boric acid (48-hour LC_{50} = 133 mg B/L). Another study indicates that the larval freshwater midge *Chironomus decorus* is more tolerant than daphnids to acute boron exposure, with a 48-hour LC_{50} value of 1376 mg B/L.

Two chronic toxicity studies in daphnids conducted with boric acid reported similar results. The lowest 21-day LC_{50} value reported is 52.2 mg B/L. The lowest NOAEC value reported for reproductive parameters is 6 mg B/L, with a LOAEC for reproductive parameters of 13 mg B/L.

Aquatic Plants - Although no studies on the effects of borax in aquatic macrophytes were identified in the available literature, there are a few studies on the effects of boric acid. Short-term exposure studies were conducted with boric acid in water milfoil, water buttercup, and waterweed, with similar LC_{50} values reported for all three-plant species (water milfoil and waterweed: 5 mg B/L; water buttercup 10 mg B/L).

A chronic exposure study of boric acid in common reed *(Phragmites australis*) reported a 2-3 month NOAEC of 8 mg B/L and a 2-year NOAEC of 4 mg B/L.

In algae, the 72-hour LC_{50} values for exposure to boron reported for *Scenedesmus subpicatus* range from 34 mg B/L to 52 mg B/L and the 72-hour NOAEC values range from 10 mg B/L to 24 mg B/L, with similar NOAEC values reported for *Scenedesmus quadricauda* and *Microcystis aeruginosa*.

Data reviewed by the WHO on the effects of boron exposure to several species of non-algal aquatic microorganisms reported 72-hour NOAEC values ranging from 0.3 mg B/L in *Entosiphon sulfacum*, a flagellate, to 291 mg B/L in *Pseudomonas putida*.

1.3.2.2.4.2 Clopyralid (Sources: FS WSM ver. 6.00.07 & 6.00.10; SERA 2004a)

As stated in SERA 2004a:

> *The toxicity of clopyralid is relatively well characterized in experimental mammals but few wildlife species have been assayed relative to the large number of non-target species that might be potentially affected by the use of clopyralid. Within this admittedly substantial reservation, clopyralid appears to be relatively non-toxic to aquatic animals. Thus, the potential for substantial effects on non-target species appears to be remote.*

As with terrestrial species, the available data on aquatic species, both plants and animals, suggest that clopyralid is relatively non-toxic.

Mammals - How clopyralid causes toxicity in mammals has not been determined. No consistent toxic effect or set of toxic effects to an organ or an organ system has been attributed to clopyralid.

The toxicity of clopyralid is relatively well characterized in experimental mammals (rats, mice, rabbits, and dogs) and appears to be relatively non-toxic, although it is likely to be more toxic to dogs. Although few wildlife species have been assayed relative to the large number of non-target species that might be potentially affected, the potential for substantial effects on non-target species appears to be remote.

Birds - Most of the acute toxicity studies of clopyralid involve dietary administration over short periods of time (i.e., 5 days) using mallard ducks and bobwhite quail. These studies suggest that the dietary LC_{50} values for both clopyralid and the monoethanolamine salt of clopyralid are above 6000 ppm.

A study of direct spray of bobwhite quail eggs at up to 0.56 kg a.e./ha (0.50 lb a.e./acre) caused no gross effects (i.e., viability, hatchability, body weight) and no effects on immune function (humoral or cell-mediated) in chicks. In California, the maximum allowable application rate for clopyralid is 0.25 lb a.e./acre, well under the quantity applied in the study. Clopyralid is considered only slightly toxic to birds.

Reptiles and Amphibians (Terrestrial-Phase) - No acute or chronic toxicity studies were found for reptiles or terrestrial-phase amphibians.

Terrestrial Invertebrates - Clopyralid is *practically nontoxic* to bees and other invertebrates tested.

In several studies involving oral and direct contact exposure to honeybees, no significant increase in mortality was noted at doses of up to 0.1 mg/bee.

Based on a large series of bioassays and field trials of Lontrel 100, a formulation of clopyralid that is no longer marketed commercially, clopyralid was classified as *harmless* (less than 30% mortality) to 14 insect parasites and 17 predatory mites in contact bioassays. It was classified as *slightly harmful* (25-50% mortality) to *Semiadalia 11-notata* (Coccinellidae), *Anthocoris nemoralis* (Anthocoridae), and *Chryosperla carnea* (Chrysopidae). A 2002 study of direct application effects of Lontrel on spiders reported an acute (96-hour) mortality of less than 10%.

Based on the results of a static bioassay on earthworms, the soil LC_{50} of clopyralid to earthworms is greater than 1000 ppm soil.

Terrestrial Plants (Macrophytes) - The toxicity of clopyralid to terrestrial plants has been examined in substantial detail. Because clopyralid is rapidly absorbed across leaf surfaces but much less readily absorbed by roots, it is much more toxic in post-emergent treatments (i.e., foliar applications) than in pre-emergent treatments (i.e., application to soil). Clopyralid appears to be highly selective in its toxicity to terrestrial plants, being highly toxic to broadleaf plants but relatively non-toxic to grasses or grains. The potential for substantial effects on non-target species appears to be remote.

An 8-year follow-up study of plots treated with Stinger, which like Transline contains the monoethanolamine salt of clopyralid, at a rate of 0.28 kg a.e./ha (0.25 lb a.e./acre) (by backpack sprayer for the control of spotted knapweed (*Centaurea maculosa*) showed no substantial or statistically significant effect on species diversity or species richness in plants. Some plant families, such as *Asteraceae* and *Fabaceae*, were impacted. Clopyralid was not detected in soil below 25 cm (9.8 inches). In California, the maximum allowable application rate for clopyralid is 0.25 lb a.e./acre, the same as applied in the study.

Terrestrial Microorganisms - What little information is available on the toxicity of clopyralid to terrestrial microorganisms appears to support that there are little to no toxic effects.

Fish - Only standard 96-hour acute toxicity bioassays are available for fish. The lowest reported LC_{50} for clopyralid is 103 mg a.e./L in trout. The monoethanolamine salt of clopyralid appears to be substantially less toxic than technical clopyralid, with 96-hour LC_{50} values in the range of 2000 mg a.i./L to 4700 mg a.i./L (equivalent to 700–1645 mg a.e./L).

No chronic toxicity studies on the toxicity of clopyralid to fish eggs or fry have been done for clopyralid, but such studies done for daphnids indicate that clopyralid is *practically nontoxic* to fish.

Amphibians (Aquatic Phase) - No acute or chronic toxicity data for amphibians was found in either U.S. EPA files or published literature.

Aquatic Invertebrates - *Daphnia magna* (water flea) is the only species of aquatic invertebrate on which toxicity data are available. The lowest reported acute toxicity LC_{50} for technical clopyralid is 225 mg/L (208–245 mg/L), about 2 times higher than the lowest reported LC_{50} in fish. Unlike with fish, the monoethanolamine salt appears to only marginally reduce the toxicity of clopyralid (LC_{50} of 350 mg a.e./L for the salt and 225 mg a.e./L for the acid).

A standard chronic reproduction bioassay conducted in *Daphnia magna* using the monoethanolamine salt resulted in a NOEC of 66 mg a.i./L (equivalent to 23.1 mg a.e./L).

Aquatic Plants - The available data on aquatic plants suggest that clopyralid is relatively non-toxic.

As might be expected, aquatic macrophytes are more sensitive to clopyralid than fish or aquatic invertebrates. The EC_{50} for growth inhibition in duckweed is 89 mg/L. At lower concentrations, in the range of 0.01 to 0.1 mg/L, growth of other aquatic macrophytes is stimulated. The lowest reported EC_{50} for growth inhibition of green algae is 6.9 mg/L.

There are no published or unpublished data regarding the toxicity of clopyralid to aquatic bacteria or fungi.

1.3.2.2.4.3 Glyphosate (Sources: FS WSM v. 6.00.10; SERA 2003a, 2011b; U.S. EPA. 2009c)

Relatively complete sets of studies are available in birds, terrestrial-phase amphibians, and terrestrial invertebrates, plants, and microorganisms for technical grade glyphosate and some formulations (Roundup and Rodeo) used in the United States.

The U.S. EPA has done an extensive review of toxicity of glyphosate to aquatic organisms. Relatively complete sets of studies are available aquatic organisms for technical grade glyphosate and some formulations (Roundup and Rodeo) used in the United States. The toxicity of the original Roundup and other formulations containing the surfactant POEA is far greater than technical grade glyphosate, Rodeo, and other formulations that do not contain POEA. Fish, amphibians, and aquatic invertebrates are about equally sensitive to technical grade glyphosate and some formulations.

Mammals - There is a large body of studies, including those published and those required by the U.S. EPA for pesticide registration, on the effects in test mammals of glyphosate and its formulations. There is less information on the toxicity of glyphosate or its formulations to wildlife and domestic mammals. Glyphosate is considered *slightly toxic* to mammals.

In terms of acute toxicity, there seems to be little difference in toxicity between tested species. Studies have resulted in intrperitoneal LD_{50} values for deer mice, chipmunks, shrews, and voles for glyphosate IPA in the range of 800 to 1370 mg/ kw bw and 1100 mg/ kw bw for lab mice. The LD_{50} for Roundup in rats is approximately 5400 mg/ kw bw, similar to that for humans.

Based on two developmental studies of 2-week sublethal dosing of rabbits and rats with glyphosate and glyphosate formulations, it is thought that larger mammals may be more sensitive than small mammals, as the NOAEL for rabbits (100 mg/kg bw/day) was a factor of 10 less than that for rats (1000 mg/kg bw/day).

An unpublished repeated-dose study (over 7 days) indicates that cattle may be more susceptible to Roundup than rats, as some cattle died at doses of 790 mg/kg bw/day and others exhibited additional signs of toxicity (including diarrhea and decreased food intake) at doses of 500, 630, and 790 mg/kg bw/day. No adverse effects were noted at 400 mg/kg bw/day, equivalent to 160 mg a.e./kg bw.

Decreased food consumption and body weight gain in experimental mammals, including three wildlife species, exposed to high dietary concentrations of glyphosate indicates toxicity, taste aversion, or a combination of these two. However, studies of exposure by dermal, gavage, or drinking water support that toxicity may be the dominant factor.

Most field studies on the effects of applications of glyphosate formulations show no adverse effects to populations of mammalian species. Following application of about 2.7 lb a.e./acre of Roundup, reproduction of deer mice and voles was comparable or better over a 3-year period on the treated site than on the untreated control site.

Birds - The U.S. EPA classifies technical grade glyphosate as *practically nontoxic* to birds. This is based on an acute gavage study in bobwhite quail that determined an LD_{50} of >2000 mg/kg bw. Additional gavage LD_{50} values range from 1130 mg/kg bw for the monoammonium salt of glyphosate to >3190 mg/kg bw for an unspecified salt. No adverse effects were seen in reproduction studies on mallard ducks or bobwhite quail at dietary concentrations up to 833 ppm.

Two acute dietary studies of Roundup PRO (41% glyphosate IPA and 14.5% POEA) determined NOAELs of 1760 ppm a.e., which is not considered highly toxic to birds.

In two open literature dietary studies on Roundup, one 7-day and one 21-day, at doses of 5000 mg a.e. zebra finches experienced substantial weight loss (20-60% over controls) and all died after 7 days and chickens lost 45% of their weight compared to controls at doses of 4500 mg a.e. No adverse effects were noted at lower doses in either study.

There are no standard reproduction tests for Roundup formulations. Two studies where eggs were immersed in a solution of Roundup for 5-30 seconds suggest that it is not likely to cause developmental effects in chicks.

Reptiles and Amphibians (Terrestrial-Phase) - No acute or chronic toxicity studies were found for reptiles.

There is little information on the toxicity of technical grade glyphosate to terrestrial-phase amphibians. Based on intraperitoneal studies on several species of amphibians, LD_{50} values for glyphosate IPA range from 790 to 925 mg a.e./kg bw. This is similar to several species of small mammals.

Direct spray of amphibians is of concern, as frog skin is 26 times as permeable as pig skin to glyphosate acid. The results of two direct spray studies are inconsistent. In one study, species of tree and wood frogs and a toad were sprayed with Roundup Weed and Grass Killer at a rate of about 0.011 lb a.e./acre, with >50% mortality after 24 hours. This is inconsistent with the findings of all the following studies. In the other study, three glyphosate formulations (one being Roundup WeatherMax) were applied to two toad species at a rate of 15 lbs a.e./acre, with no significant mortality.

In a lab bioassay of newly metamorphosed frogs misted with Vision (41% glyphosate IPA and a POEA surfactant) at a rate of 1.6 lbs a.e./acre, there was no mortality.

In a series of mesocosm studies in which Glyphos (with the adjuvant Cosmo-Flux) was applied at a rate of about 1.7-26 lbs a.e./acre, functional NOEC values ranged from ~0.3-6.3 lb a.e./acre. Substantial mortality would not be expected with application rates in the range of about 1-2 lb a.e./acre and some species would be tolerant of much higher application rates.

A study involving aerial application of glyphosate (formulation and units unspecified) to clearcuts at a rate of 1.2 lbs/acre resulted in no adverse effects (based on capture rates) on six species of amphibians (rough-skin newt, ensatina, Pacific giant salamander, Dunn’s salamander, western redback salamander, and red-legged frog), as compared with controls.

In a study where newts were given an intraperitoneal dose of glyphosate IPA at a rate of 50mg/kg bw and then released, the movements of the dosed animals did not differ substantially from that of the controls.

Terrestrial Invertebrates - There is a standard set of tests of glyphosate on the honeybee, as well as studies on earthworms, isopods, snails, spiders, butterflies, and other arthropods. Glyphosate appears to be *practically nontoxic* to bees and other invertebrates tested.

Standard oral and contact bioassays have determined a LD_{50} for honeybees of >100 µg/bee. The NOEC for Roundup PRO is also 100 µg.

Glyphosate IPA was ineffective as an insecticide in controlling spider mites at application rates of 0.593-4.74 mg a.i./leaf, based on mortality to eggs, larva, nymphs, and adults. A series of lab and field studies of the effects of glyphosate on the spider *Lepthyphantes tenuis* resulted in low mortality rates.

Two acute dietary studies of an Argentinean formulation of glyphosate IPA (Glifoglex 48), one with spiders and one with lacewings, at a rate of 192 mg a.e./L resulted in adverse effects in the spiders in food consumption, web building, and reproductive capacity. In the lacewings there was increased mortality, reduced reproductive capacity, and malformed offspring.

While data on other arthropods is less detailed, it appears to indicate that there is a low potential for direct adverse effects from exposure to glyphosate. The soil LC_{50} of glyphosate for a common Libyan earthworm is 177-246 mg/kg soil dry weight over 8-37 days of exposure. In a 14-day dietary study, there was no mortality to the Brown garden snail (*Helix aspersa*) exposed to glyphosate at a rate of 1500 mg/kg bw.

Terrestrial Plants (Macrophytes) - There are toxicity studies on vegetative vigor for both technical grade glyphosate IPA and glyphosate formulations and on seedling emergence for glyphosate formulations.

Glyphosate is much less toxic to plants when they are exposed through soil than when exposed through foliage, probably because glyphosate binds tightly to some types of soil. Soil application rates of 4-5 lbs a.e./acre for three formulations of glyphosate were relatively nontoxic to seedlings. Foliar applications of glyphosate IPA resulted in NOAEC values for monocots of 0.56-0.70 lb a.e./acre and values for dicots of 0.035-0.46 lb a.e./acre. Studies on a wetable powder formulation of glyphosate gave a similar relationship between NOAEC values (monocots 0.07-0.45 lb a.e./acre, dicots 0.02-0.45lb a.e./acre).

Several spray drift studies have been conducted. In one, transient (30-day) damage to soybeans occurred at a spray deposition concentration of 0.03 lb/acre (1/33 of the application rate of 1.121 kg/ha), but did not affect yield at harvest time. Grapes only experienced damage at deposition concentrations of 1/3 the application rate. A grass and a dicot experienced substantial damage at a spray deposition concentration of 1.8 lbs/acre. Canola, smartweed, soybeans, and sunflowers experienced no marked effects at a deposition concentration of 0.003 lb/acre (1/125 of the application rate).

A study determined that some bryophytes and fungi may be sensitive to long-term exposure to glyphosate. The EC_{50} value for a decrease in abundance two years after exposure was 0.7 lb/acre and the decrease was apparent six weeks after the application.

Terrestrial Microorganisms - A substantial body of information indicates that glyphosate is likely to enhance or have no effect on soil microorganisms, with little information indicating adverse effects under field conditions. However, under laboratory conditions, a number of

studies indicate adverse effects. At high concentrations (845-3380 mg/L) glyphosate might inhibit growth in soil algae and cyanobacteria, although other studies show no inhibitory effects on fungi and only slight effects on some species of bacteria at more realistic concentrations (2-20 ppm). Another study resulted in direct toxicity to soil fungi in a culture medium at concentrations of 10 ppm or greater. Apparently, glyphosate acid is the least inhibitory, followed by glyphosate IPA and Roundup.

Field applications of glyphosate resulted in only a short-term (2 month) decrease in fungal and bacterial counts at an application rate of 0.54 kg/ha, no effect on soil fungi or bacteria after 10-14 months after an application of 3.23 kg/ha, and only a transient decrease in soil microbial activity after an application of 5 kg/ha.

Fish - The U.S. EPA classifies glyphosate acid and glyphosate IPA as *slightly toxic* (LC_{50} >10-100 mg/L) to *practically nontoxic* (LC_{50} >100 mg/L) to fish. One study found two LC_{50} values of 10 mg a.e./L, but these appear to be related to the pH of the water, with LC_{50} values decreasing (toxicity increases) as water pH decreases (becomes more acidic). Although this acute toxicity study in five species of fish in five different types of water spans a range from 24 to 96 hours, in many of the bioassays most of the fish died on the first day, so 24-hour and 96-hour LC_{50} values are the same, or only marginally different.

The same study found that the acidity of water has a much greater effect on the toxicity of glyphosate to fish than does the variability of sensitivity between species (relatively minor at the same pH). Coho salmon were the least sensitive species to pH variance, with LC_{50s} of 27 mg a.e./L at a pH of 6.3 and 174 mg a.e./L at a pH of 8.2, differing by a factor of 6. Rainbow trout were the most sensitive species, with LC_{50s} of 10 mg a.e./L at a pH of 6.3 and 197 mg a.e./L at a pH of 8.2, differing by a factor of 20.

As the temperature of water increases there is a corresponding increase in the toxicity of Roundup. An increase of 10° C resulted in a decrease in the LC_{50} by a factor of 2 in rainbow trout and bluegill sunfish. As might be expected, smaller fingerlings and fry were more susceptible to changes in temperature than larger fingerlings or eggs.

Most acute toxicity studies require fish not to be fed (i.e., fasted) before (for 48 hours) and during the bioassay. A 96-hour bioassay would require fish to fast for 6 days. In a study of flagfish that were both fasted and fed, fasting increased the toxicity of glyphosate to 8-day old flagfish by a factor of 10 (LC_{50s} of 2.94 mg a.e./L for fed fish and 29.6 mg a.e./L for fasted fish).

In a 12-hour field simulation study, rainbow trout were exposed to either glyphosate IPA or Roundup at concentrations of 0.02, 0.2, or 2 mg/L, equivalent for IPA salt of 0.015, 0.15, and 1.5 mg a.e./L and for Roundup of 0.006, 0.06, and 0.6 mg a.e./L. Following exposure, the trout were held for 30 days in uncontaminated water, showing no adverse effects, based on gonadal weight in males and the number of eggs per female.

Most of the extensive studies on glyphosate formulations have been done on Roundup. Earlier studies were mostly done on Monsanto's Roundup (41% aqueous solution glyphosate IPA, 15% POEA surfactant). For glyphosate formulations with POEA and for POEA itself, toxicity to fish increases with increasing pH (alkalinity), although increases in toxicity are modest. For five species of salmonids, over a range of pHs from 6.3 to 8.2, the range of 96-hour LC_{50s} was about 2 to 3 (6 to 20 for glyphosate). The increase in toxicity of glyphosate formulations with increasing pH is due to the effects on the surfactant POEA, as the glyphosate itself decreases in toxicity.

Various studies have reported a range of LC_{50} values for Roundup formulations (those used in the U.S.), which contain, or appear to contain the POEA surfactant, of 1-10 mg a.e./L. The toxicity of Vision, a glyphosate formulation equivalent to Roundup, varies by a factor of 4 as the concentration of surfactant varies from 7.5% to 15%.

The only clearly documented study on Rodeo reported a LC_{50} of 429 mg a.e./L for trout. However, Rodeo and similar formulations require surfactants (less toxic than POEA), which may increase the toxicity of the formulation by a factor of 4. Roundup Biactive, an Australian formulation that contains a surfactant at a concentration of 10-20%, has an LC_{50} of 800 mg a.e./L for rainbow trout and is less toxic than Rodeo and much less toxic than Roundup with POEA.

The manufacturing process for POEA surfactants and the chemical composition is proprietary, so the variability of the surfactants in different glyphosate formulations is unavailable.

Monsanto's product code for the original Roundup surfactant is MON 0818 (75% POEA). As with Roundup, the toxicity of MON 0818 to fish increases with increasing water pH. Over a pH range of 6.3 to 8.2, the LC_{50} values for five species of salmonids decreased by factors of 1.2 to 3.2. Typical LC_{50} values for trout are 1-3 mg/L, but the upper and lower bounds for MON 0818 are 0.65 mg/L and 7.4 mg/L respectively. It has also been determined that the joint action of Roundup with MON 0818 is less than additive.

Acute toxicity values provided by Monsanto for surfactants mostly used with Rodeo and similar formulations are mostly in the range of 1-10 mg/L, similar to MON 0818. The U.S. EPA classifies Syndets (anionic surfactant), Activator 90, Entry II, Frigate, Induce, No Foam A, R-11, S. Spreader 200, Widespread, X-77 as *moderately toxic* to fish. Liqua-Wet, Passage, and Spreader-Sticker are *slightly toxic* and Agri-Dex, LI 700, and Geronol CF/AR are *practically nontoxic*.

Sub-lethal exposures to Roundup formulations sometimes, but not always, result in a broad spectrum of stress effects in fish. Roundup formulations, most likely the surfactants in the formulations, have been shown to cause damage to the gills of fish. In one study, trout and bluegill sunfish were exposed to technical grade glyphosate at a purity (62%) that is much

lower than used in commercial formulations. Damage to gill occurred at concentrations of 5 mg/L over 14 days and damage to both gills and livers at concentrations of 10 mg/L.

Trout can sense but will not avoid Roundup formulations in water until concentrations approach or exceed 96-hour LC_{50} concentrations. At concentrations, as low as 25% of this LC_{50} value, trout exhibit one or more of the following behavioral effects: changes in coughing and ventilation rates, swimming, and coloration and loss of equilibrium.

Two acute toxicity studies of Roundup involving short (10 minute) exposures to a high concentration (100 mg/L or 30 mg a.e./L) of a 41% Roundup formulation resulted in adverse effects on fish immune systems.

The only full life-cycle chronic toxicity study for any form of glyphosate is for the fathead minnow. Using 87.3% pure technical grade glyphosate, no effect was apparent on mortality or reproduction at a concentration of 25.7 mg/L. Given that the differences in the acute toxicity of technical grade glyphosate, glyphosate formulations, and glyphosate-surfactant mixtures are substantial, the merit of this finding is questionable. However, since the surfactants used with glyphosate are less persistent under field conditions, it is likely that glyphosate-surfactant mixtures over longer term exposures will not exhibit the toxicity of acute exposures.

Four long-term studies (2-3 months) of various types of exposure to various species of fish using Roundup formulations in a wide range of concentrations found no overt signs of toxicity and only sublethal adverse effects, primarily to livers, but in some cases to gills and kidneys.

Amphibians (Aquatic Phase) - Numerous acute toxicity studies have been done on the effects of technical grade glyphosate and glyphosate formulations to aquatic-phase amphibians, most over a 96-hour period. However, the U.S. EPA “Pesticide Effects Determination” for the risks of glyphosate use to the federally threatened California red-legged frog (U.S. EPA 2008b) listed only one toxicity study using an aquatic-phase amphibian (leopard frog) as a study organism.

Relative to the skin of fish, amphibian skin is highly permeable to glyphosate. However, based on acute toxicity data, there is no indication that amphibians are substantially more sensitive than fish to glyphosate, glyphosate formulations, or the POEA surfactant in Roundup.

Definitive LC_{50} values for glyphosate acid range from 75.2 to 121 mg a.e./L, similar to those for fish (43 to 100 mg a.e./L at neutral pH). Non-definitive LC_{50} values for glyphosate IPA range from >17 to >466 mg a.e./L, indicating that it is much less toxic than the acid. Based on intraperitoneal studies on several species of amphibians, LD_{50} values for glyphosate IPA range from 790 to 925 mg a.e./kg bw. This is similar to several species of small mammals.

The formulation Rodeo is essentially an aqueous solution of glyphosate IPA, with an LC_{50} of 7297 mg a.e./L in African clawed frogs (*Xenopus laevis*) embryos. It may be that frog embryos are less sensitive to glyphosate and surfactants than larvae, as in fish.

Studies on the effect of water pH on the toxicity of glyphosate and a surfactant to *Xenopus laevis* larvae indicate that as pH increases (decreasing acidity) the toxicity of Rodeo, Roundup, and the surfactant MON 0818 increases. For Rodeo, the 96-hour LC_{50} was 7-11 times more toxic at a pH of 8.0 than at 6.5. The stage of frog development also affects sensitivity, with the embryos of four species being less sensitive than the larvae. Sensitivity also varies between species, ranging from factors of 2 to 3 in *Bufo americanus* to a factor of 7 in *Xenopus laevis* and *Rana pipiens*.

Another study (Chen 2003) found that “multiple stress interactions may exacerbate chemical effects on aquatic biota in natural systems”. For two common wetland species, zooplankton and Ranid tadpoles, significant effects of the herbicide Vision® (glyphosate) were measured at concentrations lower than the calculated worst-case value for the expected environmental concentration ([EEC], 1.40 mg a.e./L). High pH (7.5) increased the toxic effects of the herbicide on all response variables for both species. This finding corroborates those from other studies and supports the premise that laboratory studies are inadequate to assess the hazard of chemicals to wild species in their natural environment. It should be noted that although Vison® is not registered for use in California, it is similar to Roundup® (Vison® = 41% glyphosate, 59% other ingredients; Roundup® = 41% glyphosate, 15% polyethoxylated-tallowamine surfactant, and 44% water).

A study on *Ranidella signifera* tadpoles exposed to glyphosate IPA, with the surfactant Geronol CF/AR (classified as *practically nontoxic* to fish by the U.S. EPA) at concentrations of 10-45%, resulted in indefinite LC_{50} values ranging from >100 to >450 mg a.e./L, which are considered NOAELs. Amphibians appear to be less sensitive to this formulation than trout. It is postulated that more toxic surfactants will increase the toxicity of glyphosate IPA, Rodeo, and similar formulations to amphibians.

Roundup Biactive, an Australian formulation that contains a surfactant at a concentration of 10-20%, has a range of non-definitive LC_{50} values of >17.9 to >494 mg a.e./L. Glyfos BIO, a formulation containing 3-7% POEA surfactant, has a LC_{50} of 17.9 mg a.e./L and is less toxic than typical Roundup formulations. It is unclear whether this is due to a less toxic form of POEA, to a smaller quantity of POEA, or to a combination of the two. Glyfos AU, which also contains 3-7% POEA, has a LC_{50} of 8.9 mg a.e./L, in the range of the upper bounds of more toxic Roundup formulations. An unspecified Glyphos formulation containing 15% POEA has a LC_{50} of 0.93 mg a.e./L, in the range of the lower bounds for Roundup formulations

More toxic formulations of glyphosate include various Roundup, Vision, and Glyphos formulations. Roundup Original Max has a LC_{50} value of 3.2 mg a.e./L. The upper bound for

other formulations of Roundup and Vision range from 8.0 to 51.8 mg a.e./L. Absence matched bioassays, it cannot be determined whether higher LC_{50} values reported in other studies are due to species sensitivity, experimental conditions, or random variability.

Rick Relyea found through his research (Relyea 2005) that Roundup®, "the most commonly used herbicide in the world, is deadly to tadpoles at lower concentrations than previously tested; that the presence of soil (in water) does not mitigate the chemical's effects; and that the product kills frogs in addition to tadpoles." Relyea wrote that "The most striking result from the experiments was that a chemical designed to kill plants killed 98 percent of all tadpoles within three weeks and 79 percent of all frogs within one day." Previous studies (Howe 2003) have determined that the surfactant polyethoxylatedtallowamine (POEA), an inert ingredient added to enhance herbicide penetration into plant leaves, and not the active ingredient (glyphosate) is lethal to amphibians.

A study (Howe 2003) in California on the effects of glyphosate formulations to four Ranid frog species in the Sierra found that "acute toxicity values in order of decreasing toxicity were POEA > Roundup Original > Roundup Transorb® >Glyfos AU®; no significant acute toxicity was observed with glyphosate technical material or the glyphosate formulations Roundup Biactive®, Touchdown®, or Glyfos BIO®." Data from this study indicated that the composition of surfactants must be considered when the toxicity of glyphosate-based herbicides are evaluated.

Differences in the toxicity of the more toxic formulations of Roundup and similar formulations to amphibians and fish appear to be negligible, with 96-hour LC_{50} values for amphibians ranging from 8.0 to 51.8 mg a.e./L and for fish from 0.96 to 11.26 mg a.e./L.

Studies on the toxicity to amphibians of the surfactant POEA (MON 0818) report a range of 96-hour LC_{50s} of 1.1 mg/L in the green frog (*Rana clamitans*) to 6.8 mg/L in the African clawed frog (*Xenopus laevis*). These values are comparable to those in fish (1-3 mg/L).

Studies by Relyea on green frog tadpoles indicate that growth is sometimes a more sensitive endpoint than mortality, but that the difference in glyphosate concentration that causes these effects is only ~1 ppm (~1 ppm for adverse growth effects and ~2 ppm for mortality).

A frog (*Xenopus laevis*) embryo teratogenesis assay for malformations after exposure for 96 hours to glyphosate IPA, Roundup, and the surfactant POEA found no statistically significant increases in abnormalities between embryos exposed to nonlethal concentrations and the control group.

Another study tested Kleeraway Grass and Weed Killer RTU (Monsanto) (0.75% glyphosate IPA and an ethoxylated tallowamine surfactant) exposure to tadpoles of the western chorus frog (*Pseudacris triseriata*) and the plains leopard frog (*Rana blairi*). Tadpoles were exposed to concentrations of 0.56, 5.6, 56, or 560 mg a.e./L for 24 hours. At a concentration of 0.56

mg a.e./L, 55% of the western chorus frog tadpoles died and at greater concentrations, all died. In an initial experiment, all plains leopard frog tadpoles died at all concentrations, but in a repeat experiment on older tadpoles, all tadpoles survived when exposed to a concentration of 0.56 mg a.e./L. In both species, normal growth and development occurred in survivors.

Some data indicate that frogs will avoid laying eggs in pools contaminated with Roundup at a concentration of 2.4 mg a.e./L, within the 96-hour LC_{50} range for frogs. Similar to fish, frogs appear to avoid waters contaminated with acutely toxic concentrations of glyphosate-surfactant mixtures, but avoidance of waters contaminated at sub-toxic concentrations has not been demonstrated.

A study of the effect on the immune function of green frog tadpoles (*Rana clamitans*) of exposure to a concentration of 3.7 mg a.e./L technical grade glyphosate found no adverse effects.

In a chronic study (42-day) on *Rana pipiens* larvae of exposure to glyphosate IPA at a concentration of 1.8 mg a.e./L, no adverse effects were noted. Tadpoles were also exposed to Roundup Original and Roundup Transorb at concentrations of 0.6 and 1.8 mg a.e./L (the surfactant MON 0818 POEA in those formulations was 0.3 and 0.9 mg a.e./L) as well as to MON 0818 by itself. With all exposures, adverse effects were noted, including an increase in the length of time for development of tadpoles, a decrease in survival, a decrease in the length of tadpoles, and an increase in the number of tadpoles with intersex gonads. Roundup Transorb appeared to be more toxic than Roundup Original and MON 0818 POEA surfactant alone caused the same effects as the formulations.

Another chronic study (43-days) on *Rana cascadae* larvae of exposure to Roundup at concentrations of 1 or 2 mg a.e./L found a substantial decrease in survival at the lower concentration and no survival to day 43 at the higher concentration.

A 16-day exposure study by Relyea on the interaction of Roundup and predator stress on six species of frogs found LC_{50} values in the absence of predator stress of 1.32 to 2.52 mg a.e./L. Based on other studies, as with fish there does not appear to be a substantial concentration-duration relationship for glyphosate-surfactant formulations.

Several mesocosm studies by Relyea and coworkers with Roundup formulations at concentrations of 1.3 to 2.8 mg a.e./L found decreases in survival (only 21% at the end of day 1) and biomass of three species of frog tadpoles.

Aquatic Invertebrates - Acute toxicity studies on aquatic invertebrates are typically done for 48 hours and results are expressed in terms of EC_{50} (immobility) rather than LC_{50} (mortality), as an immobilized invertebrate in an aquatic ecosystem is considered to be functionally dead.

As with fish and amphibians, most Roundup and similar formulations are much more toxic to invertebrates than glyphosate or glyphosate salts, with EC_{50} values for the former formulations of 1 to 50 mg a.e./L and for the latter of 100 to 650 mg a.e./L. Studies that show the joint action of glyphosate and POEA indicate a less than additive effect. For some Accord formulations that contain POEA the EC_{50} values range from 20 to 25 mg a.e./L. EC_{50} values for Rodeo, Roundup formulations with other surfactants, and other non-USA formulations range from 50 to >500 mg a.e./L. As there are few acute toxicity studies on Accord formulations with surfactants, it is unclear whether it is less toxic than most Roundup formulations.

Although for technical grade glyphosate there is a relationship between duration of exposure and response, there does not appear to be a substantial relationship for glyphosate formulations.

Acute toxicity studies are available on two species of daphnid, a copepod, midge larvae, and a bivalve. Studies on *Daphnia magna* report EC_{50} values for glyphosate acid of 128 to 647 mg a.e./L. Studies of copepods and *Ceriodaphnia* found that glyphosate acid is somewhat more toxic than glyphosate IPA. Sensitivity to glyphosate acid is about equal for midges (LD_{50} = 55 mg/L), *Ceriodaphnia* (LD_{50} = 147 mg/L), and copepods (LD_{50} = 35.3 mg/L).

An acute toxicity study of freshwater mussels (*Lampsilis siliquoidia*) exposed to glyphosate acid, glyphosate IPA, and isopropanol amine found that glyphosate acid was relatively non toxic (LC_{50} = >200 mg a.e./L) to larvae and juvenile mussels and that glyphosate IPA and isopropanol amine were much more toxic (LC_{50} = 5 to 7 mg a.e./L).

Formulations of Roundup are much more toxic (LC_{50} = 1.5 to 62 mg a.e./L) than Rodeo (essentially an aqueous solution of the IPA salt of glyphosate) and similar formulations (LC_{50} = 200 to >4,000 mg a.e./L) to aquatic invertebrates. This is attributable to the POEA surfactant in Roundup formulations, which is lacking in Rodeo and similar formulations.

Studies specifically on the toxicity of the POEA surfactant MON 0818 to aquatic invertebrates indicates an LC_{50} of 0.5 to 13 mg/L. Studies on the effect of water pH on the toxicity of MON 0818 have not been done (as they have for fish), so the lower LC_{50} value may be a reflection of a higher water pH (8.2) rather than a greater sensitivity to POEA of invertebrates relative to fish. The surfactants Activator 90, Entry II, and X-77 appear to be as toxic as MON 0818. Geronol CF/AR surfactant is much less toxic than MON 0818, with an EC_{50} for *Daphnia magna* of 48 mg/L, and the EC_{50} values for most other surfactants range from 10 to 100 mg/L. The surfactant Agri-Dex is virtually non toxic to aquatic invertebrates.

Based on studies of the joint action of glyphosate and the POEA surfactant used in Roundup (MON 0818), there was an additive toxic effect in two species of fish and in midge larvae and a less-than additive effect in a daphnid and copepod.

It appears that as the concentration of clay suspended in water increases the acute toxicity of Roundup to *Daphnia pulex* increases. In one 48-hour study the LC_{50} when there was no suspended clay was 7.9 (7.2-8.6) mg a.i./L while the LC_{50} when there was 50 mg/L of suspended clay was 3.2 (3.0-3.4) mg a.i./L. Another study found a decrease in the LC_{50} of *Ceriodaphnia dubia* from 5.38 mg a.e./L when there was no suspended clay to 0.59 mg a.e./L when there was 200mg/L of suspended clay. It is speculated that since daphnids are efficient filter feeders, they may intake and absorb greater quantities of Roundup and POEA attached to suspended clay particles.

Comparative sediment assays with *Ceriodaphnia dubia* of Roundup and Roundup Biactive found the latter formulation much less acutely toxic. The surfactant in Roundup Biactive evidently has a lesser affinity to sediment than POEA.

A study on the impact of glyphosate and Roundup on the acute toxicity of heavy metals to *Ceriodaphnia dubia* found that with most metals (Cd, Cu, Cr, Ni, Pb, Se, and Zn) there was an antagonistic effect.

A sublethal study on the effects of glyphosate to mosquito larvae found that pre-exposure to nonlethal concentrations resulted in a significant increase in cytochrome P450 levels after 72 hours, a positive outcome. Sublethal exposure of a freshwater annelid to glyphosate and Roundup Ultra resulted in oxidative stress.

Longer term toxicity studies indicate a duration-response relationship to glyphosate IPA salt in daphnids. A standard chronic bioassay study in *Daphnia magna* found a NOEC of 37 mg a.e./L and a corresponding LOEC of 74 mg a.e./L. However, a study on Roundup showed only a transient duration-response in *Daphnia pulex*. A study of glyphosate acid and the IPA salt of glyphosate in mussels showed no duration-response relationship, nor did a study of POEA or Roundup Ultramax. There was a relationship with Aqua Star. The effects of long-term exposure of the aquatic snail *Pseudosuccinea columella* to technical grade glyphosate found mixed effects, with egg-hatching being inhibited while egg-laying was enhanced, resulting in negligible effects on reproductive capacity.

Various field studies have found no adverse effects on aquatic invertebrates from applications of Rodeo or Roundup. Following applications of Roundup at rates of 2.2, 22, and 220 kg/ha to a forest pond mesocosms, there were no differences in survival rates of aquatic invertebrates. Following Roundup applications that resulted in concentrations of ~3 mg a.e./L in water, Relyea reported no effect on predatory insects or snails, although there were significant reductions in some species of dragonfly and backswimmers. An artificial stream mesocosms treated with Vision had an increase in periphyton populations.

Aquatic Plants - Acute toxicity is determined for algae and macrophytes, with EC_{50} endpoints determined for growth inhibition. Most EC_{50} values for algae are for 48-hour exposures and

for macrophytes are for 7-14 days. Duration-response relationships for macrophytes are not pronounced.

Sensitivity (EC_{50}) to glyphosate acid and glyphosate IPA varies widely between species of algae, from 2 to 600 mg a.e./L, spanning a factor of 260. The acid appears to be more toxic than the IPA salt by a factor of 2. Although there is variability in inter and intraspecies duration-response data for algae, it is apparent that for 2- and 4-day exposure durations there are substantial duration-response relationships.

The pattern of toxicity of glyphosate formulations to algae is similar to that for animals, with most glyphosate-surfactant formulations being more toxic than Rodeo without a surfactant and technical grade glyphosate. A Glyphos (IPA) formulation appears to be the most toxic, with EC_{50} values ranging from 0.12 to 0.68 mg a.e./L. The most toxic formulation (Glyphos IPA) and the least toxic (technical grade glyphosate) differ by a factor of 20.

A study exposed two species of algae to the POEA surfactant used in some formulations of Roundup for 96 hours. The EC_{50} values ranged from 3.35 to 4.1 mg/L. Tests of several surfactants, including MON 0818 (POEA), on giant salvinia, found no toxicity at concentration of 2500 mg/L. Optima was the only surfactant that enhanced the toxicity of glyphosate to salvinia.

Field studies have shown growth inhibition of algae by Roundup at concentrations of 44.4-69.7 mg/L. But growth stimulation has been observed at 10 mg a.e./L. Other studies have shown no or equivocal effects at application rates ranging from 0.4 to 2 lbs/acre. A study of the effect of Roundup on phytoplankton found a decrease in abundance on day one at concentrations of 6 and 12 mg a.e./L, but an increase after that up to the end of the experiment, on day eleven.

There is little data available on the toxicity of glyphosate acid and salts on aquatic macrophytes. EC_{50} values span a range of 10 to 200 mg a.e./L between species of macrophytes. In two species of duckweed, EC_{50} values for 7- to 10-day exposure to glyphosate acid ranged from 10 mg a.e./L for *Lemna gibba* to 47 mg a.e./L for *Lemna minor*. For glyphosate IPA exposure to *Lemna paucicostata*, the 7-day EC_{50} value was 42 mg a.e./L.

Two 14-day exposure studies (to glyphosate acid) are available for submerged macrophytes. In the watermilfoil study, the EC_{50} for reduction in root length was 1.56 mg a.e./L. For eelgrass, the NOAEC for growth inhibition was 170 mg a.e./L, with a stimulation of growth at 17 mg a.e./L.

Data on the toxicity of Rodeo, Roundup, and Glyphos on *Lemna* show 7-day EC_{50} values differing by only a factor of 2 for Roundup (3.4 mg a.e./L) and Glyphos (7.7 mg a.e./L). Based on 14-day EC_{50} values, Roundup and Rodeo differ by a factor of only 1.5 for watermilfoil and

1.7 for *Lemna gibba*. These differences are insubstantial. Other studies show only a modest duration-response relationship over 7- to 14-day exposures of *Lemna* to Roundup.

In a study of the influence of suspended clay (50 mg/L) on the toxicity of Roundup to macrophytes, a NOEC of 10 mg a.i./L was determined, as opposed to a NOEC of 2 mg a.i./L for water without clay. Evidently Roundup and the surfactant POEA bind with the clay particles, making them less available to macrophytes.

Aquatic Microorganisms - Most studies on aquatic microorganisms indicate that they are not very sensitive to glyphosate. Short-term (15-30 minutes) studies on the aquatic ciliate *Vibrio fischeri* determined EC_{50} values ranging from 17.5-44.2 mg a.e./L for glyphosate acid and 24.9-36.4 mg a.e./L for Roundup. The differences in toxicity between glyphosate acid and Roundup were slight.

A 48-hour bioassay of two other aquatic ciliates, *Euplotes vannu* (a freshwater protozoan) and *Tetrahymena pyriformis* (a marine protozoan) found large differences in sensitivity to glyphosate acid (10.1 mg a.e./L for the former and 648 mg a.e./L for the latter) and similar toxicity results for glyphosate IPA. Sensitivity to Roundup was similar (23.5 mg a.e./L for *Euplotes vannu* and 29.5 mg a.e./L for *Tetrahymena pyriformis*. The sensitivity of aquatic microorganisms to glyphosate acid appears to be similar to that of algae but less than algae for Roundup.

An aquatic mesocosm study of the effect of Roundup on cyanobacteria found an increase in abundance by a factor of up to 40, at concentrations of 6 and 12 mg a.e./L. Other bacteria were not substantially affected.

1.3.2.2.4.4 Hexazinone (Sources: FS WSM v. 6.00.10; SERA 2005)

As stated in SERA 2005, p. 4-1: *Most of the information on the toxicity of hexazinone to mammals as well as other species comes from unpublished bioassays submitted to the U.S. EPA for the registration of hexazinone. These studies as well as other studies submitted for registration are conducted using methods specified by the U.S. EPA (e.g., U.S. EPA/OPP 2005). While some studies may be conducted directly by the registrant, most toxicity studies are performed by commercial testing laboratories. All studies submitted for registration are independently reviewed by U.S. EPA*. All toxicity studies on mammals and other species that are cited in the Forest Service risk assessment for hexazinone were obtained and reviewed.

Mammals - Although the mode of action of hexazinone in mammals is unclear, the toxicity of hexazinone to mammals is relatively well-characterized in a large number of standard acute, subchronic, and chronic toxicity studies on mice, rats, rabbits, and dogs, an acute toxicity study in guinea pigs, and a number of standard skin sensitization studies in guinea pigs. (SERA 2005, p. 4-2)

The acute oral toxicity to mammals is classified by the U.S. EPA as Category III, the second lowest oral toxicity category. Assays for chronic toxicity indicate that dogs may be somewhat more sensitive than rats and mice. However, it is not clear whether patterns in sensitivity among different species are true differences or an artifact of differences in experimental design.

Hexazinone is considered to be slightly toxic to mammals, although it can cause severe, irreversible damage to the eyes of terrestrial organisms.

Birds - The available toxicity studies in birds include acute gavage studies, avian acute oral dietary studies, and two avian reproductive toxicity studies. Based on the U.S. EPA classification system, hexazinone is practically nontoxic to birds. Based on an acute gavage LD50 in quail of 2258 (1628-3130) mg/kg, birds appear to be somewhat less sensitive than mammals to hexazinone.

Reptiles and Amphibians (Terrestrial-Phase) - No acute or chronic toxicity studies were found for reptiles or terrestrial phase amphibians.

Terrestrial Invertebrates - Relatively little information is available on the toxicity of hexazinone to terrestrial invertebrates. The U.S. EPA assumes that herbicides are generally not directly toxic to insects, so only required one direct contact bioassay using honeybees. No clear dose response relationship was apparent and the highest observed mortality was only marginally significant.

In a field study conducted in northern California, hexazinone was applied to pine plantations at a rate of 2.7 lb a.i./acre (Busse et al., 2001). No significant differences were found between treated and control plots in the numbers of mites, spiders, beetles, or springtails (SERA 2005, p. 4-4). Hexazinone is considered to be *slightly to practically* nontoxic to invertebrates.

Terrestrial Plants (Macrophytes) - The toxicity to and mode of action of hexazinone are well characterized. Hexazinone is readily absorbed by plant roots and is readily translocated in most species. Differences in sensitivity to hexazinone among different types of plants is related to differences in absorption and rates of metabolism. The metabolites of hexazinone are much less toxic then hexazinone itself.

Based on standard pre-emergence and post-emergence bioassays in sensitive species, soil treatments are more toxic than direct spray treatments. Hexazinone has relatively little effect on seed germination, with Pronone 10 perhaps having more effect than Velpar L.

Many field studies on terrestrial vegetation are available. These studies are typically conducted at or above the recommended application rates and tend to focus on efficacy rather than unintended adverse effects. Hexazinone is used effectively in management of pine stands to control hardwoods and shrubs, as it causes only minor mortality in pines.

Terrestrial Microorganisms - Standard laboratory culture bioassays indicate that hexazinone can inhibit microbial growth at both low and relatively high concentrations, depending on the species. However, field studies have demonstrated no effects on mixed fungal and bacterial populations following application rates of up to 8 kg/ha (about 7 lbs/acre).

Fish - The U.S. EPA classifies technical grade hexazinone as *practically nontoxic* to fish. This is based specifically on acute LC_{50} values reported for rainbow trout (>320 mg/L), fathead minnow (274 mg/L) and bluegill sunfish (>370 mg/L and 505 mg/L). It also classifies Velpar L as *practically nontoxic* to fish, with acute LC_{50} values of >1000 mg/L in bluegills and >585.6 mg/L in trout.

Although the U.S. EPA does not discuss studies on Pronone, Pronone 10G appears to be less toxic than Velpar L and both Velpar L and Pronone 10G are less toxic than technical grade hexazinone. This is true even when comparisons are made on an mg a.i./L basis. The inerts in both Velpar L and Pronone 10G appear to lower the toxicity of hexazinone to fish. The Pronone 10G carrier and the Velpar L carrier (mainly ethanol) are essentially nontoxic to fish.

The only longer term toxicity study of hexazinone in fish is an egg-and-fry study that defined a clear NOEC of 17 mg/L and an LOEC of 35.5 mg/L. Consistent with this finding is a 4-week assay for bioconcentration in bluegill sunfish that found no signs of toxicity at concentrations of 0.1 or 1 mg/L.

Amphibians (Aquatic Phase) - Very little information is available on the toxicity of hexazinone to amphibians. The U.S. EPA Pesticide Effects Determinations for the risks of hexazinone use to the federally threatened California red-legged frog (U.S. EPA 2008c) did not list any toxicity studies using aquatic-phase amphibians as study organisms.

In one study, a hexazinone concentration of 100 mg/L over an 8-day exposure period was associated with transient reduced avoidance behavior in newly hatched tadpoles. These exposure levels had no effect on hatching success.

Aquatic Invertebrates - Toxicity information is limited to studies submitted to the U.S. EPA for pesticide registration. Based on acute toxicity studies on *Daphnia magna*, the 48-hour LC_{50} for technical grade hexazinone was 151.6 (125.2-172.8) mg/L and for Velpar L it was 110 (83-130) mg a.i./L. The U.S. EPA classifies both hexazinone and Velpar L as *practically nontoxic* to freshwater invertebrates. There is no indication that the inerts in Velpar L reduce the toxicity of hexazinone to daphnids.

The U.S. EPA classifies hexazinone as moderately toxic to saltwater crustaceans, based on the sensitivity of grass shrimp, which appear to be about equally sensitive as daphnids to hexazinone (48-hour LC_{50} value of 94 [50-176] mg/L). The fiddler crab is much less sensitive, with a NOEC for mortality of over 1000 mg/L. The only data available on mollusks, for embryos

of the eastern oyster, indicate a NOEC of 320 mg/L, substantially above the LC_{50} values for small crustaceans.

Although there were reporting deficiencies in the only available reproduction studies, in *Daphnia magna*, the U.S. EPA did accept those studies. The NOEC discussed by the U.S. EPA is 29 mg/L, however a NOEC of 10 mg/L may be a more appropriate for this risk assessment.

As stated in SERA 2005 (p. 4-9):

> *Additional information on the effects of hexazinone on aquatic invertebrates is also available in field or field simulation assays (Appendix 10). In one such study, 13 species of stream macroinvertebrates were exposed to very high concentrations of hexazinone, 70 mg/L to 80 mg/L, for one hour in an artificial stream followed by a 48-hour observation period. The most sensitive species were two species of Ephemeroptera, an Isonychia sp and Epeorus vitrea, both of which exhibited 14% mortality. Mortality in all other species ranged from 0% to 4% (Kreutzweiser et al., 1992). In a subsequent study (Kreutzweiser et al., 1995), no effects were noted on invertebrate drift in five stream channels over a 14 day period of observation after 12 hour exposures to hexazinone at concentrations that ranged from 3.1 to 4.1 mg/L. At the end of the 14-day observation period, no significant pair-wise differences between treated and control channels were noted for 14 taxa of macroinvertebrates. Overall, however, there was a significant increase in abundance of invertebrate taxa in treated versus control channels (Kreutzweiser et al., 1995). In a similarly designed study, no effects on stream invertebrates were observed after the application of Velpar L at a level that resulted in hexazinone levels of 0.145-0.432 mg/L over a 24-hour exposure period (Schneider et al., 1995). In addition, Mayack et al., (1982) reported no effects on stream macroinvertebrates at water concentrations of 0.008 mg/L to 0.044 mg/L. These concentrations were the result of the application of hexazinone pellets (formulation not specified but consistent with Pronone 10G) at a rate of 16.8 kg/ha in four small watersheds located in mixed hardwood-pine stands. One additional watershed served as an untreated control.*

Aquatic Plants - Based on the standard bioassays submitted to the U.S. EPA for registration and published studies, there are relatively substantial differences in sensitivity to hexazinone among species of freshwater algae. The differences span a factor of approximately 24 based on the EC_{25} values and 38 based on the NOEC values, with *Selenastrum capricornutum* (a freshwater green alga) being the most sensitive (5-day EC_{50} = 0.0068 [0.0063-0.0072] mg/L; NOEC of 0.004 mg/L) and the least sensitive species being *Anabaena flos-aquae* (a freshwater blue-green alga) (5-day EC_{25} = 0.16 [0.02-0.24] mg/L; NOEC 0.15 mg/L).

In one study on the toxicity of hexazinone to macrophytes (i.e., duckweed - *Lemna* sp.), adverse effects (a reduction in frond count and reduced biomass) were noted at the lowest concentration tested (0.026 mg/L), with exposures over a 14-day period. The EC_{25} for the most sensitive endpoint (frond count) was estimated at 0.027 mg/L. In another study the NOEC is estimated to be 0.012 mg/L.

The carriers and/or inerts in formulations of Velpar L do not appear to reduce the toxicity of hexazinone to aquatic plants.

It appears that in two of the field trials (Kreutzweiser et al 1995 and Schneider et al., 1995) described under *Aquatic Invertebrates*, reductions in algal photosynthesis were temporary and recovery was rapid following clearing of hexazinone from stream channels.

1.3.2.2.4.5 Imazapyr (Sources: FS WSM v. 6.00.10; SERA 2011c; U.S. EPA 2006d)

Mammals - Although acute, subchronic, and chronic toxicity studies on imazapyr do not demonstrate adverse effects that are unequivocally attributable to exposure, *this uncertainty or a lack of knowledge has a relatively minor impact on this risk assessment, because the available toxicity studies are relatively complete—chronic studies in three mammalian species (dogs, rats, and mice) and several reproduction studies in two mammalian species (rats and rabbits)—and indicate that imazapyr is not likely to be associated with adverse effects at relatively high-dose levels* (SERA 2011c, p. 54). Imazapyr is considered *slightly toxic* to mammals.

Birds - The available avian studies on imazapyr (acute gavage, acute dietary, and reproduction studies in both bobwhite quail and mallard ducks), all of which were conducted up to limit doses, do not report any signs of toxicity. Imazapyr is considered *slightly toxic* to birds.

Reptiles and Amphibians (Terrestrial-Phase) - No acute or chronic toxicity studies were found in open literature or in studies submitted to the U.S. EPA for reptiles or terrestrial-phase amphibians.

Terrestrial Invertebrates - Two studies (oral and contact) on honeybees suggest that imazapyr is *practically nontoxic* to honeybees. Whether this is true for all the diverse species of invertebrates found in the environment is unknown.

Terrestrial Plants (Macrophytes) - After foliar application, imazapyr is transported via the phloem and inhibits acetolactate synthase, an enzyme that catalyzes the biosynthesis of three branched-chain amino acids, which are essential for protein synthesis and plant growth. Imazapyr does not appear to be extensively metabolized by plants.

Imazapyr has been shown to translocate to plant roots and exude from the roots into the surrounding soil, posing a risk to nearby plants (SERA 2011c, p. 58), in a process known as

allelopathy. However, given that imazapyr moves relatively rapidly in soil, the potential for allelopathic effects may not have a practical or substantial impact on potential risk to non-target plants.

Imazapyr formulations are labeled for both post-emergence and pre-emergence control of both broadleaf vegetation (dicots) and grasses (monocots). Based on standard toxicity studies of foliar applications of technical grade imazapyr, dicots appear to be substantially more sensitive than monocots in assays for both vegetative vigor and seedling emergence

Terrestrial Microorganisms - What little information is available on the toxicity of imazapyr to terrestrial microorganisms indicates that it is highly species specific, with variations in sensitivity of up to a factor of 100. It is not clear whether these effects, which are based on laboratory cell culture studies at very high concentrations of imazapyr, would occur in field populations of microorganisms.

As stated in SERA 2011c, p 61:

> *In peak soil concentrations, imazapyr inhibited cellulose decomposition and carboxymethyl cellulase activity when applied at 0.25 to 1 kg/ha, equivalent to about 0.22 to 0.9 lb/acre, to a predominantly peat soil (Ismail and Wong 1994). These investigators speculate that "the reduction in cellulose degradation is likely to be only a temporary effect" (Ismail and Wong 1994, p. 122) and that the activity of imazapyr on terrestrial microorganisms may decline as the herbicide is adsorbed to soil and thereby becomes less bioavailable to microorganisms. On the other hand, imazapyr may persist in soil for a prolonged period of time, particularly in relatively arid regions, and will not bind tightly to alkaline soils with low organic matter. Thus, in at least some areas, a potential for longer-term effects on soil microorganisms seems possible. This effect, however, has not been demonstrated in field studies. In a greenhouse study, Busse et al., (2004) noted no effects on the infectivity of mycorrhizal fungi to pine seedlings following application of imazapyr at rates of 0.82 to 1.6 lb a.e./acre (i.e.,, rates that caused clear signs of toxicity in the pine seedlings).*

Fish - The U.S. EPA classifies both imazapyr acid and isopropylamine salt as *practically non-toxic* to fish. One commonly used formulation of imazapyr, Arsenal Herbicide (27.8% a.i, 22.6% a.e. isopropylamine salt and 72.2% inerts, which include an unspecified solvent), appears to be substantially more toxic to trout relative to imazapyr and isopropylamine salt of imazapyr. This is evidently due to one or more of the inerts in the formulation. The 96-hour LC_{50} of Arsenal Herbicide is about 41 mg a.e./L in bluegills and 21 mg a.e./L in trout.

Longer-term toxicity studies have been done on imazapyr but not on its formulations. This is problematic, as the acute NOAEC of the isopropylamine salt of imazapyr in rainbow trout is 110 mg a.e./L while for the Arsenal Herbicide formulation it is 10.4 mg a.e./L. The acute

NOAEC for the Arsenal Herbicide formulation in rainbow trout is below the longer-term NOAEC for imazapyr acid by a factor of about 4.

The longer-term toxicity of imazapyr acid to fathead minnows has been assayed in an early life-stage study and a full life cycle study. Neither study detected adverse effects at concentrations of up to about 120 mg a.e./L. Rainbow trout appear to be the most sensitive species, as at a concentration of 92.4 mg a.e./L in an early life-stage study there was a reduction in hatch and fry survival, judged by the researcher as a "…*nearly significant effect on hatching.*" No effects, however, were noted at a concentration of 43.1 mg a.e./L. The U.S. EPA determined that the 92.4 mg a.e./L concentration is a LOAEC (lowest observed adverse effect concentration) rather than a NOAEC.

Amphibians (Aquatic Phase) - No acute or chronic toxicity testing on aquatic-phase amphibians was found for imazapyr.

The U.S. EPA Pesticide Effects Determinations for the risks of imazapyr use to the federally threatened California red-legged frog (U.S. EPA 2007b) did not list any toxicity studies using aquatic-phase amphibians as study organisms.

Aquatic Invertebrates - The U.S. EPA classifies both imazapyr acid and isopropylamine salt of imazapyr as *practically non-toxic* to *Daphnia magna* and saltwater invertebrates (oysters and pink shrimp). The Arsenal Herbicide formulation of imazapyr is more toxic than either imazapyr acid or the isopropylamine salt. In *Daphnia magna* the EC_{50} for Arsenal Herbicide is 79 mg a.e./L while the EC_{50} for isopropylamine salt of imazapyr is 614 mg a.e./L, lower by a factor of about 8.

The only longer-term toxicity study on imazapyr, a standard life cycle study in *Daphnia magna*, resulted in no effects at concentrations of up to 97.1 mg a.e./L. This chronic NOAEC is above the acute NOAEC of 59.3 mg a.e./L for Arsenal Herbicide.

As stated in SERA 2011c (p. 64):

> *Concern for longer-term effects of exposures of aquatic invertebrates is at least somewhat diminished by the mesocosm study by Fowlkes et al., (2003). As summarized in Appendix 5 (Table 4), the study involved exposures of mixed macroinvertebrates to mesocosms treated with Arsenal Applicators Concentrate at concentrations of 0.184, 1.84, or 18.4 mg a.e./L. No impacts were noted on species richness or abundance after a 2-week exposure period, which is comparable to the exposure period in chronic daphnid studies. The apparent NOAEC of 18.4 mg a.e./L is consistent with the acute NOAEC of 59.3 mg a.e./L for Arsenal Herbicide (Forbis et al., 1984b) as well as the chronic NOAEC of 97.1 mg a.e./L in daphnids (Manning 1989c).*

Aquatic Plants - Based on the geometric means of the EC_{50} values in algae (37.2 mg a.e./L) and aquatic macrophytes (0.023 mg a.e./L), imazapyr is more toxic to aquatic macrophytes than to algae by a factor of over 1600. The differences in 7-day EC_{50} values for imazapyr acid among different species of algae span a factor of about 8, ranging from 12.2 to 92 mg a.e./L. The isopropylamine salt of imazapyr (EC_{50} = 11.5 mg a.e./L) is more toxic than imazapyr acid (EC_{50} = 71 mg a.e./L) by a factor of about 6.

Three standard bioassays in aquatic macrophytes (duckweed [*Lemna gibba*] and water milfoil [*Myriophyllum sibiricum*]) suggest little variability in the sensitivity of aquatic macrophytes to imazapyr acid and Arsenal (isopropylamine salt of imazapyr). These bioassays resulted in similar EC_{50} values for growth inhibition, ranging from 0.018 mg a.e./L for the salt of imazapyr in duckweed to 0.029 mg a.e./L for the Arsenal formulation in water milfoil. However, efficacy studies suggest variability in the tolerance of species to imazapyr.

1.3.2.2.4.6 NP9E (Sources: FS WS ver. 2.02; USDA/FS 2003b; U.S. EPA 2010e)

NP (nonylphenol) is one of the parent chemicals of NPE (nonylphenolpolyethyoxylate), a chemical group that is part of many herbicide surfactants. NPs are used widely in the U.S. About 80% of this use is for industrial and institutional surfactants and liquid detergents (USDA/FS 2003b). As stated in U.S. EPA.2010e:

> *NP and certain oligomeric NPEs are highly toxic to aquatic organisms, are moderately bioaccumulative in mollusks, are persistent in the aquatic environment, and accumulate in soils and sediments (EPA, 2005). (ibid, p. 1)*

Many herbicide surfactants used by the USFS, analyzed in USDA/FS 2003b (p. v), and likely to be used under the VTP and alternatives, contain from 20-80% NPE. The chemical group of NPEs that are used in herbicide surfactants, NP9E, are of relative low acute toxicity to fish, as are the metabolites (the NPECs) likely to be found in water. As stated in USDA/FS 2003b (p. 43), “The NPECs would appear to be slightly more acutely toxic to fish than NP9E. NP is an order of magnitude more toxic to fish than the NP9E or NPECs.” NP9E surfactants are generally mixed with herbicides and water carriers at dilution rates of 0.25% to 2.5% (USDA/FS 2003b, p. 1). The percentage of NP9E in a tank mix would therefore range from 0.0005% to 0.02%.

Mammals - NP9E is classified by the U.S EPA as *slightly toxic to practically non-toxic* to mammals (toxicity category III or IV). Although the acute toxicity of NP, the parent compound of NP9E, is somewhat higher, it is also classified in category III or IV.

NP9E is minimally to severely irritating to rabbit skin and moderately to severely irritating to rabbit eyes. It can cause severe, irreversible eye damage to the eyes of terrestrial organisms.

The liver and kidney are the organs most likely to be affected by chronic and subchronic exposures to NPE and NP. These compounds have been determined to be weakly estrogenic in both *in vitro* and *in vivo* tests involving aquatic and terrestrial organisms. Non-reproductive effects appear to be the more sensitive endpoint.

No evidence of carcinogenicity was reported in 2-year chronic oral toxicity studies of NP9E with rats and dogs. However, ethylene oxide and 1, 4-dioxane, sometimes found as impurities in NP9E at low levels, are classified as carcinogens. Ethylene oxide is also a mutagen.

NP9E appears to be rapidly metabolized and excreted, based on one study. It does not appear to be immunotoxic or neurotoxic at doses considered protective of kidney or liver effects.

Birds - Published literature has no data on the effects of NP or NPEs to birds.

Reptiles and Amphibians (Terrestrial-Phase) - No acute or chronic toxicity studies were found for reptiles.

Terrestrial Invertebrates - The only study found in the literature on the effects of NPE on terrestrial insects (honeybees) does not provide sufficient data to characterize the risk to terrestrial invertebrates.

Terrestrial Plants (Macrophytes) - There is no data in the published literature on the toxicity of NPEs to plants. Since NP9E surfactants would be mixed with herbicides, any potential toxic effects would be masked by the effects of the herbicides.

There is only limited data on the toxicity of NP to plants. It appears that NP is quickly mineralized by soil microorganisms, uptake of NP from soil is slow to non-existent, and there is little to no toxic effect on plants.

Terrestrial Microorganisms - There is no toxicity of NPE and NP to soil microorganisms at application rates of NP in soil up to 250 mg/kg.

Fish - As stated in U.S. EPA.2010e:

> *The available acute and chronic toxicity data of NP to aquatic organisms indicates NP is highly toxic to fish, aquatic invertebrates, and aquatic plants. (ibid, p. 4)*
>
> *However, For NPEs, toxicity to aquatic organisms tends to decrease with increasing degree of ethoxylation. For example, acute toxicity to killifish was 1.4 mg/L, 3 mg/L, 5.4 mg/L, 12 mg/L and 110 mg/L for NP, NP1EO (i.e., NPE with one ethoxylate group), NP6.4EO (i.e., NPE mixture with an average of 6.4 ethoxylate groups), NP9EO and NP16.6EO, respectively (Canada, 2002). Environment Canada, based on a comprehensive analysis of available toxicity data for NP and NPEs, developed Toxic Equivalency Factors (TEFs) for NP and NPEs, as follows: NP =1; NP1EO and NP2EO*

> *=0.5 (i.e., half as toxic as NP); NP3EO to NP8EO also = 0.5 (a conservative estimate because of inadequate data); NP9EO and greater = 0.005 (i.e., 100 times less toxic than NP) (Canada 2002). (ibid)*

As stated above, acute toxicity varies with the degree of ethoxylaytion. For NP8E, 96-hour LC_{50} values for juvenile rainbow trout range from 4,100 to 5,400 ppb. For NP8.9E, 48-hour LC_{50} values for the Japanese medaka (*Oryzias latipes*) range from 11,200 to 14,000 ppb. For NP9E, 96-hour LC_{50} values for fathead minnows (*Pimephales promelas*) range from 4,000 to 6,600 ppb. These acute toxicity values for NP8-9E are at least 1 order of magnitude less than NP. For NP10E, 96-hour LC_{50} values for adult cod (*Gadus moorhua*) and flounder (*Pleuronectes flesus*) range from 2,500 to 6,000 ppb, depending upon water temperature.

Most 96-hour LC_{50} values for acute toxicity of NP to tested fish species range from 100 to 460 ppb. The lowest tested 96-hour LC_{50} was for the salt-water species flounder (*Pleuronectes americanus*), with a value of 17 ppb. Other species tested were the fathead minnow (128 to 320 ppb), rainbow trout (190 to 270 ppb), Atlantic salmon (130 to 900 ppb), and sheepshead minnow (460 ppb). In the Japanese medaka, the 48-hour LC_{50} for NP is 1,400 ppb.

A study comparing the acute toxicity of NP in surrogate fish species against threatened or endangered species found that the Apache trout, greenback cutthroat trout, and Lahontan trout were all similar to the rainbow trout surrogate (96-hour LC_{50} values of 150 to 180 ppb as compared to the rainbow trout 190 ppb). Correlations were less good between warm water threatened or endangered fish (bonytail chub, Colorado pikeminnow, and razorback sucker) and the fathead minnow surrogate (96-hour LC_{50} values of 170 to 290 ppb as compared to 270 ppb). The authors of the study concluded that a safety factor of 2X should be sufficient to provide a conservative estimate for listed cold and warm freshwater fish species.

The acute toxicity of the environmental metabolite NP1EC to fathead minnows indicates a 96-hour EC_{50} of 2,000 ppb while in Japanese medaka or killifish, a 48-hour EC_{50} for NP1EC was determined to be 9,600 ppb and for NP2EC, 8,900 ppb.

As stated in USDA/FS 2003b (p. 43):

> *It would appear that in terms of acute toxicity to fish, NP9E is of relatively low acute toxicity, as are the likely environmental metabolites that would be found in water (the NPECs). The NPECs would appear to be slightly more acutely toxic to fish than NP9E. NP is an order of magnitude more toxic to fish than the NP9E or NPECs.*

There is little data on NPEs regarding sub-chronic and chronic toxicity. In a 7-day study of NP9E on fathead minnows, a NOEC of 1,000 ppb was determined, based on growth. In a 42-day study where fathead minnows were exposed to NP9E at rates up to 5.5 ppb, there was no mortality and no effects to secondary sex characteristics. A 14-day study of rainbow trout exposed to NP8E resulted in a LC_{50} of 4,250 ppb. Sublethal effects (impaired locomotor

activity and breathing rate) from exposure to NP10E in codfish (*Gadus morhua*) have been demonstrated at rates of >1 mg/L (1,000 ppb), with the effects remaining reversible over a long period of time. This exposure rate was three orders of magnitude higher than needed to elicit the same response from NP (2 µg/L or 2 ppb) in the same species.

For NP, the subchronic NOEC varies with species, with lab-determined 28- to 90-day values ranging from 1-23 ppb.

Exposure to the environmental metabolite NP1EC at rates up to 50 ppb for 35 days after hatch in rainbow trout had no dose-dependent effects on growth or ovosomatic index, as measured after 108 or 466 days. In an unpublished study with fathead minnows, a NOEC of 1000 ppb was established for NP1EC.

Further, as stated in USDA/FS 2003b (p. 45):

Bioconcentration potential of the short-chain ethoxylates (NP, NP1E, NP2E) in freshwater fish and other aquatic biota appears to be low to moderate ranging up to about 740 (Ahel et al 1993; Liber et al 1999b; Snyder et al 2001; US EPA 1996). Little data exists on the bioconcentration of longer chain NPEs, but based on their structure they are not expected to bioaccumulate (Environment Canada 2001a, Servos 1999)

Amphibians (Aquatic Phase) - No acute or chronic toxicity studies on adult amphibians were found for NP9E. Acute toxicity studies on amphibians were found for NP/NPE. These studies are generally of a limited nature and are limited to frog or toad embryos or tadpoles.

Two studies on NP8E tested embryos of three species and tadpoles of six species. In the embryo study, 96- to 140- hour LC_{50} values ranged from 3.9 to 9.2 ppm, comparable to values for freshwater fish. Developmental EC_{50} values ranged from 2.8 to 8.8 ppm. The minimum NP8E concentration inhibiting growth (an LOEC) ranged from 1 to 4 ppm. In the tadpole study, mild narcosis EC_{50} values ranged from 2.3 to <10.6 ppm. Water temperature increases did not affect EC_{50} values, but reduced dissolved oxygen in water reduced EC_{50} values by about half, as compared to normal levels of oxygen. Tadpoles recovered from narcosis during the life of the test.

For NP, acute toxicity 96-hour to 14-day LC_{50} values for amphibians ranged from 75 to 120 ppb in water and 10 to 30 day LC_{50} values of 260 mg/kg for dosed sediments. When *Xenopus laevis* was exposed to NP, there was a 14-day NOEC for tail resorption of 25 ppb. NP exposure for 12 weeks to *X. laevis* tadpoles at 22 ppb caused a significant increase in the percentage of female frogs, but this effect was not seen at 2.2 ppb.

Aquatic Invertebrates - NP9E toxicity to aquatic invertebrates is less than for NP, demonstrating the same relationship as is found in fish and amphibians. The 48-hour EC_{50} for *Daphnia magna*, is 14,000. In two subchronic studies, a *Daphnia* 7-day NOEC (growth) value

of 10,000 ppb was determined. For mysid shrimp, the 48-hour LC_{50} value ranges from 900 – 2,000 ppb.

After exposure to NP10E, sublethal effects to mussels, cockles, and barnacles were seen at 2-5 mg/L (ppm) while effects to locomotion of a decapod, hermit crab and shore crab were seen at 20-40 mg/L (ppm).

To determine the toxic effects to invertebrates of a tank mix of X-77, an NPE-based surfactant, mixed with the Rodeo formulation of glyphosate, in-lab toxicity tests were done as well as field applications to freshwater wetlands. For four species of invertebrates, 48- and 96-hour LC_{50} values for X-77 ranged from 2.0 to 14.1 mg/L, about two orders of magnitude greater than the acute toxicity of Rodeo alone. However, mortality patterns were similar between the treated and untreated wetlands, indicating a lack of acute toxicity of the tank mix at the application rate. But potential chronic effects of such applications are unknown.

One study of the exposure of *Daphnia* to the metabolites NP2E and NP2EC derived a 48-hour LC_{50} of 115 to 198 ppb for NP2E and 770 to 1,295 ppb for NP2EC.

Tests of NP on various species of freshwater and marine invertebrates have resulted in 96-hour LC_{50} values ranging from about 20 to about 775 ppb. For *Daphnia*, the LC_{50} for NP and NP2E are similar.

For mysid shrimp after exposure to NP, the 28-day chronic NOEC (growth) is 4 ppb. *Daphnia* have a slightly higher 21-day NOEC (reproduction) of 24 and 116 ppb while the NOEC (embryotoxicity) occurs at 44 ppb. The marine copepod *Tisbe battagliai* had a 53-day NOEC of 20 ppb. In littoral enclosure studies, no effects were seen on macroinvertebrates at levels of NP up to 23 ppb and no effects to zooplankton at levels of 5 ppb.

In a study of NP applied to outdoor microcosms at average concentrations of 5, 23, 76, and 243 µg/L, only the highest concentration caused significant declines in zooplankton abundance and insect emergence, although there were sensitive taxa affected at 23 µg/L. However, in terms of abundance, the overall zooplankton community structure was relatively unaffected.

Aquatic Plants - For NP9E exposure to green algae, the NOEC (growth) value is 8,000 ppb and the 96-hour EC_{50} (growth) value is 12,000 ppb.

For NP, a marine alga has been the most sensitive aquatic plant species tested, with a 96-hour EC_{50} (growth) of 27 ppb and a NOEC of 10 ppb. Green algae and duckweed have 96-hour NOEC (growth) values ranging from 90 to 900 ppb. Duckweed seems to be more tolerant than the algae. In a littoral enclosure study there were no effects to aquatic macrophytes (*Chara* and *Potamogeton*) while there was a small increase in periphyton biomass at the highest mean average concentration of 243 µg/L over 20 days.

1.3.2.2.4.7 Sulfometuron methyl (Sources: FS WSM v. 6.00.10; SERA 2004c; U.S. EPA 2008a, 2009g)

Mammals - Sulfometuron methyl has low acute and chronic oral toxicity to mammals. Although there is relatively little information on the effects in non-target wildlife species, it is reasonable to assume that the effects will mirror those in experimental mammals.

Birds - Based on acute exposure studies, birds appear to be somewhat less sensitive than experimental mammals to the toxic effects of sulfometuron methyl. No chronic exposure studies were identified in the available literature.

Reptiles and Amphibians (Terrestrial-Phase) - No acute or chronic toxicity studies were found for reptiles.

Terrestrial Invertebrates - Sulfometuron methyl is *practically nontoxic* to bees. It is not clear from available data whether this low level of toxicity is true for other invertebrates.

Terrestrial Plants (Macrophytes) - Non-target plants are sensitive to sulfometuron methyl. Based on pre-emergence applications, rape, tomato, sorghum, wheat, and corn were the most sensitive species (onion, pea, cucumber, and soybean were the least sensitive). Based on post-emergence applications, corn was the most sensitive species. Adverse effects were observed in most broadleaved plants and grasses tested. Field reports indicate “substantial and prolonged damage to crops or ornamentals after the application of sulfometuron methyl in both an arid region, presumably due to the transport of soil contaminated with sulfometuron methyl by wind, and in a region with heavy rainfall, presumably due to the wash-off of sulfometuron methyl contaminated soil” (SERA 2004c, p. 4-5).

Terrestrial Microorganisms - Sulfometuron methyl appears to inhibit the growth of several soil microorganisms at low concentrations.

Fish - Studies on acute toxic effects of sulfometuron methyl in fish suggest that effects are not likely to be observed at concentrations less than or equal to 150 mg/L (SERA 2004c). Available acute toxicity data for freshwater fish and invertebrates indicate that sulfometuron methyl is *practically non-toxic* on an acute exposure basis, with all EC_{50}s / LC_{50}s >100 mg/L. For marine and estuarine fish, available acute toxicity data indicate that sulfometuron is at most *slightly toxic* on an acute exposure basis (EC_{50}s / LC_{50}s range from >38 to >45 mg a.i./L) (U.S. EPA 2008a).

Based on 30-day chronic exposure assays of fathead minnow embryo hatch, larval survival, or larval growth, no adverse effects would be expected at concentrations of up to 1.17 mg/L.

Amphibians (Aquatic Phase) - In acute and chronic exposure studies, the most sensitive aquatic species tested appears to be the African clawed frog, with exposure to sulfometuron

methyl producing alterations in limb development, organogensis, and metamorphosis, with the lowest NOEL of 0.001 mg/L for metamorphosis.

Aquatic Invertebrates - Based on acute bioassays in daphnids, crayfish, and field-collected species of other aquatic invertebrates, sulfometuron methyl appears to be relatively non-toxic to aquatic invertebrates. As stated in SERA 2004c (p. 4-8):

> *One daphnid reproduction study noted a decrease in the number of neonates at 24 mg/L but not at 97 mg/L or any of the lower concentrations tested. The authors report the NOEL as 6.1 mg a.i./L. Although the effect observed at 24 mg/L may have been a random variation, it is treated as an LOAEL for the purpose of this risk assessment. While this approach may be regarded as conservative, in the absence of additional studies regarding reproductive effects in aquatic invertebrates, the approach seems prudent.*

Available acute toxicity data for invertebrates indicate that sulfometuron methyl is *practically non-toxic* on an acute exposure basis, with all EC_{50}s / LC_{50}s >100 mg/L. For marine and estuarine invertebrates, available acute toxicity data indicate that sulfometuron is at most *slightly toxic* on an acute exposure basis (EC_{50}s / LC_{50}s range from >38 to >45 mg ai/L) (U.S. EPA 2008a).

Aquatic Plants - As might be expected, aquatic plants are much more sensitive than aquatic animals to the effects of sulfometuron methyl, although the effects on aquatic plants have not been extensively studied. EC_{50} values for growth inhibition range from 0.462 g/L in duckweed to 10 g/L in hydilla. EC_{50} values in algae for growth inhibition range from 4.6 g/L in *Selenastrum capricornutum* to > 370 g/L (the NOEC value) in *Navicula pelliculosa*. Macrophytes appear to be generally more sensitive than unicellular algae.

As stated in SERA 2004c (p. 4-2):

> *There are no published or unpublished data regarding the toxicity of sulfometuron methyl to aquatic bacteria or fungi. By analogy to the effects on terrestrial bacteria and aquatic algae, it seems plausible that aquatic bacteria and fungi will be sensitive to the effects of sulfometuron methyl.*

1.3.2.2.4.8 Triclopyr (Sources: FS WSM v. 6.00.10; SERA 2011d)

The hazard identification for nontarget organisms is concerned with triclopyr acid, triclopyr TEA, triclopyr BEE, and 3,5,6-trichloro-2-pyridinol (TCP) a metabolite of triclopyr. In terrestrial animals, triclopyr TEA and triclopyr BEE appear to be bioequivalent to triclopyr. Few systematic differences in species sensitivity in terrestrial animals are apparent. In aquatic organisms, triclopyr BEE is much more toxic than triclopyr TEA or triclopyr acid.

Mammals - Triclopyr is only *slightly toxic* to mammals. Triclopyr is a weak acid and is therefore likely to be more toxic to dogs than to most other mammals. Based on very clear and consistent patterns in both subchronic and chronic studies involving dietary exposures, sensitivity to triclopyr is greater in larger mammals.

The primary target tissue for triclopyr toxicity in mammals is the kidney. Triclopyr causes developmental effects only at doses that cause maternal toxicity. Triclopyr will not accumulate in mammals on repeated dosing. Available studies on wildlife do not report adverse effects attributable to the toxicity of triclopyr.

Triclopyr TEA can cause severe, irreversible eye damage to the eyes of terrestrial organisms.

Birds - Based on studies in mallard ducks and bobwhite quail, triclopyr is only slightly toxic to birds (triclopyr acid *practically non-toxic* to *slightly toxic* and triclopyr TEA and BEE [Garlon 4] *practically non-toxic*). In ducks, the acute oral toxicity of triclopyr acid and triclopyr TEA are substantially similar. In quail, the toxicity of triclopyr BEE is lower than the toxicity of triclopyr acid and triclopyr TEA to ducks by a factor of about 2.5.

In two field studies using triclopyr applications in the range of application rates that may potentially be used under the PEIR or alternatives, no adverse effects were observed in birds.

TCP is less toxic to birds than triclopyr BEE, triclopyr TEA, and triclopyr acid (SERA 2011d, p. 90).

Reptiles and Amphibians (Terrestrial-Phase) - The toxicity of triclopyr or TCP to reptiles or terrestrial phase amphibians is not included in "either the recent EPA ecological risk assessment on triclopyr (U.S. EPA/OPP 2009a) or in the database on amphibian and reptile toxicity data maintained by the Canadian National Wildlife Research Centre (Pauli et al., 2000)" (SERA 2011d, p. 90).

No acute or chronic toxicity studies were found for reptiles.

Terrestrial Invertebrates - Triclopyr acid and triclopyr TEA are *practically nontoxic* to bees while triclopyr BEE is *slightly more toxic*.

One study on earthworms suggests that triclopyr TEA may be moderately toxic to earthworms relative to triclopyr acid. However, the toxic concentrations in this study were far higher than soil concentrations of triclopyr that would occur in the environment. A chronic effects study indicated no adverse effects from exposure to Garlon 4 on earthworm reproduction or growth. A field study of the effects of Garlon 3A to earthworms and other invertebrates resulted in no significant reduction in mixed earthworm populations, mites, springtails, or ants in turf and soil core samples.

A series of field studies suggest that effects to invertebrates were attributable to changes in vegetation rather than direct toxic effects of triclopyr.

Terrestrial Plants (Macrophytes) - Triclopyr BEE is bioequivalent to triclopyr TEA in foliar applications to terrestrial plants. With foliar applications, triclopyr is effective for controlling dicots and relatively ineffective in controlling monocots. Pines tend to be tolerant to triclopyr exposures after fall dormancy but more sensitive during the spring and summer.

In seedling emergence studies, triclopyr BEE is much more toxic than triclopyr TEA, at least in some species, such as alfalfa.

One study suggests that some bryophytes and lichens may be sensitive to long-term effects after triclopyr exposure, which raises a concern that exposure to substantial triclopyr drift may have long- term impacts on bryophyte and lichen communities. Since triclopyr BEE is much more volatile than triclopyr TEA, it can cause damage to nontarget plants through vapor transport. Although none of the field studies involving triclopyr BEE document damage to nontarget plant species through volatilization, anecdotal reports from the Forest Service suggest that volatilization of triclopyr may damage nontarget plants if triclopyr BEE is applied under a poorly ventilated canopy and high temperatures.

Terrestrial Microorganisms - Diverse studies on the toxicity of triclopyr to terrestrial microorganisms suggest that it is not likely to have an impact on soil microorganisms.

Fish - Based on acute toxicity studies, triclopyr TEA is much less toxic to fish than either triclopyr BEE or TCP. The median of the LC_{50} values for triclopyr TEA is about 131 mg a.e./L while the median for corresponding values of TCP is 3.19 mg/L. Triclopyr TEA is less toxic than TCP by a factor of about 40. The median for corresponding values of triclopyr BEE is 0.539 mg a.e./L. Triclopyr TEA is less toxic than triclopyr BEE by a factor of about 240 and TCP is less toxic than triclopyr BEE by a factor of about 6.

Based on chronic studies, the NOAEC for triclopyr TEA is about 32.4 mg a.e./L and the NOAEC for TCP is 0.178 mg/L. TCP is more toxic than triclopyr TEA by a factor of about 180. Based on a standard egg-to-fry study in trout, the NOAEC for triclopyr BEE is 0.017 mg a.e./L. Based on chronic exposures, triclopyr BEE is more toxic than TCP to fish by a factor of about 10.

To summarize, triclopyr BEE is more toxic to fish than triclopyr TEA by a factor of about 240, based on acute toxicity. TCP is more toxic to fish than triclopyr TEA by a factor of about 40, based on acute toxicity, and by a factor of 180, based on chronic toxicity. TCP is less toxic to fish than triclopyr BEE by a factor of 6, based on acute toxicity, and less toxic to fish than triclopyr BEE by a factor of 10, based on chronic toxicity. There do not seem to be any significant differences among fish species in terms of sensitivity to the forms or formulations of triclopyr covered in this risk assessment

TCP is of concern in applications of triclopyr TEA, although this concern is somewhat lessened by the lower concentrations of TCP relative to triclopyr. However, for fish exposures, the risks associated with TCP are assessed quantitatively in U.S. Forest Service risk assessments.

Studies on the sublethal effects of Garlon 4 on rainbow trout showed that at concentrations of 0.32-0.43 mg/L, about a factor of 2 below the 96-hour LC_{50} determined in this study, fish were lethargic. At levels ≤0.1 mg/L, fish were hypersensitive over 4-day periods of exposure. This is reasonably consistent with the threshold for behavioral changes in rainbow trout for Garlon 4 of 0.6 mg/L found in another study, which also found a corresponding threshold for behavioral changes to Garlon 3A of 200 mg/L, consistent with the relative acute lethal potencies of these two agents.

Amphibians (Aquatic Phase) - There is only one acute toxicity value for triclopyr TEA, the 96-hour LC_{50} of 84 mg a.e./L in *Xenopus laevis* exposed to Garlon 3A. This is lower than the median LC_{50} in fish (~130 mg a.e./L) but well within the range of LC_{50} values (~40 to 420 mg a.e./L).

The only acute toxicity values for triclopyr BEE are for the Release or Garlon 4 formulations. Tadpoles are more sensitive than embryos, with differences in sensitivity spanning about an order of magnitude (median LC_{50} values of about 2 mg a.e./L in tadpoles and 20 mg a.e./L in embryos). Based on the LC_{50} value for tadpoles, the most sensitive stage, amphibians appear to be less sensitive than fish by a factor of about 4.

A large body of literature on reproductive toxicity in mammals indicates that triclopyr is not likely to cause reproductive or teratogenic effects at sublethal concentrations. 96-hour teratogenesis assays of Garlon 3A and Garlon 4 for malformations in frog (*Xenopus laevis*) embryos found no statistically significant increases in abnormalities in any groups exposed to Garlon 3A or Garlon 4 at levels that were not lethal.

As stated in SERA 2011d (p. 99):

> *Berrill et al., (1994) also assayed the toxicity of Garlon 4 using embryos and tadpoles of Rana pipiens (leopard frog), Rana clamitans (green frog), and Rana catesbeiana (bullfrog) in a static assay with aeration, which was conducted in darkness to prevent hydrolysis of triclopyr BEE. Exposures to 0.6, 1.2, and 4.6 mg a.e./L had no effect on hatching success, malformations, or subsequent avoidance behavior of embryos. Newly hatched tadpoles died or became immobile after exposure to the two higher concentrations. The approximate EC_{50} values for response to prodding were between 1.2 and 4.6 mg a.e./L after a 24-hour exposure period. As summarized in Table 34, these EC_{50} values for response to stimuli are very close to the LC_{50} values for frog larvae and probably reflect signs of nearly lethal exposures rather than sublethal effects on behavior.*

Data on the toxicity of TCP to aquatic phase amphibians were not identified in the conduct of the current risk assessment.

Aquatic Invertebrates - Based on the median acute 48-hour LC_{50} values, triclopyr BEE is more toxic than triclopyr TEA to aquatic invertebrates, by a factor of about 140, which is less than the difference in toxicity to fish (240X) between these two chemical forms. This difference in sensitivity is due almost entirely to the greater tolerance of aquatic invertebrates to triclopyr TEA. For triclopyr TEA, aquatic invertebrates are more tolerant than fish by a factor of about 3while for triclopyr BEE, aquatic invertebrates are more tolerant than fish by a factor of about 5. Based on acute bioassays of aquatic invertebrates exposed to triclopyr BEE, daphnids appear to be more sensitive than aquatic insects, with other aquatic arthropods displaying intermediate sensitivity. Snails may be more tolerant to triclopyr than aquatic arthropods.

In a standard 48-hour LC_{50} determination in *Daphnia magna*, TCP appears to be more toxic than triclopyr TEA but less toxic than triclopyr BEE.

As stated in SERA 2011d (p. 99):

> *Kreutzweiser et al., (1992) conducted a series of 1-hour bioassays of triclopyr BEE in several species of stream invertebrates. Based on these bioassays (Kreutzweiser et al., 1992, Table 4), LC_{50} values for these aquatic invertebrates were greater than 290 mg/L (≈200 mg a.e./L). These LC_{50s} are higher than the standard 48-hour LC_{50}s for triclopyr BEE by about 2 orders of magnitude. While 1-hour LC_{50} values are not typically available and are not routinely used in Forest Service risk assessments, these data from Kreutzweiser et al., (1992) are considered further in the risk characterization for aquatic invertebrates (Section 4.4.3.4).*

Aquatic Plants - In aquatic plants, triclopyr TEA is more toxic to dicots than monocots, while the differences in the toxicity of triclopyr BEE is less pronounced. Triclopyr TEA appears to be more toxic than triclopyr BEE to aquatic macrophytes while triclopyr BEE appears to be about equally toxic to both monocots and dicots.

Of the six species of algae that have been assayed with triclopyr TEA, it appears that the filamentous or rod shaped algae (species of *Ankistrodesmus*, *Anabaena*, and *Skeletonema*) may be somewhat more sensitive than more spherical species of algae (*Chlorella* species). Triclopyr BEE is more toxic than triclopyr TEA to algae by a factor of about 10 and appears to be as toxic if not slightly more toxic to fish than to algae. Investigations into the effects of triclopyr acid on carbon fixation in algae noted no or relatively little inhibition in carbon fixation at concentrations of 2.6 mg/L.

The only two bioassays on the toxicity of TCP to algae report EC_{50s} of 1.8 mg/L. TCP appears to be more toxic to algae than triclopyr TEA. Data also suggest that TCP may be as phytotoxic as triclopyr BEE as to aquatic macrophytes.

1.3.2.3 Exposure Assessment

1.3.2.3.1 Introduction

Non-target organisms could be affected by chemicals if they are exposed to them. To assess exposure the SERA and USDA/FS RAs use both plausible and highly conservative exposure scenarios unique to each chemical and non-target species and based upon available data. The exposure scenarios used in this risk assessment to determine the amount of chemical an organism could be exposed to are determined by the application method and the chemical and toxicological properties of the compound being applied. Scenarios for foliar applications include acute and chronic oral exposure (food or drinking water) and dermal exposure, soil contamination, direct spray, and spray drift. Scenarios for other application methods, such as soil treatment or cut surface applications, use only a subset of the standard exposure scenarios for foliar applications. As stated in SERA 2012 p. 85, "*The exposure assessment for aquatic species typically relies on the estimated peak and longer-term concentrations in water that are used in the human health risk assessment, as well as the exposure assessments for terrestrial wildlife from the consumption of contaminated water.*" As with the human health exposure assessment, the computational details for each exposure assessment are presented in the 2012 EXCEL "F series" workbooks created by WorksheetMaker and summaries are in "G series" workbooks. Rather than showing these in detail here, the reader is referred to the specific SERA or USDA/FS RAs for each chemical. These RAs can be downloaded from the USFS, Forest Health Protection website (http://www.fs.fed.us/foresthealth/pesticide/risk.shtml). The most current version of WorksheetMaker can be downloaded directly from the SERA website (www.sera-inc.com).

As stated in SERA 2012 p. 86,

> *Given the large number of species that could be exposed to pesticides and the varied diets in each of these species, a very large number of different exposure scenarios could be generated. For the generic risk assessments, an attempt is made to limit the number of exposure scenarios. The* specific exposure scenarios presented in the general risk assessments are designed as conservative *screening scenarios that may serve as guides for more detailed site-specific assessments by identifying the groups of organisms and routes of exposure that are of greatest concern.*

For chemicals to adversely affect offsite, non-target organisms they must be transported from the treatment site in sufficient quantities to expose those organisms to doses that could harm them. Chemicals are mobile to varying degrees, in both similar and different ways, and for different lengths of time.

It needs to be emphasized that to minimize risks to non-target, off-site organisms, the U.S. EPA requires language on chemical product labels to minimize drift or runoff. The following

language for sulfometuron methyl is illustrative of that found on other chemical product labels (U.S. EPA 2009g, pp. 15 &17):

> *For terrestrial uses, except for under the forest canopy: Do not apply directly to water, or to areas where surface water is present or to intertidal areas below the mean high water mark. Do not contaminate water when disposing of equipment washwater or rinsate.*
>
> *Exposure to (Brand Name) can injure or kill plants. Damage to susceptible plants can occur when soil particles are blown or washed off target onto cropland. Applications may not be made to soil that is subject to wind erosion when less than a 60% chance of rainfall is predicted to occur in the treatment area within 48 hours. Soils that are subject to wind erosion usually have a high silt and/or fine to very fine sand fractions. Soils with low organic matter also tend to be prone to wind erosion.*
>
> *Applications must be made using extremely coarse or coarser droplet size spectrum per ASABE (S572) definition.*
>
> *Do not apply when wind speed is greater than 10 mph.*
>
> *Do not make aerial or ground applications into temperature inversions.*
>
> *Inversions are characterized by stable air and increasing temperatures with height above the ground. Mist or fog may indicate the presence of an inversion in humid areas. The applicator may detect the presence of an inversion by producing smoke and observing a smoke layer near the ground surface.*
>
> *For ground boom applications, apply spray at lowest height that is consistent with pest control objectives to minimize drift.*

1.3.2.3.2 Terrestrial Organisms

Terrestrial organisms could be exposed to chemicals from direct spray, ingestion of contaminated materials (vegetation, prey species, soil, or water), grooming activities, or by indirect contact with contaminated vegetation. The greatest exposure to chemicals for terrestrial vertebrates is most likely to occur from consumption of contaminated vegetation or insects. The greatest exposure for terrestrial invertebrates is by direct spray or by indirect contact with contaminated vegetation.

The highest exposure level for non-target terrestrial plants will be from direct spraying within the treatment area. Direct spraying will result in an exposure level equivalent to the application rate. Off-site drift is also a significant route of exposure, but spray drift will decrease with increasing distance from the boundaries of treatment areas.

Exposures of soil organisms to a pesticide are typically based on the Gleams-Driver modeling and/or available monitoring data. Exposures to terrestrial plants are estimated both as concentrations in soil and direct foliar contamination either from direct spray or drift. For some species of terrestrial animals (typically insects), standard toxicity studies may report units that are not readily converted to mg agent/kg body weight. For example, some contact toxicity studies express exposure only in mass of agent per unit surface area – e.g., lb/acre or mg/m^2. In such a case, some dose-response assessments may be based on units of mass of agent per unit surface area and the exposure assessment is simply expressed as the application rate, or some fraction of the application rate to account for drift. In other cases, such as honeybees, body weight data may be used to convert exposures in mg/organism to mg/kg bw.

As stated in SERA 2012 (p. 85):

> *Estimates of oral exposure are expressed in the same units as the available toxicity data. As in the human health risk assessment, these units are usually expressed as mg of agent per kg of body weight and abbreviated as mg/kg for terrestrial animals. For dermal exposures to terrestrial animals, the units of measure usually are expressed in mg of agent per cm^2 of surface area of the organism and abbreviated as mg/cm^2. In estimating dose, however, a distinction is made between the exposure dose and the absorbed dose. The exposure dose is the amount of material on the organism (i.e., the product of the residue level in mg/cm^2 and the amount of surface area exposed), which can be expressed either as mg/organism or mg/kg body weight. The absorbed dose is the proportion of the exposure dose that is actually taken in or absorbed by the animal.*

For any given type of exposure, small animals (and insects) will generally receive a higher dose (mg/kg body weight) relative to larger animals due to the relationship between body weight to surface area and to the amount of food and water consumed relative to size. Mammals of five sizes are considered: small- (20 g) and medium-sized (400 g) omnivores, a 5 kg canid, a 70 kg herbivore, and a 70 kg carnivore while birds of four standard sizes are considered: a 10 g passerine, a 640 g predatory bird, a 2.4 kg piscivorous bird, and a 4 kg herbivorous bird. Because of dietary differences, all of the mammals and birds are not considered in all of the exposure scenarios, since, for instance, predatory birds don't eat vegetation.

As toxicity data are not generally available on reptiles or terrestrial-phase amphibians, exposure assessments are typically not developed. When toxicity data are available, custom exposure scenarios are developed.

1.3.2.3.2.1 Terrestrial Mammals, Birds, Reptiles, and Amphibians (Terrestrial Phase)

Exposure assessments for terrestrial mammals, birds, reptiles, and amphibians (terrestrial phase) are typically done for direct spray, dermal contact with contaminated vegetation,

ingestion of contaminated vegetation or prey, ingestion of contaminated water, and ingestion of contaminated fish.

Direct Spray - This scenario is similar to the accidental exposure scenarios for the general public, involving exposure to direct spray. The amount of chemical absorbed depends on the application rate, the surface area of the organism, and the rate of absorption.

For foliar applications, two direct spray scenarios are conducted. The first scenario is the direct spray of half of the body surface of a 20g mammal. This exposure assessment assumes first-order dermal absorption. The second scenario assumes complete absorption during the first day of exposure. This assessment is included to encompass increased exposures due to grooming.

There are substantial uncertainties associated with all direct spray scenarios. For example, first-order dermal absorption estimates do not consider losses of applied herbicides from the surface of the animal and may overestimate the absorbed dose. Birds, mammals, and other animals may groom frequently and such grooming may contribute to the total absorbed dose by direct ingestion of any herbicide on fur or feathers. Amphibians and some other vertebrates may have skin that is much more permeable than the skin of most mammals. When data are available on dermal absorption and toxicity in amphibians, direct spray scenarios may be developed in risk assessments involving foliar applications.

Direct spray scenarios are not generally given for large mammals as allometric relationships dictate that they will be exposed to lesser amounts of a herbicide than smaller mammals. Direct spray scenarios may be given when toxicity data indicate that large mammals are more sensitive than small mammals.

Dermal Contact with Contaminated Vegetation - To estimate the potential effect of indirect dermal contact with an herbicide, a relationship is assumed between the application rate and dislodgeable foliar residue. However, rates of transfer of herbicides from foliage to organisms are unavailable for wildlife species. Wildlife are likely to be in contact with contaminated vegetation for longer periods than humans, so it is reasonable to assume that an equilibrium is reached between levels on the skin, rates of absorption, and levels on contaminated vegetation. Assuming this, the absorbed dose resulting from contact with contaminated vegetation might be on the order of one-tenth (10%) that associated with comparable direct spray scenarios. Because this assumption is speculative, it is not generally used to quantify exposures in the risk assessments. The potential for effects from contact with contaminated vegetation is only addressed qualitatively. For most herbicides, this adds relatively little uncertainty to the risk assessment, because the dominant route of exposure will be the consumption of contaminated vegetation, which is addressed in the following scenario. Therefore, dermal contact with contaminated vegetation will not be addressed in the chemical-specific section below.

Ingestion of Contaminated Vegetation or Prey - Exposure assessments for the consumption of contaminated vegetation are developed for small- and medium-sized omnivores, a canid, an herbivore, a passerine bird, a piscivorous bird, and a herbivorous bird, but not for a large carnivorous mammal or a predatory bird, as they are primarily meat eaters. Both acute and chronic exposure scenarios are developed for the consumption of contaminated fruit and the consumption of short grass. Fruit and short grass are selected to encompass the range of plausible concentrations of herbicide residues in vegetation, with fruit having the lowest concentration and short grass the highest.

For both the acute and chronic exposure scenarios it is assumed that 100% of the diet is contaminated. For some acute exposures, this may not be a realistic assumption and is probably unlikely in chronic exposures, as animals may feed only sporadically in treated areas. Rather than incorporating into the exposure assessment arbitrary adjustments in the proportion of the diet that is contaminated, the impact of variations is discussed further in the risk characterization section, because the proportion of the diet that is contaminated is linearly related to the resulting Hazard Quotients (HQs).

Allometric relationships of the estimated food consumption rates by various species of mammals and birds are based on field metabolic rates (kcal/day) and account for much of the variability in food consumption among mammals and birds. Estimates of field metabolic rates are used to calculate food consumption based on the caloric value (kcal/day dry weight) of the food items considered in risk assessments and estimates of the water content of the various foods. Residual variability is remarkably constant among different groups of organism. Estimates from the allometric relationships may differ from actual field metabolic rates by approximately ±70%. In all worksheets involving the use of the allometric equations for field metabolic rates, the lower bound is taken as 30% of the estimate and the upper bound is taken as 170% of the estimate.

Exposure scenarios like those for the consumption of contaminated vegetation are provided for the consumption of small mammals by either a predatory mammal or a predatory bird as well as for the consumption of contaminated insects by a small mammal, a medium-sized mammal, and a small bird.

As stated in SERA 2012 (p. 89), “For aquatic applications, the consumption of contaminated vegetation is not typically considered. For soil treatments, the consumption of contaminated vegetation may be considered if compound-specific data are available on the relationship between concentrations of the pesticide in soil and the resulting concentration of the pesticide in plants.”

Ingestion of Contaminated Water - Both the human health and the ecological effects risk assessments use the same methods for estimating concentrations of herbicides in water, with a major difference that the estimates of exposure for the ecological effects risk assessment

involves the weight of the animal and the amount of water consumed. Water consumption rates are well characterized in terrestrial vertebrates and are based on allometric relationships in mammals and birds. Based on these estimates, exposure scenarios involving the consumption of contaminated water are developed for mammals and birds for accidental spills, expected peak concentrations, and expected longer-term concentrations. For both acute and chronic exposures, for the chemicals analyzed in this PEIR, ingestion of contaminated water leads to dose estimates far below those associated with consumption of contaminated vegetation. This is a common pattern following terrestrial application of many herbicides and reflects the direct application of the herbicides to vegetation.

Along with many other factors, water consumption in birds and mammals varies substantially with diet and season, but there are no well-documented quantitative estimates of this variability. Therefore, the variability in water consumption rates is not considered in the exposure assessments. For both acute and chronic exposures to herbicides, the upper and lower bound estimates of concentrations in surface water typically vary substantially. Therefore, quantitative consideration of the variability in water consumption rates would not typically have a substantial impact on the risk characterization.

As stated in (USDA/FS 2006a, p. 4-17):

> *Unlike the human health risk assessment, estimates of the variability of water consumption are not available. Thus, for the acute scenario, the only factors affecting the estimate of the ingested dose include the field dilution rates (i.e., the concentration of the chemical in the solution that is spilled) and the amount of solution that is spilled. As in the acute exposure scenario for the human health risk assessment, the amount of the spilled solution is taken as 200 gallons for liquid formulations. In the exposure scenario involving contaminated ponds or streams due to contamination by runoff or percolation, the factors that affect the variability are the water contamination rate, (see Section 3.2.3.4.2) and the application rate.*

Ingestion of Contaminated Fish - Since the consumption of contaminated fish by species that eat fish is a viable route of exposure to herbicides, sets of exposure scenarios are developed for an accidental spill, expected peak exposures, and estimated longer-term concentrations. These exposure scenarios are applied to a 5kg canid, a 70kg carnivorous mammal (typified by a black bear), and a piscivorous bird.

Herbicides exposures from the consumption of contaminated fish are dependent on both the concentration of the herbicide in water and the bioconcentration factor for the herbicide. The concentrations of herbicides in water are the same as used in the scenarios for ingestion of contaminated water. Bioconcentration factors for wildlife are usually based on whole-body bioconcentration factors in fish, under the assumption that mammalian or avian predators will

typically consume the entire fish. If chemical and species-specific data indicate that this is not the case, alternative custom exposure scenarios may be developed.

1.3.2.3.2.2 Terrestrial Invertebrates

Exposure assessments for terrestrial invertebrates are typically done for direct spray and drift, ingestion of contaminated vegetation or prey, contact with contaminated soil, and honeybees foraging for nectar.

Direct Spray and Drift - Honeybees are typically used as a surrogate for other terrestrial insects. Exposure levels from broadcast applications are modeled based on the herbicide application rate and the surface area of the bee (1.42 cm^2 for a bee with a body length of 1.44 cm). Doses in units of mg/bee are converted to units of mg/kg bw, with a typical mean body weight for worker bees of 116 mg.

Honeybee exposure to an herbicide during or shortly after application depends on how close the bee is to the application site and how much of the herbicide is intercepted by foliage prior to deposition on the bee. AgDRIFT is used to estimate the proportion of the nominal application rate deposited at various distances (0 to 900 feet) downwind from the treated site. The impact of foliar interception varies per the nature of the vegetative canopy. Foliar interception rates of 0% (no interception), 50%, and 90% are used in the exposure assessment.

Broadcast applications of an herbicide will most likely expose other terrestrial invertebrates to direct spray. If toxicity data on other terrestrial invertebrates is available and supports a dose-response assessment, an exposure scenario may be elaborated.

Ingestion of Contaminated Vegetation or Prey - Terrestrial invertebrates may be exposed to foliar applications of herbicides by consuming contaminated vegetation or prey. Estimated residue rates (mg/kg residues per lb applied) are calculated for contaminated vegetation or prey.

An estimate of food consumption by a foraging herbivorous insect is required to calculate a dose level. But since food consumption varies greatly, depending on the caloric requirements in each life stage or activity and the caloric value of the food to be consumed, the derivation of consumption values for specific species, life stages, activities, and food items is beyond the scope of the current analysis. However, based on studies on food consumption patterns of various insects, the risk assessments will typically use food consumption factors of 1.3 (0.6 to 2.2) kg food /kg bw.

Contact with Contaminated Soil - Some herbicides may be broadcast applied to soil, in which case soil concentrations from Gleams-Driver and/or monitoring data are used directly in the exposure assessment. For some herbicides, earthworm subchronic toxicity tests are

available. There may also be field studies or other studies that provide toxicity data on terrestrial invertebrates that are based on soil exposures.

Honeybees Foraging for Nectar – U.S. Forest Service risk assessments develop an exposure assessment on honeybees foraging for nectar, if sufficient data are available. This is generally done only when information on the concentration of the pesticide in nectar is available or can be reasonably estimated. Exposure assessments are generally limited only to nectar foragers, because this is the subgroup estimated to be exposed to the highest doses. None of the chemicals analyzed in this PEIR have sufficient data on the concentration of the chemical in pollen or nectar to support the development of an exposure assessment.

The basis of the exposure assessments is the sugar demand of the honeybee. Studies have found that the concentration of pesticides per unit of sugar in nectar are sometimes greater than in honey, despite honey having more sugar than nectar. If this is generally true, exposure assessments based on nectar consumption could overestimate pesticide exposure from honey residue.

1.3.2.3.2.3 Terrestrial Plants

Exposure assessments for terrestrial plants are typically done for direct spray, spray drift, runoff, wind erosion and the use of contaminated irrigation water.

Direct Spray - Direct spray will result in an exposure level equivalent to the application rate. Direct spray of non-target plants immediately adjacent to the application site is modeled in the worksheets that assess off-site drift.

Off-Site Drift - Off-site drift depends primarily on spray droplet size and meteorological conditions rather than specific properties of the compound being sprayed. Estimates of off-site drift are modeled using AgDRIFT and are summarized for foliar applications. Custom worksheets may be used to assess ground broadcast and backpack applications.

As stated in SERA 2012 (p. 94):

> *The drift estimates used in the current risk assessment are based on AgDRIFT (Teske et al., 2002) using Tier 1 analyses for aerial and ground broadcast applications. The term Tier 1 is used to designate relatively generic and simple assessments that may be viewed as plausible upper limits of drift. Aerial drift estimates are based on Tier 1 using ASAE Fine to Medium drop size distributions. Tier 1 estimates of drift for ground broadcast applications are modeled using both low boom and high boom options in AgDRIFT. For both types of applications, the values are based on Very Fine to Fine drop size distributions and the 90th percentile values from AgDRIFT.*
>
> *Drift associated with backpack applications (directed foliar applications) are likely to be much less than drift from ground broadcast applications. Few studies, however, are*

available for quantitatively assessing drift after backpack applications. For the current risk assessment, estimates of drift from backpack applications are based on an AgDRIFT Tier 1 run of a low boom ground application using Fine to Medium/Coarse drop size distributions (rather than very fine to fine) as well as 50th percentile estimates of drift (rather than the 90th percentile used for ground broadcast applications).

The values for drift used in generic (i.e.,, not site-specific) risk assessments should be regarded as little more than generic estimates similar to the water concentrations modeled using GLEAMS (Section 3.2.3.4.3). Actual drift will vary per a number of conditions—e.g., the topography, soils, weather, and the pesticide formulation. These factors cannot be considered in generic risk assessments.

Typical backpack ground spray droplet sizes are greater than 100 µ and the distance from the spray nozzle to the ground is 3 feet or less. Mechanical sprays may use raindrop nozzles that generate droplets that are usually greater than 400 µ, with a maximum distance above the ground of about 6 feet. In both cases, the sprays are directed downward.

For most applications, the wind velocity will be no more than 5 mph (~7.5 feet/second). Assuming a wind direction perpendicular to the line of application, 100 µ particles falling from 3 feet above the surface could drift as far as 23 feet. A raindrop or 400 µ particle applied at 6 feet above the surface could drift about 3 feet.

For backpack applications, wind speeds of up to 15 mph are allowed in U.S. Forest Service programs. The VTP and alternatives are limited to windspeeds of no more than 7 mph (SPR HAZ-9). At a 15-mph wind speed, a 100 µ droplet can drift as far as 68 feet. Smaller droplets will drift further, so the proportion of this size particle in the spray as well as the wind speed and turbulence will affect the proportion of the applied herbicide that drifts off-site.

Runoff and Soil Mobility - Herbicides can be transported off-site from the soil by runoff, sediment loss, or percolation, so these are considered in estimating contamination of ambient water. Only runoff and sediment loss are considered in assessing contamination of off-site soil that might affect plants. Percolation is not considered in this case as it represents the amount of herbicide that is transported below the root zone. While it may impact water quality, it will likely not affect off-site vegetation. Runoff estimates are modeled using GLEAMS for clay, loam, and sand at nine sites that are representative of different temperatures and rainfall patterns.

When results from a runoff study of sulfometuron methyl were compared with GLEAMS modeling predictions, GLEAMS under-predicted runoff, in some cases by a factor of more than 30. The greatest discrepancies were apparent for heavy rainfall events. These discrepancies are likely attributable to the 1-day time step used by GLEAMS, which fails to account for rapid water and herbicide movement during short-term but intense rainfall events.

In any case, if herbicides are applied during or shortly before heavy rainfall events, concentrations in runoff of some herbicides could reach levels toxic to sensitive plant species.

Contaminated Irrigation Water - This scenario is unlikely to occur with potential herbicide application under this PEIR and the alternatives, as applications will primarily be to non-irrigated rangelands and forests. Levels of exposure will depend on the amount of irrigation water used and the herbicide concentration in the ambient water used for irrigation, based on the peak concentrations modeled in the human health risk assessment.

The selection of an irrigation rate is somewhat arbitrary and depends on the climate, soil type, topography, and plant species under cultivation. The application of 1 inch of irrigation water with a range of 0.25 to 2 inches is used in U.S. Forest Service risk assessments.

The product labels for some herbicides may note that water contaminated with the herbicide should not be used for irrigation. In these cases, the standard exposure scenario is included in the risk assessment with a comment indicating that it is not relevant except to evaluate the consequences of disregarding the labeled use restrictions.

Wind Erosion - Wind erosion can be a major mechanism for off-site movement of herbicides and is highly site-specific. The amount of herbicide that might be transported depends on several factors, including application rate, depth of incorporation into the soil, persistence in the soil, wind speed, and topographical and surface conditions of the soil. It is unlikely that herbicide transport would be substantial with relatively deep (4 inches) soil incorporation, low wind speed, and surface conditions which inhibit wind erosion.

As stated in SERA 2012 (p. 94):

> *For Forest Service risk assessments, the potential effects of wind erosion are estimated in Worksheet G06b. In this worksheet, it is assumed that the pesticide is incorporated at a depth that is identical to the depth of incorporation used in Gleams-Driver modeling, typically 1 cm. Average soil losses are estimated to range from 1 to 10 tons/ha/year with a typical value of 5 tons/ha/year. These estimates are based on the results of agricultural field studies which found that wind erosion may account for annual soil losses ranging from 2 to 6.5 metric tons/ha (Allen and Fryrear 1977).*
>
> *As noted in Worksheet G07b, the use of the above values typically results in estimates of offsite losses at about 0.014% of the application rate. Larney et al., (1999), however, report that wind erosion of other herbicides could be associated with losses up to 1.5% of the nominal application rate following soil incorporation or 4.5% following surface application. This difference appears to be a due to the much higher soil losses noted by Larney et al., (1999)—i.e., up to 56.6 metric tons/ha from a fallow field. The losses reflected in Worksheet G06b may be somewhat more realistic for forest or rangeland applications, because herbicide applications are rarely made to fallow areas. In any*

event, the higher offsite losses reported by Larney et al., (1999) are generally comparable to exposures associated with offsite drift at distances of about 50 feet from the application site following low boom (0.017) and high boom (0.05) ground broadcast applications (Worksheet G05). All the estimates for wind erosion and offsite drift are likely to vary dramatically per site conditions and weather conditions.

Volatilization - Volatilization may be an important route of exposure to some herbicides for off-site, non-target plants. As general methods for estimating exposures from volatilization have not been developed, this section is included only when the chemical-specific information is adequate to support both an exposure assessment and a dose-response assessment. None of the chemicals analyzed in this PEIR have such chemical-specific information, so no exposure scenarios have been developed.

1.3.2.3.3 Aquatic Organisms

Aquatic organisms could be exposed from direct spray, ingestion of contaminated materials (aquatic vegetation, prey species, or water), or by indirect contact with contaminated vegetation or water.

The greatest exposure for aquatic organisms is most likely to occur following an accidental chemical spill directly into a water body. The exposure assessment is based on the concentrations of the pesticide in surface water that are used in the exposure assessment for terrestrial vertebrates, which is in turn equivalent to the concentrations used in the human health risk assessment.

1.3.2.3.4 Chemical-Specific Exposure Assessments

1.3.2.3.4.1 Borax (Sources: FS WSM ver. 6.00.10; SERA 2006a)

Terrestrial Mammals, Birds, Reptiles, and Amphibians (Terrestrial Phase)

As stated in the Overview in SERA 2006a, p 4-8:

As discussed in Section 3.2, Sporax is applied directly to the surfaces of freshly cut tree stumps. Sporax is not applied using backpack, broadcast or aerial spray methods and it is not applied directly to vegetation. Therefore, many of the standard exposure scenarios that are typically considered for Forest Service risk assessments, such as direct spray, oral exposure via ingestion of contaminated prey or vegetation, are not applicable for this risk assessment. The exposure scenarios used in this risk assessment are those expected to result in substantial exposure considering the atypical application method for Sporax.

For terrestrial vertebrates, two exposure scenarios are considered for this risk assessment: acute exposure via consumption of Sporax applied to tree stumps, and acute as well as chronic exposure via exposure to contaminated pond water.

Ingestion of Sporax from Tree Stumps – A field study found that deer licked borax (Sporax) applied to the surface of tree stumps, but also licked the surface of untreated stumps. Therefore, it is unclear whether Sporax attracts deer. But the study suggests that the consumption of Sporax from treated stumps is a plausible exposure scenario for deer and perhaps other species.

As little information is available to estimate the amount of Sporax that terrestrial mammals or birds are likely to consume from tree stumps, exposures developed for this scenario are highly uncertain. For large (70 kg) mammals, such as a deer, exposure is based on the underlying assumption that a deer might consume all the Sporax applied to a tree stump that is 1 foot in diameter, with amounts consumed estimated as 40 mg (lower bound), 242 mg (central bound), and 807 mg (upper bound). Although direct consumption of Sporax from a stump by a large (4 kg) bird, such as a goose or heron, is implausible, as they typically consume either vegetation or fish, a similar scenario is developed for a Canada goose. For smaller species, it seems less plausible that the animal would consume all the Sporax on a treated stump. The body weights that are used are 20 grams for a small mammal and 10 grams for a small bird.

For small mammals and birds, exposure values for acute exposure via consumption of Sporax applied to a tree stump are essentially identical, as follow: 0.0056 mg B/kg/event (lower bound), 0.011 mg B/kg/event (central bound), and 0.011 mg B/kg/event (upper bound). For large mammals and birds, exposure values for the same scenario are also essentially identical, as follow: 0.575 mg B/kg/event (lower bound), 3.43 mg B/kg/event (central bound), and 11.5 mg B/kg/event (upper bound). A summary of exposure assessments for terrestrial animals is displayed in Worksheet G01 in FS WSM ver. 6.00.10.

Ingestion of Contaminated Water – After application of granular Sporax to tree stumps, runoff from rainfall could contaminate standing water or streams. Accidental spills of Sporax could also contaminate a small body of water. Exposure assessments are developed for terrestrial animals for both scenarios. However, the use of Sporax in stump treatments is not likely to have a substantial affect on concentrations of boron in ambient water, so this is not considered a relevant scenario (Worksheet G01 in FS WSM ver. 6.00.10). For chronic exposures of a small mammal by consuming water contaminated by runoff, exposure values are 0.00146 mg B/kg/day (lower bound), 0.0102 mg B/kg/day (central bound), and 0.0512 mg B/kg/day (upper bound).

Terrestrial Invertebrates

There is no information in SERA or USDA/FS risk assessments on exposure of terrestrial invertebrates to borax. Since Sporax is not applied as a spray, wide-spread exposure of insects is not expected.

Terrestrial Plants (Macrophytes)

As stated in the Overview in SERA 2006a, p 4-8: "*Since Sporax is not applied to vegetation, the only exposure scenario considered for terrestrial macrophytes is exposure to boron that reaches soil via runoff. Based on the results of GLEAMS modeling, peak concentrations of boron in soil range from 0.0026 ppm for the lowest value associated with an application rate of 0.1 lb Sporax/acre to 2.29 ppm in soil for the highest value associated with an application rate of 5 lbs Sporax/acre.*"

Aquatic Organisms

As stated in SERA 2006a, p. 4-11: "The potential for effects on aquatic species is based on estimated concentrations of borax (as boron equivalent) in water that are identical to those used in the human health risk assessment. For this risk assessment, contamination of water is considered for two scenarios – accidental spill of a bag of Sporax (containing an amount ranging from 6.25 to 25 pounds Sporax) into a small pond and contamination of pond water and contamination of a small pond by runoff. For an accidental spill of Sporax into a small pond, the peak estimated concentration of boron in ambient water is 0.64 mg B/L (0.32 - 1.28) mg B/L (ppm). Details of this calculation are provided in Worksheet F05.

Contamination of a small pond by runoff, the peak estimated concentration of boron in ambient water is 30 (6 to 100) µg boron/L after a single application of 1 lb Sporax/acre (0.11 lb boron/acre). For longer-term exposures, the corresponding longer term concentrations in ambient water are estimated at about 14 (2 to 70) µg boron/L. (ibid)

1.3.2.3.4.2 Clopyralid (Sources: FS WSM ver. 6.00.07 & 6.00.10; SERA 2004a)

Exposure values for the scenarios displayed below are summarized in the "G" series Worksheets in FS WSM ver. 6.00.10: for mammals (G01a) and birds (G01b). For the analysis in this PEIR, all exposure values for clopyralid have been computed for the typical application rate of 0.25 lb. a.e./acre, which is also the highest application rate that is legal in California.

By far the highest short-term acute exposures to clopyralid are associated with the consumption of contaminated grass by a small mammal (173 mg/kg bw/event) and a small bird (427 mg a.e./kg bw/event). The corresponding maximum chronic exposures are 90.9 mg/kg bw/day for a small mammal and 225 mg a.e./kg bw/day for a small bird. For both acute and chronic exposures, consumption of contaminated water leads to dose estimates far below those associated with consumption of contaminated vegetation. This pattern is common in many herbicide exposure assessments, reflecting the consequences of direct applications to vegetation.

Hexachlorobenzene is a contaminant of clopyralid that may be of concern to terrestrial and aquatic animals. Per the SERA risk assessment for clopyralid (SERA 2004a, p. 3-23), hexachlorobenzene is: "*…. ubiquitous and persistent in the environment. The major sources of general exposure for the public to hexachlorobenzene involve industrial emissions,*

proximity to hazardous waste sites, and the consumption of contaminated food. Virtually all individuals are exposed to hexachlorobenzene and virtually all individuals have detectable concentrations of hexachlorobenzene in their bodies (ATSDR 2002)."

Hexachlorobenzene is found at average concentrations of less than 2.5 ppm in technical grade clopyralid. It has a higher potential for human exposure than clopyralid itself, because the body is better able to absorb it. Hexachlorobenzene will bioconcentrate in fish and has a BCF that ranges from 2,000 to 20,000. For the Forest Service RA a BCF of 2,000 was used for acute exposure and a BCF of 20,000 for chronic exposure (SERA 2004a, p. 3-22).

Terrestrial Mammals, Birds, Reptiles, and Amphibians (Terrestrial Phase)

Direct Spray – At the typical application rate, accidental acute exposure scenarios lead to upper bound estimates of exposure for mammals ranging from 0.435 mg/kg/event (first-order absorption of direct spray by a small mammal) to 12.1 mg/kg/event (100% absorption of direct spray by a small mammal). For birds, no exposure scenarios for direct spray are developed, as it is assumed that most birds will fly away during herbicide applications.

Dermal Contact with Contaminated Vegetation - Based on data for clopyralid, dislodgeable residue from the surface of contaminated vegetation will be approximately 10 times less than the highest application rate of 0.25 lb. a.e./acre. Since direct spray scenarios result in exposure levels below the estimated NOAEL, details of the exposure scenarios for contaminated vegetation are not elaborated. This adds relatively little uncertainty to the risk assessment, because the dominant route of exposure will be the consumption of contaminated vegetation.

Ingestion of Contaminated Vegetation or Prey - At the typical application rate, non-accidental acute exposure scenarios lead to upper bound estimates of exposure for mammals ranging from 1.15 mg/kg/event (consumption of a small mammal by a canid) to 173 mg/kg/event (consumption of grass by a small mammal). For birds, estimates of exposure range from 1.37 mg/kg/event (consumption of a small mammal by a carnivorous bird) to 427 mg/kg/event (consumption of short grass by a small bird).

Chronic exposure scenarios lead to upper bound estimates of exposure for mammals ranging from 0.99 mg/kg/day (consumption of fruit by a large mammal) to 90.9 mg/kg/day (consumption of short grass by a small mammal). For birds, estimates of exposure range from 1.90 mg/kg/day (consumption of fruit by a large bird) to 225 mg/kg/day (consumption of short grass by a small bird).

Ingestion of Contaminated Water - At the typical application rate, accidental acute exposure scenarios for consumption of contaminated water lead to upper bound estimates of exposure for mammals ranging from 0.735 mg/kg/event (large mammal) to 1.66 mg/kg/event (small

mammal). For birds, estimates of exposure range from 0.424 mg/kg/event (large bird) to 3.06mg/kg/event (small bird).

Scenarios of non-accidental acute exposure and chronic exposure(latter values in parentheses) for consumption of contaminated water lead to upper bound estimates of exposure for mammals ranging from 0.00113 (0.00021) mg/kg/event(day) (large mammal) to 0.00256 (0.000476) mg/kg/event(day) (small mammal). For birds, estimates of exposure range from 0.000653 (0.000121) mg/kg/event(day) (large bird) to 0.00472 (0.000876) mg/kg/event(day) (small bird).

Ingestion of Contaminated Fish - Ambient water and fish are exposure pathways for clopyralid. As clopyralid has a low potential to bioconcentrate in fish, the bioconcentration factor for fish is taken as 1.0 L/kg for chronic exposure scenarios. For the scenario of accidental acute exposure from a spill into a pond, the upper bound estimates of exposure are 1.9 mg/kg/event (large mammalian carnivore), 2.74 mg/kg/event (canid), and 3.18 (fish-eating bird). The non-accidental acute exposure scenario for a large mammalian carnivore or a canid (value for canid in parentheses) consuming contaminated fish results in doses of 0.00293 (0.00422) mg/kg/event at the upper bound at the highest application rate. The corresponding value for a fish-eating bird is 0.0049 mg/kg/event. Chronic exposure values at the upper bound at the highest application rate are 0.000545 mg/kg/day (large mammalian carnivore) and 0.000784 mg/kg/day (canid). The corresponding value for a fish-eating bird is 0.000911 mg/kg/day.

Terrestrial Invertebrates

Concentrations of clopyralid in clay, loam, and sand over a wide range of rainfall rates are summarized in Table 4-2 in SERA 2004a (p. Tables-12). At the highest application rate of 0.25 lb a.e./acre, the estimated maximum concentrations of clopyralid in clay soil would range from about 0.066 lb. a.e./acre at an annual rainfall of 10 inches to 0.07 lb. a.e./acre at an annual rainfall of 100 inches. Due to percolation, concentrations in loam and sand soils would be less.

Only limited data is available on the toxicity of clopyralid to soil invertebrates and soil microorganisms. Since there is no information regarding the dermal absorption rate of clopyralid by bees or other invertebrates, an exposure scenario (100% absorption over one day) for a honeybee with a body weight of 0.093 g is used.

Terrestrial Plants (Macrophytes)

Direct Spray and Off-Site Drift - Unintended direct spray will result in an exposure level equivalent to the application rate. Estimates of off-site drift for ground applications of clopyralid, which is typically applied by low boom ground spray, are used in the SERA risk assessment. At the typical and maximum application rate of 0.25 lb. a.e./acre, drift is

estimated to result in concentrations of clopyralid of 0.00875 lb. a.e./acre 25 feet from the application site to 0.00237 lb. a.e./acre 100 feet from the application site, the furthest distance away where there is still a concern for toxicity to non-target, sensitive plant species. A summary of both the exposure assessment and risk characterization for terrestrial plants from direct spray and off-site drift is in Worksheet G05 in FS WSM ver. 6.00.10.

Runoff and Soil Mobility – Runoff of minor amounts of clopyralid following broadcast applications, at the typical and highest application rate of 0.25 lb. a.e./acre, is estimated to begin occurring on clay soils at an annual rainfall rate of 15 inches (50 inches on loams and >250 inches on sand). Runoff is estimated to result in concentrations of clopyralid of 0.01075 lb. a.e./acre at 15 inches of rain to 0.09125 lb. a.e./acre at 100 inches, the annual rainfall rate where toxicity to non-target, sensitive plant species becomes problematic. A summary of both the exposure assessment and risk characterization for terrestrial plants from runoff is in Worksheet G04 in FS WSM ver. 6.00.10.

Based on the GLEAMS modeling, clopyralid may penetrate to about 18 inches in clay. In loam or sand, detectable residues are modeled to occur at 60 inches. Because the GLEAMS modeling used a 60-inch root zone, the actual penetration in loam or sand could be greater than 60 inches.

Contaminated Irrigation Water - Clopyralid is relatively mobile and contamination of ambient water is plausible. Based on the estimated concentrations of clopyralid in ambient water at the typical and highest application rate of 0.25 lb. a.e./acre, the estimated functional application rate of clopyralid to the irrigated area is 0.0011 lb. a.e./acre at an irrigation rate of 1 inch per day and 0.0079 lb. a.e./acre at an irrigation rate of 2 inches per day. Relative to off-site drift and runoff, this level of exposure is inconsequential. A summary of both the exposure assessment and risk characterization for terrestrial plants from contaminated irrigation water is in Worksheet G06a in FS WSM ver. 6.00.10.

Wind Erosion - Although no specific incidents of non-target damage from wind erosion have been encountered in the literature for clopyralid, this mechanism has been associated with the environmental transport of other herbicides. Wind erosion of minor amounts of clopyralid following broadcast applications, at the typical and highest application rate of 0.25 lb. a.e./acre, is estimated to result in concentrations of clopyralid of 0.000017 lb. a.e./acre at the central bound to 0.000034 lb. a.e./acre at the upper bound. Relative to off-site drift and runoff, this level of exposure is inconsequential and well below a LOC for non-target, sensitive plant species. A summary of both the exposure assessment and risk characterization for terrestrial plants from wind erosion is in Worksheet G06b in FS WSM ver. 6.00.10.

Aquatic Organisms

At the typical (and highest) application rate of 0.25 lb a.e./acre the peak estimated rate of contamination of ambient water associated with the normal application of clopyralid is 0.005

(0.00125 to 0.0175) mg a.e./L, while the average estimated rate of contamination for longer-term exposures is 0.00175 (0.00025 to 0.00325) mg a.e./L.

1.3.2.3.4.3 Glyphosate (Sources: FS WSM v. 6.00.10; SERA 2011b; U.S. EPA. 2009c)

The SERA risk assessment for glyphosate (SERA 2011b) displays a standard set of exposure assessments. All workbooks use a unit application rate of 1 lb. a.e./acre, but the exposure assessment in this PEIR uses a typical application rate of 2 lbs. a.e./acre. Values displayed in SERA 2011b can be easily converted by multiplying them by whatever application rate is anticipated. Summaries of the exposure assessments are in Worksheet G01a (mammals), G01b (birds), and G08a (insects) in FS WSM ver. 6.00.10.

By far the highest short-term acute exposures to glyphosate are associated with the consumption of contaminated grass by a small mammal (1,380 mg/kg bw/event) and a small bird (3,420 mg a.e./kg bw/event). The corresponding maximum chronic exposures are 221 mg/kg bw/day for a small mammal and 547 mg a.e./kg bw/day for a small bird. For both acute and chronic exposures, consumption of contaminated water leads to dose estimates far below those associated with consumption of contaminated vegetation. This pattern is common in many herbicide exposure assessments, reflecting the consequences of direct applications to vegetation.

Terrestrial Mammals, Birds, Reptiles, and Amphibians (Terrestrial Phase)

The SERA risk assessment for terrestrial mammals and birds displays a standard set of exposure assessments (accidental, acute non-accidental, and chronic) for foliar applications of glyphosate, in Attachment 1a for backpack applications and in Attachment 1b for ground broadcast applications. As stated above, values displayed in those attachments can be easily converted by multiplying by 2, to reflect the typical rate of application.

The exposure assessments for terrestrial mammals, birds, reptiles, and amphibians (terrestrial phase) do not distinguish between the more or less toxic forms of glyphosate. Apparently, glyphosate becomes more toxic to aquatic species when certain surfactants are added to the formulation, most notably POEA. In this analysis, “more toxic” glyphosate includes such formulations.

Direct Spray – At the typical application rate, accidental acute exposure scenarios lead to upper bound estimates of exposure for mammals ranging from 1.15 mg/kg/event (first-order absorption of direct spray by a small mammal) to 97 mg/kg/event (100% absorption of direct spray by a small mammal). For birds, no exposure scenarios for direct spray are developed, as it is assumed that most birds will fly away during herbicide applications.

Ingestion of Contaminated Vegetation or Prey - At the typical application rate, non-accidental acute exposure scenarios lead to upper bound estimates of exposure for mammals ranging

from 9.23 mg/kg/event (consumption of a small mammal by a canid) to 1,380 mg/kg/event (consumption of grass by a small mammal). For birds, estimates of exposure range from 11 mg/kg/event (consumption of a small mammal by a carnivorous bird) to 3,420 mg/kg/event (consumption of short grass by a small bird).

Chronic exposure scenarios lead to upper bound estimates of exposure for mammals ranging from 2.41 mg/kg/day (consumption of fruit by a large mammal) to 221 mg/kg/day (consumption of short grass by a small mammal). For birds, estimates of exposure range from 4.61 mg/kg/day (consumption of fruit by a large bird) to 547 mg/kg/day (consumption of short grass by a small bird).

Ingestion of Contaminated Water - At the typical application rate, accidental acute exposure scenarios for consumption of contaminated water lead to upper bound estimates of exposure for mammals ranging from 2.35 mg/kg/event (large mammal) to 5.32 mg/kg/event (small mammal). For birds, estimates of exposure range from 1.36 mg/kg/event (large bird) to 9.8 mg/kg/event (small bird).

Scenarios of non-accidental acute exposure and chronic exposure(values in parentheses) for consumption of contaminated water lead to upper bound estimates of exposure for mammals ranging from 0.0107 (0.000751) mg/kg/event(day) (large mammal) to 0.0243 (0.0017) mg/kg/event(day) (small mammal). For birds, estimates of exposure range from 0.0062 (0.000433) mg/kg/event(day) (large bird) to 0.0448 (0.00313) mg/kg/event(day) (small bird).

Ingestion of Contaminated Fish - Ambient water and fish are exposure pathways for glyphosate. As glyphosate has a low potential to bioconcentrate in fish, the bioconcentration factor for fish is taken as 0.52 L/kg for chronic exposure scenarios.

At the typical application rate, accidental acute exposure scenarios for consumption of contaminated fish lead to upper bound estimates of exposure for mammals ranging from 3.17 mg/kg/event (large mammalian carnivore) to 4.56 mg/kg/event (canid). For birds, the estimated exposure of a fish-eating bird is 5.29 mg/kg/event. Non-accidental acute exposures lead to upper bound estimates of exposure for mammals ranging from 0.0145 mg/kg/event (large mammalian carnivore) to 0.0208 mg/kg/event (canid). For birds, the estimated exposure of a fish-eating bird is 0.0242 mg/kg/event. Chronic exposures lead to upper bound estimates of exposure for mammals ranging from 0.0010 mg/kg/event (large mammalian carnivore) to 0.0015 mg/kg/event (canid). For birds, the estimated exposure of a fish-eating bird is 0.00169 mg/kg/event.

Terrestrial Invertebrates

The exposure assessments for terrestrial invertebrates do not distinguish between the more or less toxic forms of glyphosate. Honeybees are used as a surrogate for other terrestrial

insects as available toxicity data on terrestrial invertebrates do not support the derivation of separate toxicity values for different groups of terrestrial insects.

Direct Spray and Off-Site Drift – A summary of the exposure assessments and risk characterization for the honeybee for the scenarios of direct spray and drift is in G09 in FS WSM ver. 6.00.10. Exposure from direct spray is shown for three scenarios (0%, 50%, and 90% foliar interception), none of which lead to a HQ above the LOC. The absorbed doses are 137.2, 68.6, and 13.7 mg/kg bw/event, respectively. The absorbed doses from spray drift 25 feet from the application site are 4.8, 2.4, and 0.5 mg/kg bw/event, respectively.

Ingestion of Contaminated Vegetation or Prey - Four non-accidental acute exposure scenarios of a herbivorous insect consuming vegetation were developed. For a large insect, consuming fruit, the estimated dose at the typical application rate of 2 lbs. a.e./acre, is 18.2 mg/kg bw/event (central bound) and 66 mg/kg bw/event (upper bound). For a small insect consuming broadleaf foliage, the estimated dose is 117 mg/kg bw/event (central) and 594 mg/kg bw/event (upper). For an insect consuming tall and short grass (the latter value in parentheses), the estimated dose is 93.6 (221) mg/kg bw/event (central) and 484 (1,056) mg/kg bw/event (upper).

Contact with Contaminated Soil - Concentrations of glyphosate in clay, loam, and sand over a wide range of site conditions are summarized in Table 4-2 in SERA 2004a (p. Tables-12). At the typical application rate of 2 lb a.e./acre, the estimated maximum concentrations of glyphosate in the top 12 inches of clay soil would range from about 0.283 lb. a.e./acre in dry, warm locations to 0.243 lb. a.e./acre in wet, cool locations. Due to percolation, concentrations in loam and sand soils would be less; 0.176 lb. a.e./acre in dry, warm locations to 0.172 lb. a.e./acre in wet, cool locations.

Terrestrial Plants (Macrophytes)

Direct Spray and Off-Site Drift - Unintended direct spray will result in an exposure level equivalent to the application rate. Estimates of off-site drift for broadcast ground applications of glyphosate are calculated in the SERA risk assessment. At the typical application rate of 2 lb. a.e./acre, drift is estimated to result in concentrations of clopyralid of 0.01664 lb./acre 25 feet from the application site to 0.00482 lb./acre 100 feet from the application site, the furthest distance away where there is still a concern for toxicity to non-target, sensitive plant species. A summary of both the exposure assessment and risk characterization for terrestrial plants from direct spray and off-site drift is in Worksheet G05 in FS WSM ver. 6.00.10.

Runoff and Soil Mobility – For glyphosate, there is no rainfall-specific information for runoff displayed in Worksheet G04 in FS WSM ver. 6.00.10. Information on the relationship between site conditions and runoff rates is displayed in SERA 2011b, Appendix 10, Table 1, p. 116. The effective off-site application rate from runoff in clay soils ranges from 0.000104 lb a.e./acre in dry and warm locations to 0.036 lb a.e./acre in wet and cool locations. In loam

and sand soils (values for sand in parentheses) these values range from 0.0 lb a.e./acre in dry and warm locations to 0.0058 (0.00057) lb a.e./acre in wet and cool locations.

Based on the GLEAMS modeling, detectable residues of glyphosate may penetrate to a depth of about 4-12 inches in clay soils, resulting in concentrations in the top 12 inches of soil of 0.283 ppm in dry and warm locations to 0.243 ppm in wet and cool locations. In loam soils, detectable residues may penetrate to about 4-12 inches (4-18 inches for sandy soils), resulting in concentrations of 0.176 ppm in dry and warm locations to 0.172 ppm in wet and cool locations.

Contaminated Irrigation Water - Glyphosate is not likely to contaminate ambient water. Based on the estimated concentrations of glyphosate in ambient water at the typical application rate of 2 lb. a.e./acre, the estimated functional application rate of glyphosate to the irrigated area is 0.0050 lb. a.e./acre at an irrigation rate of 1 inch per day and 0.075 lb. a.e./acre at an irrigation rate of 2 inches per day. Relative to off-site drift and runoff, this level of exposure is inconsequential. A summary of both the exposure assessment and risk characterization for terrestrial plants from contaminated irrigation water is in Worksheet G06a in FS WSM ver. 6.00.10.

Wind Erosion - Although no specific incidents of non-target damage from wind erosion have been encountered in the literature for glyphosate, this mechanism has been associated with the environmental transport of other herbicides. Wind erosion of minor amounts of glyphosate following broadcast applications, at the typical application rate of 2 lb. a.e./acre, is estimated to result in concentrations of glyphosate of 0.000137lb. a.e./acre at the central bound to 0.000274 lb. a.e./acre at the upper bound. Relative to off-site drift and runoff, this level of exposure is inconsequential and well below a LOC for non-target, sensitive plant species. A summary of both the exposure assessment and risk characterization for terrestrial plants from wind erosion is in Worksheet G06b in FS WSM ver. 6.00.10.

Aquatic Organisms

The plausibility of effects on aquatic species is assessed based on estimated concentrations of glyphosate in water that are identical to those used in the human health risk assessment. At the typical application rate of 2 lb a.e./acre, the peak estimated rate of contamination of ambient water associated with the normal application of glyphosate is 0.042 (0.0013 to 0.083) mg a.e./L, while the average estimated rate of contamination for longer-term exposures is 0.0029 (0.000088 to 0.0058) mg a.e./L.

1.3.2.3.4.4 Hexazinone (Sources: FS WSM v. 6.00.10; SERA 2005)

Exposure values for the scenarios displayed below are summarized in the “G” series Worksheets in FS WSM ver. 6.00.10: for mammals (G01a) and birds (G01b). For the analysis

in this PEIR, all exposure values for liquid and granular hexazinone have been computed for the typical application rate of 2 lb. a.i./acre.

In the SERA 2005 risk assessment, no exposure scenarios were developed for granular formulations of hexazinone, as the clay pellets were thought not to stick to mammals or other ecological receptors. Also, data for adjusting estimates of pellet deposition were not available. It was thought that risks were far below a LOC and any overestimate of exposure would have no impact on the characterization of risk.

However, two sets of exposure scenarios are provided in the 2012 version of the EXCEL workbooks. One workbook covers Velpar L, the only liquid formulation considered in this risk assessment, and the other covers the granular formulations. Although these assessments are generally similar in nature, some of the computational details differ in ways that are mandated by differences between granular and liquid formulations. There is also a substantial difference in the amount of residue on contaminated vegetation, with much higher residues expected after the application of Velpar L compared to the granular formulations.

By far the highest short-term acute exposures to liquid and granular (the latter values in parentheses) formulations of hexazinone are associated with the consumption of grass, 1,380 (55.3) mg/kg bw/event (small mammal) and 3,420 (137) mg a.e./kg bw/event (small bird). The corresponding maximum chronic exposures are 581 (23.3) mg/kg bw/day for a small mammal and 1,440 (57.5) mg a.e./kg bw/day for a small bird. For both acute and chronic exposures, consumption of contaminated water leads to dose estimates far below those associated with consumption of contaminated vegetation. This pattern is common in many herbicide exposure assessments, reflecting the consequences of direct applications to vegetation.

Terrestrial Mammals, Birds, Reptiles, and Amphibians (Terrestrial Phase)

Direct Spray – At the typical application rate, accidental acute exposure scenarios for liquid and granular (values in parentheses) formulations of hexazinone lead to upper bound estimates of exposure for mammals ranging from 5.28 (0.0109) mg/kg/event (first-order absorption of direct spray by a small mammal) to 97 (3.0) mg/kg/event (100% absorption of direct spray by a small mammal). For birds, no exposure scenarios for direct spray are developed, as it is assumed that most birds will fly away during herbicide applications.

Ingestion of Contaminated Vegetation or Prey – Residues on vegetation are likely to be much greater after applications of Velpar L compared to applications of the granular formulations. Standard residue rates are used directly in the Velpar L worksheets but are divided by a factor of 25 for applications of granular formulations.

At the typical application rate, non-accidental acute exposure scenarios for liquid and granular (values in parentheses) formulations of hexazinone lead to upper bound estimates of exposure for mammals ranging from 9.23 mg/kg/event (consumption of a small mammal by

a canid) to 1,380 (55.3) mg/kg/event (consumption of grass by a small mammal). The lower estimate for the granular formulation is 0.602 mg/kg/event (consumption of fruit by a large mammal). For birds, estimates of exposure range from 11 mg/kg/event (consumption of a small mammal by a carnivorous bird) to 3,420 (137) mg/kg/event (consumption of short grass by a small bird). The lower estimate for the granular formulation is 1.15 mg/kg/event (consumption of fruit by a large bird).

Chronic exposure scenarios for liquid and granular (values in parentheses) formulations of hexazinone lead to upper bound estimates of exposure for mammals ranging from 6.33 (0.253) mg/kg/day (consumption of fruit by a large mammal) to 581 (23.3) mg/kg/day (consumption of short grass by a small mammal). For birds, estimates of exposure range from 12.1 (0.485) mg/kg/day (consumption of fruit by a large bird) to 1,440 (57.5) mg/kg/day (consumption of short grass by a small bird).

Ingestion of Contaminated Water – Since estimates of the variability of water consumption by mammals, birds, reptiles, and terrestrial amphibians are not available, for the acute scenario, the only factors affecting the estimate of the ingested dose include the amount of solution that is spilled and the field dilution rates. For liquid formulations (Velpar L), the amount of the spilled solution is the standard amount used for exposure assessments, 200 gallons. For granular formulations, the amount spilled is calculated in pounds based on the number of acres that would be treated with the corresponding liquid formulation(s) and the range of application rates covered by this risk assessment. Variability in the exposure scenario involving ponds or streams contaminated by runoff or percolation is affected by the water contamination rate and the herbicide application rate.

At the typical application rate, accidental acute exposure scenarios for both formulations of hexazinone for consumption of contaminated water lead to upper bound estimates of exposure for mammals ranging from 2.35 mg/kg/event (large mammal) to 5.32 mg/kg/event (small mammal). For birds, estimates of exposure range from 1.36 mg/kg/event (large bird) to 9.8 mg/kg/event (small bird).

Scenarios of non-accidental acute exposure and chronic exposure (values in parentheses) for both formulations of hexazinone for consumption of contaminated water lead to upper bound estimates of exposure for mammals ranging from 0.0518 (0.0205) mg/kg/event(day) (large mammal) to 0.117 (0.00906) mg/kg/event(day) (small mammal). For birds, estimates of exposure range from 0.0299 (0.00523) mg/kg/event(day) (large bird) to 0.216 (0.0378) mg/kg/event(day) (small bird).

Ingestion of Contaminated Fish - Ambient water and fish are exposure pathways for hexazinone. As hexazinone has a low potential to bioconcentrate in fish, the bioconcentration factor for fish is taken as 2 L/kg for chronic exposure scenarios.

At the typical application rate, accidental acute exposure scenarios for both formulations of hexazinone for consumption of contaminated fish lead to upper bound estimates of exposure for mammals ranging from 12.2 mg/kg/event (large mammalian carnivore) to 17.5 mg/kg/event (canid). For birds, the estimated exposure of a fish-eating bird is 20.4 mg/kg/event. Non-accidental acute exposure scenarios lead to upper bound estimates of exposure for mammals ranging from 0.268 mg/kg/event (large mammalian carnivore) to 0.386 mg/kg/event (canid). For birds, the estimated exposure of a fish-eating bird is 0.0448 mg/kg/event. Chronic exposure scenarios lead to upper bound estimates of exposure for mammals ranging from 0.0469 mg/kg/event (large mammalian carnivore) to 0.0676 mg/kg/event (canid). For birds, the estimated exposure of a fish-eating bird is 0.0785 mg/kg/event.

Terrestrial Invertebrates

Direct Spray and Drift – No specific information on exposure to terrestrial invertebrates from direct spray or off-site drift of hexazinone is available in the SERA 2005 risk assessment. The application rate and the amount of drift will be the same as for plants (see below) and will determine the maximum dose that terrestrial invertebrates could be exposed to.

Ingestion of Contaminated Vegetation or Prey - No specific information on exposure to terrestrial invertebrates from ingestion of contaminated vegetation or prey of hexazinone is available in the SERA 2004c risk assessment. It seems likely that the routes of exposure modeled for some other herbicides analyzed in this PEIR would be similar, with similar exposure levels. For those herbicides, four non-accidental acute exposure scenarios were developed for herbivorous insects consuming vegetation contaminated by herbicide residues. The highest anticipated dose was to a small insect consuming broadleaf vegetation, followed by an insect consuming tall or short grass, and lastly, by a large insect consuming fruit.

Contact with Contaminated Soil - Only limited data are available on the toxicity of hexazinone to soil invertebrates and microorganisms. The data on soil invertebrates are only semi-quantitative and the effects reported are not associated with soil concentrations of hexazinone.

Concentrations of hexazinone in clay, loam, and sand over a wide range of site conditions are summarized in Table 4-3 in SERA 2005 (p. Tables 1-25). At the typical application rate of 2 lb a.i./acre, the estimated maximum concentrations of hexazinone in the top 12 inches of clay soil would range from about 0.147 ppm at 10 inches of annual rainfall to 0.0752 ppm at 100 inches. Due to percolation, concentrations in loam and sand soils would be less; 0.139 (0.119) ppm at 10 inches of annual rainfall and 0.215 (0.168) ppm at 100 inches.

Terrestrial Plants (Macrophytes)

Direct Spray and Off-Site Drift - Unintended direct spray will result in an exposure level equivalent to the application rate. Estimates of off-site drift for ground applications of the liquid formulation hexazinone, which is typically applied by low boom ground spray, are used in the SERA risk assessment. At the typical application rate of 2 lb. a.i./acre, drift is estimated to result in concentrations of hexazinone of 0.07 lb./acre 25 feet from the application site to 0.01896 lb./acre 100 feet from the application site, the furthest distance away where there is still a concern for toxicity to non-target, sensitive plant species. A summary of both the exposure assessment and risk characterization for terrestrial plants from direct spray and off-site drift is in Worksheet G05 in FS WSM ver. 6.00.10 for the liquid formulation (but not granular) of hexazinone.

Runoff and Soil Mobility – Runoff of minor amounts of both the liquid and granular formulations of hexazinone following broadcast applications, at the typical application rate of 2 lb. a.i./acre, is estimated to begin occurring on clay soils at an annual rainfall rate of 15 inches (50 inches on loams and >250 inches on sand). Runoff is estimated to result in concentrations of hexazinone of 0.10 lb. a.e./acre at 15 inches of rain to 0.894 lb. a.e./acre at 100 inches. Toxicity to non-target, sensitive plant species from runoff from clay soils becomes problematic at an annual rainfall rate of 15 inches and severe at 100 inches. Even for tolerant species, exposures become problematic at an annual rainfall rate of 15 inches, but are much less severe. A summary of both the exposure assessment and risk characterization for terrestrial plants from runoff is in Worksheet G04 in FS WSM ver. 6.00.10.

Based on the GLEAMS modeling, detectable residues of hexazinone may penetrate to a depth of about 18-36 inches in clay soils, 42->60 inches in loam soils, and >60 inches in sand at annual rainfall rates of 15-100 inches (SERA 2005, Table 4-5). The detectable concentrations of hexazinone in the top 12 inches of clay soil average from 0.274 ppm (rainfall 15”) to 0.1504 ppm (rainfall 100”). In loam soil, concentrations average 0.25 ppm (rainfall 15”) and 0.0836 ppm (rainfall 100”) and in sandy soils, concentrations average 0.1924 ppm (rainfall 15”) and 0.0248 ppm (rainfall 100”) (SERA 2005, Table 4-3). These estimates are consistent with the field monitoring studies reporting soil penetration.

Contaminated Irrigation Water - Hexazinone is highly mobile and contamination of ambient water may be anticipated. Based on the estimated concentrations of hexazinone in ambient water at the typical application rate of 2 lb. a.i./acre, the estimated functional application rate of hexazinone to the irrigated area is 0.0453 lb. a.e./acre at an irrigation rate of 1 inch per day and 0.3625 lb. a.e./acre at an irrigation rate of 2 inches per day. A summary of both the exposure assessment and risk characterization for terrestrial plants from contaminated irrigation water is in Worksheet G06a in FS WSM ver. 6.00.10.

Wind Erosion - Although no specific incidents of non-target damage from wind erosion have been encountered in the literature for hexazinone, this mechanism has been associated with the environmental transport of other herbicides. While somewhat speculative, it seems

plausible that granular formulations would be more susceptible to wind erosion than liquid formulations. Since no data have been located that would permit a quantitative adjustment in estimates of off-site transport, the worksheets for the two formulations are identical.

Wind erosion of minor amounts of hexazinone following broadcast applications, at the typical application rate of 2 lb. a.i./acre, is estimated to result in concentrations of hexazinone of 0.000137 lb. a.e./acre at the central bound to 0.000274 lb. a.e./acre at the upper bound. Relative to off-site drift and runoff, this level of exposure is inconsequential and well below a LOC for non-target, sensitive plant species. A summary of both the exposure assessment and risk characterization for terrestrial plants from wind erosion is in Worksheet G06b in FS WSM ver. 6.00.10.

Aquatic Organisms

The plausibility of effects on aquatic species is based on estimated concentrations of hexazinone in water that are identical to those used in the human health risk assessment. At the typical application rate of 2 lb a.i./acre, the peak estimated rate of contamination of ambient water associated with the normal application of hexazinone is 0.200 (0.0005 to 0.4) mg a.e./L, while the average estimated rate of contamination for longer-term exposures is 0.035 (0.00001 to 0.07) mg a.e./L.

1.3.2.3.4.5 Imazapyr (Sources: FS WSM v. 6.00.10; SERA 2011c; U.S. EPA 2006d)

Exposure values for the scenarios displayed below are summarized in the “G” series Worksheets in FS WSM ver. 6.00.10: for mammals (G01a), birds (G01b), and insects (G08a). For the analysis in this PEIR, all exposure values for imazapyr have been computed for the typical application rate of 0.30 lb. a.e./acre.

By far the highest short-term acute exposures to imazapyr are associated with the consumption of contaminated grass by a small mammal (207 mg/kg bw/event) and a small bird (513 mg a.e./kg bw/event). The corresponding maximum chronic exposures are 100 mg/kg bw/day for a small mammal and 248 mg a.e./kg bw/day for a small bird. For both acute and chronic exposures, consumption of contaminated water leads to dose estimates far below those associated with consumption of contaminated vegetation. This pattern is common in many herbicide exposure assessments, reflecting the consequences of direct applications to vegetation.

Terrestrial Mammals, Birds, Reptiles, and Amphibians (Terrestrial Phase)

Direct Spray – At the typical application rate, accidental acute exposure scenarios lead to upper bound estimates of exposure for mammals ranging from 0.489 mg/kg/event (first-order absorption of direct spray by a small mammal) to 14.5 mg/kg/event (100% absorption of direct

spray by a small mammal). For birds, no exposure scenarios for direct spray are developed, as it is assumed that most birds will fly away during herbicide applications.

Ingestion of Contaminated Vegetation or Prey - At the typical application rate, non-accidental acute exposure scenarios lead to upper bound estimates of exposure for mammals ranging from 1.38 mg/kg/event (consumption of a small mammal by a canid) to 207 mg/kg/event (consumption of grass by a small mammal). For birds, estimates of exposure range from 1.65 mg/kg/event (consumption of a small mammal by a carnivorous bird) to 513 mg/kg/event (consumption of short grass by a small bird).

Chronic exposure scenarios lead to upper bound estimates of exposure for mammals ranging from 1.09 mg/kg/day (consumption of fruit by a large mammal) to 100 mg/kg/day (consumption of short grass by a small mammal). For birds, estimates of exposure range from 2.09 mg/kg/day (consumption of fruit by a large bird) to 248 mg/kg/day (consumption of short grass by a small bird).

Ingestion of Contaminated Water - At the typical application rate, accidental acute exposure scenarios for consumption of contaminated water from a spill lead to upper bound estimates of exposure for mammals ranging from 0.353 mg/kg/event (large mammal) to 0.798 mg/kg/event (small mammal). For birds, estimates of exposure range from 0.204 mg/kg/event (large bird) to 1.47 mg/kg/event (small bird).

Scenarios of non-accidental acute exposure and chronic exposure (values in parentheses) for consumption of contaminated water lead to upper bound estimates of exposure for mammals ranging from 0.00505 (0.00233) mg/kg/event(day) (large mammal) to 0.0114 (0.00527) mg/kg/event(day) (small mammal). For birds, estimates of exposure range from 0.00291 (0.00134) mg/kg/event(day) (large bird) to 0.0210 (0.00971) mg/kg/event(day) (small bird).

Ingestion of Contaminated Fish - Ambient water and fish are exposure pathways for imazapyr. As imazapyr has a low potential to bioconcentrate in fish, the bioconcentration factor for fish is taken as 0.5 L/kg f for chronic exposure scenarios.

At the typical application rate, accidental acute exposure scenarios for consumption of contaminated fish lead to upper bound estimates of exposure for mammals ranging from 0.457 mg/kg/event (large mammalian carnivore) to 0.658 mg/kg/event (canid). For birds, the estimated exposure of a fish-eating bird is 0.764 mg/kg/event. Non-accidental acute exposure scenarios lead to upper bound estimates of exposure for mammals ranging from 0.00654 mg/kg/event (large mammalian carnivore) to 0.00941 mg/kg/event (canid). For birds, the estimated exposure of a fish-eating bird is 0.0109 mg/kg/event. Chronic exposure scenarios lead to upper bound estimates of exposure for mammals ranging from 0.00302 mg/kg/day (large mammalian carnivore) to 0.00434 mg/kg/day (canid). For birds, the estimated exposure of a fish-eating bird is 0.00504 mg/kg/day.

Terrestrial Invertebrates

Direct Spray and Drift - A summary of the exposure assessments and risk characterization for the honeybee for the scenarios of direct spray and drift of imazapyr is in G09 in FS WSM ver. 6.00.10. Exposure from direct spray is shown for three scenarios (0%, 50%, and 90% foliar interception), none of which lead to a HQ above the LOC. The absorbed doses are 20.6, 10.3, and 2.1 mg/kg bw/event, respectively. The absorbed doses from spray drift 25 feet from the application site are0.72, 0.36, and 0.07 mg/kg bw/event, respectively.

Ingestion of Contaminated Vegetation or Prey - Four non-accidental acute exposure scenarios were developed for herbivorous insects consuming vegetation contaminated by residues of imazapyr. For a large insect, consuming fruit, the estimated dose at the typical application rate of 0.30 lbs. a.e./acre, is 2.73 mg/kg bw/event (central bound) and 9.9 mg/kg bw/event (upper bound). For a small insect consuming broadleaf foliage, the estimated dose is 17.6 mg/kg bw/event (central) and 89.1 mg/kg bw/event (upper). For an insect consuming tall and short grass (the latter value in parentheses), the estimated dose is 14.04 (33.2) mg/kg bw/event (central) and 72.6 (158) mg/kg bw/event (upper).

Contact with Contaminated Soil - Based on the GLEAMS modeling, imazapyr may penetrate to 36 inches in clay, loam, and sand soils. Because the GLEAMS modeling used a 36-inch root zone, the actual penetration of imazapyr could be greater than 36 inches.

Terrestrial Plants (Macrophytes)

Direct Spray and Off-Site Drift - Unintended direct spray will result in an exposure level equivalent to the application rate. Estimates of off-site drift for broadcast ground applications of imazapyr are used in the SERA risk assessment. At the typical application rate of 0.30 lb. a.e./acre, drift is estimated to result in concentrations of imazapyr of 0.0105 lb./acre 25 feet from the application site to 0.000327 lb./acre 900 feet from the application site, where adverse effects to non-target, sensitive plant species are still plausible. There are no concerns for tolerant species, even at the application site. A summary of both the exposure assessment and risk characterization for terrestrial plants from direct spray and off-site drift is in Worksheet G05 in FS WSM ver. 6.00.10.

Runoff and Soil Mobility – For imazapyr, there is no rainfall-specific information for runoff displayed in Worksheet G04 in FS WSM ver. 6.00.10. Information on the relationship between site conditions and runoff rates is displayed in SERA 2011c, Appendix 7, Table 1, p. 196. The effective off-site application rate from runoff in clay soils ranges from 0.00106 lb a.e./acre in dry and warm locations to 0.12 lb a.e./acre in wet and cool locations. In loam and sand soils (values for sand in parentheses) these values range from 0.0 (0.0) lb a.e./acre in dry and warm locations to 0.0093 (0.0) lb a.e./acre in wet and cool locations.

Based on the GLEAMS modeling, detectable residues of imazapyr may penetrate to a depth of about 4-36 inches in clay soils, resulting in concentrations in the top 12 inches of soil of 0.27 ppm in dry and warm locations and 0.211 ppm in wet and cool locations. In loam and sand soils (values for sand in parentheses), detectable residues may penetrate to about 4-36 inches, resulting in concentrations of 0.241 (0.209) ppm in dry and warm locations to 0.198 (0.17) ppm in wet and cool locations.

Contaminated Irrigation Water - Imazapyr is relatively mobile and contamination of ambient water may be anticipated. Based on the estimated concentrations of imazapyr in ambient water at the typical application rate of 0.30 lb. a.e./acre, the estimated functional application rate of imazapyr to the irrigated area is 0.00136 lb. a.e./acre at an irrigation rate of 1 inch per day and 0.0353 lb. a.e./acre at an irrigation rate of 2 inches per day. Relative to off-site drift and runoff, this level of exposure is inconsequential. A summary of both the exposure assessment and risk characterization for terrestrial plants from contaminated irrigation water is in Worksheet G06a in FS WSM ver. 6.00.10.

The Re-registration Eligibility Decision for imazapyr notes that water that contains imazapyr residues should not be used for irrigation. Product labels for the formulations listed in SERA 2011c (Table 2) include restrictions to limit the use of water for crop irrigation that may contain imazapyr residues. While perhaps not relevant to imazapyr, the exposure assessment in this PEIR is included for consistency with other herbicide risk assessments and to enable assessment of the consequences of disregarding the labeled use restrictions.

Wind Erosion - Although no specific incidents of non-target damage from wind erosion have been encountered in the literature for clopyralid, this mechanism has been associated with the environmental transport of other herbicides. Wind erosion of minor amounts of imazapyr following broadcast applications, at the typical application rate of 0.30 lb. a.e./acre, is estimated to result in concentrations of imazapyr of 0.000055 lb. a.e./acre at the central bound to 0.000041 lb. a.e./acre at the upper bound. Relative to off-site drift and runoff, this level of exposure is inconsequential and well below a LOC for non-target, sensitive plant species. A summary of both the exposure assessment and risk characterization for terrestrial plants from wind erosion is in Worksheet G06b in FS WSM ver. 6.00.10.

Aquatic Organisms

At the typical application rate of 0.30 lb a.e./acre, the peak estimated rate of contamination of ambient water associated with the normal application of imazapyr is 0.13 (0.000009 to 0.26) mg a.e./L, while the average estimated rate of contamination for longer-term exposures is 0.06 (0.000003 to 0.12) mg a.e./L.

1.3.2.3.4.6 NP9E (Sources: FS WS ver. 2.02; USDA/FS 2003b; U.S. EPA 2010e)

Exposure values for the scenarios displayed below are summarized in the Worksheet in FS WSM ver. 6.00.10: for mammals and birds (WL Ex1). For the analysis in this PEIR, all exposure values for NP9E have been computed for the typical application rate of 1.67 lb. a.i./acre.

By far the highest short-term acute exposures to NP9E are associated with the consumption of contaminated vegetation by a large mammal (324 mg/kg bw/event) and a large bird (508 mg a.e./kg bw/event). The corresponding maximum chronic exposures are 0.0822 (off-site), 520 (on-site) mg/kg bw/day for a large mammal and 0.129 (off-site), 8.14 (on-site) mg a.e./kg bw/day for a large bird. For both acute and chronic exposures, consumption of contaminated water leads to dose estimates far below those associated with consumption of contaminated vegetation. This pattern is common in many herbicide exposure assessments, reflecting the consequences of direct applications to vegetation. Because of the apparently low toxicity of NP9E to animals, the rather substantial variations in the different exposure assessments have little impact on the assessment of risk to terrestrial animals.

Terrestrial Mammals, Birds, Reptiles, and Amphibians (Terrestrial Phase)

Direct Spray – At the typical application rate, accidental acute exposure scenarios lead to upper bound estimates of exposure for mammals ranging from 0.00107 mg/kg/event (100% absorption by a honeybee) to 3.46 mg/kg/event (first-order absorption of direct spray by a small mammal) to 162 mg/kg/event (100% absorption of direct spray by a small mammal). For birds, no exposure scenarios for direct spray are developed, as it is assumed that most birds will fly away during herbicide applications.

Dermal Contact with Contaminated Vegetation - Neither the bioconcentration data on NP9E or the estimated rates of dermal absorption in humans indicate that NP9E is likely to preferentially partition from the surface of contaminated vegetation to the surface of skin, feathers, or fur, which supports a plausible partition coefficient of unity (i.e., the concentration of the chemical on the surface of the animal will be equal to the dislodgeable residue on the vegetation).

Ingestion of Contaminated Vegetation or Prey – As stated in USDA/FS 2003b, p. 50: "*For estimating the effects of longer-term exposures, time-weighted average concentrations are used, which is similar to the approach taken in the human health risk assessment and using the same estimates of foliar halftime as were used in the corresponding human health risk assessment. Also, the longer-term exposure scenario is based on a 90-day post-spray period and uses the geometric mean over this period as the central estimate of the exposed dose, as in the human health risk assessment. Like the acute exposure scenario, this exposure scenario assumes that 100% of the diet is contaminated.*"

At the typical application rate, non-accidental acute exposure scenarios lead to upper bound estimates of exposure for mammals ranging from 17.9 mg/kg/event (consumption of vegetation by a small mammal) to 324 mg/kg/event (consumption of vegetation by a large mammal). For birds, the estimated exposure for consumption of vegetation by a large bird is 508 mg/kg/event.

Chronic exposure scenarios lead to upper bound estimates of exposure for mammals ranging from 0.000906 (off-site), 0.0574 (on-site) mg/kg/day (consumption of vegetation by a small mammal) to 0.0822 (off-site), 520 (on-site) mg/kg/day (consumption of vegetation by a large mammal). For birds, the estimated exposure is 0.129 (off-site), 8.14 (on-site) mg/kg/event (consumption of vegetation by a large bird).

Ingestion of Contaminated Water - At the typical application rate, the accidental acute exposure scenario for a small mammal drinking from a pond after a spill leads to an estimated dose of 2.22 mg/kg/event. The non-accidental scenario of a small mammal drinking from a stream contaminated by runoff or percolation through the soil leads to an upper bound estimate of exposure of 0.00457 mg/kg/event. For chronic exposure, for a small mammal, the dose is 0.00205 mg/kg/day.

Ingestion of Contaminated Fish - Ambient water and fish are exposure pathways for NP9E. As NP9E has a low potential to bioconcentrate in fish, the bioconcentration factor for fish is taken as 1 L/kg for chronic exposure scenarios. The only scenario for ingestion of contaminated fish involves a predatory bird. The acute accidental dose is 2.27 mg/kg/event and the chronic dose is 0.0021 mg/kg/day.

Terrestrial Invertebrates

Direct Spray and Drift – There is no information for NP9E in the Worksheet or in USDA/FS 2003b specific to these scenarios. For other herbicides analyzed in this PEIR, exposure from direct spray and off-site drift is shown for three scenarios (0%, 50%, and 90% foliar interception). In the case of imazapyr, none of these scenarios leads to absorbed doses above the LOC at the application site. At 25 feet from the application site, absorbed doses are close to 30 times lower. It is plausible that NP9E would follow a similar pattern.

Ingestion of Contaminated Vegetation or Prey - There is no information for NP9E in the Worksheet or in USDA/FS 2003b specific to these scenarios. For other herbicides analyzed in this PEIR, four non-accidental acute exposure scenarios were developed for herbivorous insects consuming contaminated fruit, broadleaf vegetation, and grass.

Contact with Contaminated Soil - There is some concern that surfactants might increase the movement of herbicides into soils. In one study, levels of nonionic NPE-based surfactants at concentrations below 1000 mg/L caused little or no decrease in sorption of a fungicide, but at 10,000 mg/L, an increase in sorption was seen.

Terrestrial Plants (Macrophytes)

Direct Spray and Off-Site Drift - Unintended direct spray will result in an exposure level equivalent to the application rate of 1.67 lb. a.i./acre. There is no information for NP9E in the Worksheet or in USDA/FS 2003b specific to off-site drift or to the toxicity of NP9E to terrestrial plants. Since NP9E-based surfactants would not be applied alone, but would be applied in a mix with an herbicide, the herbicide would determine the effects to terrestrial plants.

Runoff and Soil Mobility – The dose-response assessment in USDA/FS 2003b did not support a quantitative assessment and no GLEAMS modeling was conducted, so no information is available for an assessment of NP9E. Since NP9E-based surfactants would not be applied alone, but would be applied in a mix with an herbicide, the herbicide would determine the effects to terrestrial plants.

Contaminated Irrigation Water - There is no information for NP9E in the Worksheet or in USDA/FS 2003b specific to the effects of contaminated irrigation water. Since NP9E-based surfactants would not be applied alone, but would be applied in a mix with an herbicide, the herbicide would determine the effects to terrestrial plants.

Wind Erosion - There is no information for NP9E in the Worksheet or in USDA/FS 2003b specific to the effects from wind erosion. Since NP9E-based surfactants would not be applied alone, but would be applied in a mix with an herbicide, the herbicide would determine the effects to terrestrial plants.

Aquatic Organisms

As stated in USDA/FS 2003b, p. 51:

> *The potential for effects on aquatic species are based on estimated concentrations of NP9E or NP1-2EC in water that are identical to those used in the human health risk assessment. The estimated rate of contamination of ambient water associated with the normal application of NP9E is 0.0125 mg a.e./L (12.5 ppb). For acute exposure scenarios, the highest estimated concentration of NP9E in water after an accidental spill is about 6.1 mg a.e./L (ppm) with a range of about 3.0 to 15.1 mg a.e./L. As another exposure scenario, if the Forest Service were to overspray an herbicide mixture with an 80% NPE-based surfactant into a small pond or stagnant stream reach, with no foliar interception, instantaneous levels of NP9E could approach 1.5 mg/L (1,500 ppb) and the concentration of NP and the short-chain ethoxylates (NP1E and NP2E) could approach (0.075 mg/L (75 ppb) (refer to worksheet 1 in Appendix 1). Assuming a more realistic live stream, these levels would be quickly lowered as water is mixed through stream flow.*

As discussed in section 3.2.3.3, the breakdown of NPE would likely not liberate NP, and any free NP in the surfactant would be broken down in the forested environment or bound to soil particles. Therefore, it is very unlikely that NP would be found in forest streams above the level that might be found in the NP9E mixture originally. As stated in section 4.3, the acute toxicity of NP9E includes this small percentage of NP and short-chain NPEs, so no adjustment for acute exposures is necessary.

Based on environmental fate, the toxicological compound of interest is more likely to be the short chain NPECs (NP1EC, NP2EC), as they will be formed in the forested environment and their persistence would make them more available for aquatic wildlife exposure and for exposure to terrestrial wildlife through water consumption. As stated in section 3.2.3.3.2, the assumed levels of NP1-2EC in water will be based on water monitoring and set at 0.007 mg/L (with a range of 0 to 0.014 mg/L).

1.3.2.3.4.7 Sulfometuron methyl (Sources: FS WSM v. 6.00.10; SERA 2004c; U.S. EPA 2008a, 2009g)

Exposure values for the scenarios displayed below are summarized in the "G" series Worksheets in FS WSM ver. 6.00.10: for mammals (G01a) and birds (G01b). For the analysis in this PEIR, all exposure values for sulfometuron methyl have been computed for the typical application rate of 0.045 lb. a.e./acre.

By far the highest short-term acute exposures to sulfometuron methyl are associated with the consumption of contaminated grass by a small mammal (31.1 mg/kg bw/event) and a small bird (76.9 mg a.e./kg bw/event). The corresponding maximum chronic exposures are 4.97mg/kg bw/day for a small mammal and 12.3 mg a.e./kg bw/day for a small bird. For both acute and chronic exposures, consumption of contaminated water leads to dose estimates far below those associated with consumption of contaminated vegetation. This pattern is common in many herbicide exposure assessments, reflecting the consequences of direct applications to vegetation.

Terrestrial Mammals, Birds, Reptiles, and Amphibians (Terrestrial Phase)

Direct Spray – At the typical application rate, accidental acute exposure scenarios lead to upper bound estimates of exposure for mammals ranging from 0.0254 mg/kg/event (first-order absorption of direct spray by a small mammal) to 2.18 mg/kg/event (100% absorption of direct spray by a small mammal). For birds, no exposure scenarios for direct spray are developed, as it is assumed that most birds will fly away during herbicide applications.

Ingestion of Contaminated Vegetation or Prey - At the typical application rate, non-accidental acute exposure scenarios for sulfometuron methyl lead to upper bound estimates of exposure for mammals ranging from 0.208 mg/kg/event (consumption of a small mammal by a canid) to 31.1 mg/kg/event (consumption of grass by a small mammal). For birds, estimates of

exposure range from 0.247 mg/kg/event (consumption of a small mammal by a carnivorous bird) to 76.9 mg/kg/event (consumption of short grass by a small bird).

Chronic exposure scenarios lead to upper bound estimates of exposure for mammals ranging from 0.0542 mg/kg/day (consumption of fruit by a large mammal) to 4.97 mg/kg/ day (consumption of short grass by a small mammal). For birds, estimates of exposure range from 0.104 mg/kg/ day (consumption of fruit by a large bird) to 12.3 mg/kg/ day (consumption of short grass by a small bird).

Ingestion of Contaminated Water - At the typical application rate, accidental acute exposure scenarios for consumption of contaminated water lead to upper bound estimates of exposure for mammals ranging from 0.0539 mg/kg/event (large mammal) to 0.122 mg/kg/event (small mammal). For birds, estimates of exposure range from 0.0311 mg/kg/event (large bird) to 0.225 mg/kg/event (small bird).

Scenarios of non-accidental acute exposure and chronic exposure (values in parentheses) for consumption of contaminated water lead to upper bound estimates of exposure for mammals ranging from 0.0000583 (0.000000204) mg/kg/event(day) (large mammal) to 0.000132 (0.000000461) mg/kg/event(day) (small mammal). For birds, estimates of exposure range from 0.0000336 (0.000000118) mg/kg/event(day) (large bird) to 0.000243 (0.000000849) mg/kg/event(day) (small bird).

Ingestion of Contaminated Fish - Ambient water and fish are exposure pathways for sulfometuron methyl. Sulfometuron methyl may bioconcentrate to a small degree in the muscle and viscera of fish. The bioconcentration factor for fish is taken as 7 L/kg for chronic exposure scenarios.

At the typical application rate, accidental acute exposure scenarios for consumption of contaminated fish lead to upper bound estimates of exposure for mammals ranging from 0.977 mg/kg/event (large mammalian carnivore) to 1.41 mg/kg/event (canid). For birds, the estimated exposure of a fish-eating bird is 1.63 mg/kg/event. Non-accidental acute exposure scenarios lead to upper bound estimates of exposure for mammals ranging from 0.00106 mg/kg/event (large mammalian carnivore) to 0.00152 mg/kg/event (canid). For birds, the estimated exposure of a fish-eating bird is 0.00177 mg/kg/event. Chronic exposure scenarios lead to upper bound estimates of exposure for mammals ranging from 0.0000037 mg/kg/day (large mammalian carnivore) to 0.00000532 mg/kg/day (canid). For birds, the estimated exposure of a fish-eating bird is 0.00000618 mg/kg/day.

Terrestrial Invertebrates

Direct Spray and Drift – No specific information on exposure to terrestrial invertebrates from direct spray or off-site drift of sulfometuron methyl is available in the SERA 2004c risk assessment. The application rate and the amount of drift will be the same as for plants (see

below) and will determine the maximum dose that terrestrial invertebrates could be exposed to.

Ingestion of Contaminated Vegetation or Prey - No specific information on exposure to terrestrial invertebrates from ingestion of contaminated vegetation or prey of sulfometuron methyl is available in the SERA 2004c risk assessment. It seems likely that the routes of exposure modeled for some other herbicides analyzed in this PEIR would be similar, with similar exposure levels. For those herbicides, four non-accidental acute exposure scenarios were developed for herbivorous insects consuming vegetation contaminated by herbicide residues. The highest anticipated dose was to a small insect consuming broadleaf vegetation, followed by an insect consuming tall or short grass, and lastly, by a large insect consuming fruit.

Contact with Contaminated Soil - Only limited data are available on the toxicity of sulfometuron methyl to microorganisms. The maximum detectable concentrations of sulfometuron methyl in clay soil averages from 0.27 ppm (mg/kg) (rainfall 10") to 0.05 ppm (rainfall 100"). In loam soil, concentrations average 0.387 ppm (rainfall 10") and 0.23 ppm (rainfall 100") and in sandy soils, concentrations average 0.287 ppm (rainfall 10") and 0.014 ppm (rainfall 100") (SERA 2004c, Table 4-2).

Terrestrial Plants (Macrophytes)

Direct Spray and Off-Site Drift - Unintended direct spray will result in an exposure level equivalent to the application rate. Estimates of off-site drift for broadcast and backpack applications of sulfometuron methyl are used in the SERA risk assessment. At the typical application rate of 0.045 lb. a.e./acre in a broadcast application, drift is estimated to result in concentrations of sulfometuron methyl of 0.001575 lb./acre 25 feet from the application site to 0.000094 lb./acre 500 feet from the application site, the furthest distance away where there is still a concern for toxicity to non-target, sensitive plant species. There is only minor concern for tolerant plants at up to 25 feet from the application site. A summary of both the exposure assessment and risk characterization for terrestrial plants from direct spray and off-site drift is in Worksheet G05 in FS WSM ver. 6.00.10.

Runoff and Soil Mobility – Runoff of minor amounts of sulfometuron methyl following broadcast applications, at the typical application rate of 0.045 lb. a.i./acre, is estimated to begin occurring on clay soils at an annual rainfall rate of 15 inches (50 inches on loams and >250 inches on sand). Runoff is estimated to result in concentrations of sulfometuron methyl of 0.000756 lb. a.e./acre at 15 inches of rain to 0.01494 lb. a.e./acre at 100 inches. Adverse effects in sensitive species are plausible at an annual rainfall rate of 15 inches (100 inches for loam soils, with concentrations of 0.00039 lb. a.e./acre) and severe effects are likely at a rate of 100 inches. Runoff becomes problematic for tolerant species at a rainfall rate of 20

inches. A summary of both the exposure assessment and risk characterization for terrestrial plants from runoff is in Worksheet G04 in FS WSM ver. 6.00.10.

Various studies on runoff losses of sulfometuron methyl generally support the supposition that at least 1% could run off from the application site to adjoining areas after a moderate rain and up to 50% could run off in the case of a heavy rain (200 inches), especially in an extremely heavy rain on a steep slope. Runoff will be negligible in relatively arid environments (5-20 inches annual rainfall) as well as in sandy or loam soils, but in regions of California with very high rainfall rates (100 inches), in clay soils, off-site loss may reach up to about 35% of the applied amount.

Contaminated Irrigation Water - There are no studies in the literature addressing the impact of sulfometuron methyl in contaminated irrigation water, but since it is relatively mobile, contamination of ambient water may be anticipated. Based on the estimated concentrations of sulfometuron methyl in ambient water at the typical application rate of 0.045 lb. a.i./acre, the estimated functional application rate of sulfometuron methyl to the irrigated area is 0.0000102 lb. a.e./acre at an irrigation rate of 1 inch per day and 0.000408 lb. a.e./acre at an irrigation rate of 2 inches per day. Relative to off-site drift and runoff, this level of exposure is inconsequential. A summary of both the exposure assessment and risk characterization for terrestrial plants from contaminated irrigation water is in Worksheet G06a in FS WSM ver. 6.00.10.

Wind Erosion - Although no specific incidents of non-target damage from wind erosion have been encountered in the literature for sulfometuron methyl, this mechanism has been associated with the environmental transport of other herbicides. Wind erosion of minor amounts of sulfometuron methyl following broadcast applications, at the typical application rate of 0.045 lb. a.i./acre, is estimated to result in concentrations of sulfometuron methyl of 0.00000308 lb. a.e./acre at the central bound to 0.00000606 lb. a.e./acre at the upper bound. Relative to off-site drift and runoff, this level of exposure is inconsequential and well below a LOC for non-target, sensitive plant species. A summary of both the exposure assessment and risk characterization for terrestrial plants from wind erosion is in Worksheet G06b in FS WSM ver. 6.00.10.

Aquatic Organisms

At the typical application rate of 0.045 lb a.i./acre, the peak estimated rate of contamination of ambient water associated with the normal application of sulfometuron methyl is 0.010 (0.00006 to 0.02) mg a.e./L, while the average estimated rate of contamination for longer-term exposures is 0.00004 (0.00001 to 0.00007) mg a.e./L. sulfometuron methyl is highly soluble in water and is likely to dilute quickly.

1.3.2.3.4.8 Triclopyr (Sources: FS WSM v. 6.00.10; SERA 2011d)

Exposure values for the scenarios displayed below are summarized in the "G" series Worksheets in FS WSM ver. 6.00.10: for mammals (G01a), birds (G01b), honeybee (G09), and insects (G08a). For the analysis in this PEIR, exposure values for triclopyr have been computed for the typical application rate of 1 lb. a.e./acre. Triclopyr TEA and BEE appear to have similar effects on terrestrial organisms.

By far the highest short-term acute exposures to triclopyr are associated with the consumption of contaminated grass by a small mammal (691 mg/kg bw/event) and a small bird (1,710 mg a.e./kg bw/event). The corresponding maximum chronic exposures are 164 mg/kg bw/day for a small mammal and 404 mg a.e./kg bw/day for a small bird. For both acute and chronic exposures, consumption of contaminated water leads to dose estimates far below those associated with consumption of contaminated vegetation. This pattern is common in many herbicide exposure assessments, reflecting the consequences of direct applications to vegetation.

Terrestrial Mammals, Birds, Reptiles, and Amphibians (Terrestrial Phase)

The highest exposures are associated with the consumption of contaminated grasses, and the lowest exposures are associated with the consumption of contaminated water. The exposure assessment for mammals is somewhat more detailed to encompass more diverse body weights. Larger mammals appear to be substantially more sensitive than smaller mammals to triclopyr, experiencing adverse effects at lower doses. As toxicity data on terrestrial phase amphibians are unavailable, exposure assessments for these organisms are not developed.

Direct Spray – At the typical application rate for triclopyr TEA and BEE (values in parentheses), accidental acute exposure scenarios lead to upper bound estimates of exposure for mammals ranging from 1.47 (4.28) mg/kg/event (first-order absorption of direct spray by a small mammal) to 48.5 (48.5) mg/kg/event (100% absorption of direct spray by a small mammal). For birds, no exposure scenarios for direct spray are developed, as it is assumed that most birds will fly away during herbicide applications.

Ingestion of Contaminated Vegetation or Prey - At the typical application rate for both formulations of triclopyr and TCP, non-accidental acute exposure scenarios lead to upper bound estimates of exposure for mammals ranging from 4.62 mg/kg/event (consumption of a small mammal by a canid) to 691 mg/kg/event (consumption of grass by a small mammal). For birds, estimates of exposure range from 5.49 mg/kg/event (consumption of a small mammal by a carnivorous bird) to 1,710 mg/kg/event (consumption of short grass by a small bird).

Chronic exposure scenarios lead to upper bound estimates of exposure for mammals ranging from5.06 mg/kg/day (consumption of fruit by a large mammal) to 164 mg/kg/day (consumption of short grass by a small mammal). For birds, estimates of exposure range from 9.69 mg/kg/day (consumption of fruit by a large bird) to 404 mg/kg/day (consumption of short grass by a small bird).

Fruit and short grass are the food items that define the upper and lower bounds of residue rates. They are not necessarily intended to be interpreted literally, but do encompass the range of triclopyr and TCP concentrations in food items likely to be consumed by a variety of mammals and birds.

Ingestion of Contaminated Water – At the typical application rate, accidental acute exposure scenarios for both formulations of triclopyr and TCP for consumption of contaminated water lead to upper bound estimates of exposure for mammals ranging from1.18 mg/kg/event (large mammal) to 2.66 mg/kg/event (small mammal). For birds, estimates of exposure range from 0.678mg/kg/event (large bird) to 4.90 mg/kg/event (small bird).

Scenarios of non-accidental acute exposure and chronic exposure (values in parentheses) for triclopyr TEA for consumption of contaminated water lead to upper bound estimates of exposure for mammals ranging from 0.0155 (0.00388) mg/kg/event(day) (large mammal) to 0.0351 (0.00878) mg/kg/event(day) (small mammal). For birds, estimates of exposure range from 0.00896 (0.00224) mg/kg/event(day) (large bird) to 0.0647 (0.0162) mg/kg/event(day) (small bird).

Scenarios of non-accidental acute exposure and chronic exposure (values in parentheses) for triclopyr BEE for consumption of contaminated water lead to upper bound estimates of exposure for mammals ranging from 0.00194 (0.00000453) mg/kg/event(day) (large mammal) to 0.00439 (0.0000102) mg/kg/event(day) (small mammal). For birds, estimates of exposure range from 0.00112 (0.00000261) mg/kg/event(day) (large bird) to 0.00809 (0.0000189) mg/kg/event(day) (small bird).

Scenarios of non-accidental acute exposure and chronic exposure (values in parentheses) for TCP for consumption of contaminated water lead to upper bound estimates of exposure for mammals ranging from 0.00181 (0.000129) mg/kg/event(day) (large mammal) to 0.00410 (0.000293) mg/kg/event(day) (small mammal). For birds, estimates of exposure range from 0.00105 (0.0000747) mg/kg/event(day) (large bird) to 0.00755 (0.000539) mg/kg/event(day) (small bird).

For both acute and chronic exposures, contaminated water leads to dose estimates far below those associated with contaminated vegetation. The upper and lower bounds of the estimated concentrations of both triclopyr and TCP in surface water vary by several orders of magnitude (see Table 26 in SERA 2011d). Given this variability, it seems likely that a quantitative

consideration of the variability in water consumption rates of birds and mammals would not have a substantial impact on the risk characterization.

Ingestion of Contaminated Fish - Ambient water and fish are exposure pathways for triclopyr. As triclopyr has a low potential to bioconcentrate in fish, the bioconcentration factor for fish is taken as 0.83 L/kg for chronic exposure scenarios.

At the typical application rate for both formulations of triclopyr and TCP, accidental acute exposure scenarios for consumption of contaminated fish lead to upper bound estimates of exposure for mammals ranging from 2.53 mg/kg/event (large mammalian carnivore) to 3.64 mg/kg/event (canid). For birds, the estimated exposure of a fish-eating bird is 4.23 mg/kg/event.

Non-accidental acute exposure scenarios for triclopyr TEA and BEE (values in parentheses), lead to upper bound estimates of exposure for mammals ranging from 0.0334 (0.00418) mg/kg/event (large mammalian carnivore) to 0.0481 (0.00601) mg/kg/event (canid). For birds, the estimated exposure of a fish-eating bird is 0.0558 (0.00698) mg/kg/event. Chronic exposure scenarios lead to upper bound estimates of exposure for mammals ranging from 0.00835 (0.00000974) mg/kg/day (large mammalian carnivore) to 0.012 (0.000014) mg/kg/day (canid). For birds, the estimated exposure of a fish-eating bird is 0.0000163 mg/kg/day.

Non-accidental acute exposure scenarios for TCP lead to upper bound estimates of exposure for mammals ranging from 0.0039 mg/kg/event (large mammalian carnivore) to 0.00561 mg/kg/event (canid). For birds, the estimated exposure of a fish-eating bird is 0.00651 mg/kg/event. Chronic exposure scenarios lead to upper bound estimates of exposure for mammals ranging from 0.000278 mg/kg/day (large mammalian carnivore) to 0.000401 mg/kg/day (canid). For birds, the estimated exposure of a fish-eating bird is 0.000465 mg/kg/day.

Terrestrial Invertebrates

Direct Spray and Drift - A summary of the exposure assessments and risk characterization for the honeybee for the scenarios of direct spray and drift of both formulations of triclopyr is in worksheet G09 in FS WSM ver. 6.00.10. Exposure from direct spray is shown for three scenarios (0%, 50%, and 90% foliar interception), none of which lead to a HQ above the LOC. The absorbed doses are 68.6, 34.3, and 6.9 mg/kg bw/event, respectively. The absorbed doses from spray drift 25 feet from the application site are2.4, 1.2, and 0.24 mg/kg bw/event, respectively.

Ingestion of Contaminated Vegetation or Prey - Four non-accidental acute exposure scenarios were developed for herbivorous insects consuming vegetation contaminated by residues of both formulations of triclopyr. For a large insect, consuming fruit, the estimated

dose at the typical application rate of 1.0 lb. a.e./acre, is 9.1 mg/kg bw/event (central bound) and 33 mg/kg bw/event (upper bound). For a small insect consuming broadleaf foliage, the estimated dose is 58.5 mg/kg bw/event (central) and 297 mg/kg bw/event (upper). For an insect consuming tall and short grass (the latter value in parentheses), the estimated dose is 46.8 (111) mg/kg bw/event (central) and 242 (528) mg/kg bw/event (upper).

Contact with Contaminated Soil - Only limited data are available on the toxicity of triclopyr to microorganisms. No GLEAMS information was found in SERA 2011d specific to soil concentrations. Based on the GLEAMS modeling, triclopyr TEA may penetrate to about 36 inches in clay, loam, and sand. Because a 36-inch root zone was used in the GLEAMS modeling, the actual penetration in loam or sand could be greater than 60 inches. Triclopyr BEE is much less likely to penetrate the soil column, with a maximum penetration of 24 inches occurring only in sandy soils, cool temperatures, and heavy rainfall. In relatively arid locations, the maximum penetration is estimated at 4-8 inches.

Terrestrial Plants (Macrophytes)

Direct Spray and Off-Site Drift - Unintended direct spray will result in an exposure level equivalent to the application rate. Estimates of off-site drift from broadcast ground applications of triclopyr TEA and BEE are calculated in the SERA 2011d risk assessment. At the typical application rate of 1 lb. a.e./acre, drift is estimated to result in concentrations of triclopyr of 0.035 lb./acre 25 feet from the application site to 0.0177 lb./acre 50 feet from the application site, the furthest distance away where there is still a concern for toxicity to non-target, sensitive plant species. The modeled concentrations of off-site drift are not problematic for tolerant plants at any distance from the application site. A summary of both the exposure assessment and risk characterization for terrestrial plants from direct spray and off-site drift is in Worksheet G05 in FS WSM ver. 6.00.10.

Runoff and Soil Mobility – As stated in SERA 2011d, p. 110: “The runoff for triclopyr TEA as a proportion of the application rate is taken as 0.00266 (0.00001 to 0.108) rounded to 0.0027 to 0.11. The central estimate and upper bound is taken directly from the Gleams-Driver modeling—i.e., the median and empirical upper 95% bound. The lower limit is the approximate lower bound for clay soils in areas with moderate to heavy rain. Although lower loss rates of 1x10-6 to 1x10-8 are plausible, they have no impact on the risk characterization. For triclopyr BEE, the rates, which are similarly derived, are much lower due to the binding of triclopyr BEE to soil—i.e., rates of 0.0006 (2x10-7 to 0.046).” A summary of both the exposure assessment and risk characterization for terrestrial plants from runoff is in Worksheet G04 in FS WSM ver. 6.00.10.

Based on the GLEAMS modeling, triclopyr TEA may penetrate to about 36 inches in clay, loam, and sand. Because a 36-inch root zone was used in the GLEAMS modeling, the actual penetration in loam or sand could be greater than 60 inches. Triclopyr BEE is much less likely

to penetrate the soil column, with a maximum penetration of 24 inches occurring only in sandy soils, cool temperatures, and heavy rainfall. In relatively arid locations, the maximum penetration is estimated at 4-8 inches.

Contaminated Irrigation Water - Triclopyr is slightly mobile and contamination of ambient water is plausible. Based on the estimated concentrations of triclopyr in ambient water at the typical application rate of 1 lb. a.e./acre, the estimated functional application rate of triclopyr TEA, BEE, and TCP (values for BEE and TCP in parentheses) to the irrigated area is 0.00068 (BEE - 0.0000906, TCP – 0.000204) lb. a.e./acre at an irrigation rate of 1 inch per day and 0.1087 (BEE - 0.0136, TCP – 0.0127) lb. a.e./acre at an irrigation rate of 2 inches per day. Relative to off-site drift and runoff, this level of exposure is inconsequential, although at the highest rate of irrigation, adverse effects are plausible to sensitive plants. A summary of both the exposure assessment and risk characterization for terrestrial plants from contaminated irrigation water is in Worksheet G06a in FS WSM ver. 6.00.10.

Wind Erosion - Although no specific incidents of non-target damage from wind erosion have been encountered in the literature for triclopyr, this mechanism has been associated with the environmental transport of other herbicides. Wind erosion of minor amounts of both triclopyr TEA and BEE following broadcast applications, at the typical application rate of 1 lb. a.e./acre, is estimated to result in concentrations of triclopyr of 0.0000685 lb. a.e./acre at the central bound to 0.000137 lb. a.e./acre at the upper bound. Relative to off-site drift and runoff, this level of exposure is inconsequential and well below a LOC for non-target, sensitive plant species. A summary of both the exposure assessment and risk characterization for terrestrial plants from wind erosion is in Worksheet G06b in FS WSM ver. 6.00.10.

Aquatic Organisms

The plausibility of effects on aquatic species is assessed based on estimated concentrations of triclopyr and TCP in water that are identical to those used in the human health risk assessment. At the typical application rate of 1 lb a.e./acre, the peak estimated rate of contamination of ambient water associated with the normal application of triclopyr TEA is 0.12 (0.000001 to 0.24) mg a.e./L, while the average estimated rate of contamination for longer-term exposures is 0.03 (0.0000000002 to 0.06) mg a.e./L. Corresponding values for triclopyr BEE are 0.015 (0.00000015 to 0.03) mg a.e./L, while the average estimated rate of contamination for longer-term exposures is 0.000035 (0.00000000002 to 0.00007) mg a.e./L. Corresponding values for TCP are 0.014 (0.00000001 to 0.028) mg a.e./L, while the average estimated rate of contamination for longer-term exposures is 0.001 (0.0000000000012 to 0.002) mg a.e./L.

1.3.2.4 Dose-Response Assessment

U.S. Forest Service risk assessments attempt to define dose-response relationships for all classes of organisms discussed in the hazard identification section, such as mammals, birds,

reptiles, amphibians (terrestrial and aquatic phases), terrestrial and aquatic invertebrates and macrophytes, microorganisms, fish, and algae (SERA 2012). When there is enough acceptable data to permit doing so, sensitivity differences between species within each class are also considered in USDA/FS risk assessments for each chemical. Additional relationships are also evaluated, as specified below.

Studies report toxicological effect results in several ways. For example, some studies are designed to identify acute hazards while determining the dose or concentration of a chemical that will cause death in an "X" percentage (i.e., most commonly 25% or 50%) of a defined experimental animal population over a specific observation period. When doses for such a study are administered through gavage, diet, or dermal methods, results are expressed as a "Lethal Dose" or LD. When aquatic organisms are exposed to chemically treated water, or terrestrial organisms are dosed through inhalation of chemically treated air for such a study, the results are recorded as "Lethal Concentration" or LC. The LD or LC is then followed by a subscripted percentage of lethality. Thus, if 1,500 milligrams of a chemical (i.e., per kilogram of body weight) had been fed to a population of experimental rats and proved fatal to 50% of that population, the lethal dose would be LD_{50} = 1500 mg/kg bw. However, if 1500 mg was the maximum dose tested in the study and the dose was not lethal to any rats, then the infinite lethal dose, LD_{50} >1500 mg/kg bw, would be assigned. Similarly, sublethal effects may be recorded as "effect dose" or "effect concentration", with a subscript percent to indicate the dose causing "X"% inhibition of a process.

Results may also be recorded in terms of *lowest-observed-adverse-effect-level* or *concentration* (LOAEL or LOAEC), as well as by *no-observed-adverse-effect-level* or *concentration* (NOAEL or NOAEC). As implied, LOAEL values indicate the lowest dose an adverse effect occurred and, by contrast, the NOAEL is the lowest dose administered that did not result in an adverse effect. It should be noted that in some studies both the *no-observed-effect-level* or *concentration* (NOEL/NOEC) and associated LOEL and LOEC values are recorded. These values indicate *any* effect, though for all practical purposes these terms may be considered synonymous with respective NOAEL and NOAEC or LOAEL and LOAEC terms. In reference to wildlife, results reported using the terms "dose" and "level" generally refer to studies on terrestrial organisms, whereas results expressed as "concentration" are usually reserved for aquatic organisms. All such results that function to define the occurrence of toxicological effects, or lack thereof, are collectively referred to as endpoints.

The USDA/FS predominantly utilizes five different methods to assess dose-response relationships. In order of increasing complexity, these methods include 1) Point Estimates and 2) Extreme Values (SERA 2012). Point estimates involve making use of only values that specifically evaluate for sublethal effects rather than just for lethality. Ideally, to establish point estimates, "the study should define both a NOAEL and a LOAEL and there should be reasonable confidence that the NOAEL involves endpoints that would not impair the ability of the organism to function normally over a short-term period" (SERA 2012, p. 98). In cases

where LD_{50} or LC_{50} values are the only ones available, an LD_{50} is divided by 10 to estimate an NOAEL for mammals and birds, whereas an LC_{50} is divided by 20 to estimate an NOAEL for aquatic organisms. The extreme value method involves making use of a range of values that include a central estimate, with upper and lower bounds, for toxicity and exposures. This approach also applies when evaluating studies of the same taxonomic group to decide if the highest and lowest NOAEL values represent, respectively, the most tolerant and sensitive species.

The next three methods commonly utilized in U.S. Forest Service risk assessments include: 3) Relative Potency, 4) Species Sensitivity Distributions, and 5) Allometric Relationships (SERA 2012). The relative potency method makes use of ratios for toxicity to calculate values for missing data. If a data set is complete for a tolerant species, for example including both acute and chronic endpoints, but only acute information is available for the sensitive species, the ratio of acute to chronic data for the sensitive species can be used to calculate an estimated chronic endpoint. Species sensitivity distributions are utilized when data are occasionally available to suggest more refined estimates in gradations of sensitivity within and among species. It should be noted that the dose differences between tolerant and sensitive species within the same class of organisms are often limited by how many species have been tested. Allometric relationships are those that relate body size or mass to any number of characteristics (i.e., anatomical, physiological, or pharmacological). One example of an allometric relationship applicable to this risk assessment is that larger mammals are more sensitive to adverse effects associated with triclopyr exposure than their smaller counterparts (SERA 2011d).

Most toxicological endpoints applied in U.S. Forest Service risk assessments are typically those used by the U.S. EPA, which are obtained from registrant-submitted studies. These endpoints, however, are altered, supplemented or replaced in USDA/FS risk assessments when evidence warrants that changes are necessary. The USDA/FS apply endpoints used by the U.S. EPA whenever possible, though there are some distinct differences in how values are used, as discussed in SERA 2012 (p. 97): "*As in the human health risk assessment, the Forest Service will consider, discuss, and sometimes defer to dose-response assessments developed in ecological risk assessments developed by the U.S. EPA/OPP. Also, as in the human health risk assessment, this approach avoids a duplication of effort, capitalizes on the substantial expertise of U.S. EPA/OPP, and decreases the size, complexity, and cost of Forest Service risk assessments. There are, however, important differences between the approach taken by U.S. EPA/OPP and the approach preferred by the Forest Service. The Forest Service prefers to use NOEC values for both acute and chronic exposures. This differs from the U.S. EPA/OPP which will base dose-response assessments for acute exposures on LC_{50} or EC_{50} values. Nonetheless, the Forest Service assessment will adapt (slightly modify) the methods used by U.S. EPA/OPP, as detailed further below, for data sets in which only LC_{50} or EC_{50} values are available.*"

As briefly mentioned above when discussing the five methods (relationships) utilized by the U.S. Forest Service, there are several ways that values reported by the U.S. EPA may be adapted in U.S. Forest Service risk assessments. The risk assessment for triclopyr (SERA 2011d) describes some examples of how these adaptations may be done with aquatic organism data. These modifications are also applicable to terrestrial organism results that are reported as NOAEL, LD_{50}, and ED_{50} values:

> *If NOAECs are not available, LC_{50} or EC_{50} values may be multiplied by 0.05 to approximate an NOAEC. This procedure is based on the U.S. EPA/OPP general approach of using LC_{50} or EC_{50} values with levels of concern (LOC) of 0.05 for the ratio of exposure to the LC_{50} or EC_{50} for endangered species (e.g., U.S. EPA/OPP 2009a, Appendix C). It should be noted that this is a very conservative approach, equivalent to treating all aquatic species as endangered species.*
>
> *As noted in several instances below, an intermediate approach can be taken to estimate NOAECs for sensitive and tolerant species. When there is not an NOAEC for the most sensitive or most tolerant species within a group of organisms, but there is either an LC_{50} or EC_{50} with a corresponding NOAEC for one or more other species in the group, the ratio of the available NOAEC to the available LC_{50} or EC_{50} can be used to estimate an NOAEC for the most sensitive or tolerant species.*
>
> *Few chronic NOAECs are available for any group of aquatic organisms. For some groups (e.g., algae), the lack of a chronic NOAEC is not a concern, because chronic is not meaningful in the context of exposure for organisms with very short lifespans. For fish and invertebrates, however, attempts are made to incorporate the very well-documented variability in acute data into the chronic dose-response assessment. Consequently, acute-to-chronic ratios are developed for the species on which both acute and chronic toxicity data are available; furthermore, these ratios are used to estimate chronic NOAECs for sensitive and tolerant species. As detailed below, this approach is used only when it appears to be sensible given the available species-specific data. (SERA 2011d, p. 118)*

Endpoints are established in U.S. Forest Service dose-response assessments using a few more approaches. Values from one organism class may be applied to an organism from a different class, as a surrogate endpoint. If acute data for mammals and birds, for example, indicate that a chemical is equally toxic to each class of organism, but there is no chronic NOAEL established for birds, a rat NOAEL may be used as a surrogate endpoint, if all other data supports the assumption of equivalency. Additionally, in limited instances LOAEL or LOAEC may also be used in the absence of other, more conservative data. In cases when there is not enough data to support a dose response assessment using U.S. Forest Service methods, or data is limited for a class of organisms, qualitative information from available

studies will be discussed in depth in the risk characterization section of the applicable risk assessment.

This section functions to summarize the endpoint values for class of organisms, by chemical. Endpoints for terrestrial organisms including mammals, birds, invertebrates, and plants (macrophytes) are disclosed in tables for each chemical when data is available (Tables 5.17.30 – 5.17.42). Likewise, each table also includes values for aquatic organisms, such as fish, amphibians, invertebrates, plants (macrophytes) and algae (microphytes). When information is available from U.S. Forest Service risk assessments, these tables also summarize test species (aka receptor), the form of active ingredient used in a study, and how the endpoints were derived or adapted. Any additional information particularly pertinent to dose-response values will be briefly paraphrased from U.S. Forest Service overviews in the dose-response section of each chemical. For information regarding studies evaluated, explanations regarding the choice of endpoints, or details regarding how chosen values were adjusted for USDA/FS risk assessments, consult the appropriate SERA risk assessments. For background information regarding SERA risk assessment methodology, refer to SERA 2012. For this PEIR, dose response values determined to be appropriate by the U.S. Forest Service are adopted without reservation, for similar reasons that the U.S. Forest Service opts to rely on information released by the U.S. EPA.

After exposures are calculated in the exposure assessment and maximum doses that lack adverse effects are determined for each chemical in the dose-response assessment, risk will be evaluated in the risk characterization section, in part using Hazard Quotient (HQ) values. A HQ is the ratio of an exposure level to a toxicity value and is analogous to the Risk Quotient (RQ) values used to assess risk to human health in U.S. EPA risk assessments. Both HQ and RQ values function to quantitatively express risk characterization. As with human risk studies, ecological risk studies used by the U.S. EPA are acceptable under specific guidelines and protocols for each organism being assessed for risk. For the human health assessment, NOEL, NOEC, or other toxicity values are divided by an uncertainty factor to derive a reference dose for each endpoint. By contrast, uncertainly factors are not used for ecological risk assessment. Instead, values are often used directly, or in some instances divided by factors to account for a level of concern (LOC) or an endangered species.

1.3.2.4.1 Terrestrial Organisms

For terrestrial organisms, the dose-response assessment is most complete for mammals and terrestrial plants. This is likely due to the direct applicability of mammal studies to human health risks, and to the chemicals evaluated in this PEIR being predominately used to alter terrestrial plant growth. Other terrestrial organisms often have little to no dose-response information available relative to plants and mammals. Acceptable lifetime or chronic studies are seldom available for these other classes of terrestrial organism. Details regarding each class and the assumptions used by the U.S. Forest Service are summarized from each

applicable chemical risk assessment, as well as SERA (2012). The latter document provides details of USDA/FS methodology.

Mammals and Birds – The dose-response assessment for mammals is generally based on the same values used to derive reference doses (RfDs) in the human health dose response section. Typically, these data are on non-canine mammals, such as rats and rabbits, since dogs are unable to excrete weak acids to the same extent and thus are often more severely affected than most other mammals. When considering the comparability of different types of mammalian and avian studies, gavage application methods tend to produce greater toxicological effects compared to dietary ingestion of a chemical. When available, results from dietary studies are usually preferred over those involving gavage applications. This is in part because gradual intake through consumption of food is most ecologically relevant in most cases.

Reptiles and Amphibians (Terrestrial Phase) – The U.S. EPA does not require standard toxicity studies on reptiles or terrestrial-phase amphibians. Currently, no information is available regarding toxicity to reptiles for any of the chemicals proposed in the PEIR. If no acceptable studies are available for risk characterization for terrestrial-phase amphibians, no formal dose-response assessment is developed. Information regarding terrestrial phase amphibians is very limited and contributes most to dose-response assessment of aquatic phase amphibians. Thus, all information regarding amphibian exposure is discussed under the aquatic section for each chemical in this PEIR.

Terrestrial Invertebrates – Acute toxicity values from honey bees are often used as surrogate values for other terrestrial insects. Given the numerous species of terrestrial invertebrates, the use of this single acute toxicity value on a single species obviously leads to uncertainty in the risk assessment. U.S. Forest Service risk assessments also attempt to characterize risks to terrestrial invertebrates from the consumption of contaminated vegetation following broadcast applications (i.e., direct spray). The results of oral toxicity studies in honeybees are typically used to assess risks associated with this scenario (SERA 2011c). Results of contact toxicity studies in honeybees are often used as surrogate toxicity values to characterize risks to herbivorous insects from the consumption of contaminated vegetation (SERA 2011c). Most honeybee results are reported in units of µg chemical/bee, and in USDA/FS risk assessments that value is divided by the average honeybee body weight (bw) of 116 mg to convert the result into units of mg/kg bw for risk characterization.

Terrestrial Plants (Macrophytes) – The assessment of potential effects in plants is based on standard toxicity studies required for pesticide registration, involving pre-emergence and post-emergence exposures. All the herbicides are designed to adversely affect specific plant physiological processes in specific ways. Each herbicide is targeted to specific plant groups, as specified on the herbicide labels. Non-targeted plant groups will generally experience fewer adverse effects than those that are targeted. To assess the potential consequences of

exposures to nontarget plants via transport of runoff or sediment or through direct soil treatment, the values reported from seedling emergence (pre-emergence application) bioassays are used (USDA/FS 2006a). To assess the impact of drift (accidental direct spray) on nontarget terrestrial vegetation, the values reported from the post-emergent (vegetative vigor) bioassays are used (USDA/FS 2006a).

Terrestrial Microorganisms - For the purposes of this risk assessment, terrestrial microorganism refers to terrestrial bacteria, fungi and in some cases, heterotrophic algae and green algae. Given the limited testing done to evaluate toxicological effect of proposed chemicals on such organisms, little specific endpoint data will be presented in tables, but instead a summary will be included in this subsection for each chemical, when information is available.

1.3.2.4.2 Aquatic Organisms

For some aquatic species, as well as other groups of organisms, sensitive life-stage studies are often available. Such studies include egg-and-fry studies in fish and life-cycle toxicity studies in Daphnia magna, both of which are typically required by the U.S. EPA for the registration of herbicides. U.S. EPA toxicity categories assigned to aquatic species have the same caveats regarding the limitations of applying data from surrogate species tested in controlled situations to wild populations (see SERA 2005, p. xviii). Note that variation in toxicity values for aquatic species may be based more on the conditions of exposure, particularly the pH of water, than on differences between species (SERA 2011b).

Fish - The three general types of relatively standardized studies most commonly used by the U.S. Forest Service, which follow standard U.S. EPA study protocols, include acute toxicity studies, egg-and-fry studies, also referred to as early lifestage studies, and full life cycle studies (SERA 2012, p. 4-8). There is also extensive open source literature available on fish species that is consulted as needed for U.S. Forest Service risk assessments.

Amphibians (Aquatic Phase) –While studies are not required by the U.S. EPA at this time, the U.S. Forest Service uses the following approach to evaluating risks of chemical exposure to amphibians: *Because of the relative sparsity of data available on toxic effects to amphibians and the high level of concern with effects on amphibians because they may be good indicator species, any available information on effects to amphibians are typically reviewed in some detail. If the data are sufficient, these data are used in the dose-response assessment* (SERA 2012, p. 4-8). See also the “Reptiles and Amphibians” section above.

Aquatic Invertebrates – As stated in SERA (2012 p. 4-8): *Many aquatic invertebrates are relatively simple organisms to culture and test in aquatic toxicity studies, and standard acute toxicity protocols from U.S. EPA/OPPTS (2005) are available on a number of invertebrate species: daphnids (OPTTS 850.1010), gammarids (OPTTS 850.1020), oysters (OPTTS 850.1025), mysid shrimp (OPTTS 850.1035), penaeid shrimp (OPTTS 850.1045), and*

several species of bivalves (OPTTS 850.1055). These tests are similar in design to acute toxicity studies in fish (Section 4.1.3.1), although some may involve somewhat shorter periods of exposure – e.g., the daphnid study typically only lasts for 48 hours.

Aquatic Plants (Algae and Macrophytes) – As stated in SERA (2012 p. 4-9): Aquatic plants comprise both macrophytes (large multicellular plants) and algae (small microscopic plants). Bioassays in aquatic algae typically involve freshwater green alga (Selenastrum capricornutum or Raphidocelis subcapitata), a freshwater diatom (Navicula pelliculosa), a marine diatom (Skeletonema costatum), and a blue-green alga or cyanobacterium (Anabaena flos-aquae). Bioassays on macrophytes typically use a species of duck weed (e.g., Lemna gibba). The duration of exposure for algae is typically 48-hours and the duration for duckweed is typically about 7-days. Both types of studies measure growth (either as cell count or gross weight) and express results as effective concentrations (e.g., EC_{50}) rather than lethal concentrations (e.g., LC_{50}). As with most other types of bioassays, the studies often report NOEC and LOEC values, and NOEC values are typically used in the dose-response assessment.

Aquatic Microorganisms – The assessment of aquatic microorganisms is the same as for terrestrial microorganisms, except that algae are included in the assessment for aquatic plants.

1.3.2.4.3 Chemical-Specific Dose-Response Assessment

1.3.2.4.3.1 Borax (Sources: FS WSM ver. 6.00.10; SERA 2006a)

Dose-response endpoints for borax are summarized in Table D.3-19. Dose response assessments are supported for ten classes of organisms in the U.S. Forest Service risk assessment for borax: terrestrial mammals, birds, non-target terrestrial invertebrates, terrestrial macrophytes, fish, aquatic invertebrates, amphibians, aquatic macrophytes, algae, and aquatic microorganisms.

There is relatively little difference in acute toxicity values between fish and aquatic invertebrates. For chronic exposures, however, fish appear more sensitive than aquatic invertebrates to boron exposure.

Mammals and Birds - Borate compounds are relatively non-toxic to mammals and birds. For mammals, the toxicity values used in the ecological risk assessment are identical to those used in the human health risk assessments: the 95% lower bound on the dose corresponding to the benchmark response (BMR) level, i.e., the $BMDL_{05}$, of 10.3 mg B/kg/day (the *critical dose*) for decreased fetal body weight. The acute NOAEL for birds was taken at the highest dose given during a 5-day dietary study, as no clinical signs of toxicity occurred. For chronic exposure of birds, the limited data available suggest that longer-term exposure to boron compounds can cause testicular toxicity in avian species. However, the available studies did

not rigorously investigate the potential for boron compounds to produce testicular toxicity. Therefore, the mammalian *critical dose* of 10.3 mg B/kg/day will be used to characterize the risk of chronic exposure to boron compounds in birds.

Terrestrial Invertebrates – A honey bee study that evaluated mortality relative to a single contact was used as a NOAEL for this class of organism.

Terrestrial Plants (Macrophytes) - Boron is known to be an essential element for plants, though data specifically evaluating the effects of borax on seedling emergence and vegetative vigor are limited. It is likely that a wider range of plant sensitivity exists.

Terrestrial and Aquatic Microorganisms – *No formal dose-response assessment was completed for terrestrial microbes due to a lack of acceptable studies. Available microbe studies will be used to qualitatively assess these organisms in the risk characterization section.* In terms of terrestrial organisms, borax is used as an anti-fungal treatment, so some soil microbes could be affected by borax exposure, though such data is limited. For aquatic microorganisms, the NOAEC values of 0.3 mg B/L and 291 mg B/L are used to assess the consequences of both acute and longer-term exposures for sensitive and tolerant species of aquatic microorganisms.

Fish - In fish the range of NOAEC values is relatively narrow, with the difference between sensitive and tolerant species being only 0.05 ppm (1.0 - 0.5 ppm).

Table D.3-19

Ecological Endpoints for Borax

			Endpoint	Receptor, Study & Endpoint Details
Canine mammals	Acute		***N/A***	No data available
	Chronic		***N/A***	
Medium mammals	Acute		***NOAEL = 10.3 mg B/kg bw***	chronic endpoint is surrogate
	Chronic		***adjusted NOAEL = 10.3 mg B/kg bw/day***	rat, borates
Small mammals	Acute		***N/A***	No data available
	Chronic		***N/A***	
Large herbivore mammals	Acute		***N/A***	
	Chronic		***N/A***	
Birds	Acute		***NOAEL = 136 mg B/kg bw***	bobwhite quail, borax
	Chronic		***surrogate NOAEL = 10.3 mg B/kg bw/day***	rats, borates; based on a benchmark response (BMR) level and used as the critical dose.
Terrestrial Invertebrates	Acute		***single contact NOAEL = 677 mg B/kg bw***	honey bees, boric acid, for mortality; also used as a surrogate for herbivorous insects
Terrestrial Plants (Macrophytes)	Seedling Emergence	sensitive spp.	***NOAEC = 5 B/kg soil***	potato, boric acid
		tolerant spp.	***NOAEC = 20 B/kg soil***	sugar beet, boric acid
	Vegetative vigor	sensitive spp.	***N/A***	seedling emergence values are equivalent, as the only method of application is direct stump application for borax
		tolerant spp.	***N/A***	
Aquatic Microorganism	sensitive spp.		***NOEC =0.3 mg/L***	*Entosiphon sulfacum*, a flagellate
	tolerant spp.		***NOEC =291 mg/L***	*Pseudomonas putida*
Fish	Acute	sensitive spp.	***LC50 = 233 mg B/L***	razorback sucker swimup fry, boric acid

		tolerant spp.	***LC50 > 1,100 mg B/L***	rainbow trout, boric acid
	Chronic	sensitive spp.	***NOAEC = 0.5 mg B/L***	goldfish, borax
		tolerant spp.	***NOAEC = 1 mg B/L***	rainbow trout /channel catfish, borax
Amphibians	Acute	sensitive spp.	***NOAEC = 1.0 mg B/L***	leopard frog larvae, borax, NOAEC = 1.0, sensitive vs. tolerant species not identified
		tolerant spp.		
	Chronic	sensitive spp.	***N/A***	No chronic exposure studies were identified or surrogate values in the risk assessment; chronic NOAEC values were listed in FS WSM
		tolerant spp.	***N/A***	
Aquatic invertebrate	Acute	sensitive spp.	***LC50 = 133 mg B/L***	*Daphnia magna,* boric acid
		tolerant spp.	***LC50 = 1,376 mg B/L***	*Chironomas decorus*, freshwater midge, borax
	Chronic	sensitive spp.	***NOEC = 6.0 mg/L***	*Daphnia magna*, boric acid
		tolerant spp.	***surrogate NOEC = 61.8 mg/L***	*Chironomas decorus*, midges are more tolerant than daphnids by a factor of 10.3 (1,376/133) [derived by daphnid NOAEC of 6 mg B/L x 10.3]
Aquatic Plants (Macrophytes)	Acute	sensitive spp.	***EC50 = 5 mg/L***	water milfoil and waterweed, boric acid, 21-day study
		tolerant spp.	***EC50 = 10 mg/L***	water buttercup, boric acid, 21-day study
	Chronic	sensitive/ tolerant spp.	***N/A***	No data available
Aquatic Algae (Microphytes)	sensitive spp.		***NOEC = 10 mg/L***	green alga, (*Scenedesmus subpicatus)*, unspecified chemical spp. of boron
	tolerant spp.		***NOEC = 20.3 mg/L***	blue-green alga, (*Microcystis aeruginosa*), unspecified chemical spp. of boron

All endpoints are in terms of a.i. ED/C = Effect Dose/Concentration, LD/C = Lethal Dose/Concentration, N/A = Not Applicable, NOAEL/C = no-observed-adverse-effect-level/concentration.

Amphibians – To characterize acute risk in amphibians, only a single study in leopard frog larvae is used. Appropriate chronic data is lacking.

Aquatic Invertebrates – Unlike in fish, the dose range for sensitivity of aquatic invertebrates is much wider, with a difference of about 55 mg B/L (61.8 – 6 mg B/L).

Aquatic Plants (Algae and Macrophytes) – Sensitivity of algae ranged from 10 to 20.3 mg B/L. These sensitive and tolerant concentrations were applied to both short and long-term concentrations due to the short lifespan of individual algal cells. For aquatic macrophytes, 21-day exposure studies yield a range of values from 5 to 10 mg B/L. These values will be used to assess acute exposure risk to sensitive and tolerant aquatic macrophytes.

1.3.2.4.3.2 Clopyralid (Sources: FS WSM ver. 6.00.07 & 6.00.10; SERA 2004a)

Dose-response endpoints for clopyralid are summarized in Table D.3-20. Dose response assessments are fully supported for a few classes of organisms in the U.S. Forest Service risk assessment for clopyralid: terrestrial mammals, terrestrial macrophytes, fish, aquatic invertebrates, amphibians, aquatic macrophytes, algae, and aquatic microorganisms. There is only acute data for several classes, such as birds, bees, fish, aquatic macrophytes and algae. Currently, there is a lack of data regarding toxicological effects of clopyralid on amphibians.

Mammals and Birds – A comparison of gavage studies between mammals and birds suggest that birds may be more sensitive than mammals by a factor of about 3. However, based on a comparison of short-term dietary NOAELs, birds appear to be somewhat less sensitive, with an acute dietary NOAEL of about 670 mg/kg/day, a factor of about 9 above the acute NOEL of 75 mg/kg/day for mammals. These more ecologically relevant dietary NOAEL values are those chosen for dose response. No chronic toxicity studies have been completed in birds at dosages as high as the chronic NOAEL of 15 mg/kg/day for rats, which are used as a surrogate for chronic exposure of birds.

Terrestrial Invertebrates – Values relating to honey bee exposure are used to represent the effects clopyralid may have on terrestrial invertebrates.

Terrestrial Plants (Macrophytes) - Clopyralid is more toxic to broadleaf plants than to grains or grasses and is more toxic in post-emergence applications (i.e., foliar spray) than pre-emergence applications (i.e., soil treatment). For assessing the potential consequences of exposures to nontarget plants via runoff, the NOEC values for seed emergence are used for sensitive species (0.025 lb a.e./acre) and tolerant species (0.5 lb a.e./acre). For assessing the impact of drift, bioassays on vegetative vigor are used, with NOEC values of 0.0005 lb/acre for sensitive species and 0.5 lb/acre for tolerant species.

Terrestrial and Aquatic Microorganisms – No formal dose-response assessment was completed for terrestrial or aquatic microbes due to a lack of acceptable studies. Available terrestrial microbe studies will be used to qualitatively assess these organisms in the risk characterization section. A NOEC for soil microorganisms was established for clopyralid at concentrations of 10 ppm, based on effects relating to nitrification, nitrogen fixation, and degradation of carbonaceous material. This NOEC is much higher than anticipated for concentrations of clopyralid in soil.

Fish - No chronic studies, or even long-term studies, on fish egg- and-fry have been encountered. The dose-response assessment uses admittedly limited data, suggesting that at least some fish species may be more sensitive to clopyralid than daphnids. The chronic value for tolerant species was adopted directly from the daphnid study.

Amphibians – Neither the published literature nor the U.S. EPA files include data regarding the toxicity of clopyralid to amphibian species. No formal dose-response assessment was completed for amphibians due to a lack of acceptable studies.

Aquatic Invertebrates – A limited dataset may indicate that daphnia may be more tolerant than some fish species.

Aquatic Plants (Algae and Macrophytes) - For sensitive aquatic plants, risk is characterized using the lowest reported EC_{50} of 6.9 mg a.e./L. Conversely, for tolerant aquatic plants, the highest reported EC_{50} of 449 mg/L is used. The available data on aquatic plants are not sufficient to support separate dose-response assessments for macrophytes and algae.

Table D.3-20

Ecological Endpoints For Clopyralid

			Endpoint	Receptor, Study & Endpoint Details
Canine mammals	Acute		***N/A***	Dog studies resulted in inconsistent results; no canine endpoints established
	Chronic		***N/A***	
Medium mammals	Acute		***NOAEL = 75 mg/kg bw***	rat, 11-day gavage study
	Chronic		***NOAEL = 15 mg/kg bw/day***	rat, 2-year dietary study
Small mammals	Acute		***N/A***	No data available
	Chronic		***N/A***	
Large herbivore mammals	Acute		***N/A***	
	Chronic		***N/A***	
Birds	Acute		***NOAEL = 670 mg/kg bw***	quail and ducks, 5-day dietary studies [NOAEL rounded from 696 mg/kg/day]
	Chronic		***surrogate NOAEL = 15 mg/kg bw/day***	No lifetime toxicity studies in birds, and thus the chronic mammal exposure NOAEL is applied, surrogate is a 2-year dietary study with rats
Terrestrial Invertebrates	Acute		***NOAEL = 909 mg/kg bw***	honey bee (*Apis mellifera)*
Terrestrial Plants (Macrophytes)	Seedling Emergence	sensitive spp.	***NOEC = 0.025 lb/acre***	soy bean
		tolerant spp.	***NOEC = 0.5 lb/acre***	several spp.
	Vegetative vigor	sensitive spp.	***NOEC = 0.0005 lb/acre***	soybean, snap bean, tomato, sunflower
		tolerant spp.	***NOEC = 0.5 lb/acre***	barley, corn, radish, canola

Fish	Acute	sensitive spp.	***LC50 = 103 mg/L***	rainbow trout, (*Salmo gairdneri*) clopyralid acid
		tolerant spp.	***LC50 = 1,645 mg/L***	rainbow trout, bluegill sunfish, and fathead minnows, clopyralid monoethanolamine salt
	Chronic	sensitive spp.	***surrogate NOEC = 10 mg/L***	no fish data [derived from daphnid study: 23.1 mg a.e./L divided by 2, then rounded to 1 significant digit]
		tolerant spp.	***NOEC = 23.1 mg/L***	no chronic fish studies, *Daphnia* value accepted directly
Amphibians	Acute	sensitive spp.	***N/A***	No data available
		tolerant spp.	***N/A***	
	Chronic	sensitive spp.	***N/A***	
		tolerant spp.	***N/A***	
Aquatic invertebrate	Acute	sensitive spp.	***NOEC = 23.1 mg/L***	*Daphnia magna*, used from one existing study, which examined chronic exposure. Sensitivity was not specified
		tolerant spp.		
	Chronic	sensitive spp.	***NOEC = 23.1 mg/L***	
		tolerant spp.		
Aquatic Plants (Macrophytes)	Acute	sensitive spp.	***NOEC = 0.1 mg/L***	water milfoil (*Myriophyllum sibiricum)* and sago pondweed, (*Potamogeton pectinatus)*
		tolerant spp.	***NOEC = 0.1 mg/L***	
	Chronic	sensitive/ tolerant spp.	***NOEC = 0.1 mg/L***	available data on aquatic plants are not sufficient to support dose-response assessments for macrophytes
Aquatic Algae (Microphytes)	sensitive spp.		***EC50 = 6.9 mg/L***	green algae (*Selanastrum capricornutum)*
	tolerant spp.		***EC50 = 449 mg/L***	green alga

All endpoints are in terms of a.e. ED/C = Effect Dose/Concentration, LD/C = Lethal Dose/Concentration, N/A = Not Applicable, NOAEL/C = no-observed-adverse-effect-level/concentration.

1.3.2.4.3.3 Glyphosate (Sources: FS WSM v. 6.00.10; SERA 2011b; U.S. EPA. 2009c)

As discussed in several sections, there are often substantial differences between the toxicity of some formulations that contain surfactants like POEA and those that do not, such as technical grade glyphosate, Accord, and Rodeo. While the available information does not permit formulation-specific toxicity values, an attempt is made in the U.S. Forest Service risk assessment to discriminate between less toxic and more toxic formulations. For details regarding what and how formulations were categorized, see SERA 2011b. In general, formulations clearly identified as *Low Toxicity* are less toxic, while all other formulations are regarded as more toxic.

For most ecological receptors, apart from plants, separate toxicity values can be derived for less and more toxic glyphosate formulations, as indicated in Tables D.3-21 and Table D.3-22. The dose-response assessment for terrestrial plants assumes that the surfactants added to all formulations of glyphosate will result in equal efficacy among formulations. While less toxic formulations typically do not contain surfactants, labels on these formulations specify that surfactants must be added to the field solution prior to application. The surfactants added have the potential to be more toxic than the initial formulation, and thus may become the dominant toxicological concern, especially for aquatic species. The impact of using surfactants with less toxic formulations of glyphosate is discussed in the risk characterization. The dose-response assessments for the less toxic surfactants are based on the toxicity of glyphosate, salts of glyphosate, and the information on the toxicity of the less toxic formulations of glyphosate.

Mammals and Birds – Whether evaluating toxic formulations, chronic exposure to glyphosate appears to be somewhat more toxic to mammals than birds. For chronic toxicity, the difference between more and less toxic formulations is narrower for mammals (325 mg/kg bw/day) than for birds (960 mg/kg bw/day).

Terrestrial Invertebrates – Studies indicate that more toxic formulations have a greater oral and contact exposure toxicity to honey bees than less toxic formulations, by factors of >3 and 2 respectively.

Terrestrial Plants (Macrophytes) – The glyphosate formulations are more toxic to plants than technical grade glyphosate. It is reasonable to assume that the increased toxicity is attributable to the surfactants in the formulations. The dose-response assessment for terrestrial plants assumes that the surfactants added to all formulations result in equal efficacy among formulations. No distinction is made between less toxic and more toxic surfactants, and the assessment is based only on the toxicity data involving glyphosate formulations. Foliar exposures in the range of 0.7 lbs/acre may have long-term impacts on bryophyte and lichen communities. Glyphosate is much less toxic and less effective as an herbicide in soil exposures.

Terrestrial and Aquatic Microorganisms *– No formal dose-response assessment was completed for either group of microbes due to a lack of acceptable studies. Available terrestrial microbe studies will be used to qualitatively assess these organisms in the risk characterization section.* For terrestrial organisms, studies show that glyphosate inhibits microbial growth in laboratory cultures, causes transient decreases in populations of soil fungi and bacteria after field applications of ~0.5 lbs/acre), and results in increases in soil microorganisms or microbial activity.

Fish - There is no indication of a pronounced duration-response relationship in fish from glyphosate or glyphosate formulations. Any sublethal effects that were observed from chronic exposure to more toxic formulations were encompassed by the 0.048 and 0.5 mg a.e./L surrogate NOEC values derived for acute toxicity for more toxic formulations. Similarly, chronic exposure to less toxic formulations did not indicate a dose response relationship. Thus, the acute values for both more and less toxic formulations were maintained for respective chronic exposure values.

Amphibians - Based on the acute bioassays with the more toxic formulations of glyphosate, the sensitivities of fish and aquatic-phase amphibians to glyphosate appear to be virtually identical. For the more toxic formulations of glyphosate, the dose-response assessment for amphibians is developed in the same manner as for fish, which involves the LOC approach used by the U.S. EPA (i.e., multiplying by a RQ of 0.05 and rounding the outcome). As with the dose-response assessment for fish, for more toxic formulations the surrogate acute NOAEC values for amphibians are applied to longer-term exposures. The dose-response assessment for acute exposures of amphibians to less toxic formulations is similar to that of fish.

Evidence indicates that glyphosate IPA is less acutely toxic than glyphosate acid to amphibians and that the differences between the toxicity of glyphosate IPA and glyphosate acid relates to the pH of water. Unlike with fish, the above data are sufficiently compelling to assert that the lower toxicity values for glyphosate acid are not appropriate for the dose-response assessment. All the less toxic formulations of glyphosate likely to be used in U.S. Forest Service programs contain glyphosate IPA as the active ingredient. Consequently, for amphibians the dose-response assessment for less toxic formulations are based on studies using glyphosate IPA. No sublethal toxicity studies have been identified on glyphosate IPA, Rodeo, or equivalent formulations. The lack of more detailed sublethal toxicity studies on glyphosate IPA, Rodeo, and other similar formulations is treated qualitatively as a data gap. The dose-response assessment for longer-term exposures of amphibians to less toxic formulations is extremely simple, in that only one longer-term study (i.e., using glyphosate IPA) is available. Given the limited data, sensitive and tolerant species could not be distinguished for chronic exposure to less toxic formulations.

Aquatic Invertebrates - As with fish and amphibians, for more toxic formulations the first approximation to estimating NOAEC values is made by multiplying the range of acute EC_{50} values by a factor of 0.05. Existing data for more toxic formulations does not indicate a dose-response relationship, and thus the acute values are also applied to chronic exposure for less toxic formulations. As discussed above, the acute toxicity data for glyphosate acid and glyphosate IPA in amphibians indicate that glyphosate IPA is less toxic than glyphosate acid, probably due to variable water pH. For aquatic invertebrates, the studies on the toxicity of glyphosate acid relative to glyphosate IPA are not consistent. While all evidence is evaluated in the U.S. Forest Service risk assessment, some is not used when calculating dose response values, as discussed in depth in the risk assessment. For long-term exposure to less toxic formulations, the NOAEC for sensitive species was maintained for chronic exposure, though a different NOAEC was used for tolerant species.

Aquatic Plants (Algae and Macrophytes) – For exposure of a tolerant algal species to a more toxic formulation of glyphosate, the U.S. Forest Service risk assessment applied an EC_{10} of 3.78 mg a.e./L. While EC_5 values are typically used to approximate a NOAEL value, a conversion was unnecessary because an EC_{10} of 3.78 was considered a reasonable approximation of a minimal effect level. EC_{50} values for algae exposed to less toxic formulations of glyphosate, however, were converted by dividing by a factor of 10 for approximate EC_5 (and estimated NOAEC) values to account for endangered species. For macrophytes, there are no substantial differences between the sensitivity of macrophytes to the formulations of glyphosate that are generally classified as more toxic or less toxic formulations in the current U.S. Forest Service risk assessment. Consequently, and as with terrestrial macrophytes, separate dose-response assessments for more and less toxic formulations of glyphosate are not developed for aquatic macrophytes.

Table D.3-21

Ecological Endpoints for Less Toxic Glyphosate Formulations

		Endpoint	Receptor, Study & Endpoint Details
Canine mammals	Acute	***N/A***	No data available
	Chronic	***N/A***	
Medium mammals	Acute	***NOAEL = 500 mg/kg bw***	rat, dietary exposure of glyphosate 97.67% a.i., 2-generation reproduction study
	Chronic	***NOAEL = 500 mg/kg bw/day***	
Small mammals	Acute	***N/A***	No data available

	Chronic		***N/A***	
Large herbivore mammals	Acute		***N/A***	
	Chronic		***N/A***	
Birds	Acute		***NOAEL = 1,500 mg/kg bw***	bobwhite quail/mallard duck, technical grade acid [converted from NOAEC = ~5,000 ppm]
	Chronic		***NOAEL = 58 mg/kg bw/day***	bobwhite quail, technical grade acid [converted from NOAEL = 830 ppm]
Terrestrial Invertebrates	Acute		***oral/contact NOEAL = 860 mg/kg bw***	honey bees, technical grade glyphosate [converted oral/contact LD50 values >100 µg/bee] [1]
Terrestrial Plants (Macrophytes)	Seedling Emergence	sensitive spp.	***NOEC = 3.6 lb/acre***	The dose-response assessment for terrestrial plants assumes that the surfactants added to all formulations, resulting in equal efficacy among formulations. For study details, see the more toxic glyphosate formulations table below.
		tolerant spp.	***NOEC = 5 lb/acre***	
	Vegetative vigor	sensitive spp.	***NOEC = 0.0013 lb/acre***	
		tolerant spp.	***NOAEC = 0.445 lb/acre***	
Fish	Acute	sensitive spp.	***surrogate NOAEC of 0.5 mg/L***	several spp. - i.e., chum salmon and rainbow trout, Rodeo at pH 6.3, [derived from an LC50 of 10 mg a.e./L a factor of 0.05]
		tolerant spp.	***surrogate NOAEC = 21 mg/L***	rainbow trout, Rodeo without surfactant at pH 7.8 [derived from an LC50 of 429.2 mg a.e./L * a factor of 0.05]
	Chronic	sensitive spp.	***surrogate NOAEC = 0.5 mg/L***	A duration-response relationship is not evident from the few chronic toxicity studies. As with the more toxic formulations of glyphosate, the surrogate acute NOAECs are applied to longer-term exposure scenarios
		tolerant spp.	***NOAEC = 21 mg/L***	
Amphibians	Acute	sensitive spp.	***NOAEC = 340 mg/L***	glyphosate IPA, tadpoles (*Litoria moorei*) [derived from indefinite LC50 (343 mg a.e./L)]
		tolerant spp.	***NOAEC = 470 mg/L***	tadpole (*Crinia insignifera*), glyphosate IPA [derived from indefinite LC50 (466 mg a.e./L)]
	Chronic	sensitive spp.	***NOAEC = 1.8 mg/L***	

		tolerant spp.	***NOAEC = 1.8 mg/L***	glyphosate IPA, leopard frogs. Note: difference in risk between sensitive and tolerant spp. could not be distinguished
Aquatic invertebrate	Acute	sensitive spp.	***surrogate NOAEC = 2.7 mg/L***	midge larvae (*Chironomous plumosus*), acid (96.7%) [derived from 53.2 mg a.e./L x 0.05 = 2.66 mg a.e./L]
		tolerant spp.	***surrogate NOAEC = 210 mg/L***	midge (*Chironomus riparius*), Rodeo (glyphosate IPA: 53.5% a.i.) [derived from 4140 mg a.e./L x 0.05 = 207 mg a.e./L]
	Chronic	sensitive spp.	***NOAEC = 1.0 mg/L***	No duration-response relationship is evident for glyphosate, Glyphosate acid, 97%, snails (*Pseudosuccinea columella*)
		tolerant spp.	***surrogate NOAEC = 210 mg/L***	No duration-response relationship is evident for glyphosate, so the acute endpoint is maintained for longer-term exposures
Aquatic Plants (Macrophytes)	Acute	sensitive spp.	***surrogate NOAEC = 0.082 mg/L***	As with terrestrial plants, there are no substantial differences between the sensitivity of macrophytes to the formulations of glyphosate that are generally classified as more toxic or less toxic formulations. For study details, see the *more toxic glyphosate formulations* table below.
		tolerant spp.	***NOAEC = 170 mg/L***	
	Chronic	sensitive/ tolerant spp.	***N/A***	
Aquatic Algae (Microphytes)	sensitive spp.		***surrogate NOAEC = 0.23 mg/L***	*Skeletonema costatum,* technical grade glyphosate [EC10 derived from EC50 of 2.27 mg a.e./L divided by a factor of 10]
	tolerant spp.		***surrogate NOAEC = 59 mg/L***	*Chlorella pyrenoidosa*, glyphosate acid (96.7%) [EC10 derived from a EC50 of 590 mg a.e./L divided by a factor of 10]

All endpoints are in terms of a.e. ED/C = Effect Dose/Concentration, IPA = isopropyl amine (salt), LD/C = Lethal Dose/Concentration, N/A = Not Applicable, NOAEL/C = no-observed-adverse-effect-level/concentration. [1] The oral toxicity values for the honey are used as a surrogate for herbivorous insects.

Table D.3-22

Ecological Endpoints for More Toxic Glyphosate Formulations

		Endpoint	**Receptor, Study & Endpoint Details**
Canine mammals	Acute	***N/A***	No data available

	Chronic		***N/A***	
Medium mammals	Acute		***NOAEL = 175 mg/kg bw***	Rabbit, developmental study, dietary exposure of glyphosate acid
	Chronic		***NOAEL = 175 mg/kg bw/day***	
Small mammals	Acute		***N/A***	No data available
	Chronic		***N/A***	
Large herbivore mammals	Acute		***N/A***	
	Chronic		***N/A***	
Birds	Acute		***NOAEL = 540 mg/kg bw***	bobwhite quail/mallard duck, likely RoundUp PRO (a.i. IPA salt) [converted from NOAEC = ~1800 ppm]
	Chronic		***NOAEL = 43 mg/kg bw/day***	broiler chickens, RoundUp (a.i. IPA salt) [converted from NOAEC = 450 ppm]
Terrestrial Invertebrates	Acute		***contact NOAEC = 260 mg/kg bw***	honey bee, MON 77360 (containing POEA) [30µg/bee divided by 0.000116 kg, rounded]
			oral NOAEC = 430 mg/kg bw	honey bee, MON 77360 (containing POEA); also representative of herbaceous insects [15µg/bee divided by 0.000116 kg, rounded]
Terrestrial Plants (Macrophytes)	Seedling Emergence	sensitive spp.	***NOEC = 3.6 lb/acre***	80WDG, 75% a.i., crop monocots and dicots
		tolerant spp.	***NOEC = 5 lb/acre***	oat, rice, sorghum, barnyard grass, soybean, sugar beet, buckwheat, cocklebur, crabgrass, panicum grass, downy brome, velvetleaf, smartweed, morning glory, lambsquarter, hemp, CP-70139, IPA, 50% a.i.
	Vegetative vigor	sensitive spp.	***NOEC = 0.0013 lb/acre***	daisy, Roundup Bio (European formulation)[derived from NOAEC of 0.02 lb/acre x a factor of 15]
		tolerant spp.	***NOAEC = 0.445 lb/acre***	purple nut sedge, formula 80WDG, 48.3% a.i
Fish	Acute	sensitive spp.	***surrogate NOAEC = 0.048 mg/L***	rainbow trout, Roundup formulation with surfactants (i.e., POEA) [derived from an LC50 of 0.96 mg a.e./L x an RQ of 0.05]

		tolerant spp.	***surrogate NOAEC = 0.5 mg/L***	Glyphosate technical from Monsanto [derived from an LD50 of 10 mg a.e./L x an RQ of 0.05]
	Chronic	sensitive spp.	***surrogate NOAEC = 0.048 mg/L***	A duration-response relationship is not evident from the few chronic toxicity studies, and significant effects in such studies were within the range of acute LD50 doses (0.96 and 10 mg a.e./L) for acute studies. Thus, the NOAEC range for acute exposure (i.e., 0.048 to 0.5) is used for chronic exposure to more toxic formulations.
		tolerant spp.	***surrogate NOAEC = 0.5 mg/L***	
Amphibians	Acute	sensitive spp.	***surrogate NOAEC = 0.04 mg/L***	American bullfrog larvae, Roundup Original Max [derived from an LC50 of 0.8 mg a.e./L x an RQ of 0.05]
		tolerant spp.	***surrogate NOAEC = 2.6 mg/L***	metamorph (*Crinia insignifera*), Roundup with POEA surfactant (MON 2139), [derived from LC50 of 51.8 mg a.e./L x an RQ of 0.05]
	Chronic	sensitive spp.	***surrogate NOAEC = 0.04 mg/L***	Acute data used for both sensitive and tolerant spp.
		tolerant spp.	***surrogate NOAEC = 2.6 mg/L***	
Aquatic invertebrate	Acute	sensitive spp.	***surrogate NOAEC = 0.075 mg/L***	amphipods, Roundup formulation from Monsanto USA, [derived from LC50 1.5 mg a.e./L x an RQ of 0.05]
		tolerant spp.	***surrogate NOAEC = 2.3 mg/L***	amphipods, original Roundup formulation, [derived from LC50 46 mg a.e./L x an RQ of 0.05]
	Chronic	sensitive spp.	***surrogate NOAEC = 0.075 mg/L***	A duration-response relationship is not indicated for limited data of more toxic glyphosate formulations, so the surrogate acute NOAECs for sensitive and tolerant species are used for chronic exposures.
		tolerant spp.	***surrogate NOAEC = 2.3 mg/L***	
Aquatic Plants (Macrophytes)	Acute	sensitive spp.	***NOAEC = 0.082 mg/L***	Macrophytes seem equally sensitive to more and less toxic formulations of glyphosate. The algae endpoint is protective for sensitive species of aquatic macrophytes.

		tolerant spp.	***NOAEC = 170 mg/L***	Macrophytes seem equally sensitive to more and less toxic formulations, marine eelgrass, (Zostera marina), acid
	Chronic	sensitive/ tolerant spp.	***N/A***	No data available
Aquatic Algae (Microphytes)	sensitive spp.		***NOAEC = 0.082 mg/L***	*Navicula pelliculosa,* Glyphos
	tolerant spp.		***EC10 = 3.8 mg/L***	*Pseudokirchneriella subcapitata*, Roundup *[an EC5 or NOAEC is not warranted]*

All endpoints are in terms of a.e. ED/C = Effect Dose/Concentration, IPA = isopropyl amine (salt), LD/C = Lethal Dose/Concentration, N/A = Not Applicable, NOAEL/C = no-observed-adverse-effect-level/concentration.

1.3.2.4.3.4 Hexazinone (Sources: FS WSM v. 6.00.10; SERA 2005)

Dose-response endpoints for hexazinone are summarized in Table D.3-23. The available toxicity data support separate dose-response assessments in eight classes of organisms: terrestrial mammals, birds, terrestrial invertebrates, terrestrial plants, fish, aquatic invertebrates, aquatic algae, and aquatic macrophytes.

Mammals and Birds - Based on dietary and gavage toxicity studies, mammals appear to be somewhat more sensitive to hexazinone than birds. For example, the acute dietary NOAEL for birds is 550 mg/kg/day, a factor of about 1.4 above the acute NOEL of 400 mg/kg/day that is used for mammals. No lifetime toxicity studies in birds have been encountered. Based on the reproduction study, the chronic NOAEL for birds is set at 150 mg/kg/day. This is about a factor of 30 above the NOAEL of 5 mg/kg/day used for mammals.

Terrestrial Invertebrates **-** Relatively little information is available on terrestrial insects. A contact toxicity value of 1075 mg/kg bw is taken as a marginal LOEC***.***

Terrestrial Plants (Macrophytes) - Hexazinone is relatively ineffective in inhibiting seed germination, but is toxic after either direct spray or soil application. Based on toxicity studies in which exposure can be characterized as an application rate, hexazinone is more toxic in pre-emergent soil applications than direct spray (post-emergent application).

Terrestrial and Aquatic Microorganisms *– No formal dose-response assessment was completed for aquatic microbes due to a lack of acceptable studies.* For terrestrial microbes, there is extensive literature regarding toxicity of hexazinone towards soil bacteria and fungi, though most information is from laboratory cultures. However, some field studies have shown hexazinone to have no adverse effects on these organisms at application rates up to about 7 lbs/acre. This information is used directly in the risk characterization for terrestrial microorganisms.

Fish - The acute NOEC values for sensitive and tolerant species of fish cover a very narrow range, 160 mg/L to 370 mg/L. For longer term exposures, the data are not sufficient to identify tolerant and sensitive species, so a single NOEC value of 17 mg/L is used.

Amphibians – *No formal dose-response assessment was completed for amphibians due to a lack of acceptable studies.*

Aquatic Invertebrates - Somewhat greater variability is apparent in aquatic invertebrates compared to fish, with acute NOEC values ranging from 20.5 mg/L to 320 mg/L. However, this may be an artifact of comparisons between freshwater and saltwater species. An NOEC of 10 mg/L from a reproduction study in daphnids is used to assess the effects of longer-term exposures in sensitive aquatic invertebrates. No longer-term NOEC is available for tolerant invertebrates, so the relative potency from acute studies is used to estimate a longer-term NOEC for tolerant species at 160 mg/L.

***Aquatic Plants (Algae and Macrophytes*) -** Aquatic plants are much more sensitive to hexazinone than fish and aquatic invertebrates, with much greater toxicity variability. Aquatic macrophytes appear to fall within the range of algae, so a single NOEC of 0.012 mg/L is used for this group.

Table D.3-23

Ecological Endpoints for Hexazinone

		Endpoint	**Receptor, Study & Endpoint Details**
Canine mammals	Acute	***N/A***	No data available
	Chronic	***NOAEL = 5 mg/kg bw/day***	dog, 1-dietary study for chronic toxicity
Medium mammals	Acute	***NOAEL = 400 mg/kg bw/day***	rat, developmental study, endpoint for offspring; at dose evidence of maternal toxicity
	Chronic	***N/A***	No data available
Small mammals	Acute	***N/A***	
	Chronic	***N/A***	
Large herbivore mammals	Acute	***N/A***	
	Chronic	***N/A***	

Birds	Acute		***surrogate NOAEL = 550 mg/kg bw***	bobwhite quail, (derived from 2,500 ppm * food consumption rate of 22% bw/day), dietary study
	Chronic		***surrogate NOAEL = 150 mg/kg bw/day***	bobwhite quail, derived from 1,000 ppm * food consumption rate of 15% bw/day, reproduction study
Terrestrial Invertebrates	Acute		***LOEC = 1075 mg/kg bw***	honey bee, derived from LD50 > 0.1 mg/bee and functions as a marginal LOEC
Terrestrial Plants (Macrophytes)	Seedling Emergence	sensitive spp.	***NOEC = 0.000348 mg/kg bw***	tomato, for all effects
		tolerant spp.	***NOEC = 0.0234 mg/kg bw***	corn, for all effects
	Vegetative vigor	sensitive spp.	***NOEC = 0.00391 mg/kg bw***	cucumber, for all effects
		tolerant spp.	***NOEC = 0.0625 mg/kg bw***	corn, for all effects
Fish	Acute	sensitive spp.	***NOEC = 160 mg/L***	flathead minnows, for mortality
		tolerant spp.	***NOEC = 370 mg/L***	trout, for mortality
	Chronic	sensitive spp.	***NOEC = 17 mg/L***	flathead minnows, egg-and-fry development, used given the narrow range for acute NOEC and LD50 values and that flatheads appear to be most sensitive.
		tolerant spp.		
Amphibians	Acute	sensitive spp.	***N/A***	Data is not adequate enough to propose an independent toxicity value for amphibians
		tolerant spp.	***N/A***	
	Chronic	sensitive spp.	***N/A***	
		tolerant spp.	***N/A***	
Aquatic invertebrate	Acute	sensitive spp.	***NOEC = 20.5 mg/L***	*Daphnia magna*
		tolerant spp.	***NOEC = 320 mg/L***	Oyster embryos
	Chronic	sensitive spp.	***NOEC = 10 mg/L***	*Daphnia magna*, reproduction study

		tolerant spp.	***NOEC = 160 mg/L***	[derived by multiplying relative potency from acute studies (320 divided by 20.5 mg/L) x 10 mg/L]
Aquatic Plants (Macrophytes)	Acute	sensitive/ tolerant spp.	***NOEC = 0.012 mg/L***	Lemna minor, 7-day growth study
	Chronic			
Aquatic Algae (Microphytes)	sensitive spp.		***NOEC = 0.004 mg/L***	green algae, Selenastrum capricornutum, 5-day growth inhibition study
	tolerant spp.		***NOEC = 0.15 mg/L***	blue-green algae (*Anabaena flos-aquae*), 5-day growth inhibition study

All endpoints are in terms of a.i. LD/C = Lethal Dose/Concentration, N/A = Not Applicable, NOAEL/C = no-observed-adverse-effect-level/concentration. LOEC = lowest-observed-effect-level.

*1.3.2.4.3.5*Imazapyr (Sources: FS WSM v. 6.00.10; SERA 2011c)

Dose-response endpoints are summarized in Table D.3-24 for imazapyr. Dose response assessments are supported for eight classes of organisms in the U.S. Forest Service risk assessment for imazapyr: terrestrial mammals, birds, terrestrial invertebrates, terrestrial plants, fish, aquatic invertebrates, aquatic algae, and aquatic macrophytes. The dose-response assessments for terrestrial and aquatic animals are limited, primarily because imazapyr is relatively nontoxic to animals and the number of animal species tested is so few. Consequently, sensitive and tolerant species are not defined for either terrestrial animals or for most groups of aquatic animals.

Mammals and Birds - The standard array of studies used to assess the acute, subchronic, and chronic toxicity of pesticides, including effects on reproduction and development, indicate that imazapyr causes adverse effects in mammals only at doses of 1000 mg a.e./kg or more. The use of a NOAEL in dogs to characterize risks for all terrestrial mammals, however, may be overly conservative. Imazapyr is a weak acid, and, like most weak acids, is excreted primarily in the urine. Because dogs have a limited capacity to excrete weak acids, they are more sensitive than other mammals to certain weak acids. Imazapyr has a low order of acute toxicity in birds. Both acute and chronic NOAEL values for toxicity of birds are free-standing—i.e., adverse effects may occur at higher, yet undetermined, doses.

Terrestrial Invertebrates - The standard contact toxicity study in honeybees is used to represent this class of organisms. Likewise, the standard oral toxicity study using honey bees is used as a surrogate toxicity value to characterize risks to herbivorous insects from the consumption of vegetation contaminated with imazapyr.

Terrestrial Plants (Macrophytes) - Like other imidazolinone herbicides, imazapyr appears to be more toxic to terrestrial monocots than to dicots.

Terrestrial and Aquatic Microorganisms - *No formal dose-response assessment was completed for either group of microbes due to a lack of acceptable studies. Available studies will be used to qualitatively assess terrestrial microbes in the risk characterization section.* Liquid culture solutions of imazapyr were toxic to various soil bacteria, with LC_{50} values ranging from about 10 to 1000 µM (2.61 to 261 mg/L - ppm). Because these concentrations involve liquid cultures and because bioavailability of imazapyr is likely to be substantially less in a soil matrix, these values are not appropriate for direct use, analogous to other NOAEL and NOAEC values discussed in this risk assessment. Imazapyr had only a slight effect on the breakdown of cellulose at a concentration in soil of 20 mg/kg but had a substantial impact at a concentration of 150 mg/kg. These values are relevant to the functional effect of imazapyr on soil microorganisms.

Fish **-** Studies consistently indicate that Arsenal, the only formulation on which toxicity data are available, is more toxic than imazapyr acid or the isopropylamine salt of imazapyr.

Amphibians **-** *No formal dose-response assessment was completed for terrestrial phase or aquatic phase amphibians due to a lack of toxicity data.*

Aquatic Invertebrates - Studies consistently indicate that, as for fish, the formulation Arsenal is more toxic than imazapyr acid or the isopropylamine salt of imazapyr.

Aquatic Plants (Algae and Macrophytes**) -** Like other imidazolinone herbicides, imazapyr appears to be more toxic to aquatic macrophytes than to algae and more toxic to terrestrial monocots than to dicots.

Table D.3-24

Ecological Endpoints for Imazapyr

		Endpoint	Receptor, Study & Endpoint Details
Canine mammals	Acute	***NOAEL = 250 mg/kg bw/day***	Chronic endpoint applied
	Chronic	***NOAEL = 250 mg/kg bw/day***	dog, 1-year dietary study
Medium mammals	Acute	***NOAEL = 738 mg/kg bw/day***	Chronic endpoint applied
	Chronic	***NOAEL = 738 mg/kg bw/day***	rat, reproduction (dietary) study
Small mammals	Acute	***N/A***	No data available
	Chronic	***N/A***	
Large herbivore mammals	Acute	***N/A***	
	Chronic	***N/A***	

Birds	Acute		***NOAEL = 2,510 mg/kg bw***	Mallard ducks, technical grade (93% a.e.) used in gavage study; Also supported by Northern bobtail quail studies
	Chronic		***NOAEL = ~610 mg/kg bw***	Northern bobwhite quail, acid, based on measured food consumption and body weights, reproductive (dietary) study [derived from 1,670 ppm a.e.]
Terrestrial Invertebrates	Acute		***contact/oral NOAEL > 860 mg/kg bw***	honey bee, oral functionally surrogate for herbivorous insects [derived from an LD50 = 100µg/bee]
Terrestrial Plants (Macrophytes)	Seedling Emergence	sensitive spp.	***NOAEL = 0.00017 lbs/acre***	sugar beet (a dicot), technical grade
		tolerant spp.	***NOAEL = 0.0156 lbs/acre***	oat (a monocot), for growth (height)
	Vegetative vigor	sensitive spp.	***NOAEL = 0.000064 lb/acre***	cucumber (a dicot)
		tolerant spp.	***NOAEL = 0.4 lb/acre***	pumpkin (a dicot)
Fish	Acute	sensitive spp.	***NOAEC = 10.4 mg/L***	trout, formulation
		tolerant spp.	***N/A***	No data available
	Chronic	sensitive spp.	***surrogate NOAEC = 4.0 mg/L***	formulation, [derived from the chronic NOAEC of 43.1 mg a.e./L from a trout study that is divided by 10 and rounded]
		tolerant spp.	***estimated NOAEC = 12 mg/L***	formulation, [derived from the NOAEC of 118 mg a.e./L in flathead minnows divided by 10 and rounded]
Amphibians	Acute	sensitive spp.	***N/A***	no data available
		tolerant spp.	***N/A***	
	Chronic	sensitive spp.	***N/A***	
		tolerant spp.	***N/A***	
Aquatic invertebrate	Acute	sensitive spp.	***N/A***	No data available
		tolerant spp.	***NOAEC = 41 mg/L***	Daphnia magna, Arsenal formulation
	Chronic	sensitive spp.	***N/A***	No data available

		tolerant spp.	***surrogate NOAEC = 12 mg/L***	Daphnia magna, formulation [derived from NOAEC of 97.1 mg a.e./L divided by 8.0 to account potentially greater long-term toxicity of formulations]
Aquatic Plants (Macrophytes)	Acute	sensitive spp.	***NOEC = 0.003 mg/L***	water milfoil *(Myriophyllum sibiricum*), Arsenal formulation
		tolerant spp.	***surrogate NOEC = 0.1 mg/L***	giant salvinia (*Salvinia molesta*)
	Chronic	sensitive/ tolerant spp.	***N/A***	No data available
Aquatic Algae (Microphytes)	sensitive spp.		***NOEC = 7.6 mg/L***	*Selenastrum capricornutum*, acid
	tolerant spp.		***NOEC = 50.9 mg/L***	*Skeletonema costatum*, acid

All endpoints are in terms of a.e. ED/C = Effect Dose/Concentration, LD/C = Lethal Dose/Concentration, N/A = Not Applicable, NOAEL/C = no-observed-adverse-effect-level/concentration.

1.3.2.4.3.6 NP9E (Sources: FS WS ver. 2.02; USDA/FS 2003b)

Dose-response endpoints are summarized in Table D.3-25 for NP9E and associated compounds.

Although NP is of higher toxicity to aquatic organisms than NPE or NPEC, there is sufficient information in the literature to assume that in a forested environment, contamination of surface water is more likely to involve NP9E in the short-term and NP1-2EC in the long-term. As such, indicators of risk will be based upon these two compounds, not on NP.

Mammals and Birds - Mammalian toxicity is well characterized for NP, but less so for NP9E and the carboxylate metabolites. The acute NOEL value of 10 mg/kg bw was taken from a 90-day rat feeding study and should be considered a conservative value, as the NOEL values from similar tests range up to 40 mg/kg/day, with LOELs beginning at 50 mg/kg/day. The chronic toxicity value is also 10 mg/kg bw/ day, though it was derived from an NP multigenerational study with rats; it will be used for both NP and NPE. For birds, mammal data is protective of birds and is thus used for surrogate values.

Terrestrial Invertebrates – *No formal dose-response assessment was completed due to a lack of acceptable studies.*

Terrestrial Plants (Macrophytes) - Since NP9E-based surfactants would not be applied alone, but would be applied in a mix with an herbicide, the herbicide would determine the effects to terrestrial plants. Thus, a dose-response assessment is not appropriate.

Terrestrial and Aquatic Microorganisms *- No formal dose-response assessment was completed due to a lack of acceptable studies.*

Fish - For NP9E, the value that will be used to establish the aquatic acute no-effect level is the 7-day NOEC (growth) for minnows of 1,000 ppb. Species that have been tested with the longer chain NPEs all have similar values, so no interspecies factor will be used. It is assumed that acute toxicity tests involving NP9E included a small percentage of the short-chain ethoxylates, as well as small amounts of NP. For NP1EC and NP2EC, the NOEC value of 100 ppb in fathead minnows will be applied.

Amphibians - Frogs seem similar in sensitivity or somewhat less sensitive than fish. Therefore, levels of exposure that result in low levels of risk to fish should be similarly protective of frogs.

Aquatic Invertebrates – For aquatic invertebrates, the 7-day NOEC for NP9E of 10 mg/L for *Daphnia* spp. will be used for acute exposures. For chronic exposures, since no testing has been done using the NP1-2ECs, the 21-day NP NOEC for *Daphnia magna* will be used (0.024 mg/L).

Aquatic Plants (Algae and **Macrophytes) -** For aquatic plants, the 96-hour NP9E NOEC (growth) of 8 mg/L for green algae will be used for acute exposures. There are no chronic exposure studies for aquatic plants.

Table D.3-25

Ecological Endpoints for NP9E

		Endpoint	**Receptor, Study & Endpoint Details**
Canine mammals	Acute	***N/A***	No data available
	Chronic	***N/A***	
Medium mammals	Acute	***NOAEL = 10 mg/kg***	rat, NP9E dietary study
	Chronic	***NOAEL = 10 mg/kg/day***	rat, NP oral gavage multigeneration study
Small mammals	Acute	***N/A***	No data available
	Chronic	***N/A***	
Large herbivore mammals	Acute	***N/A***	
	Chronic	***N/A***	

Birds	Acute		***surrogate NOAEL = 10 mg/kg***	rat data, acute and chronic mammal endpoints are used as surrogate values for avian species
	Chronic		***surrogate NOAEL = 10 mg/kg/day***	
Terrestrial Invertebrates	Acute		***N/A***	No data available
Terrestrial Plants (Macrophytes)	Seedling Emergence	sensitive spp.	***N/A***	NP9E-based surfactants would not be applied alone, but applied with an herbicide, and the herbicide would determine effects to plants. Thus, a dose-response assessment is NA.
		tolerant spp.	***N/A***	
	Vegetative vigor	sensitive spp.	***N/A***	
		tolerant spp.	***N/A***	
Fish	Acute	sensitive spp.	***NP9E: NOEC = 1.0 mg/L***	Fathead minnow (*Pimephales promelas*), 7-day growth study (based on growth), [converted from 1,000 ppb]
		tolerant spp.		
	Chronic	sensitive spp.	***NP1EC/NP2EC: NOEC = 0.1 mg/L***	flathead minnow (*Pimephales promelas*), [derived from 1,000 ppb, dividing by an interspecies factor of 10 for NOEC = 100 ppb which is then converted to mg/L]
		tolerant spp.		
Amphibians	Acute	sensitive spp.	***surrogate NP9E: NOEC = 1.0 mg/L***	flathead minnow (Pimephales promelas) data, limited amphibian data suggests NP9E is equally or less toxic to amphibians compared to fish.
		tolerant spp.		
	Chronic	sensitive spp.	***surrogate NP1EC/NP2EC: NOEC = 0.1 mg/L***	
		tolerant spp.		
Aquatic invertebrate	Acute	sensitive spp.	***NOEC = 10 mg/L***	sensitive and tolerant spp. not specified; Daphnia spp., 7-day study using NP9E
		tolerant spp.	***NOEC = 10 mg/L***	
	Chronic	sensitive spp.	***NOEC = 0.024 mg/L***	species sensitivity not specified; *Daphnia magna* 21-day study using NP
		tolerant spp.	***NOEC = 0.024 mg/L***	
Aquatic Plants (Macrophytes)	Acute	sensitive spp.	***surrogate NOEC = 8 mg/L***	algal values applied: green algae (*Selenastrum capricornutum*), NP9E study
		tolerant spp.	***surrogate NOEC = 8 mg/L***	
	Chronic	sensitive/ tolerant spp.	***N/A***	No data available

Aquatic Algae (Microphytes)	sensitive spp.	***NOEC = 8 mg/L***	sensitive and tolerant species not specified; exposure to NP9E, green algae (*Selenastrum capricornutum*)
	tolerant spp.	***NOEC = 8 mg/L***	

All endpoints are in terms of a.i. N/A = Not Applicable, NOAEL/C = no-observed-adverse-effect-level/concentration.

1.3.2.4.3.7 Sulfometuron methyl (Sources: FS WSM v. 6.00.10; SERA 2004c)

Dose-response endpoints are summarized in Table D.3-26 for sulfometuron methyl.

Mammals and Birds **-** All the potential longer-term and acute exposures of terrestrial mammals to sulfometuron methyl are substantially below the NOAEL values of 2 mg/kg/day and 87 mg/kg/day respectively. Birds appear to exhibit the same low order of toxicity to sulfometuron methyl as mammals, with an acute NOAEL of 312 mg/kg based on changes in body weight observed following a single gavage administration to mallard ducks. No chronic exposure studies of birds to sulfometuron methyl were identified in the available literature. Since results of acute exposure studies suggest that the sensitivity of birds to sulfometuron methyl is similar to that of mammals, in the absence of chronic exposure data in birds the chronic NOAEL for rats is used for birds.

Terrestrial Invertebrates - For terrestrial invertebrates, based on direct spray studies in honey bees, no mortality would be expected following acute exposure to doses up to 1075 mg/kg.

Terrestrial Plants (Macrophytes) - Sulfometuron methyl is a potent herbicide that causes adverse effects in a variety of target and non-target plant species.

Terrestrial and Aquatic Microorganisms - No formal dose-response assessment was completed for either group of microbes due to a lack of acceptable studies. Available studies will be used to qualitatively assess terrestrial microbes in the risk characterization section. Regarding terrestrial microbes, soil microorganisms appear sensitive to sulfometuron methyl at concentrations of about 70 µg/L. No specific NOEC was determined, though the chemical has been found to inhibit growth in some species (e.g. *Salmonella typhimurium*) and microbe species may develop resistance to the chemical, while other bacteria species (e.g. *Streptomyces griseolus*) metabolize the compound.

Fish - The data on toxicity to fish and aquatic invertebrates was obtained for several species. Fish do not appear to be highly sensitive to sulfometuron methyl toxicity. However, investigations of acute toxicity have been hampered by the limited water solubility of sulfometuron methyl. Both acute values were the highest concentration tested in both studies, so identification of a most sensitive and a most tolerant species cannot be made with certainty. Toxicity values for chronic toxicity may be based on the available egg-and-fry/early life stage studies. Only one study of chronic exposure in fish is available, a 30-day exposure of fathead

minnow yielding an NOAEC of 1.17 mg a.i./L. This value is used for both the most sensitive and tolerant species for chronic exposure.

Amphibians – The toxicity of acute and chronic exposure to sulfometuron methyl to amphibians has been evaluated in a single study in African Clawed frogs (*Xenopus laevis*). In this report, the author did not state whether data were reported in terms of mg sulfometuron methyl/L or mg Oust/L. Taking the most conservative approach, values are assumed to be expressed in terms of mg a.i./L. Since no studies on other amphibian species were identified in the available literature, it is not possible to identify a most tolerant and most sensitive amphibian species.

Aquatic Invertebrates **-** For acute exposure of aquatic invertebrates, the most sensitive species appear to be *Alonella* sp. and *Cypria* sp., with *Daphnia* the most tolerant species. *Daphnia* are 32 times more tolerant than *Alonella* and *Cypria* to acute exposure of sulfometuron methyl. For chronic exposure of aquatic invertebrates, data are only available from a single study in Daphnia, with a NOAEC of 6.1 mg/L. This value is used for the most tolerant species for chronic exposure. Although no data are available to determine the most sensitive species for chronic exposures, parallels can be drawn to the acute exposure studies. Using the relative potency factor for acute exposures of 32 and the chronic NOEC in *Daphnia* of 6.1 mg/L, a NOAEC for *Alonella* and *Cypria* is estimated to be 0.19 mg/L. This surrogate NOAEC for chronic exposure in *Alonella* and *Cypria* will be used to estimate the chronic NOAEC for the most sensitive species.

***Aquatic Plants (Algae and Macrophytes*)** **-** Aquatic plants appear to be much more sensitive to sulfometuron methyl than aquatic animals. A NOAEC for growth inhibition of 0.00021 mg/L in duckweed is used to quantify effects for both acute and chronic exposure in aquatic macrophytes. Based on the limited data available as well as difference in experimental protocols, it is not possible to identify a most sensitive and most tolerant species for aquatic macrophytes. For algae, the most sensitive algal species appears to be *Selenastrum capricornutum* and the most tolerant species appears to be *Navicula pelliculosa*.

Table D.3-26

Ecological Endpoints for Sulfometuron Methyl

		Endpoint	**Receptor, Study & Endpoint Details**
Canine mammals	Acute	***N/A***	No data available
	Chronic	***N/A***	
Medium mammals	Acute	***NOAEL = 87 mg/kg bw***	rat, diets containing 1000 ppm convert to ~86.6 mg/kg/day

	Chronic		***NOAEL = 2 mg/kg bw/day***	rats, 2-year feeding study
Small mammals	Acute		***N/A***	No data available
	Chronic		***N/A***	
Large herbivore mammals	Acute		***N/A***	
	Chronic		***N/A***	
Birds	Acute		***NOAEL = 312 mg/kg bw***	mallard duck, technical grade, gavage administration
	Chronic		***NOAEL = 2 mg/kg bw/day***	Acute values for birds and mammals had comparable magnitude. Chronic mammal endpoint applied as surrogate chronic bird endpoint. rats, from a 2-year feeding study.
Terrestrial Invertebrates	Acute		***NOAEL = 1075 mg/kg bw***	honey bee, [derived from an LD50 of 100 µg/bee divided by bee bw of 0.093 g
Terrestrial Plants (Macrophytes)	Seedling Emergence	sensitive spp.	***NOEC = 0.0000086 lb/acre***	rape, tomato sorghum, wheat and corn
		tolerant spp.	***NOEC = 0.00026 lb/acre***	onion, pea, cucumber and soybean
	Vegetative vigor	sensitive spp.	***NOEC = 0.000024 lb/acre***	corn
		tolerant spp.	***NOEC = 0.00078 lb/acre***	pea
Fish	Acute	sensitive spp.	***NOEC = 7.3 mg/L***	acute LC50 result hampered by limited water solubility of sulfometuron methyl, flathead minnows
		tolerant spp.	***NOEC = 150 mg/L***	acute LC50 result hampered by limited water solubility of sulfometuron methyl, bluegill sunfish and rainbow trout
	Chronic	sensitive spp.	***NOEC = 1.17 mg/L***	flathead minnow larvae; identification of sensitivity by species not possible
		tolerant spp.		
Amphibians	Acute	sensitive spp.	***NOEC = 0.38 mg/L***	African Clawed frogs (*Xenopus laevis*), Oust formulation, sensitivity by spp. is NA for 1 study. This NOAEC and assoc. LOAEC
		tolerant spp.		

				value are for lethality and malformations during metamorphosis
	Chronic	sensitive spp.	***NOEC = 0.00075 mg/L***	African Clawed frog (*Xenopus laevis*) study, Oust formulation, sensitivity by spp. is NA for 1 study. This NOEC is for changes in tail resorption rates during a 14-day study
		tolerant spp.		
Aquatic invertebrate	Acute	sensitive spp.	***LOEC = 75 mg/L***	Alonella spp. and Cypria spp. [the lowest concentration tested]
		tolerant spp.	***NOEC = 1,800 mg/L***	*Daphnia*
	Chronic	sensitive spp.	***surrogate NOEC = 0.19 mg/L***	[derived from tolerant chronic NOEC of 6.1 mg/L ÷ relative potency of 32 that is based on ratio of Daphnia to Alonella and Cypria acute LOAEC values (2,400/75)]
		tolerant spp.	***NOEC = 6.1 mg/L***	*Daphnia*
Aquatic Plants (Macrophytes)	Acute	sensitive/ tolerant spp.	***NOEC = 0.00021 mg/L***	duckweed (Lemna spp.), technical grade, 14-day study; most conservative (lowest) NOEC of both acute and chronic values
	Chronic	sensitive/ tolerant spp.	***NOEC = 0.00021 mg/L***	
Aquatic Algae (Microphytes)	sensitive spp.		***NOEC = 0.0025 mg/L***	alga (*Selenastrum capricornutum*), based on cell density
	tolerant spp.		***NOEC = 0.37 mg/L***	alga (*Navicula pelliculosa*), based on growth inhibition

All endpoints are in terms of a.i. LD/C = Lethal Dose/Concentration, LOEC = lowest-observed-effect-level, N/A = Not Applicable, NOAEL/C = no-observed-adverse-effect-level/concentration.

1.3.2.4.3.8 Triclopyr (Sources: FS WSM v. 6.00.10; SERA 2011a,d)

Triclopyr acid and salts are considered separately from esters**.** Dose response is also considered for the compound 3,5,6-trichloro-2-pyridinol (TCP) within this section, as TCP is a metabolite of triclopyr of concern. Dose-response endpoints are summarized in Table D.3-27 for TCP, Table D.3-28 for acid and triethylamine salt of triclopyr, and Table D.3-29 for butoxyethyl esters of triclopyr.

Data on triclopyr TEA are typically included in the dose-response assessment for triclopyr acid, because these two forms of triclopyr appear to be bioequivalent in most groups of organisms. Data on triclopyr BEE and formulations of triclopyr BEE are discussed separately

for some groups of organisms, primarily because the toxicity of triclopyr BEE formulations (expressed in units of triclopyr a.e.) and technical grade triclopyr BEE (also expressed in units of triclopyr a.e.) appears to be the same. In other words, the inerts used in the triclopyr BEE formulations do not have an obvious impact on the toxicity of the triclopyr BEE formulations on which data are available (primarily Garlon 4). The toxicity values for TCP span much narrower ranges than the toxicity values for triclopyr. This difference is almost certainly due to the fewer number of studies available on TCP.

The dose-response assessments for triclopyr acid and triclopyr BEE in terrestrial animals are relatively standard and uncomplicated, except for mammals. For TCP, the available data limit the dose-response assessment for terrestrial organisms to mammals. The dose-response assessment for aquatic species is somewhat detailed, because triclopyr acid and triclopyr BEE are not bioequivalent in aquatic organisms. Except for aquatic dicots, triclopyr BEE is much more toxic than triclopyr acid or triclopyr TEA. Within most groups of aquatic organisms, the toxicity values differ substantially for both triclopyr TEA and triclopyr BEE. Typically, this high variability reflects differences among bioassays conducted by different investigators at different times rather than true underlying differences in species sensitivity. A possible exception involves the toxicity of triclopyr BEE to aquatic arthropods.

Mammals and Birds - The available toxicity data on triclopyr indicate that larger mammals are substantially more sensitive than smaller mammals, and this relationship can be characterized quantitatively. Most U.S. Forest Service risk assessments consider only small mammals and canids, however, the dose-response assessment for mammalian wildlife is elaborated to include a large herbivorous mammal, such as a deer. There is no remarkable difference in the toxicity of triclopyr acid, triclopyr TEA, and triclopyr BEE to birds. Similarly, the toxicity data, available only on a few avian species, do not indicate substantial or systematic differences in species sensitivities to triclopyr. The current U.S. Forest Service risk assessment relies on the EPA review of the toxicity of TCP and available open literature. Relatively little information is available on the toxicity of TCP to mammals or birds (U.S. EPA/OPP 2002b as referenced in SERA 2011d).

Terrestrial Invertebrates – For triclopyr, an indefinite LD_{50} was used rather than a well-documented NOAEC for the calculation of hazard quotients, though the risk characterization for insects is based primarily on field studies rather than the HQs. A dose-response assessment of the toxicity of TCP to terrestrial invertebrates cannot be proposed due to the lack of pertinent data.

Terrestrial Plants (Macrophytes) The dose-response assessments in terrestrial plants are also relatively standard for triclopyr acid and the triclopyr ester. Foliar studies do not suggest any remarkable differences in potency between triclopyr TEA and triclopyr BEE formulations. Dicots are more sensitive than monocots to both formulations. A dose-response assessment

of the phytotoxicity of TCP is not proposed because no data are available on the toxicity of TCP to terrestrial plants.

Terrestrial and Aquatic Microorganisms *– No formal dose-response assessment was completed for either group of microbes due to a lack of acceptable studies. Available field studies will be used to qualitatively assess terrestrial organisms in the risk characterization section.*

Fish - Acute LC_{50} values for triclopyr TEA range from 40.1 to 422.8 mg a.e./L and encompass the more limited number of LC_{50} values available on triclopyr acid. The acute sublethal toxicity of triclopyr acid and triclopyr TEA is not well documented, either in standard acute toxicity studies or field studies. There are more toxicity data for triclopyr BEE than for triclopyr TEA, including more acute toxicity studies, many of which report both LC_{50} values and NOAECs. Acute LC_{50} values for triclopyr BEE range from 0.2 to 1.5 mg a.e./L. As with triclopyr TEA, there is only one chronic study available. For TCP, there are two sets of studies, which are obviously inconsistent and reflect experimental variability or other unidentified factors rather than any differences in species sensitivity.

Amphibians - Information on the toxicity of triclopyr to amphibians is much less abundant than the information on fish. Since there are no chronic bioassays involving amphibian exposure to triclopyr, explicit longer-term NOAECs are not developed. Nonetheless, a field study involving longer-term observations of amphibian populations following forestry applications of triclopyr BEE is used in the development of acute NOAECs and is discussed further in the risk characterization for amphibians.

Aquatic Invertebrates - There is no apparent basis, given admittedly limited data, for asserting that non-arthropod aquatic invertebrates are substantially different from aquatic arthropods in their sensitivity to triclopyr. Within this group, cladocerans appear to be more sensitive than aquatic insects to triclopyr BEE, though no such species sensitivity is clearly documented for triclopyr TEA or the TCP metabolite.

Aquatic Plants (Algae and Macrophytes) – Data regarding toxicity to algae are available for triclopyr acid, triclopyr BEE, and TCP. As with most other groups of aquatic organisms, algae are more sensitive to triclopyr BEE than to triclopyr TEA. Based on median EC_{50} values, triclopyr BEE is more toxic to algae than triclopyr TEA by a factor of 10. When considering toxicity to aquatic macrophytes, relative sensitivity to triclopyr TEA is assessed based on an analogy to differences in the sensitivity of monocots and dicots, with dicots comprising the sensitive species and monocots comprising the tolerant species. There is not a substantial difference in the toxicity of triclopyr BEE to monocots and dicots. Dicots are the only group of aquatic organisms in which triclopyr TEA is substantially more toxic than triclopyr BEE. A dose-response assessment of the toxicity of TCP to macrophytes is not proposed because no data are available on the toxicity of TCP to aquatic macrophytes.

Table D.3-27

Ecological Endpoints for TCP, a Metabolite of Triclopyr

			Endpoint	Receptor, Study & Endpoint Details
Canine mammals	Acute		***NOAEL = 25 mg/kg bw***	rabbit, LOAEL endpoint: birth defects
	Chronic		***surrogate NOAEL = 12 mg/kg bw***	dog study, chronic NOAEL
Medium mammals [1]	Acute		***surrogate NOAEL = 25 mg/kg bw***	rabbit study, acute NOAEL
	Chronic		***NOAEL = 12 mg/kg bw***	dog, LOAEL endpoint: clinical chemistry
Small mammals	Acute		***N/A***	No data available
	Chronic		***N/A***	
Large herbivore mammals	Acute		***N/A***	
	Chronic		***N/A***	
Birds	Acute		***LOAEL = 116 mg/kg bw***	5-day dietary study
	Chronic		***N/A***	No data available
Terrestrial Invertebrates	Acute		***N/A***	No data available
Terrestrial Plants (Macrophytes)	Seedling Emergence	sensitive spp.	***N/A***	No data available
		tolerant spp.	***N/A***	
	Vegetative vigor	sensitive spp.	***N/A***	
		tolerant spp.	***N/A***	
Fish	Acute	sensitive spp.	***surrogate NOAEC = 0.18 mg/L***	rainbow trout, [see chronic NOAEC; conservatively applied]
		tolerant spp.	***estimated NOAEC = 0.63 mg/L***	rainbow trout, [LC50 of 1.26 mg TCP/L x LOC of 0.5]
	Chronic	sensitive spp.	***adjusted NOAEC = 0.18 mg/L***	rainbow trout, fry to egg study, variation for trout (see acute) may be related to environmental and experimental variability (i.e., pH), unidentifiable
		tolerant spp.		

				factors, and/or chance [rounded from 0.178 mg/L].
Amphibians	Acute	sensitive spp.	***N/A***	No data available
		tolerant spp.	***N/A***	
	Chronic	sensitive spp.	***N/A***	
		tolerant spp.	***N/A***	
Aquatic invertebrate	Acute	sensitive spp.	***estimated NOAEC = 0.55 mg/L***	[LC50 of 10.9 mg/L x 0.05]
		tolerant spp.	***estimated NOAEC = 0.55 mg/L***	
	Chronic	sensitive spp.	***NOAEC = 0.058 mg/L***	*Daphnia magna* study
		tolerant spp.	***NOAEC = 0.058 mg/L***	
Aquatic Plants (Macrophytes)	Acute	sensitive spp.	***N/A***	No data available
		tolerant spp.	***N/A***	
	Chronic	sensitive/ tolerant spp.	***N/A***	
Aquatic Algae (Microphytes)	sensitive spp.		***NOAEC = 0.36 mg/L***	*Anabaena flos-aquae*, 5-day study
	tolerant spp.		***NOAEC = 0.65 mg/L***	*Kirchneria subcapitata*, 5-day study

All toxicity values for 3,5,6-trichloro-2-pyridinol (TCP) metabolite of triclopyr are expressed as mg TCP/kg bw or mg TCP/L. LD/C = Lethal Dose/Concentration, LOC = level of concern, LOAEL = lowest-observed-adverse-effect-level, N/A = Not Applicable, NOAEL/C = no-observed-adverse-effect-level/concentration. [1] Due to lack of data for species sensitivity of mammals to TCP, the NOAELs of 25 mg/kg bw for acute exposures and 12 mg/kg bw for longer-term term exposures are used to characterize risks of TCP exposure to small mammals.

Table D.3-28

Ecological Endpoints for Triclopyr Acid and TEA

		Endpoint	**Receptor, Study & Endpoint Details**
Canine mammals	Acute	***NOAEL = 20 mg a.e./kg bw***	estimated relative to rat [derived by 100 mg/kg bw ÷ factor of 5]
	Chronic	***NOAEL = 1 mg a.e./kg bw***	estimated relative to rat [derived by 5 mg/kg bw ÷ factor of 5]
Medium mammals	Acute	***NOAEL = 100 mg/kg bw***	rat
	Chronic	***NOAEL = 5 mg/kg bw***	rat

Small mammals	Acute		***NOAEL = 440 mg/kg bw***	estimated relative to rat [derived by 100 mg/kg bw x factor of 4.4]
	Chronic		***NOAEL = 22 mg/kg bw***	estimated relative to rate [derived by 5 mg/kg bw x factor of 4.4]
Large herbivore mammals	Acute		***NOAEL = 8 mg/kg bw***	estimated relative to rat [derived by 100 mg/kg bw ÷ factor of 13 ≈ 7.69]
	Chronic		***NOAEL = 0.4 mg/kg bw***	estimated relative to rat [derived by 5 mg/kg bw ÷ factor of 13 ≈ 0.38]
Birds	Acute		***NOAEL = 126 mg/kg bw***	Northern bobwhite quail, BEE gavage study
	Chronic		***NOAEL =7.5 mg/kg bw/day***	Northern bobwhite quail, reproduction study
Terrestrial Invertebrates	Acute		***indefinite oral LD50 = 620 mg/kg bw***	honey bees, [derived by LD50 of >72 µg (0.072 mg) ÷ 0.000116 kg bee bw ≈ 620.68 mg/kg bw]
Terrestrial Plants (Macrophytes)	Seedling Emergence	sensitive spp.	***NOEC = 0.0028 lb/acre***	soybean (a dicot), TEA, based on shoot length
		tolerant spp.	***NOEC = 0.23 lb/acre***	barley (a monocot), TEA, based on shoot length
	Vegetative vigor	sensitive spp.	***NOEC = 0.0028 lb/acre***	sunflower (a dicot), TEA and BEE
		tolerant spp.	***NOEC = 2.0 lb/acre***	oat (a monocot), BEE [converted from >2242 g a.i./ha]
Fish	Acute	sensitive spp.	***estimated NOAEC = 20 mg/L***	acid [derived from LC50 of 40.1 mg a.e./L x LOC of 0.5]
		tolerant spp.	***estimated NOAEC = 210 mg/L***	acid [LC50 of 210 mg a.e./L x LOC of 0.5]
	Chronic	sensitive spp.	***estimated NOAEC = 7.4 mg/L***	acid, [acute NOAEC 20 mg a.e./L x acute-to chronic ratio 0.37 = 7.4 mg a.e./L]
		tolerant spp.	***estimated NOAEC = 78 mg/L***	acid [acute NOAEC 210 mg a.e./L x acute-to-chronic ratio 0.37 =77.7 mg a.e./L]
Amphibians	Acute	sensitive spp.	***NOAEC = 125 mg/L***	African clawed frog (*Xenopus laevis*), embryos, TEA, for growth (only study)
		tolerant spp.	***NOAEC = 125 mg/L***	
	Chronic	sensitive spp.	***N/A***	No data available
		tolerant spp.	***N/A***	
Aquatic invertebrate	Acute	sensitive spp.	***adjusted NOAEC = 25 mg/L***	[estimated acute NOAEC of 5 mg a.e./L is adjusted upward to 25 mg a.e./L given the chronic NOAEC]

		tolerant spp.	***estimated NOAEC = 320 mg/L***	[LD50 of 6,400 mg/L x LOC factor of 0.05]
	Chronic	sensitive spp.	***NOAEC = 25 mg/L***	daphnid, cannot be classified as sensitive, tolerant, or intermediate
		tolerant spp.		
Aquatic Plants (Macrophytes)	Acute	sensitive spp.	***marginal NOAEC = 0.0005 mg/L***	Eurasian water milfoil (*Myriophyllum sibiricum*; a dicot), NOAEL is a biochemical indicator of an adverse effect but no overt effect found.
		tolerant spp.	***NOEC = 5.6 mg/L***	duckweed (*Lemna minor*; a monocot) Garlon 3A (32.3% a.e.)
	Chronic	sensitive/ tolerant spp.	***N/A***	No data available
Aquatic Algae (Microphytes)	sensitive spp.		***NOEC = 0.23 mg+E41/L***	*Ankistrodesmus* spp., 5-day study
	tolerant spp.		***estimated NOEC = 4.0 mg/L***	*Chlorella pyrenoidosa*, 4-day study [upper bound EC50 of 80 mg a.e./L x factor of 0.05]

All endpoints are in terms of a.e. BEE = butoxyethyl ester, LD/C = Lethal Dose/Concentration, LOC = level of concern, NOAEL/C = no-observed-adverse-effect-level/concentration, TEA = triethylamine salt.

Table D.3-29

Ecological Endpoints for Triclopyr BEE

		Endpoint	**Receptor, Study & Endpoint Details**
Canine mammals	Acute	***NOAEL = 20 mg a.e./kg bw***	estimated relative to rat [derived by 100 mg/kg bw ÷ 5]
	Chronic	***NOAEL = 1 mg a.e./kg bw***	estimated relative to rat: [derived by 5 mg/kg bw ÷ 5]
Medium mammals	Acute	***NOAEL = 100 mg/kg bw***	rat
	Chronic	***NOAEL = 5 mg/kg bw***	rat
Small mammals	Acute	***NOAEL = 440 mg/kg bw***	estimated relative to rat [derived by 100 mg/kg bw x 4.4]
	Chronic	***NOAEL = 22 mg/kg bw***	estimated relative to rate [derived by 5 mg/kg bw x 4.4]
	Acute	***NOAEL = 8 mg/kg bw***	estimated relative to rat [derived by 100 mg/kg bw ÷ 40 13 ≈ 7.69]

Large herbivore mammals	Chronic		***NOAEL = 0.4 mg/kg bw***	estimated relative to rat [derived by 5 mg/kg bw ÷ 13 ≈ 0.38]
Birds	Acute		***NOAEL = 126 mg /kg bw***	Northern bobwhite quail, BEE gavage study
	Chronic		***NOAEL =7.5 mg/kg bw/day***	Northern bobwhite quail, reproduction study
Terrestrial Invertebrates	Acute		***indefinite oral LD50 = 620 mg/kg bw***	honey bees [derived by LD50 of >72 µg (0.072 mg) ÷ 0.000116 kg bee bw ≈ 620.68 mg/kg bw]
Terrestrial Plants (Macrophytes)	Seedling Emergence	sensitive spp.	***NOEC = ~0.022 lb/acre***	soybeans (a dicot); BEE, equivalent to 35 g a.i/ha, based on shoot weight
		tolerant spp.	***NOEC = 2.0 lb/acre***	corn, oats, sunflowers, wheat, BEE study, [converted from >2242 g a.i./ha based on shoot weight]
	Vegetative vigor	sensitive spp.	***NOEC = 0.0028 lb/acre***	sunflower (a dicot), TEA and BEE
		tolerant spp.	***NOEC = 2.0 lb/acre***	oat (a monocot), BEE [converted from >2242 g a.i./ha]
Fish	Acute	sensitive spp.	***NOAEC = 0.091 mg/L***	bluegills, BEE [converted from a.i. to a.e.]
		tolerant spp.	***adjusted NOAEC = 0.75 mg/L***	flathead minnows, BEE [LC50 of 1.5 mg a.e./L x LOC of 0.5]
	Chronic	sensitive spp.	***U.S. EPA adjusted NOAEC = 0.019 mg/L***	rainbow trout, BEE [Chronic exposure to BEE are far below this dose, and thus protective of all spp. sensitivity]
		tolerant spp.		
Amphibians	Acute	sensitive spp.	***surrogate NOAEC: sublethal EC10 = 0.1 mg/L,***	*Rana clamitans* larvae, TEA, abnormal avoidance response.
		tolerant spp.	***estimated NOAEL = 4.2 mg/L***	*Rana clamitans* embryos, TEA [LC50 of 24.6 mg a.e./L x 0.17 (factor resulting from ratio of an NOAEC to LC50)]
	Chronic	sensitive spp.	***N/A***	No data available
		tolerant spp.	***N/A***	
Aquatic invertebrate	Acute	sensitive spp.	***estimated NOAEC = 0.045 mg/L***	[LD50 of 0.25 mg a.e./L x a factor of 0.18 (lower bound of mean that resulted from ratios of NOAEC to LC50 values)]

		tolerant spp.	***estimated NOAEC = 3.6 mg/L***	[LD50 of 20.0 mg a.e./L x a factor of 0.18 (lower bound of mean that resulted from ratios of NOAEC to LC50 values)]
	Chronic	sensitive spp.	***LOAEC = 0.25 mg/L***	*Simocephalus vetulus*, concentration-related decreases in reproduction
		tolerant spp.	***estimated LOAEC = 20 mg/L***	[chronic LOAEC of 0.25 mg a.e./L x factor of 80 (ratio of LD50 values for tolerant and sensitive species)]
Aquatic Plants (Macrophytes)	Acute	sensitive spp.	***estimated NOEC = 0.043 mg/L***	[EC50 of 0.86 mg a.e/L x a factor of 0.05]
		tolerant spp.	***estimated NOEC = 0.31 mg/L***	[EC50 of 6.25 mg a.e/L x a factor of 0.05]
	Chronic	sensitive/ tolerant spp.	***N/A***	No data available
Aquatic Algae (Microphytes)	sensitive spp.		***U.S. EPA estimated NOEC = 0.0014 mg/L***	*Navicula pelliculosa,*[~0.002 mg a.i./L]
	tolerant spp.		***NOEC = 1.0 mg/L***	*Skeletonema costatum*

All endpoints are in terms of a.e. BEE = butoxyethyl ester, ED/C = Effect Dose/Concentration, LD/C = Lethal Dose/Concentration, LOEC = lowest-observed-effect-concentration, N/A = Not Applicable, NOAEL/C = no-observed-adverse-effect-level/concentration, TEA = triethylamine salt.

1.3.2.5 Risk Characterization

1.3.2.5.1 Introduction

Conceptually, risk characterization is simply the process of comparing the exposure assessment to the dose-response assessment. In this process, risk is characterized quantitatively as a ratio. Because the risk characterization flows directly from the exposure and dose-response assessments, the complexity and clarity of the risk characterization will be dependent on complexity and clarity of both the exposure and dose-response assessments. In most cases, risk will be quantitatively characterized as a ratio: a level of exposure divided by some defined effect level. In the human health risk assessment, the defined effect level is almost always the reference dose (RfD), and the ratio of the exposure to the reference dose is referred to as the hazard quotient (HQ). In the ecological risk assessments, the defined effect level may be an NOEC or a risk level. The risk level, in turn, may be a lethal dose (e.g., LD_{50} or some other response level such as an LD_{25}) or a dose causing some risk of a non-lethal effect (e.g., an ED_{50} or ED_{25}). For aquatic organisms and for some terrestrial organisms for which exposure is characterized by a concentration rather than a dose, the defined risk levels may be expressed as a lethal concentration (LC_{50} or some other response level) or a sublethal concentration that leads to some effect (e.g., an EC_{50}). In

general, the Forest Service prefers to use NOAEL or NOEC values in risk characterizations. If NOAEL or NOEC values are not available, a sublethal effective dose at some response rate may be used to approximate a NOAEL or NOAEL.

The following is a characterization of the risks associated with plausible levels of exposure to the chemicals, and in some cases metabolites and surfactants, likely to be used in the VTP and alternatives. This is a synthesis of the hazard (toxicity) of each chemical, the likelihood of exposure to non-target organisms, and the likelihood that non-target organisms would be adversely affected by plausible levels (doses) of chemicals. The characterization of risk is substantially from the most recent USDA/FS and SERA risk assessments (RAs) for each chemical analyzed. These RAs can be downloaded from the USFS Forest Health Protection website at (http://www.fs.fed.us/foresthealth/pesticide/risk.shtml). These RAs have been updated using information from the 2012 EXCEL "F" and "G" series workbooks created by WorksheetMaker. The most current version of WorksheetMaker can be downloaded directly from the SERA website (www.sera-inc.com).

As cautioned in the SERA risk assessment for clopyralid (SERA 2004a, p. xviii), when considering the risks portrayed in SERA RAs: "*The risk characterization for both terrestrial and aquatic animals is limited by the relatively few animal and plant species on which data are available compared to the large number of species that could potentially be exposed. This limitation and consequent uncertainty is common to most if not all ecological risk assessments*."

As discussed above in Section D.3.2.2, *Hazard (Toxicity) Identification*, chemicals that are not approved for aquatic use may be inadvertently applied or transported to shallow wetlands or to low volume or intermittent streams that support frogs and their larvae (tadpoles), and/or other amphibians. There is some scientific evidence that chemicals could accumulate to toxic levels in these shallow, low volume waterbodies. D.G. Thompson (Thompson 2003) measured the toxic effects on Ranid frogs of Vision® (glyphosate), which is not registered for use in California, in 51 wetlands in Canada that were 1) buffered from spraying, 2) sprayed adjacent to the wetland, and 3) over sprayed. No significant differences in mortality to Ranid frogs were observed between the treatments. However, "*vegetated buffers significantly mitigated against exposure and thus potential for acute effects. Aqueous concentrations of Vision® (glyphosate) in buffered wetlands were below analytical limits of quantification (0.02 mg acid equivalent [a.e.]/L) in 14 of 16 cases, with mean concentration (0.03 ± 0.02 mg a.e./L) significantly ($p < 0.05$) less than that of either adjacent (0.18 ± 0.06 mg a.e./L) or over sprayed wetlands (0.33 ± 0.11 mg a.e./L)*" (Thompson 2003).

A study of potential pesticide toxicity (including imazapyr and sulfometuron methyl) in Midwestern streams found that: 1) spring and early summer runoff events can contain pesticides in sufficient quantities to be toxic to non-target aquatic organisms, 2) accounting for herbicide degradates can substantially increase the estimated toxicity of stream water to

aquatic plants, and 3) the quality of this analysis is limited by the lack of acute toxicity data for many of the pesticide-organism combinations (Battaglin and Fairchild 2002). Only 10% of the water samples contained acetolactate synthase (ALS) inhibitor herbicides, a class of herbicides that includes imazapyr and sulfometuron methyl. It was thought that the data from this study might underestimate potential effects of pesticides on aquatic systems in smaller streams because peak concentrations of herbicides were generally inversely related to stream size.

Except for glyphosate formulations containing POEA, sulfometuron methyl (for amphibians), and triclopyr, the chemicals analyzed in this PEIR and potentially applied under the VTP and alternatives are only slightly toxic to practically nontoxic to aquatic organisms. However, there is little to no testing of most of the chemicals for effects on adult amphibians.

Mann et al., 2003 found that: "*Although the relative sensitivity of amphibians to the toxic effects of pesticides and other environmental contaminants has yet to be established, the perceived vulnerability of amphibians to pesticide effects may actually be attributable to their specific habitat requirements. Shallow temporary ponds, essential to the life cycles of many amphibians, are also areas where pollutants may accumulate without substantial dilution. Research in Western Australia has highlighted the potential risk that agricultural chemicals may pose to fauna that inhabit low dilution environments, and indicates that the data currently required for pre-registration assessment of pesticides may be inadequate to effectively protect these environments.*"

Raphael 2003, made the following findings in the forested systems of the western Pacific Northwest: "*While not all* [stream-dwelling amphibians] *respond the same way, there is typically a rapid decrease in population after management activity in the riparian zone, and recovery for some species can be quite slow. In some sites, the numbers are still low as much as 60 years after timber harvest.*"

> *"Potential for large-scale reduction in amphibian numbers is high, and indeed the focus on amphibian population decline worldwide is increasing. It seems clear that amphibian numbers should at least be considered as part of the buffer zone assessment and recommendation process." (ibid)*

Considering the sensitivity of amphibians to microsite conditions and some of the herbicides and surfactants likely to be used under the VTP and alternatives, it is clear that buffer zones are needed, particularly adjacent to shallow wetlands, vernal pools, and ponds and shallow, slow-moving, low-volume, and/or intermittent streams.

Chemicals will be potentially used in the VTP and alternatives to only treat terrestrial vegetation and only by ground-based application methods. Aquatic environments are buffered during spray applications through specific chemical label requirements and court orders applicable to specific chemicals, areas, and species. Buffers to protect special status aquatic

species are required by Standard Project Requirements HAZ-6, HAZ-8, HYD-3, BIO-1, BIO-7, BIO-11, and BIO-13 (see Section 2.5). Such measures will preclude the application of herbicides within watercourse buffer zones as described even when the label allows for use within these buffers.

1.3.2.5.2 Chemical-Specific Risk Characterization

1.3.2.5.2.1 Borax (Sources: FS WSM ver. 6.00.10; SERA 2006a)

Terrestrial and Aquatic Organism Overview

Three exposure scenarios are considered: 1) the direct consumption of Sporax® applied to tree stumps (acute exposure), 2) consumption of water contaminated by an accidental spill (acute exposure), and 3) acute and chronic exposure by consumption of water contaminated by runoff. Other than the direct consumption of Sporax® applied to tree stumps, none of the exposure scenarios for terrestrial organisms are associated with HQs that exceed the LOC.

For terrestrial species, risks associated with the application of Sporax® to tree stumps appear to be very low. At the application rates (lowest 0.1 lb./acre, typical 1 lb./acre, and highest 5 lbs./acre) and methods used in U.S. Forest Service programs and likely to be used under the VTP and alternatives, Sporax® will not substantially contribute to or increase boron concentrations in water or soil beyond those that are associated with its normal occurrence in the environment. The highest HQ (5.6), for the direct consumption of Sporax® from a tree stump by a large mammal, is at the upper bound at the highest application rate.

Most aquatic animals do not appear to be at risk for any of the exposure scenarios (water contaminated by accidental spill or by runoff). Accidental spill of large quantities of Sporax® into a small pond may result in toxicity in amphibians.

HQs for aquatic plants for the accidental spill scenario and for acute and longer-term exposures to water contaminated by runoff are well below the LOC. Sensitive aquatic microorganisms may be at risk following an accidental spill of a large quantity (25 pounds) of Sporax® into a small pond, but exposure *via* runoff does not present a risk.

Terrestrial Organisms

Mammals and Birds – For the direct consumption scenario, there appears to be very little risk to either mammals or birds. Only a large mammal, such as a deer, consuming Sporax® from a treated stump is at risk, with HQs exceeding the LOC at the upper bound (HQ 1.1) at the typical application rate and at the central (HQ 1.7) and upper bound (HQ 5.6) at the highest rate. However, Sporax® applied to tree stumps does not appear to attract deer and deer allowed free access to Sporax®-treated stumps showed no clinical signs of toxicity.

Risk associated with other exposure scenarios are very low, as Sporax will not substantially contribute to or increase boron concentrations in water or soil beyond those that are associated with its normal occurrence in the environment.

Terrestrial Invertebrates – Exposure assessments were not conducted for insects, so risk of exposure cannot be characterized quantitatively. Borax is used effectively to control insects, so adverse effects of environmental exposures are possible. However, given the atypical application method for Sporax®, widespread exposures are not likely.

Terrestrial Plants (Macrophytes) – Even at the at the maximum application rate potentially used under the VTP and alternatives, non-target terrestrial plants do not appear to be at risk from exposure to borax, as no HQ values exceed the LOC. However, since this risk assessment is based on data from relatively few terrestrial plant species, more sensitive species may exist and may be at risk for boron-induced toxicity.

Terrestrial and Aquatic Microorganisms – Exposure assessments were not conducted for soil microorganisms, so risk of exposure cannot be characterized quantitatively. Borax is effective as either a fungicide or an insecticide. Sporax® will be used in the VTP and alternatives as a fungicide, to control annosum root rot, so adverse effects of environmental exposures are possible. However, given the atypical application method for Sporax®, widespread exposures are unlikely.

Aquatic Organisms

Fish – HQs associated with acute exposure of fish to water contaminated by an accidental spill or runoff are all below the LOC, so there is no indication that adverse effects will occur. For chronic exposure of fish to water contaminated by runoff, HQs for both sensitive (HQ 4) and tolerant (HQ 2) species are above the LOC only at the upper bound at the highest application rate. Adverse effects on non-target fish are plausible for longer-term exposures.

Amphibians – If large amounts (25 pounds) of Sporax® accidentally contaminate surface waters, such as a small pond, amphibians may be at risk. HQs for both sensitive and tolerant species exceed one at the highest application rate and the upper bound at the typical rate.

HQs for acute and chronic exposure of amphibians to water contaminated by runoff are above the LOC for both sensitive and tolerant species at the upper bound at the highest application rate. Although HQs are below the LOC at the lower and central bounds at the highest application rate, adverse effects on amphibians are plausible for either acute or longer-term exposures at the upper bound at the highest application rate.

Aquatic Invertebrates – HQs for acute and chronic exposure of aquatic invertebrates to water contaminated by runoff are all below the LOC. There is no basis for asserting that adverse effects are likely for either acute or longer-term exposures to Sporax®.

Aquatic Plants (Algae and Macrophytes) – HQs for the accidental spill scenario and for acute and longer-term exposures to water contaminated by runoff are well below the LOC. There is no basis for asserting that effects on aquatic macrophytes or algae are likely for either acute or longer-term exposures.

Aquatic Microorganisms – HQs for the most sensitive species (but not tolerant species) of microorganisms exceed the LOC for all accidental spill scenarios. All HQs are below the LOC for both sensitive and tolerant species for acute exposure to water contaminated by runoff. More sensitive microorganisms may be at risk following an accidental spill of large quantities of Sporax® into a small pond, but exposure *via* runoff does not present a risk to aquatic microorganisms.

1.3.2.5.2.2 Clopyralid (Sources: FS WSM ver. 6.00.07 & 6.00.10; SERA 2004a)

Terrestrial and Aquatic Organism Overview

The SERA 2004a risk assessment for clopyralid uses a typical application rate of 0.35 lb a.e./acre and an upper application rate of 0.5 lb a.e./acre. In California, the maximum allowable application rate is 0.25 lb a.e./acre. Therefore, information from the SERA 2004a "Risk Characterization" section is adjusted to reflect a lower application rate.

The SERA 2004a risk assessment for clopyralid anticipated no adverse effects in terrestrial or aquatic animals from the use of clopyralid in U.S. Forest Service programs at the typical application rate of 0.35 lb a.e./acre. However, using the 2012 Excel Worksheets, at an application rate of 0.25 lb a.e./acre, HQs are above the LOC at the upper bound for some exposure scenarios for terrestrial organisms.

For aquatic organisms, HQs are only above the LOC at the central and upper bounds for the acute accidental spill exposure scenario for tolerant aquatic macrophytes (no data on sensitive species) and for sensitive algae at the upper bound.

Terrestrial Organisms

Mammals – At an application rate of 0.25 lb a.e./acre, HQs for all terrestrial organisms are above the LOC at the upper bound for all acute and chronic exposure scenarios of small mammals consuming contaminated grass and broadleaf foliage. HQs range from 1.3 to 6, with the highest HQ for a small mammal consuming contaminated short grass. The only scenario where the HQ (1.4) for a larger animal exceeds the LOC is for long-term consumption of contaminated short grass. However, the scenario of a mammal consuming vegetation on-site is essentially used as a very conservative/extreme screening scenario. It assumes that animals stay in treated areas consuming nothing but contaminated vegetation. Since most forms of vegetation would likely die after herbicide applications, or at least be substantially damaged, this exposure scenario is implausible. Still, adverse acute and chronic effects are

plausible based on consumption of contaminated vegetation, especially longer term consumption of short grass, by small mammals.

Birds – HQs for small birds are above the LOC at the upper bound for chronic exposure scenarios involving consumption of contaminated fruit, tall and short grass, and vegetation. HQs range from 1.3 to 15, with the highest HQ for a small bird consuming contaminated vegetation. The HQ (1.3) also exceeds the LOC for a small bird consuming contaminated tall grass at the central bound. The only scenario where the HQ (1.7) for a larger bird exceeds the LOC is for consumption of contaminated vegetation. However, the scenario of a bird consuming vegetation on-site is essentially used as a very conservative/extreme screening scenario. It assumes that animals stay in treated areas consuming nothing but contaminated vegetation. Since most forms of vegetation would likely die after herbicide applications, or at least be substantially damaged, this exposure scenario is implausible. Still, adverse chronic effects are plausible based on consumption of contaminated vegetation, especially longer term consumption by small birds.

Terrestrial Invertebrates – As there is a dearth of data available, values relating to honey bee exposure are used to represent the effects clopyralid may have on terrestrial invertebrates.

At the highest application rate of 0.25 lb a.e./acre, the estimated maximum concentrations of clopyralid in clay soil would range from about 0.066 mg/kg at an annual rainfall of 10 inches to 0.07 mg/kg at an annual rainfall of 100 inches. Due to percolation, concentrations in loam and sand soils would be less. Concentrations of clopyralid in clay, loam, and sand over a wide range of rainfall rates are summarized in Table 4-2 in SERA 2004a (p. Tables-12).

While the available toxicity data on soil organisms are limited, these projected maximum concentrations in soil are far below potentially toxic levels. Information on the toxicity of clopyralid to soil organisms is limited, consisting only of an acute LC_{50} value for earthworms reported as >1000 mg/kg soil and a report on soil microorganisms indicating an NOEC of 10 ppm soil for effects on nitrification, nitrogen fixation, and degradation of carbonaceous material. This information does not provide any basis for asserting that adverse effects on soil invertebrates are plausible. (SERA 2004a, p. 4-25)

Terrestrial Plants (Macrophytes) – Clopyralid is an auxin-mimicking herbicide that is formulated to control many annual and perennial broadleaf plants, particularly of the Asteraceae (sunflower), Fabaceae (legume), Polygonaceae (knotweed), and Solanaceae (nightshade) families. It has been used to control non-native invasive species such as Canada thistle, Russian knapweed, yellow star thistle, and English ivy. Like other auxin-mimicking herbicides, clopyralid has little to no effect on grasses and other monocots, plants in the Brassicaceae (mustard) family, and several other groups of broad-leaved plants. (TNC 2001)

Clopyralid is an extremely effective herbicide in trace concentrations. Studies have determined that it will bind to organic matter when treated vegetation is composted and will remain active for some time. If the compost is spread around susceptible non-target plants, they could be damaged or killed. If livestock eat clopyralid-treated vegetation, the chemical will pass through the digestive system and be eliminated in manure, still in an active form. Wherever the manure lands, susceptible non-target plants could be damaged or killed. (TNC 2001)

Drift is likely to cause adverse effects on some non-target plant species under certain application conditions and circumstances. Off-site drift of clopyralid associated with ground applications may cause damage to sensitive plant species at distances of about 300 feet (HQ 2) from the application site. Tolerant plant species would probably not be impacted and might show relatively little damage.

As stated in SERA 2004a, p. 4-25, "*The situational variability in the exposure assessments for runoff, wind erosion, and irrigation water has a substantial impact on the characterization of risk for sensitive non-target plant species. These scenarios may overestimate or underestimate risk under certain conditions.*"

The SERA 2004a (p. 4-23) risk assessment for clopyralid states that: "*Because of the tendency for clopyralid to move into soil rather than to be transported by runoff and because of the greater toxicity of clopyralid by foliar deposition compared to soil contamination, off-site movement of clopyralid by soil runoff does not appear to be substantial risk to nontarget plant species.*" Runoff does not appear to present a significant risk to sensitive or tolerant non-target plant species even under conditions in which runoff is favored (clay soil over a very wide range of rainfall rates).

Wind erosion could lead to adverse effects in sensitive plant species. Soil losses by wind erosion are substantially less than off-site losses associated with runoff from clay soils, but similar to off-site losses from drift in the range of about 200-900 feet from the treatment site. Wind erosion of contaminated soil is most plausible in relatively arid environments and if local soil surface and topographic conditions are favorable.

As stated in SERA 2004a, p. 4-25: "*The simple verbal interpretation for this quantitative risk characterization is that sensitive plant species could be adversely affected by the off-site drift of clopyralid under a variety of different scenarios depending on local site-specific conditions that cannot be generically modeled. If clopyralid is applied in the proximity of sensitive crops or other desirable sensitive plant species, site-specific conditions and anticipated weather patterns will need to considered if unintended damage is to be avoided. More tolerant plant species are not likely to be affected unless they are directly sprayed.*"

Terrestrial and Aquatic Microorganisms – At the highest application rate of 0.25 lb a.e./acre, the estimated maximum concentrations of clopyralid in clay soil would range from

about 0.066 mg/kg at an annual rainfall of 10 inches to 0.07 mg/kg at an annual rainfall of 100 inches. Due to percolation, concentrations in loam and sand soils would be less. Concentrations of clopyralid in clay, loam, and sand over a wide range of rainfall rates are summarized in Table 4-2 in SERA 2004a (p. Tables-12).

As stated in SERA 2004a, p. 4-26: "While the available toxicity data on soil organisms are limited, these projected maximum concentrations in soil are far below potentially toxic levels. The information on soil organisms is limited, however, consisting only of an acute LC_{50} value for earthworms reported as >1000 mg/kg soil (Section 4.3.2.3) and a report in soil microorganisms indicating an NOEC of 10 ppm soil for effects on nitrification, nitrogen fixation, and degradation of carbonaceous material (Section 4.3.2.5). Nonetheless, this information does not provide any basis for asserting that adverse effects on soil organisms are plausible."

Aquatic Organisms

The SERA 2004a (p. 4-23) risk assessment for clopyralid states that: "*Aquatic plants do not appear to be at any substantial risk from any plausible acute or chronic exposures. In the very extreme case of an accidental spill of a large amount of the herbicide into a relatively small body of water, sensitive aquatic plants could be damaged.*"

Clopyralid appears to have a very low potential to cause any adverse effects in any aquatic species, although there is no data available for amphibians or sensitive species of invertebrates or macrophytes, so risk is not characterized for these aquatic organisms.

Fish – There are no exposure scenarios for fish that approach a LOC. Chronic toxicity studies in fish are lacking. For the HQ in fish to reach a LOC they would have to be more sensitive than daphnids by a factor of 2500, based on the maximum HQ (0.0004) for daphnids for chronic exposures, at an application rate of 0.25 lb./acre. It is unlikely that fish would experience acute or chronic adverse effects at the maximum application rate.

Concentrations of clopyralid in ambient water with an application rate of 0.25 lb/acre are estimated to be no greater than 0.00325 mg/L over prolonged periods of time. The peak concentration associated with runoff or percolation is estimated to be no more than 0.0175 mg/L.

Amphibians – No toxicity data is available for amphibians so risk is not characterized.

Aquatic Invertebrates – There are no acute or chronic exposure scenarios for tolerant species of aquatic invertebrates where the HQ exceeds the LOC. No toxicity data is available for sensitive species of invertebrates, so risk is not characterized for these aquatic organisms. It is unlikely that aquatic invertebrates would experience acute or chronic adverse effects at the maximum application rate.

Aquatic Plants (Algae and Macrophytes) – The HQs for tolerant species of aquatic macrophytes for accidental acute exposures range from 11 at the central bound to 114 at the upper bound, well above the LOC. HQs for all other exposure scenarios for tolerant species are well below the LOC. No toxicity data is available for sensitive species of macrophytes, so risk is not characterized. The HQ for sensitive species of algae for accidental acute exposures is 1.7 at the upper bound. HQs at the central and lower bounds for both sensitive and tolerant algae are well below the LOC. There is no basis for asserting that effects on non-target aquatic plants are likely, except in cases of accidental contamination of a small body of water, when adverse effects in sensitive aquatic plants are plausible.

1.3.2.5.2.3 Glyphosate (Sources: FS WSM v. 6.00.10; SERA 2011b; U.S. EPA. 2009c)

Terrestrial and Aquatic Organism Overview

Glyphosate is a broad-spectrum, nonselective systemic herbicide that is formulated to suppress or kill many grasses, forbs, vines, shrubs, and trees. It is commonly used in natural areas to control many non-native invasive species. But because it is nonselective it should be used carefully so as not to damage or kill desirable native plants. (TNC 2001)

Glyphosate can be applied to the foliage, green stems, and cut-stems (cut-stumps) of terrestrial plants, but is unable to penetrate woody bark. Since glyphosate by itself is essentially non-toxic to submersed plants, specific formulations (e.g., Rodeo®) are registered for aquatic use. These formulations do not have the adjuvants that may be toxic to aquatic plants and animals. (ibid)

This risk characterization is based on the following ground application rates that may potentially be used under the VTP and alternatives: lowest application rate of 0.29 lb. a.e./acre, typical application rate of 2.0 lbs. a.e./acre, and highest application rate of 8.0 lbs. a.e./acre.

This risk characterization of glyphosate is designed to clearly differentiate between the more toxic and less toxic formulations. As stated in SERA 2011b, p. 201: "*While some, formulations cannot be easily classified as more or less toxic, the general approach discussed in the dose-response assessment (Section 4.3.1) is applicable to the risk characterization: any formulation that contains a POEA surfactant should be regarded as more toxic, unless there is compelling evidence to the contrary. If the presence and/or toxicity of the surfactants in the formulation cannot be determined, it is prudent to classify the formulation as more toxic.*"

For terrestrial organisms, other than plants, applications of up to 2.5 lb a.e./acre of the more toxic formulations of glyphosate do not present any apparent risks. At application rates, greater than 2.5 lb a.e./acre, risks to mammals cannot be ruled out at upper bound estimates of exposure, but are not apparent at central estimates of exposure. At application rates,

greater than approximately 3.3 lb a.e./acre, the HQs for birds modestly exceed the LOC, but there is no demonstrated evidence that these exposure levels will cause overt toxicity in birds.

Risks to terrestrial insects from dietary exposures are of greater concern than risks from direct spray. As stated in the "Overview" in SERA 2011b, p. 201, "*Based on upper bound estimates of exposure at the maximum application rate of 8 lb a.e./acre, the HQs for terrestrial insects can reach a value of 10. Concern for terrestrial invertebrates is enhanced by two toxicity studies using South American formulations of glyphosate in which adverse effects on reproduction and development were noted. While most field studies suggest that effects on terrestrial invertebrates are due to secondary effects on vegetation, the field studies do not directly contradict the South American toxicity studies or the HQs.*"

> *"The risk characterization for aquatic organisms suggests that amphibians are the group at greatest risk both in terms of sensitivity and severity of effects. At an application rate of 1 lb a.e./acre, the upper bound HQ for amphibians is 2. The corresponding HQs for other groups of aquatic organisms are 1.7 for fish, 1.1 for invertebrates, 1.0 for algae, and 0.008 for aquatic macrophytes. Concern for amphibians is enhanced by the Howe et al., (2004) study which indicates that two formulations of Roundup as well as the POAE surfactant used in some of the more toxic formulations of glyphosate are associated with the development of intersex gonads. The HQs for aquatic species will increase linearly with the application rate. Because the upper bound HQs for most groups of aquatic organisms exceeds or reaches the level of concern at the relatively low application rate of 1 lb a.e./acre, care should be exercised when applying more toxic formulations of glyphosate near surface water." (SERA 2011b, p. 202)*
>
> *"The less toxic formulations of glyphosate do not appear to present any risks to terrestrial organisms other than terrestrial plants. Unlike the case with more toxic formulations, risks to amphibians and aquatic invertebrates appear to be insubstantial. Algae appear to be the most sensitive group of nontarget aquatic organisms. At an application rate of 1 lb a.e./acre, the upper bound of the HQ for sensitive species of algae is 0.8." (ibid)*
>
> *"Risks to fish cannot be ruled out based on standard and conservative assumptions and methods for applications of less toxic formulations of glyphosate at rates in excess of about 2.5 lb a.e./acre (acute effects). It seems most likely, however, that adverse effects would be observed in stressed populations of fish and less likely that effects would be noted in otherwise healthy populations of fish." (ibid)*
>
> *"The less toxic formulations of glyphosate require the use of a surfactant. Some surfactants such as Agri-Dex (LC_{50} >1000 mg/L) are virtually nontoxic, and the use of a nontoxic surfactant would have no substantial impact on the risk characterization.*

Based on the available toxicity data in fish and aquatic invertebrates, some surfactants that may be used with the less toxic formulations of glyphosate could pose a much greater risk than the glyphosate formulation itself." (ibid)

Terrestrial Organisms

The most recent for glyphosate differentiates risk between the more toxic and the less toxic formulations. Formulations that are known to contain the surfactant POEA are considered more toxic. Formulations where the toxicity or presence of surfactants is unknown are also considered more toxic. As stated in the SERA risk assessment (SERA 2011d, p. 201):

For terrestrial organisms other than plants, applications of up to 2.5 lbs a.e./acre of the more toxic formulations do not present any apparent risk, based on upper bound estimates of exposure levels. At application rates greater than 2.5 lbs a.e./acre, risks to mammals cannot be ruled out, based on upper bound estimates of exposure; however, no risks are apparent, based on central estimates of exposure. At application rates greater than approximately 3.3 lbs a.e./acre, the HQs for birds modestly exceed the level of concern; however, there is no demonstrated evidence that these exposure levels will cause overt toxicity in birds.

The less toxic formulations of glyphosate do not appear to present any risks to terrestrial organisms other than terrestrial plants.

Mammals – For more toxic formulations of glyphosate, HQs for accidental acute exposures exceed the LOC only at the highest application rate at the central and upper bounds. For non-accidental acute exposure at the typical application rate, central bound, only small mammals have a HQ (1.6) exceeding the LOC, from consuming contaminated tall and short grass. At the upper bound, HQs range from 1.1 to 8 for non-accidental acute exposures to small mammals consuming broadleaf foliage (4), tall and short grass (both 8), and insects (1.1) and large (70 kg) mammals consuming short grass (HQs of 1.8). For chronic (long term) exposure at the typical application rate, upper bound, only small mammals consuming short grass have a HQ (1.3) exceeding the LOC.

At the highest application rate of glyphosate, the only HQ above the LOC for the accidental direct spray scenario is for a small mammal at the central (1.1) and upper (2) bounds. At the central bound non-accidental acute exposure HQs range from 3 to 7 for small mammals consuming broadleaf vegetation and at the upper bound HQs range from 3 to 32 for small mammals consuming fruit (3), broadleaf foliage (18), tall and short grass (each 32), and insects (4). HQs for larger (400g) mammals consuming vegetation or insects at the upper bound range from 3 to 7 and HQs for large mammals consuming vegetation range from 1.9 to 4, modestly greater than the LOC. For chronic (long term) exposure, HQs for small mammals consuming short grass (1.3) exceed the LOC only at the typical application rate (upper bound) and at the highest application rate central (1.1) and upper (5) bounds and for

larger mammals consuming short grass (HQ 1.2). Based on the upper bound at the highest application rate, adverse effects are plausible only for small mammals consuming contaminated tall and short grass.

For these worst-case exposure assessments, at the central bound at the typical application rate and the upper bound at the highest application rate, adverse effects are plausible only for small mammals consuming contaminated broadleaf foliage and tall and short grass. However, well-documented field studies have not identified adverse effects in populations of small mammals following applications of Roundup and an unidentified formulation of glyphosate.

For less toxic formulations of glyphosate, at the typical application rate, HQs exceed the LOC only at the upper bound, for small mammals for the scenarios of accidental acute exposure from consuming contaminated broadleaf foliage (HQ 1.6) and tall and short grass (HQs 3). HQs for most of the other scenarios at the central bound are well below the LOC. Based on the upper bound at the highest application rate, adverse effects are plausible only for small mammals consuming contaminated broadleaf foliage and tall and short grass.

Birds – For more toxic formulations of glyphosate, there are no HQs that exceed the LOC for the accidental direct spray scenario. At the typical application rate, central bound, only small birds have a HQ (1.3) exceeding the LOC, for the scenario of non-accidental acute exposure from consuming contaminated short grass. At the upper bound, HQs range from 3 to 6 for non-accidental acute exposures to small birds consuming broadleaf foliage (4), tall (3) and short (6) grass. For chronic (long term) exposure at the typical application rate, upper bound, HQs exceed the LOC for small birds consuming tall (7) and short grass (6) and for large birds consuming short grass (1.4).

At the highest application rate of glyphosate, for the non-accidental acute exposure (central bound) HQs range from 2 to 5 for small birds consuming vegetation. At the upper bound, HQs range from 1.9 to 25 for small birds consuming fruit (1.9), broadleaf foliage (14), tall (12) and short (25) grass, and insects (3). HQs for larger (400g) birds consuming vegetation at the upper bound are 1.3 for tall grass and 3 for short grass, modestly greater than the LOC. For chronic (long term) exposure, HQs at the upper bound at the highest rate range from 4 to 51 for small birds consuming fruit (4), tall grass (29), short grass (23), and contaminated vegetation (51). Based on the upper bound at the highest application rate, adverse acute effects and longer term chronic effects from exposure to the more toxic formulations of glyphosate are plausible for small birds consuming contaminated tall and short grass and vegetation. However, longer term worst-case exposure assessments are based on the assumption that 100% of the diet is contaminated, which is unlikely, as birds may feed only sporadically in treated areas.

For less toxic formulations of glyphosate, there are no HQs that exceed the LOC for the accidental direct spray scenario at either the typical or highest rate of application. At the typical application rate, HQs exceed the LOC only at the upper bound for small birds for the scenarios of non-accidental acute exposure from consuming contaminated broadleaf foliage (1.3) and short grass (2). HQs for the other exposure scenarios at the central bound are well below the LOC. At the upper bound, HQs range from 1.2 to 9 for small birds consuming broadleaf foliage (5), tall (4) and short (9) grass, and insects (1.2). For chronic (long term) exposure, HQs at the upper bound at the highest rate range from 3 to 38 for small birds consuming fruit (3), tall grass (21), short grass (17), and contaminated vegetation (38) and large birds consuming tall and short grass (both 2) and contaminated vegetation (4). Based on the upper bound at the highest application rate, adverse acute effects and longer term chronic effects from exposure to the more toxic formulations of glyphosate are plausible for small birds consuming contaminated tall and short grass and vegetation. However, longer term worst-case exposure assessments are based on the assumption that 100% of the diet is contaminated, which is unlikely, as birds may feed only sporadically in treated areas.

Terrestrial Invertebrates – Risks from direct spray and off-site drift are based on the direct spray of a honeybee. At the highest application rate of 8 lb a.e./acre, the HQ would be about 2.4, modestly higher than the LOC. As stated in SERA 2011b, p. 205, “Thus, while risks to honeybees from a direct spray cannot be excluded at the highest application rate, the effects would not be substantial and probably would not be detectable. Regardless of the application rate, no exposures associated with spray drift exceed the level of concern at any application rate.”

At the upper bound at the highest application rate, HQs exceed the LOC for terrestrial invertebrates consuming short grass (10), broadleaf vegetation and small insects (6), and long grass (5). However, the use of toxicity data on honeybees as a surrogate for other terrestrial invertebrates consuming contaminated vegetation or prey adds uncertainty to this quantitative risk characterization. Two studies raise concerns that moderate to high application rates of more toxic formulations of glyphosate could have an adverse impact on some terrestrial invertebrates. For the most part, available field studies on terrestrial invertebrates do not reinforce a concern for terrestrial invertebrates. Most field studies suggest that effects on terrestrial invertebrates will be minimal and secondary to changes in vegetation.

Terrestrial Plants (Macrophytes) –SERA 2011d (p. 201) found that: “Glyphosate is an effective post emergent herbicide. Foliar applications of glyphosate with an effective surfactant (POEA or otherwise) may pose a risk to terrestrial plants. The direct spray of a nontarget terrestrial plant at an effective application rate is likely to kill or seriously injure most plants. Nonetheless, substantial differences in sensitivity to glyphosate are apparent among different species of plants. For sensitive species, offsite drift of glyphosate can pose a risk.

The nature of the risk depends on the application rate, application method, and site-specific conditions that affect the extent of drift."

In direct foliar applications, glyphosate is an extremely effective herbicide. No distinction is made in the dose-response assessment between more and less toxic glyphosate formulations for terrestrial plants. Direct spray HQs are 1,538 for sensitive species and 4 for tolerant species at the typical application rate. At the highest application rate the HQs are three times higher. Over the range of glyphosate application rates that might potentially be used under the VTP and alternatives, the unintended direct spray of non-target terrestrial vegetation will potentially damage tolerant plant species and is certain to kill sensitive species.

The risk characterization for drift differs substantially for sensitive and tolerant species of macrophytes. At the typical application rate of 2 lb a.e./acre, risks to sensitive species from drift exceed the LOC at distances of 100 feet for backpack applications. To reach a LOC at 900 feet downwind would require glyphosate to be applied at a rate of 5 lbs. a.e./acre. For ground broadcast applications, the LOC for sensitive species would be exceeded at 900 feet (HQ 1.7) from the application site, but tolerant species would exceed the LOC only at the application site. All the HQs would increase by three times at the highest application rate. For tolerant species, risks associated with drift appear to be minimal because of backpack and ground broadcast applications.

Glyphosate is not particularly effective as an herbicide at any application rate when applied to soils. All HQs, even at the highest application rate, are substantially below the LOC, so the transport of glyphosate in runoff is of no concern. Since the central and upper bounds of the functional application rates of glyphosate in irrigation water are below those associated with runoff, the risks of contaminated irrigation water are not considered further. A similar risk characterization applies to wind erosion, as all HQs are substantially below the LOC at the highest application rate.

Terrestrial and Aquatic Microorganisms – In studies in which arthropods were fed prey contaminated with formulations of glyphosate, a spectrum of adverse effects was noted. Although glyphosate may be toxic to terrestrial microorganisms in laboratory cultures, numerous field studies fail to demonstrate adverse effects. Glyphosate is readily metabolized by soil bacteria and many species of soil microorganisms can use glyphosate as a sole carbon source. There is sufficient evidence that direct toxic effects on soil microorganism are not likely to occur due to glyphosate exposure.

Aquatic Organisms

SERA 2011d (p. 201) found that: "*Terrestrial applications of the more toxic formulations of glyphosate may pose a risk to sensitive species of aquatic plants with an upper bound HQ of 1 at the unit application rate of 1 lb a.e./acre and an HQ of 8 at an application rate of 8 lb a.e./acre.*"

The most recent SERA RA (SERA 2011d, p. 202) for glyphosate distinguishes risk based on the toxicity of the formulations. The risk from more toxic formulations is as follows:

> *The risk characterization for aquatic organisms suggests that amphibians are the group at greatest risk both in terms of sensitivity and severity of effects. At an application rate of 1 lb a.e./acre, the upper bound HQ for amphibians is 2. The corresponding HQs for other groups of aquatic organisms are 1.7 for fish, 1.1 for invertebrates, 1.0 for algae, and 0.008 for aquatic macrophytes. Concern for amphibians is enhanced by the Howe et al., (2004) study which indicates that two formulations of Roundup as well as the POEA surfactant used in some of the more toxic formulations of glyphosate are associated with the development of intersex gonads. The HQs for aquatic species will increase linearly with the application rate. Because the upper bound HQs for most groups of aquatic organisms exceeds or reaches the level of concern at the relatively low application rate of 1 lb a.e./acre, care should be exercised when applying more toxic formulations of glyphosate near surface water.*

SERA 2011d (p. 202) characterizes risk for less toxic formulations as follows:

> *Unlike the case with more toxic formulations, risks to amphibians and aquatic invertebrates appear to be insubstantial. Algae appear to be the most sensitive group of nontarget aquatic organisms. At an application rate of 1 lb a.e./acre, the upper bound HQ for sensitive species of algae is 0.8.*
>
> *Risks to fish cannot be ruled out based on standard and conservative assumptions and methods for applications of less toxic formulations of glyphosate at rates in excess of about 2.5 lbs a.e./acre (acute effects). It seems most likely, however, that adverse effects would be observed in stressed populations of fish and less likely that effects would be noted in otherwise healthy populations of fish.*
>
> *The less toxic formulations of glyphosate require the use of a surfactant. Some surfactants such as Agri-Dex (LC_{50}>1000 mg/L) are virtually nontoxic, and the use of a nontoxic surfactant would have no substantial impact on the risk characterization. Based on the available toxicity data in fish and aquatic invertebrates, some surfactants that may be used with the less toxic formulations of glyphosate could pose a much greater risk than the glyphosate formulation itself.*

Fish – For more toxic formulations of glyphosate, accidental acute exposures (from spills into small bodies of water) exceed the LOC even at the central bound at the lowest application rate of 0.29 lb. a.e./L. At the upper bound at the highest application rate the HQ for sensitive species of fish is 2,996 and for tolerant species it is 288. For non-accidental acute exposures at the upper bound at the highest application rate, HQs are much lower; 14 for sensitive

species and 1.3 for tolerant species. All chronic exposure HQs are below the LOC and most are substantially lower.

Because of concerns with sublethal effects, all the HQs are derived from surrogate NOAECs that are based on LC_{50} values. An HQ of 20, which is not exceeded in the non-accidental or chronic scenarios, would be associated with substantial mortality. However, all the LC_{50} values used in the dose-response assessment involve fasted fish, and a study has shown that the toxicity of glyphosate is reduced by about a factor of 10 in fed fish, relative to fasted fish. HQs for populations of fish in areas where the food supply is adequate could overestimate risk. Water containing suspended sediments has been shown to reduce the toxicity of glyphosate to aquatic macrophytes, so it seems reasonable to assert that suspended sediments could reduce the bioavailability to fish of glyphosate and surfactants used with glyphosate.

As stated in SERA 2011b, p. 209, "*The most reasonable qualitative risk characterization is that risks to fish cannot be ruled out based on standard and conservative assumptions and methods for applications of more toxic formulations of glyphosate. Nonetheless, it is not clear that any effects would be evident in healthy populations of fish in habitats with adequate supplies of food. Adverse effects could be more likely, however, in stressed populations of fish.*" The obvious exception to this characterization would be in the event of an accidental spill into a small body of water.

For less toxic formulations of glyphosate, accidental acute exposures exceed the LOC for sensitive (but not tolerant) species of fish even at the central bound at the lowest application rate. At the upper bound at the highest application rate the HQ for sensitive species of fish is 288 and for tolerant species it is 7. For non-accidental acute exposures at the upper bound at the highest application rate, HQs slightly exceed the LOC only at the upper bound (1.3) for sensitive species and are substantially lower for tolerant species. All chronic exposure HQs are below the LOC and most are substantially lower.

For less toxic formulations of glyphosate, risks to tolerant species of fish are not evident from non-accidental or chronic exposures. In the event of an accidental spill into a small body of water, adverse effects are plausible, especially to sensitive species of fish. Adverse effects would appear to be more likely in stressed populations of fish and less likely in otherwise healthy populations.

Since a surfactant must be added to less toxic formulations, it is plausible that the surfactant could impact the toxicity of the formulations to fish. Some surfactants are virtually nontoxic while others are similar to POEA in toxicity. The risk characterization for less toxic glyphosate formulations using more toxic surfactants would be similar to that for more toxic formulations of glyphosate. The additive toxic effect of any surfactant can be computed using custom worksheets.

Amphibians – SERA 2011b, p. 205, "*The available data on terrestrial-phase amphibians do not lend themselves to the types of dose-response assessments conducted for mammals and birds. Based on the approach used by U.S. EPA/OPPTS (2004), risks to terrestrial-phase amphibians would be characterized as the same as risks to birds.*"

For more toxic formulations of glyphosate, HQs for accidental acute exposures (from spills into small bodies of water) exceed the LOC for aquatic-phase amphibians even at the central bound at the lowest application rate of 0.29 lb. a.e./L. At the upper bound at the highest application rate the HQ for sensitive species of amphibians is 3,596 and for tolerant species it is 55. For non-accidental acute exposures at the upper bound at the highest application rate, HQs are much lower; 17 for sensitive species and 0.3 for tolerant species. Except for an upper bound HQ of 1.2 for sensitive species of amphibians, all chronic exposure HQs are substantially lower than the LOC.

At the highest application rate of 8 lb a.e./acre, the upper bound concentration of glyphosate in water is about 0.7 mg a.e./L, close to the lowest acute LC_{50} of 0.8 mg a.e./L. Mortality, perhaps substantial mortality, would be expected in sensitive species of aquatic-phase amphibians. In a toxicity study of two Roundup Original formulations, concentrations of 0.6 and 1.8 mg a.e./L were associated with decreases in growth and survival and development of intersex gonads over a 42-day exposure period. Developmental effects were not noted for glyphosate IPA and appear to be most clearly associated with the surfactants used in Roundup Original formulations rather than glyphosate itself. Several studies clearly indicate that the acute toxicity of glyphosate IPA to amphibians is very low.

For less toxic formulations of glyphosate, HQs for accidental acute exposures to aquatic-phase amphibians are all below the LOC. For non-accidental acute exposures at the upper bound at the highest application rate, sensitive species have an HQ of 1.6, the only HQ that exceeds the LOC. All chronic exposure HQs are substantially below the LOC. There is no basis for asserting that adverse effects in aquatic-phase amphibians would be apparent, even at the upper bound estimates of exposure at the highest application rate.

At the typical application rate, concerns for amphibians would be modest, and the likelihood of substantial or detectable effects appears to be low. However, as stated in SERA 2011b, p. 214, "*As application rates increase toward the maximum labeled rate of 8 lb a.e./acre, the likelihood of observing adverse effects increases. At the maximum application rate, the upper bounds of potential exposure levels suggest that mortality and/or developmental effects would be expected. Thus, if more toxic formulations of glyphosate are applied at high rates near surface water that serves as a habitat for amphibians, efforts may be warranted to refine the exposure assessment based on site-specific considerations and to minimize the likelihood of the contamination of surface water.*"

There is no information for amphibians regarding the toxicity of surfactants that may be used with the less toxic glyphosate formulations. The use of a relatively nontoxic surfactant would probably have no impact on the risk characterization, but a toxic surfactant could dominate it. Assuming a fixed concentration of a toxic surfactant in a field tank mix, low application volumes relative to high volumes will generally reduce adverse effects.

Aquatic Invertebrates – For more toxic formulations of glyphosate, HQs for accidental acute exposures (from spills into small bodies of water) exceed the LOC for sensitive aquatic invertebrates at the central (18) and upper (70) bounds at the lowest application rate of 0.29 lb. a.e./L. At the upper bound at the highest application rate the HQ for sensitive species of invertebrates is 1,918 and for tolerant species it is 63. For non-accidental acute exposures at the upper bound at the highest application rate, HQs are much lower; 9 for sensitive species and 0.3 for tolerant species. Except for an upper bound HQ of 1.2 for sensitive species of invertebrates, all chronic exposure HQs are substantially lower than the LOC. At the highest application rate of 8 lb a.e./acre, "*some studies suggest that mortality at about one-half of the EC_{50} would be quite modest and might be undetectable. This risk characterization is supported by several field studies in which very little impact was observed on aquatic invertebrates following applications of Roundup or other similar formulations.*"

For less toxic formulations of glyphosate, HQs for accidental acute exposures barely exceed the LOC for sensitive aquatic invertebrates at the upper (2) bound at the lowest application rate of 0.29 lb. a.e./L. At the upper bound at the highest application rate, the HQ for sensitive species of invertebrates is 53 and below the LOC for tolerant species. All non-accidental acute and chronic exposure HQs are substantially below the LOC. The risks associated with the less toxic formulations of glyphosate are minimal.

Aquatic Plants (Algae and Macrophytes) – The dose-response assessment for sensitive species of aquatic macrophytes is based on that for sensitive species of algae, so the risk characterizations for sensitive species (but not tolerant species) of aquatic plants are identical for both algae and macrophytes. For more toxic formulations of glyphosate, HQs for accidental acute exposures exceed the LOC for sensitive aquatic macrophytes at the lower (1.3) bound at the lowest application rate of 0.29 lb. a.e./L. At the upper bound at the highest application rate the HQ for sensitive species of macrophytes is 1,754 and for tolerant species it is 0.8 (below the LOC). For non-accidental acute exposures at the upper bound at the highest application rate, HQs are much lower; 8 for sensitive species and substantially below the LOC for tolerant species. All chronic exposure HQs are substantially below the LOC.

For less toxic formulations of glyphosate, HQs for acute accidental, acute non-accidental, and chronic exposures are the same as for the more toxic formulations, so the risk characterization is similar.

For less toxic formulations of glyphosate, HQs for accidental acute exposures exceed the LOC for tolerant algae at the central (6) bound at the lowest application rate. At the upper bound at the highest application rate the HQ for sensitive species of algae is 625 and for tolerant species it is 2. For non-accidental acute exposures at the upper bound at the highest application rate, HQs are much lower; 3 for sensitive species and substantially below the LOC for tolerant species. All chronic exposure HQs are substantially below the LOC.

Following an accidental spill, sensitive species of aquatic plants would likely be damaged or killed, but tolerant species of algae are unlikely to be killed. After non-accidental acute exposures, only at the upper bound at the highest application rate would it be plausible that sensitive, but not tolerant, aquatic plants could be damaged. Adverse effects from chronic exposures are implausible.

Several field studies found that the more toxic formulations of glyphosate, applied at up to the typical rate of 2 lb a.e./acre, did not have a substantial impact on what are presumed to be tolerant algae. Other field studies using sub-toxic concentrations of glyphosate found increases in the primary productivity of algae.

1.3.2.5.2.4 Hexazinone (Sources: FS WSM v. 6.00.10; SERA 2005; U.S. EPA 2002b and 2010d)

Terrestrial and Aquatic Organism Overview

Hexazinone is a broad-spectrum herbicide that is formulated to control annual and perennial herbaceous broadleaf weeds, some grasses, and some woody species. It is commonly used in tree plantations to control brush, in rangeland, and in pasturelands. (TNC 2001)

As stated in U.S. EPA. 2010d (p. 2): "*Hexazinone is a triazine herbicide, which is structurally and toxicologically dissimilar to the other triazines herbicides, such as atrazine. The selectivity of triazine herbicides depends on the plant's ability to degrade or metabolize the parent compound. Sensitive plants have limited ability to metabolize hexazinone. Hexazinone acts through inhibition of photosynthesis.*"

Per The Nature Conservancy (TNC 2001), "*Hexazinone is absorbed through the roots and foliage of plants, and best results are obtained for herbaceous species when applied in moist soil conditions, as either a foliage spray or basal soil treatment. Larger woody species are best controlled by injection or hack-and-squirt techniques. Species that have been controlled by hexazinone include: tansy-mustard (Descurainia pinnata), cheatgrass (Bromus tectorum), filaree (Erodium spp.), shepards-purse (Capsella bursa-pastoris), false dandelion (Hypochaeris radicata), privet (Ligustrum spp.), and Chinese tallowtree (Sapium sebiferum) (Du Pont 1993).*"

Hexazinone is registered for pre-emergent, post-emergence, directed spray and soil applications. Chemical end-use products are formulated as a liquid, soluble granules, water dispersible granules, and pellets. Products are applied by aerial, broadcast and directed spray, or injection. There are no reported impurities of toxicological concern in hexazinone. (U.S. EPA 2002b, p. 5)

The ground application rates of liquid and granular formulations of hexazinone considered in this risk assessment and potentially used under the VTP and alternatives are as follow: the lowest anticipated application rate of 0.5 lb. a.i./acre, the typical application rate of 2 lbs. a.i./acre, and the highest anticipated application rate of 4 lbs. a.i./acre.

Adverse effects on terrestrial plants due to either drift or runoff are plausible from applications of granular or liquid formulations of hexazinone at rates that are effective in weed control. Depending on local conditions and the proximity of streams or ponds to the treatment site, damage to aquatic vegetation is also plausible and could be substantial.

The potential for adverse effects in animals is somewhat dependent on the hexazinone formulation. Granular formulations appear to pose a very low risk to any terrestrial or aquatic animal. Liquid formulation applications will result in much higher concentrations of hexazinone in terrestrial vegetation than will comparable applications of granular formulations. For mammals, this has a major impact on the potential for adverse effects.

As stated in the “Overview” in SERA 2005, p. 4-25: “Over the range of application rates used in U.S. Forest Service programs [and potentially used under the VTP and alternatives], adverse effects are plausible in mammals consuming contaminated vegetation after the application of liquid formulations and adverse reproductive effects in some mammalian species could occur. There is no indication that substantial numbers of mammals would be subject to lethal exposure to hexazinone. Consequently, adverse effects such as weight loss and reproductive impairment could occur but might not be readily apparent or easy to detect. Birds appear to be much more tolerant to hexazinone than mammals and adverse effects on birds do not seem plausible. Similarly, there is no indication that direct toxic effects are likely in aquatic animals.”

Terrestrial Organisms

Mammals – Based on contaminated vegetation, there are large differences between the LOCs (HQs) for granular and liquid formulations for all exposure scenarios. These differences are attributable to the much higher estimates of hexazinone residue on contaminated vegetation following application of liquid formulations relative to granular formulations.

For granular formulations, directed or broadcast soil applications exceed the LOC only for chronic exposures, to small mammals consuming broadleaf foliage (HQ 3) and tall grass (HQ 2) at the typical application rate (upper bound) and short grass at the upper (HQ 5) bounds

and larger mammals consuming short grass at the upper bound (HQ 1.1). Since all HQs are <8, it is plausible that minor adverse effects could occur, especially to small mammals consuming vegetation applied at the typical (upper bound) and highest rates.

For liquid formulations of hexazinone, the non-accidental acute exposure HQs exceed the LOC at the typical application rate (upper bound) only for small mammals consuming contaminated vegetation (HQs 1.9 & 3). Chronic exposure HQs (1.3 to 116) are exceeded for almost all the scenarios involving small, larger, and large mammals consuming contaminated fruit and vegetation at the typical application rate (central and upper bound). At the lower bound, HQs (1.3 & 3) are only exceeded for small mammals consuming contaminated vegetation. As stated in SERA 2005, p. 4-27, "*Over the range of application rates used in Forest Service programs* [and likely to be used under the VTP and alternatives]*, adverse effects could be anticipated in mammals who consume contaminated vegetation over prolonged periods of time. It is unclear whether or not frank effects such as severe weight loss might occur or be evident. Adverse reproductive effects do not appear to be plausible.*"

Birds – Birds appear to be substantially more tolerant of both liquid and granular formulations of hexazinone than do mammals. At none of the application rates, even at the upper limit of exposure, is the LOC exceeded. There is no basis for asserting that any adverse effects are plausible in birds. As stated in SERA 2005, p. 4-27, "*This unambiguous risk characterization is consistent with the risk characterization for birds given by the U.S. EPA/OPP (1994a) in the registration document for hexazinone.*"

Terrestrial Invertebrates – The only available information on the toxicity of hexazinone to terrestrial invertebrates are two bioassays in the honey bee, which severely limits the risk characterization. Based on this, there is no basis for asserting that terrestrial insects or other terrestrial invertebrates will be directly affected by the use of hexazinone in the VTP and alternatives.

Terrestrial Plants (Macrophytes) – As stated in the SERA RA for hexazinone (SERA 2005, p. 4-25): "*Because hexazinone is an effective herbicide, unintended effects on nontarget vegetation are plausible. The effective use of hexazinone is achieved by applying the compound to target vegetation at a time and in a manner which will minimize effects on nontarget plant species. If this is done properly and with care, effects on nontarget vegetation should be minor and perhaps negligible. Nonetheless, in the normal course of applications of granular or liquid formulations at rates that are effective in weed control, adverse effects on terrestrial plants are plausible due to either drift or runoff.*"

There are few quantitative differences in the risk characterizations associated with the application of granular and liquid formulations of hexazinone. Both sensitive and tolerant plants, including special status species, could be adversely affected by off-site drift of hexazinone, sediment loss, or runoff under different scenarios, depending on local site-

specific conditions that cannot be generically modeled. Direct spray of liquid formulations by low boom ground applications is likely to damage both tolerant and sensitive plant species by off-site spray drift at distances of up to about 300 feet at the highest application rate and up to about 25 feet at the lowest application rate. Patterns of drift will vary depending upon whether granular or liquid formulations are applied.

Relatively conservative estimates of pesticide transport by wind erosion of soil suggest that this process is not likely to result in exposures that would be of concern. Off-site transport of hexazinone by runoff and sediment losses could cause substantial damage to both sensitive and tolerant plants across the range of application rates under conditions that favor runoff and sediment loss, such as high rainfall rates and clay soil. As soil textures change from clay to loam to sand, off-site runoff will become increasingly less. If hexazinone is applied in the proximity of sensitive crops or other desirable sensitive plant species, site-specific conditions and anticipated weather patterns will need to be considered if unintended damage is to be avoided.

Terrestrial and Aquatic Microorganisms – The most useful toxicity study for risk characterization found no effects on mixed fungal and bacterial populations after field application at rates of up to 7 lbs/acre, a rate that is substantially higher than potentially used under the VTP and alternatives.

Aquatic Organisms

It appears that aquatic animals are at a very low risk of direct toxic effects from granular formulations of hexazinone (such as Pronone 10G) but at more risk from liquid formulations (such as Velpar L), which are more likely to travel to aquatic environments. However, there is a much greater risk of direct toxic effects of hexazinone to aquatic vegetation, particularly following an accidental spill into a small water body. This risk may be heightened by the use of liquid formulations of hexazinone (such as Velpar L), which are more likely to travel to aquatic environments, than for granular formulations (such as Pronone 10G).

Fish – HQs did not exceed the LOC for fish for any exposure scenarios. There is no indication that hexazinone will cause direct toxic effects in fish even at the highest anticipated application rate of 4 lbs/acre.

Amphibians – The only relevant information that is available on the toxicity of hexazinone to amphibians is that a concentration of 100 mg/L in water caused transient reduced avoidance in newly hatched tadpoles. The highest estimated concentration in water after an accidental spill of the liquid formulation of hexazinone is about 36 mg/L, which might have a short-term effect on avoidance behavior. Whether or not this would result in any substantial impact on amphibian populations is unclear.

Aquatic Invertebrates – HQs did not exceed the LOC for aquatic invertebrates for any exposure scenario, although no toxicity data is indicated for sensitive species. However, a reproduction study in Daphnia magna resulted in a NOEC of 10 mg/L. As stated in SERA 2005, p. 4-31, "*Based on a conservative analysis of a reasonably complete set of standard toxicity studies, there is little basis for asserting that direct toxic effects on aquatic invertebrates are plausible.*"

Aquatic Plants (Algae and Macrophytes) – Adverse effects on aquatic plants are virtually certain unless effective measures at taken to ensure that bodies of open water are not contaminated. For accidental exposures, HQs range from 605 to 3,024 for tolerant macrophytes, from 48 to 242 for tolerant algae, and from 1,814 to 9,072 for sensitive algae. HQs for sensitive macrophytes were not calculated due to a lack of toxicity data.

For non-accidental exposures at the typical rate of exposure, HQs are 17 (central bound) and 67 (upper bound) for tolerant macrophytes, 1.3 (central bound) and 5 (upper bound) for tolerant algae, and 50 (central bound) and 200 (upper bound) for sensitive algae. HQs for sensitive macrophytes were not calculated due to a lack of toxicity data.

For chronic exposures at the typical rate of exposure, HQs are 3 (central bound) and 12 (upper bound) for tolerant macrophytes and 10 (central bound) and 35 (upper bound) for sensitive algae. HQs for sensitive macrophytes were not calculated due to a lack of toxicity data and HQs for tolerant algae are below the LOC.

*1.3.2.5.2.5*Imazapyr (Sources: FS WSM v. 6.00.10; SERA 2011c; U.S. EPA 2006d)

Terrestrial and Aquatic Organism Overview

Imazapyr is a broad-spectrum herbicide that is formulated to control: "*... terrestrial annual and perennial grasses and broadleaved herbs, woody species, and riparian and emergent aquatic species. It can be used where total vegetation control is desired or in spot applications. Imazapyr is relatively slow acting, does not readily break down in the plant, and is therefore particularly good at killing large woody species.*" (TNC 2001)

Imazapyr has been used to control saltcedar (*Tamarix ramossissima*), blackberries (*Rubus* spp.), field bindweed (*Convolvulus arvensis*), tree-of-heaven (*Ailanthus altissima*), pampasgrass (*Cortaderia selloana*), and downy brome (*Bromus tectorum*). But it can also adversely affect non-target plants. The Nature Conservancy (TNC 2001) has identified potential routes of transport of imazapyr that may cause adverse effects to non-target plants, as follows:

> *"Caution should be used when applying imazapyr, as a few reports to TNC from the field indicate that imazapyr might be exuded from the roots of target species. Some legume species, such as mesquite, may actively exude imazapyr (J. Vollmer pers.*

comm.). Imazapyr herbicide can be mobile within roots and transferred between intertwined root systems (root grafts) of many different plants and/or to several species. Movement of imazapyr via root grafts or by exudates (which is a defense mechanism of those plants) may therefore adversely affect the surrounding vegetation. This movement of herbicide may also be compounded when imazapyr is incorrectly over-applied. Movement of soil particles that contain imazapyr can also potentially cause unintended damage to desirable species."

"Imazapyr is effective for creating openings for wildlife use. It can be applied pre-emergent, but is most effective when applied as a post-emergent herbicide. Care should be taken in applying it around non-target species, as it is readily adsorbed through foliage and roots, and therefore, could be injurious by drift, runoff, or leaching from the roots of treated plants. To avoid injury to desirable trees, do not apply imazapyr within twice the drip line (tree canopy.)" (ibid)

As stated in SERA 2011c, p 87: "Imazapyr has been subject to a standard and relatively extensive series of acute, subacute, and chronic studies in mammals. There is little doubt that imazapyr is practically non-toxic (the classification assigned by the U.S. EPA/OPP) to mammals, birds, honeybees, fish, and aquatic invertebrates. None of the expected (non-accidental) exposures to these groups of animals raise substantial concern. The major uncertainties regarding toxic effects in animals are associated with the lack of toxicity data on either reptiles or amphibians.

Imazapyr is an effective herbicide for the control of both terrestrial and aquatic vegetation, so under some conditions ground application could damage non-target terrestrial and aquatic macrophytes. However, it is not an effective algaecide, so no adverse effects would be expected following ground applications.

The directed and broadcast foliar ground application rates of imazapyr considered in this risk assessment and potentially used under the VTP and alternatives are as follow: the lowest anticipated application rate of 0.125 lb. a.e./acre, the typical application rate of 0.3 lb. a.e./acre, and the highest anticipated application rate of 1.5 lbs. a.e./acre.

Terrestrial Organisms

Mammals – The only HQ (1.4) that exceeds a LOC is the non-accidental acute exposure of a small mammal consuming grass at the upper bound at the maximum application rate. This is an extreme worst-case scenario, as it assumes that a small mammal will consume nothing but contaminated grass following a direct spray. Most small mammals have a more diverse diet. For all the other exposure scenarios, HQs are substantially below the LOC for mammals. Thus, adverse effects from exposure to imazapyr are unlikely.

Birds – The only HQs that exceed a LOC are for the chronic exposure of a small bird consuming tall (HQ 1.1) and short (HQ 2) grass at the upper bound at the maximum application rate. This is an extreme worst-case scenario. For almost all the other exposure scenarios, HQs are substantially below the LOC for both small and large birds. As toxic exposure levels of imazapyr are not defined for birds, the HQs probably overestimate risk. Thus, adverse effects to birds from exposure to imazapyr are unlikely.

Terrestrial Invertebrates – The upper bounds of the highest HQs for terrestrial invertebrates are below the LOC. These HQs are for invertebrates consuming contaminated short grass, which is expected to have substantially higher imazapyr residue concentrations than in tall grass, broadleaf vegetation, or fruit. As toxicity data on terrestrial invertebrates is limited to standard acute bioassays in honeybees, the potential risk of adverse effects in terrestrial invertebrates exposed to imazapyr is not characterized. However, due to the low HQs for imazapyr, concern with adverse effects is essentially negligible.

Terrestrial Plants (Macrophytes) – The U.S. EPA RED (U.S. EPA 2006d, p. 18): " ... *has determined that there are ecological risks of concern associated with the use of imazapyr for non-target terrestrial plants and aquatic vascular plants, and potential risks to endangered species (aquatic vascular plants, terrestrial and semi-aquatic monocots and dicots).*"

As stated in SERA 2011c, p. 87: "*The exposure scenarios developed for terrestrial plants result in an extremely wide range of HQs, some of which are far below the LOC and others substantially above it. This apparent ambiguity relates to the attempt made in the exposure assessments to encompass a wide range of potential exposures associated with different weather patterns and other regional or site-specific variables. Thus, for applications of imazapyr to areas in which potential effects on non-target plants are a substantial concern, refinements to the exposure scenarios for non-target plants should be considered based on site or region specific factors.*"

Direct spraying of sensitive plants at the typical application rate of 0.3 lb. a.e./acre, the lowest anticipated application rate of 0.125 lb. a.e./acre, and the highest anticipated application rate of 1.5 lbs. a.e./acre will cause total mortality. At the typical application rate (0.3 lbs a.e./acre) used in U.S. Forest Service programs and potentially used under the VTP and alternatives, the HQ for tolerant plant species would be at the LOC, so damage to tolerant or very resistant species would probably not occur.

Off-site drift of imazapyr is likely to cause adverse effects on some species of non-target plants under certain application conditions and circumstances. Off-site drift from ground applications may cause damage to sensitive species at distances that could extend well beyond 900 feet, unless effective efforts are made to reduce drift from the application site. Tolerant species would probably show relatively little damage even close to treatment sites.

However, there is substantial uncertainty regarding drift estimates due to the numerous site-specific variables which can affect drift. The estimates for backpack applications are based on a modified set of assumptions for low-boom ground applications, so are likely to overestimate drift associated with carefully conducted applications during field conditions that do not favor drift.

The situational variability in the exposure assessments for runoff, irrigation water, and wind erosion has a substantial impact on the characterization of risk for sensitive non-target plant species. These scenarios may overestimate or underestimate risk under certain conditions.

For tolerant species of plants, HQs for exposure from runoff are 0.5 at the central bound and 22 at the upper bound at the highest application rate of 1.5 lbs. a.e./acre. The corresponding HQs for sensitive species are 49 (central) and 2,003 (upper). Since estimates of off-site transport in runoff and sediment are only crude approximations, the upper bound HQs represent estimates of exposure levels which may not be applicable to many site-specific applications potentially made under the VTP and alternatives.

As stated in SERA 2011c, p. 91: "*Appendix 7, Table A7-1 should be consulted in any consideration of the consequences of potential risks to sensitive species of nontarget vegetation in a site-specific application. In areas with predominantly sandy soils, the runoff of imazapyr following foliar applications should be negligible and risks to nontarget plants should also be negligible. Conversely, risks will be greatest in areas with predominantly clay soils and moderate to high rates of rainfall. Risks may also be relatively high in cool locations with predominantly loam soils. Further generalizations do not appear to be warranted, because the modeling conducted for the current risk assessment is inherently conservative and a number of site-specific conditions could reduce, and perhaps substantially reduce, estimates of risks to nontarget vegetation.*"

Since the EPA requires language on all product labels restricting the use of imazapyr-contaminated water for irrigation, consideration of risks associated with this scenario reflects a misuse rather than an expected event. For tolerant species of plants, HQs for exposure due to contaminated irrigation water are substantially below a LOC at the highest application rate of 1.5 lbs. a.e./acre. The corresponding HQs for sensitive species are 106 (central) and 2,761 (upper). Considering reasonable variations that might be made in the exposure scenario, there is little basis for asserting that tolerant plant species will be at risk of adverse effects. However, risks to sensitive species appear to be substantial.

For wind erosion, the HQs for tolerant species of plants are substantially below a LOC while the HQs for sensitive species of plants modestly exceed a LOC at the central (1.6) and upper (3.2) bounds at the highest application rate of 1.5 lbs. a.e./acre. While potential damage to non-target vegetation due to wind erosion of contaminated soil cannot be totally dismissed, the risks associated with this scenario are far below those for runoff or irrigation water.

Terrestrial and Aquatic Microorganisms – The peak concentrations of imazapyr expected in the top 12 inches of soil range from 0.218 to 0.46 mg a.e./kg soil, far below the range of LC_{50} values that caused adverse effects to microorganisms in several studies. As stated in SERA 2011c, p. 93, "*Thus, there does not appear to be any basis for asserting that imazapyr is likely to affect soil microorganisms adversely. This conclusion appears to be consistent with the use of imazapyr as an effective herbicide. If imazapyr were extremely toxic to terrestrial microorganisms that are important for the maintenance of soil suitable for plant growth, it seems reasonable to assume that secondary signs of injury to microbial populations would have been reported.*"

Aquatic Organisms

The U.S. EPA RED (U.S. EPA 2006d, p. 18 has determined that there are no risks of concern to aquatic invertebrates and fish: "For aquatic organisms, available acute and chronic toxicity data indicate that imazapyr acid and salt are practically non-toxic to fish, invertebrates, and non-vascular aquatic plants."

The only ecological risks of concern to the U.S. EPA were: "... associated with the use of imazapyr for non-target terrestrial plants and aquatic vascular plants, and potential risks to endangered species (aquatic vascular plants, terrestrial and semi-aquatic monocots and dicots). (ibid) However, "Registered uses of imazapyr acid and the imazapyr isopropylamine salt will have no direct effect on endangered or threatened fish, aquatic invertebrates, non-vascular aquatic plants (algae), birds or mammals." (U.S. EPA 2006d (p. 23) As per the annual Pesticide Use Reports (CDPR 2010), only imazapyr isopropylamine salt was used in forestry and rangeland applications in California during the years 2000-2010. This is the imazapyr formulation that is assessed in this PEIR.

Although there is little concern for the risk of adverse effects to most aquatic organisms, risk characterization to amphibians is limited, as per SERA 2011e (p. 87), which states that: "There is little doubt that imazapyr is practically non-toxic (the classification assigned by the U.S. EPA/OPP) to mammals, birds, honeybees, fish, and aquatic invertebrates. None of the expected (non-accidental) exposures to these groups of animals raise substantial concern. The major uncertainties regarding toxic effects in animals are associated with the lack of toxicity data on either reptiles or amphibians. While the available studies on other groups of organisms fail to suggest hazards associated with exposure to imazapyr, confidence in extending this risk characterization to reptiles and amphibians is limited."

Fish – The only HQ (3) that exceeds a LOC is the accidental acute exposure of a sensitive species of fish at the upper bound at the maximum application rate. This HQ is based on a single acute NOAEC (10.4 mg a.e./L from a trout bioassay) for the Arsenal formulation, rather than on technical grade imazapyr. In chronic studies, experimental NOAECs are adjusted downward by a factor of 10 to account for Arsenal's greater toxicity to fish relative to imazapyr

acid. As stated in SERA 2011c, p. 93, “Given the very low acute and chronic HQs in fish and the conservative assumptions used to derive these HQs, there is no basis for asserting that acute or longer-term exposure to imazapyr will cause toxic effects in fish.”

Amphibians – No toxicity data is available for either terrestrial or aquatic phase amphibians (or reptiles), so a reasonably definitive risk characterization cannot be developed. Based on the risk characterization for birds and fish, and all other groups of terrestrial and aquatic animals for which data are available, there is no basis for assuming that amphibians are likely to be at risk from exposures to imazapyr.

Aquatic Invertebrates – There are no exposure scenarios in which the HQ exceeds a LOC for tolerant aquatic invertebrates. For most scenarios HQs are substantially below the LOC. No scenarios were developed for sensitive species, as none of the 33 species on which data are available were so identified. The acute NOAEC for invertebrates is higher than that for fish (41 vs. 10.4 mg a.e./L) and the chronic NOAECs for tolerant species are identical (12 mg a.e./L). Potentially sensitive species would need to be 100 to 250 times more sensitive to imazapyr relative to tolerant species before the HQs would be high enough to suggest concern.

Aquatic Plants (Algae and Macrophytes) – The risk characterization for algae is similar to that for fish and aquatic invertebrates, as the acute NOECs for sensitive species of algae are only moderately below the acute NOAECs for sensitive species of fish (i.e., 7.6 mg a.e./L vs. 10.4 mg a.e./L) and the acute NOECs for tolerant species of algae are only moderately higher than the acute NOAECs for tolerant species of aquatic invertebrates (i.e., 50.9 mg a.e./L vs. 41 mg a.e./L). An HQ (4) is exceeded only for sensitive species in the accidental acute exposure scenario at the upper bound at the highest application rate. Most other HQs are substantially below a LOC. Imazapyr is not an effective algaecide. No adverse effects would be expected following terrestrial applications. However, in the event of a severe, accidental spill, populations of sensitive species of algae would probably be reduced.

Imazapyr is labeled for control of aquatic macrophytes, as it is highly toxic to them. The HQs for sensitive aquatic macrophytes following an accidental spill are 9 at the lower bound, 227 at the central bound, and 1,817 at the upper bound at the typical application rate. For tolerant macrophytes the lower bound is below the LOC and the HQ at the central bound is 7 and at the upper bound is 55. These HQs are substantially higher at the highest rate of application.

As stated in SERA 2011c, p. 93, “*In the event of an accidental spill, adverse effects are virtually certain in both sensitive and tolerant species of aquatic macrophytes. In the event of a severe or even a typical spill, extensive mortality would occur. In the event of a small spill, mortality would be expected in sensitive species of macrophytes. Tolerant species could also be adversely affected in areas close to the spill site.*”

For non-accidental acute exposures, the HQs for sensitive macrophytes are 2 at the central bound and 26 at the upper bound at the typical application rate and five times higher at the highest application rate. HQs for tolerant species are below the LOC, except for an HQ of 4 at the upper bound at the highest application rate.

For chronic exposures, the HQs for sensitive macrophytes are 0.7 at the central bound and 12 at the upper bound at the typical application rate, and five times higher at the highest application rate. HQs for tolerant species are below the LOC, except for an HQ of 1.8 at the upper bound at the highest application rate. In areas where the potential for water contamination is lower due to low rainfall rates, damage to aquatic macrophytes is unlikely, while in areas with moderate to high rainfall long term damage could occur to sensitive species.

1.3.2.5.2.6 NP9E (Sources: FS WS ver. 2.02; USDA/FS 2003b; U.S. EPA 2010e)

Terrestrial and Aquatic Organism Overview

Per US EPA 2010e, p. 7: "*Ecological receptors have the potential for significant exposure to NP and NPE for two reasons: 1) facilities that manufacture products containing NP or NPEs are discharging them into surface waters (Ellis et al., 1982); and 2) NP and NPEs tend to partition to sediments and accumulate (Naylor et al., 1992). Both freshwater and saltwater invertebrates, plants and fish are sensitive to this category of chemicals and have demonstrated toxicity to it in varying degrees.*"

However, it appears that there is little risk to terrestrial wildlife from the surfactant NP9E, as per USDA/FS 2003b (p. vi): "*Based on the expected chronic exposure levels, there is little risk to terrestrial wildlife at any application rate considered in this risk assessment.*" It also appears that normal applications of NP9E will not adversely affect aquatic plants, as stated in USDA/FS 2003b (p. vi): "*For aquatic plants, similar conclusions are reached; the normal applications should not represent a risk of effects, either through acute or chronic exposures, while the spill or over spray scenarios do represent a risk of effects.*"

The directed and broadcast foliar ground application rates of NP9E considered in this risk assessment and potentially used under the VTP and alternatives are as follow: the lowest anticipated application rate of 0.167 lb. a.i./acre, the typical application rate of 1.67 lbs. a.i./acre, and the highest anticipated application rate of 6.68 lbs. a.i./acre.

Terrestrial Organisms

It appears that there is little risk to terrestrial wildlife from the surfactant NP9E, as per USDA/FS 2003b (p. vi): "*Based on the expected chronic exposure levels, there is little risk to terrestrial wildlife at any application rate considered in this risk assessment. With the typical application rates, two scenarios represent a slight risk of effects to mammals: direct spray to*

a small mammal (assuming the skin affords no protection) and consumption of contaminated vegetation by a large grazing mammal, such as a deer. None of the other acute exposures at the typical rates of application represent a risk of effects to terrestrial wildlife. At the highest application rates, acute exposures from the consumption of contaminated vegetation present a risk of effects, assuming 100% of consumed vegetation is contaminated. If we assume the skin is not a barrier at all (100% absorption), then the direct spray also provides a risk of effects at the highest application rates."

Terrestrial Organisms – As stated in USDA/FS 2003b, p. 53: "Based on the Hazard Quotients in Table 4-2, several of the scenarios represent potential risk to terrestrial wildlife. With the typical application rates, two of the acute scenarios result in hazard quotients that exceed unity (direct spray with 100% absorption [HQ 16 at the upper bound] and consumption of contaminated vegetation by a large animal [HQ 32 at the upper bound]). As stated in Section 3.3.3, acute doses from 10 to 40 mg/kg/day may not represent a risk to mammals, in which case these typical scenarios may be of low risk, even though the hazard quotient exceeds unity. As stated previously it is also less likely that a large grazing mammal, such as a deer would feed exclusively in a treated area. At the highest application rates, these same two acute exposures scenarios represent a high risk of effects. At exposures above 250 mg/kg/day (an HQ>25) frank toxic effects are possible. At exposures between 100 and 250 mg/kg/day, as stated in section 3.3.3, effects are uncertain in terms of seriousness, with inconsistent results in the various studies. Both scenarios are unlikely, as discussed previously. Given the assumptions, combined with typical animal behaviors, the actual exposure rate for a directly sprayed small mammal is likely somewhere in between the two scenarios of first order absorption and 100% absorption."

USDA/FS 2003b (p. 40) found no data in published literature on NPE toxicity to plants and only limited data on NP. The few studies on NP found that there was little to no plant uptake of NP applied to the soil, uptake was slow, NP was quickly metabolized by soil microorganisms, and/or there was generally a variable biomass growth reduction, from little to none to 50%. It was also stated that: "Since NP9E-based surfactants would not be applied alone, but would be applied in a mix with an herbicide, the herbicide would determine the effects to terrestrial plants."

"Existing soil microbes are able to utilize NPE and NP with little or no lag phase (Environment Canada 2001a; Topp 2000), at application rates (of NP) in the soil of from 1 to 250 mg/kg, indicating a lack of toxicity to soil microorganisms." (ibid)

Aquatic Organisms

Per the U.S. EPA (U.S. EPA 2010e): "NP and NPEs in the freshwater and saltwater ecosystems have the potential for ecological effects on all trophic levels of aquatic species exposed to them (USEPA, 2005)."

Many of the herbicide surfactants analyzed in USDA/FS 2003b (p. v) and likely to be used under the VTP and alternatives, contain from 20-80% NPE. The chemical group of NPEs that are used in herbicide surfactants, NP9E, are of relative low acute toxicity to fish, as are the NPEC metabolites likely to be found in water. NP however, which is another environmental metabolite, is an order of magnitude more toxic to fish than the NP9E or NPECs (USDA/FS 2003b, p. 43). Commercial NPE-based surfactants contain from 20-80% NP9E and are generally mixed with herbicides and water carriers at dilution rates of 0.25% to 2.5% (ibid, p. 1). The percentage of NP9E in a tank mix would therefore range from 0.0005% to 0.02%.

Further, as stated in USDA/FS 2003b:

> *Bioconcentration potential of the short-chain ethoxylates (NP, NP1E, NP2E) in freshwater fish and other aquatic biota appears to be low to moderate ranging up to about 740 (Ahel et al 1993; Liber et al 1999b;Snyder et al 2001; US EPA 1996). Little data exists on the bioconcentration of longer chain NPEs, but based on their structure they are not expected to bioaccumulate (Environment Canada 2001a, Servos 1999). (ibid, p. 45)*
>
> *The duration of an exposure must be considered, which, in the case of aquatic environments in the National Forests, would be short; the compounds of concern are broken down and their concentration reduced through dilution, as well as binding of the compounds to stream sediments. (ibid, p. 53)*
>
> *The ambient levels of NP9E (including a small percentage of NP and NP1-2E) assumed to be present from normal operations (12.5 ppb with a range of 3.1 to 31.2 ppb) would be protective of all aquatic organisms at all application rates. For fish, these assumed levels are at least 30 times lower than the 1,000-ppb protective level for NP9E. For aquatic invertebrates, exposure levels are at least 320 times lower than the 10,000-ppb protective level for NP9E. Given the chronic exposure to NP1-2EC of 7 ppb (0 to 14 ppb range), there should be no chronic toxic risk to aquatic species. (ibid)*
>
> *Both the overspray and the spill scenarios involve levels of NP9E that could represent a risk of toxic effects. The overspray scenario exceeds the acute NP9E threshold for fish by a factor of 1.5 (typical rate), up to a factor of 4.9 (highest rate). The overspray scenario should not represent an acute risk to aquatic invertebrates. With a spill, the NP9E threshold for acute effects to fish is exceeded by a factor of 6.1 (central estimate), up to a factor of 15.1 (highest rate), while for aquatic invertebrates, the threshold for acute effects is exceeded at the highest concentration rate, by a factor of 1.5 (Refer to Worksheet D05). Aquatic plants would have values intermediate between fish and invertebrates. In a stagnant small pond or stream reach, there could be effects seen to aquatic organisms. In a live stream, the more realistic scenario would be a short-term pulse of concentrated NP9E moving downstream, mixing with water and*

being broken down into NP1-2EC and/or partitioning into sediments. The effects of a short pulse should be minor on aquatic organisms as the short exposure time would result in lower doses than are discussed here. (ibid, p. 54)

It appears that normal applications of NP9E will not adversely affect aquatic plants, as stated in USDA/FS 2003b (p. vi): "For aquatic plants, similar conclusions are reached; the normal applications should not represent a risk of effects, either through acute or chronic exposures, while the spill or over spray scenarios do represent a risk of effects."

The risks of adverse effects to aquatic organisms from the use of NP9E surfactants is slight, given that typically there is only a minor amount of surfactant in a tank mix, waterbodies will be buffered, any chemical mix that gets into moving water or waterbodies should dilute rapidly and exposure should be of short duration, and only terrestrial ground applications of chemical mixes will be made.

Fish –For fish, the assumed ambient levels of NP9E in water are at least 30 times lower than the 1,000-ppb protective level for NP9E. There should be no chronic toxic risk to aquatic species, as the chronic exposure level to NP1-2EC is 7 ppb (0 to 14 ppb range). There is also little potential for increased vitellogenin levels in fish at both acute and chronic exposure levels.

Both the overspray and the spill scenarios involve levels of NP9E that could represent a risk of toxic effects. The overspray scenario exceeds the acute NP9E threshold for fish by a factor of 1.5 at the typical application rate and up to a factor of 4.9 at the highest application rate. After an accidental spill into a small water body, the NP9E threshold for acute effects to fish is exceeded by a factor of 6.1 at the central estimate up to a factor of 15.1 at the upper estimate.

Amphibians – Limited data on aquatic amphibians suggests NP9E is equally or less toxic to aquatic amphibians compared to fish.

Aquatic Invertebrates – For aquatic invertebrates, exposure levels to NP9E are at least 320 times lower than the 10,000-ppb protective level for NP9E. The overspray scenario should not represent an acute risk to aquatic invertebrates. After an accidental spill into a small water body, the NP9E threshold for acute effects to aquatic invertebrates is exceeded by a factor of 1.5 at the highest concentration rate.

Aquatic Plants (Algae and Macrophytes) – After an accidental spill into a small water body, aquatic plants would have acute toxic threshold values intermediate between fish and invertebrates. As stated in USDA/FS 2003b, p. 54: "*In a stagnant small pond or stream reach, there could be effects seen to aquatic organisms. In a live stream, the more realistic scenario would be a short-term pulse of concentrated NP9E moving downstream, mixing with water and being broken down into NP1-2EC and/or partitioning into sediments. The effects of a*

short pulse should be minor on aquatic organisms as the short exposure time would result in lower doses than are discussed here."

1.3.2.5.2.7 Sulfometuron methyl (Sources: FS WSM v. 6.00.10; SERA 2004c; U.S. EPA 2008a, 2009g)

Terrestrial and Aquatic Organism Overview

Sulfometuron methyl is a non-selective, sulfonyl urea herbicide formulated to control the growth of broadleaf weeds and grasses. In California, it is used by the USFS primarily to control non-native invasive plants, and to a lesser extent for conifer release from competing vegetation. Oust and Oust XP are the most common formulations used and foliar applications, by backpack or boom spray, are the most common methods employed.

No recent SERA RA report is available for sulfometuron methyl. SERA 2004c (p. 4-29) found no data leading to a conclusion that this herbicide would cause adverse effects in terrestrial animals. The pertinent conclusions from the Risk Characterization "Overview" are as follow: "*There is no clear basis for suggesting that effects on terrestrial animals are likely or would be substantial. Adverse effects in mammals, birds, terrestrial insects, and microorganisms are not likely using typical or worst-case exposure assumptions at the typical application rate of 0.045 lb a.e./acre.*"

The U.S. EPA RED for sulfometuron methyl (U.S. EPA 2008a, p.19) calculated a low Risk Quotient for aquatic and terrestrial animals and determined that "direct exposure of sulfometuron is not of concern for non-plant species." The U.S. EPA RED Amendment (U.S. EPA 2009g, p. 6) states that: "*When considering options to mitigate the ecological risk, the Agency also considered the benefits of sulfometuron methyl, namely its efficacy at extremely low rates and, its low ecological toxicity profile to other non-target organisms.*"

The directed and broadcast foliar ground application rates of sulfometuron methyl considered in this risk assessment and potentially used under the VTP and alternatives are as follow: the lowest anticipated application rate of 0.03 lb. a.i./acre, the typical application rate of 0.045 lb. a.i./acre, and the highest anticipated application rate of 0.38 lbs. a.i./acre.

Terrestrial Organisms

Mammals – There are no HQs that exceed the LOC for accidental acute exposures to mammals. For non-accidental acute exposures, HQs exceed the LOC only at the upper range of the highest application rate and only for small mammals consuming tall and short grass (HQs 3) and broadleaf foliage (HQ 1.7). Adverse effects are unlikely even at the highest application rate that might be used under the VTP and alternatives.

For chronic exposures, the HQs for small mammals consuming vegetation are all ≤2 at the upper bound at the typical rate of application and the central bound at the highest rate. At the

upper bound at the highest rate of application, for all scenarios, the HQs for all mammals (small, larger, and large) range from 1.3 for a large animal consuming tall grass to 21 for a small mammal consuming short grass. These are very conservative/extreme screening scenarios that assume that animals stay in treated areas consuming nothing but contaminated vegetation, which is unlikely given that most vegetation would die or be damaged. Adverse effects are unlikely, even at the highest application rate.

Birds – There are no HQs that exceed the LOC for accidental or non-accidental acute exposures to birds. For chronic exposures, the only HQs exceeding the LOC are for small birds consuming tall grass (1.1) and short grass (2) at the upper bound at the highest application rate. HQs for all other scenarios are substantially below the LOC. Adverse effects are unlikely even at the highest application rate that might be used under the VTP and alternatives.

Terrestrial Invertebrates – Based on direct spray studies in honey bees, no mortality would be expected following acute exposure of doses up to 1075 mg/kg. For honey bees, the HQs are well below the LOC at all rates of application of sulfometuron methyl. There is no basis for anticipating the occurrence of adverse effects in bees, and perhaps other terrestrial invertebrates, at application rates that might be used under the VTP and alternatives.

Terrestrial Plants (Macrophytes) – Per SERA 2004c, the toxicity of sulfometuron methyl to terrestrial plants has been studied extensively and is well characterized: "*Results of both pre-emergent and post-emergent bioassays show that terrestrial plants are highly susceptible to the effects of sulfometuron methyl.*" (SERA 2004c, p. 4-1)

Concern for the sensitivity of non-target plant species is further increased by field reports of substantial and prolonged damage to crops or ornamentals after the application of sulfometuron methyl in both an arid region, presumably due to the transport of soil contaminated with sulfometuron methyl by wind, and in a region with heavy rainfall, presumably due to the wash-off sulfometuron methyl contaminated soil. Sulfometuron methyl exposure inhibited growth of several soil microorganisms and caused significant growth inhibition in Salmonella typhimurium after exposure periods of less than 3 hours. (ibid)

The U.S. EPA RED for sulfometuron methyl (U.S. EPA 2008a, p. 19) indicates that there is concern for adverse effects on terrestrial plants: "*RQs for direct exposure of sulfometuron to non-target aquatic and terrestrial plants range from 6.7 to >18000. These RQs exceed the LOC and show sulfometuron exposure to non-target aquatic and terrestrial plants to be of concern. Although use of 'typical' application rates would result in RQs of up to one order of magnitude lower than the maximum application rate these RQs would still exceed Agency LOC for terrestrial and aquatic plants. The conclusion of potential risks to aquatic and terrestrial plants from sulfometuron application in non-crop uses is consistent with findings*

from other sulfonylurea herbicide risk assessments and ecological incident reports associated with sulfometuron usage."

An amendment to the 2008 U.S. EPA RED (U.S. EPA 2009g, p. 5) reduced the potential risk to non-target plants for the following reasons: "No new data or comments were submitted that modified the Agency's ecological hazard profile for sulfometuron methyl and, therefore, the revised ecological risk assessment of sulfometuron methyl results from changes that reduced the estimated environmental concentrations and improved the overall risk picture. Overall, potential risk to non-target plants has been reduced due to comments and proposals submitted by stakeholders."

The dominant factor in the risk characterization for terrestrial plants is the potency of sulfometuron methyl relative to the application rate. At the typical application rate of 0.045 lb/acre, sulfometuron methyl is about 700 times higher than the NOEC in the vegetative vigor (direct spray) assay of the most sensitive non-target species (0.000064 lb/acre) and <1 times higher than the NOEC for the most tolerant species in the same assay (0.40 lb./acre).

Sulfometuron methyl is a potent herbicide, so adverse effects on some non-target, terrestrial, monocot and dicot plant species from direct spray are certain. Under unfavorable weather conditions and in areas in which drift is not reduced by foliar interception, off-site drift of sulfometuron methyl during ground broadcast applications may cause damage to sensitive plant species at distances >900 feet from the application site. However, when used in directed foliar applications (backpack spray), offsite drift could be reduced substantially. Tolerant plant species would probably not be impacted by drift and might show relatively little damage.

Runoff could pose a substantial risk to sensitive non-target plant species under conditions in which runoff is favored (clay soil over a very wide range of rainfall rates). Some tolerant plants species could be adversely affected under conditions which favor runoff and in regions with high rainfall.

Off-site losses of sulfometuron methyl due to wind erosion are substantially less than losses associated with runoff from clay or from drift at a distance of 500 feet or more from the application site. Wind erosion of contaminated soil is most plausible in relatively arid environments and if soil surface and topographic conditions favor wind erosion. In such locations wind erosion, could lead to adverse effects in sensitive plant species.

The situational variability in the exposure assessments for runoff and wind erosion has a substantial impact on the characterization of risk for sensitive nontarget plant species. These scenarios may overestimate or underestimate risk under certain conditions. As stated in SERA 2011c, p. 4-31: "The simple verbal interpretation for this quantitative risk characterization is that sensitive and tolerant plant species could be adversely affected by the off-site drift or runoff of sulfometuron methyl under a variety of different scenarios depending on local site-specific conditions. If sulfometuron methyl is applied in the proximity of sensitive

crops or other desirable plant species, site-specific conditions and anticipated weather patterns will need to [be] considered if unintended damage is to be avoided."

Terrestrial and Aquatic Microorganisms – Data regarding the toxicity of soil-incorporated sulfometuron methyl to soil microorganisms is not available. A study found that concentrations of ~73 µg/L in a liquid glucose medium inhibited the growth of soil microorganisms after exposure periods of less than 3 hours. Another study found that following light to heavy rainfalls sulfometuron methyl concentrations in runoff were <2400 µg g/L and in percolate were 100 µg g/L, at applications rates within the range used by the U.S. Forest Service and potentially used under the VTP and alternatives. While the level of sulfometuron methyl in runoff may be substantially greater than levels that might inhibit microbial growth, concentrations in the percolate are more directly relevant to soil bacteria. It is uncertain if the level used in glucose medium is relevant to soil exposure, but if it is, microbial inhibition is likely to occur and could be substantial.

Aquatic Organisms

The U.S. EPA RED for sulfometuron methyl (U.S. EPA 2008a, p.19) calculated a low Risk Quotient for aquatic and terrestrial animals and determined that "direct exposure of sulfometuron is not of concern for non-plant species." However, for aquatic plants:

> *RQs for direct exposure of sulfometuron to non-target aquatic and terrestrial plants range from 6.7 to >18000. These RQs exceed the LOC and show sulfometuron exposure to non-target aquatic and terrestrial plants to be of concern. Although use of 'typical' application rates would result in RQs of up to one order of magnitude lower than the maximum application rate these RQs would still exceed Agency LOC for terrestrial and aquatic plants. The conclusion of potential risks to aquatic and terrestrial plants from sulfometuron application in non-crop uses is consistent with findings from other sulfonylurea herbicide risk assessments and ecological incident reports associated with sulfometuron usage.*

Aquatic macrophytes appear to be at risk of adverse, but transient, effects if sulfometuron methyl is applied at the highest application rate in areas where transport to water containing aquatic macrophytes is likely. Measures should be taken to substantially reduce the anticipated levels of exposure. Algae do not appear to be at risk from non-accidental or longer term exposure to sulfometuron methyl in water, although effects may be evident in sensitive species at the upper bound of the highest application rate. Accidental spills will certainly cause adverse effects in sensitive species and may cause adverse effects in tolerant species of both aquatic macrophytes and algae.

As per SERA 2004c (p. 4-2), "*There are no published or unpublished data regarding the toxicity of sulfometuron methyl to aquatic bacteria or fungi. By analogy to the effects on*

terrestrial bacteria and aquatic algae, it seems plausible that aquatic bacteria and fungi will be sensitive to the effects of sulfometuron methyl."

To reduce the hazard of spray drift to non-target organisms, the 2009 U.S. EPA RED Amendment (U.S. EPA 2009g, p. 10) requires all sulfometuron methyl applications to be made with extremely coarse or coarser nozzles. It also requires product labels to carry the following language regarding aquatic vegetation buffer zones:

> *For broadcast ground applications, do not apply within 50 feet of aquatic vegetation including, but not limited to, lakes, reservoirs, rivers, streams, marshes, ponds, estuaries, and commercial fish ponds, or water used as an irrigation source, or crops.*
>
> *For hand held applications, do not apply within 30 feet of aquatic vegetation including but not limited to, lakes, reservoirs, rivers, streams, marshes, ponds, estuaries, and commercial fish ponds, or water used as an irrigation source, or crops.*

The U.S. EPA RED Amendment (U.S. EPA 2009g, p. 6) states that: "When considering options to mitigate the ecological risk, the Agency also considered the benefits of sulfometuron methyl, namely its efficacy at extremely low rates and, its low ecological toxicity profile to other non-target organisms."

Fish – There are no HQs that exceed the LOC for accidental or non-accidental acute exposures to fish or for chronic exposures. However, chronic exposure data are only available in one species of fish (fathead minnow), so confidence in this risk characterization is reduced by the lack of chronic toxicity studies in potentially sensitive fish. Nevertheless, adverse effects are unlikely even at the highest application rate that might be used under the VTP and alternatives.

Amphibians – Tolerant and sensitive species of amphibians could not be identified due to insufficient data. HQs in non-accidental acute exposure and chronic exposure scenarios are substantially below the LOC. HQs exceed the LOC only for the accidental acute exposure scenario at the upper bound (HQ 2) at the typical application rate and the central (HQ 3) and upper (HQ 18) bounds at the highest application rate.

The endpoints for amphibians are an acute NOEC of 0.38 mg/L and a chronic NOEC of 0.00075 mg/L. Concentrations of sulfometuron methyl in ambient water over prolonged periods of time are estimated to be no greater than 0.0000032 mg/L and peak concentration of sulfometuron methyl associated with runoff or percolation are estimated to be no more than 0.0009 mg/L. Based on available data, sulfometuron methyl appears to have a very low potential to cause any adverse effects in amphibians.

Aquatic Invertebrates – The HQs for aquatic invertebrates are extremely low and the available data are sufficient to assert that no adverse effects are anticipated.

Aquatic Plants (Algae and Macrophytes) –The risk characterization for aquatic macrophytes is based on NOEC values in a single species and a most sensitive and most tolerant species could not be identified due to a lack of data. HQs in accidental acute exposure scenarios substantially exceed the LOC, ranging from 47 at the lower bound at the typical application rate to 32,803 at the upper bound at the highest application rate. HQs for the non-accidental acute exposure scenario exceed the LOC only at the upper bound (HQ 4) at the typical application rate and the central (HQ 1.8) and upper (HQ 36) bounds at the highest application rate. HQs for chronic exposure scenarios are substantially below the LOC. Aquatic macrophytes appear to be at risk of adverse, but transient, effects if sulfometuron methyl is applied in areas where transport to water containing aquatic macrophytes is likely. Measures should be taken to substantially reduce the anticipated levels of exposure.

Algae appear to be much less sensitive to sulfometuron methyl than macrophytes. HQs for sensitive species in accidental acute exposure scenarios substantially exceed the LOC, ranging from 4 at the lower bound at the typical application rate to 2,755 at the upper bound at the highest application rate. HQs for tolerant species range from 4 at the upper bound at the typical application rate to 19 at the upper bound at the highest application rate. HQs for the non-accidental acute exposure scenario exceed the LOC only for the most sensitive species and only at the upper bound (HQ 3) at the highest application rate. Most of the other HQs, as well as all the HQs for chronic exposure scenarios, are substantially below the LOC. Algae do not appear to be at risk from non-accidental or longer term exposure to sulfometuron methyl in water, although effects may be evident in sensitive species at the upper bound of the highest application rate. Accidental spills will certainly cause adverse effects in sensitive species and may cause adverse effects in tolerant species.

1.3.2.5.2.8 Triclopyr (Sources: FS WSM v. 6.00.10; SERA 2011a & d)

Terrestrial and Aquatic Organism Overview

Triclopyr is an auxin-mimic herbicide that is formulated to control broadleaf herbs and woody species. "*It is particularly effective at controlling woody species with cut-stump or basal bark treatments. Susceptible species include the brooms (Cytisus spp., Genista spp., and Spartium spp.), the gorses (Ulex spp.), and fennel (Foeniculum vulgare). Triclopyr ester formulations are especially effective against root- or stem-sprouting species such as buckthorns (Rhamnus spp.), ash (Fraxinus spp.), and black locust (Robinia pseudoacacia), because triclopyr remains persistent in plants until they die.*" (TNC 2001)

"Even though offsite movement of triclopyr acid through surface or sub-surface runoff is a possibility, triclopyr is one of the most commonly used herbicides to control woody species in natural areas. Mr. Bill Neil, who has worked extensively on tamarisk/saltcedar (Tamarix spp.) control, concluded that Pathfinder II®, a triclopyr ester formulation by DowElanco, is the most cost effective herbicide for combating saltcedar. On preserves across the U.S., triclopyr has

provided good control of tree-of-heaven (Ailanthus altissima), salt cedar (Tamarix spp.), glossy buckthorn (Frangula alnus), common buckthorn (Rhamnus cathartica), sweet fennel (Foeniculum vulgare), Brazilian peppertree (Schinus terebinthifolius), and Chinese tallow tree (Sapium sebiferum) …Triclopyr can also be used in forest plantations to control brush without significant impacts to conifers (Kelpsas & White). Spruces (Picea spp.) can tolerate triclopyr, but some species of pine (Pinus spp.), however, can only tolerate triclopyr during the dormant fall and winter months (Jotcham et al., 1989)." (ibid)

The following summary of the risks to organisms from exposure to triclopyr comes from the "Overview" in SERA 2011a, p. 130: "Based on the HQs resulting from extreme value exposure assessments, it appears that large mammals consuming contaminated vegetation are the non-target organisms at greatest risk. The available field studies neither support nor substantially refute concerns for adverse effects in large mammals. The lack of detailed field studies involving longer-term observations in populations of large mammals following applications of triclopyr adds substantial uncertainty to the risk characterization for mammalian wildlife."

> *"Some upper bound HQs exceed the level of concern for exposure scenarios in which smaller mammals or birds consume contaminated vegetation or insects. The magnitude of these HQs, however, is much lower than the magnitude of HQs for large mammals, particularly at the upper bounds. Based on the findings of available field studies, triclopyr is not likely to cause frank adverse effects in small mammals and birds. These observations are not contradicted by the relatively moderate exceedances above the level of concern (HQ 1) in the central estimates of the HQs for small mammals and birds." (ibid)*

Terrestrial applications of triclopyr TEA do not pose substantial risks to aquatic animals across the range of labeled application rates. "*Triclopyr BEE, however, is much more toxic than triclopyr TEA to aquatic animals. At application rates in excess of about 3 lb a.e./acre, peak concentrations of triclopyr BEE in surface water could pose acute risks to sensitive species of fish and aquatic phase amphibians. Similarly, acute risks to sensitive species of aquatic invertebrates could occur if application rates exceed about 1.5 lb a.e./acre. The likelihood of acute risks to aquatic animals depends very much on site-specific conditions. In areas with low rates of rainfall, acute risks to aquatic animals would be negligible, so long as drift to surface water were minimal. In areas with high rates of rainfall, the surface water contamination is more likely. Because triclopyr BEE is not persistent in soil or surface water, longer-term risks to aquatic animals after terrestrial applications of triclopyr BEE appear to be negligible.* (ibid)

> *"Since triclopyr is an effective herbicide, damage to terrestrial vegetation is to be expected in the event of direct spray, substantial drift, and substantial runoff from the application site. Substantial runoff from the treated site would depend on the same site-specific factors that*

determine contamination of surface water. Damage to aquatic plants, particularly macrophytes, may result from terrestrial applications of triclopyr. Triclopyr is an effective aquatic herbicide and damage to sensitive species of aquatic macrophytes following effective aquatic applications is certain." (ibid)

The directed and broadcast foliar ground application rates of triclopyr considered in this risk assessment and potentially used under the VTP and alternatives are as follow: the lowest anticipated application rate of 0.1 lb. a.e./acre, the typical application rate of 1 lb. a.e./acre, and the highest anticipated application rate of 6.6 lbs. a.e./acre.

Terrestrial Organisms

The SERA 2011d (p. 130) risk assessment found that: "*Based on the HQs resulting from extreme value exposure assessments, it appears that large mammals consuming contaminated vegetation are the nontarget organisms at greatest risk.*

This assessment based on HQs is consistent with the recent EPA risk assessment, U.S. EPA/OPP (2009a). The available field studies neither support nor substantially refute concerns for adverse effects in large mammals. The lack of detailed field studies involving longer-term observations in populations of large mammals following applications of triclopyr adds substantial uncertainty to the risk characterization for mammalian wildlife.

Some upper bound HQs exceed the level of concern for exposure scenarios in which smaller mammals or birds consume contaminated vegetation or insects. The magnitude of these HQs, however, is much lower than the magnitude of HQs for large mammals, particularly at the upper bounds. Based on the findings of available field studies, triclopyr is not likely to cause frank adverse effects in small mammals and birds. These observations are not contradicted by the relatively moderate exceedances above the level of concern (HQ=1) in the central estimates of the HQs for small mammals and birds."

The application rates for triclopyr anticipated in the VTP and alternatives will be within the range of those analyzed in the SERA RA for Forest Service programs. It should be noted that the specimen labels for the two triclopyr products most commonly used in California, Garlon 3A and Garlon 4, prescribe application rates for forestry uses of up to 6 lbs a.e./acre/year (2 gallons), a smaller amount than used in the high application rate scenario in the SERA RA. For rangeland use, Garlon 3A and Garlon 4can be applied at rates of up to 2 lbs a.e./acre/growing season (2/3 gal. for 3A, 1/2 gal. for 4), again a smaller amount than used in the highest application rate scenario in the SERA RA.

The U.S. EPA/OPP database of ecological incidents associated with pesticide applications lists 63 reported incidents regarding triclopyr applications, none of which reported adverse effects in mammals. Also, none of the available field studies used in the SERA 2011f RA

associate adverse effects in mammals with the direct toxicity of triclopyr. As stated in that RA (SERA 2011d, p. 133): "*Two general factors may contribute to the apparent discrepancy between the high HQs (as well as the high RQs) and the lack of reported adverse effects in field studies or incident reports. Like the human health risk assessment, the ecological risk assessment uses the extreme value approach. The upper bound HQs represent multiple worst case exposure assumptions that may not occur frequently in the field. Also, the field study by Leslie et al., (1996) suggests that some mammals, such as deer, may avoid treated areas. As discussed in the exposure assessment, the scenarios for the consumption of contaminated vegetation assume that 100% of the diet is contaminated. If larger mammals avoid treated areas, the proportion of the contaminated diet could be much less than 100%. As the proportion of the diet that is contaminated decreases, the consequent HQs will also decrease.*"

Mammals – HQs for triclopyr exceed the LOC for accidental acute exposures in only one scenario, a small mammal (HQ 2) and a canid (HQ 1.2) consuming contaminated fish at the upper bound at the highest rate of application (6.6 lbs. a.e./acre). For non-accidental acute exposures, HQs that exceed the LOC range from 1.1 for small and larger mammals consuming contaminated broadleaf vegetation at the central bound at the highest application rate to 74 for a large mammal consuming contaminated short grass at the upper bound. At the typical rate, HQs range from 1.2 (central bound) to 11 (upper bound) for a large mammal consuming short grass. HQs for chronic exposures are somewhat higher, at the highest application rate ranging from 1.8 (central bound) for small and larger mammals consuming contaminated tall grass to 351 (upper bound) for a large mammal consuming contaminated short grass. Exposure scenarios not involving the consumption of contaminated vegetation, direct spray and the consumption of contaminated water and fish, lead to HQs substantially lower than the LOC.

In all non-accidental and chronic exposure scenarios, except for the consumption of tall grass, the HQs are identical for small (20g) and larger (400 g) mammals, ranging from 1.5 for non-accidental consumption of insects to 49 for chronic consumption of short grass. Large (70 kg) mammals appear to be much more sensitive to triclopyr, as HQs are seven times higher, ranging from 6 for non-accidental consumption of fruit to 351 for chronic consumption of short grass.

The high HQs for mammals consuming contaminated vegetation suggest that triclopyr applications may cause adverse effects in mammalian wildlife populations at application rates typically used in U.S. Forest Service programs and potentially used under the VTP and alternatives. For chronic exposures, HQs of about 4 at the typical application rate and about 26 at the highest application rate could be associated with adverse effects that could range from subclinical changes in blood chemistry to birth defects. As stated in SERA 2011a, p. 132, "This HQ-based risk characterization for mammals is similar to the EPA's RQ-based risk characterization in U.S. EPA/OPP (2009a, Table 5-9, p. 101): *Acute and chronic-dose based*

and chronic dietary-based RQs exceed the Agency's acute and chronic endangered species LOC (0.1 acute and 1.0 chronic) for all foliar application uses of triclopyr (Table 5-9). The recommended mitigated maximum foliar application rate of 9 lbs ae/A would still result in exceedances of the Agency's acute and chronic LOC of 0.1 and 1.0 respectively (Table 5-9). U.S. EPA/OPP 2009a, p. 100."

To predict the actual effects of field applications of triclopyr, the preceding quantitative risk characterization must be tempered by information from actual field applications. In the U.S. EPA/OPP database of ecological incidents associated with pesticide applications, there are a total of 63 reported incidents regarding triclopyr applications. None of these incidents reported adverse effects in mammals. In addition, none of the available field studies associate adverse effects in mammals with the direct toxicity of triclopyr.

As stated in SERA 2011a, p. 133, "*Two general factors may contribute to the apparent discrepancy between the high HQs (as well as the high RQs) and the lack of reported adverse effects in field studies or incident reports. Like the human health risk assessment, the ecological risk assessment uses the extreme value approach. The upper bound HQs represent multiple worst case exposure assumptions that may not occur frequently in the field. Also, the field study by Leslie et al., (1996) suggests that some mammals, such as deer, may avoid treated areas. As discussed in the exposure assessment, the scenarios for the consumption of contaminated vegetation assume that 100% of the diet is contaminated. If larger mammals avoid treated areas, the proportion of the contaminated diet could be much less than 100%. As the proportion of the diet that is contaminated decreases, the consequent HQs will also decrease."*

Risk to mammals exposed to triclopyr at application rates potentially used under the VTP and alternatives is as characterized in SERA 2011a, p. 133: "*Considering all of the above factors, the risk characterization for terrestrial mammals based on the HQ method does not appear to be unreasonable. Based on relatively standard methods used to estimate risks to mammals from well-conducted toxicity studies as well as reasonably well-documented estimates of exposure, it is likely that mammals will be exposed to triclopyr at doses that exceed the level of concern (HQ=1). In extreme cases, adverse effects could be anticipated in some mammals, particularly larger mammals, at application rates as low as 1 lb a.e./acre. These effects, however, might not involve overt signs of toxicity that would be observed in field studies.*"

> *"The chronic HQs for mammals are substantially higher than the acute HQs. This matter suggests that while overt signs of toxicity might not be evident shortly after triclopyr applications, longer-term adverse effects on mammalian populations, possibly involving changes in reproductive rates, could occur. While these effects are not reported or otherwise noted in field studies, it is the case that the available field studies focus on small mammals, and the available literature does not include longer-term studies on populations of larger mammals (carnivores or herbivores)."*

HQs for TCP exceed the LOC only for first-order accidental acute exposures (direct spray) for a small mammal at the central (1.6) and upper (4) bounds at the highest application rate and for 100% absorption at the highest application rate (lower 3, central 6, upper 13) and at the upper (1.9) bound at the typical rate. The only TCP non-accidental and chronic exposure scenarios for mammals that approach or exceed the LOC involve the consumption of contaminated vegetation. For non-accidental acute exposures, HQs that exceed the LOC range from 1.2 for larger mammals consuming contaminated insects at the central bound at the highest application rate to 182 for a small mammal consuming contaminated grass at the upper bound. At the typical rate, HQs range from 1.2 (upper bound) for a canid consuming a small mammal to 28 (upper bound) for a small mammal consuming grass. HQs for chronic exposures are generally lower, at the highest application rate ranging from 1.8 (central bound) for a larger mammal consuming contaminated short grass to 90 (upper bound) for a small mammal consuming short grass. Exposure scenarios not involving the consumption of contaminated vegetation, direct spray and the consumption of contaminated water and fish, lead to HQs substantially lower than the LOC.

Unlike triclopyr, the HQs associated with exposure to TCP are highest for smaller mammals, which reflect the greater food consumption rate per body size for smaller mammals, as well as the use of the same NOAEL for all mammals. For contaminated grasses and fruit, the higher HQs for grasses reflect the higher estimated residue rates in short grass relative to fruit. For chronic exposures, HQs of about 4 at the typical application rate and about 26 at the highest application rate could be associated with adverse effects, which could range from subclinical changes in blood chemistry to birth defects.

Risk to mammals exposed to TCP at application rates potentially used under the VTP and alternatives is as characterized in SERA 2011a, p. 136: "*As discussed in the previous subsection, field studies on forestry applications of triclopyr do not support the assertion that triclopyr applications in the range of about 2 lb a.e./acre will cause detectable adverse effects in populations of small mammals. These field observations are consistent with the above HQs. At the central estimate of the exposure assumptions for an application rate of 2 lb a.e./acre, the HQs would be in the range of about 0.6 to 2. The modest excursion above the level of concern (HQ = 1) would not necessarily result in detectable effects on populations of mammals. The upper bound HQs would mostly likely reflect extreme exposures which might occur only rarely.*"

Birds – Except for differences in the impact of body size on apparent risk, the risk characterization for birds is essentially identical to that for mammals. For birds, there is no clear indication of systematic differences in sensitivity with body size. Smaller birds have somewhat higher HQs than larger birds because they will consume more food per unit body weight.

No HQs for triclopyr exceed the LOC for accidental acute exposures. For non-accidental acute exposures, HQs that exceed the LOC range from 1.1 for a large bird consuming contaminated broadleaf vegetation at the central bound at the highest application rate to 90 for a small bird consuming contaminated short grass at the upper bound. At the typical rate, HQs range from 1.2 (central bound) to 14 (upper bound) for a small bird consuming tall and short grass, respectively. HQs for chronic exposures are somewhat higher, at the highest application rate ranging from 1.4 (central bound) for a large bird consuming contaminated fruit to 200 (upper bound) for a small bird consuming contaminated short grass. Exposure scenarios involving direct spray and the consumption of contaminated water and fish lead to HQs substantially lower than the LOC.

Based on the HQs, adverse effects in birds from exposure to triclopyr could be anticipated. Field studies on birds are not as numerous or as detailed as those involving mammals and neither confirm nor substantially refute concerns based on the HQs.

There is no chronic exposure data available on the toxicity of TCP to birds, so risks associated with chronic exposure to TCP residues cannot be characterized quantitatively. For acute exposures, risks are characterized based on a LOAEL of 116 mg/kg bw rather than a NOAEL. The LOAEL is based only on decreases in body weight gain and food consumption in which no overt signs of toxicity were observed, so the toxicological significance is questionable.

Terrestrial Invertebrates – The quantitative risk characterization for terrestrial invertebrates is limited by the available toxicity data. HQs for the direct spray and the consumption of contaminated vegetation scenarios are based on an indeterminate LD_{50} of >620 mg a.e./kg bw for honeybees. At the highest application rate, the only HQs above the LOC are at the central (1.2) and upper bounds (5.6) for an insect consuming short grass, the upper (2.6) bound for an insect consuming tall grass, and the upper (3.2) bound for an insect consuming broadleaf foliage. All other HQs are substantially below the LOC.

There is little indication that concentrations of triclopyr in soil are likely to adversely affect soil invertebrates. The peak concentrations of triclopyr that are likely to occur in the upper 12 inches of soil following applications of triclopyr are about 1.6 ppm a.e. following an application at the highest rate of 6.6 lbs. a.e./acre. This maximum concentration is about four times lower than the chronic NOAEC for earthworms. Numerous field studies suggest that effects on terrestrial invertebrates are most likely to be associated with changes in habitat and food availability rather than direct toxic effects.

Terrestrial Plants (Macrophytes) – These findings are supported by SERA 2011f (p. 131), which found that: “Since triclopyr is an effective herbicide, damage to terrestrial vegetation is to be expected in the event of direct spray, substantial drift, and substantial runoff from the application site. Substantial runoff from the treated site would depend on the same site-specific factors that determine contamination of surface water.”

The HQs for direct spray of terrestrial plants are the same for triclopyr TEA and BEE, but are higher for broadcast boom applications and sensitive plant species than for backpack applications and tolerant species. The HQs are 3,571 for sensitive species (5 for tolerant species) exposed by broadcast applications and 2,357 (3.3 for tolerant species) for backpack applications at the highest application rate. HQs at the typical application rate of 1 lb. a.e./acre are 357 for sensitive species and 0.5 for tolerant species, for both broadcast and backpack applications. Direct spray of triclopyr at the highest and typical application rates will kill and/or damage sensitive plants, as it is designed to do. It is plausible, but unlikely, that tolerant species of plants would be killed, but they might be damaged at the highest application rate.

Off-site spray drift of triclopyr is likely to kill or damage sensitive species of plants, with the extent of damage depending on the application rate and method and the distance from the application site. Estimates of drift used in this risk assessment are generic. Actual drift from applications in the field could vary substantially from these estimates, based on a number of site-specific conditions.

For broadcast applications of triclopyr TEA at the highest application rate, HQs for sensitive plants at various distances from the application site are as follow: 25'-125, 50'-63, 100 -34, 300'-13, 500'-7, and 900'-4. The only HQ above the LOC for tolerant plants is 5, at 25 feet from the application site. For backpack applications of triclopyr TEA at the highest application rate, HQs for sensitive plants are as follow: 25'-20, 50'-10.2, 100 -5.7, 300'-2.2, 500'-1.4, and 900'-0.7. The only HQ above the LOC for tolerant plants is 3.3, at 25 feet from the application site. HQs at the typical application rate of 1 lb a.e./acre are these HQs divided by 6.6.

For broadcast applications of triclopyr BEE at the highest application rate, HQs for sensitive plants at various distances from the application site are as follow: 25'-83, 50'-42, 100 -22, 300'-8.3, 500'-4.9, and 900'-2.6. There are no HQs above the LOC for tolerant plants. For backpack applications of triclopyr BEE at the highest application rate, HQs for sensitive plants are as follow: 25'-20, 50'-10.2, 100 -5.7, 300'-2.2, 500'-1.4, and 900'-0.7. There are no HQs above the LOC for tolerant plants. HQs at the typical application rate of 1 lb a.e./acre are these HQs divided by 6.6.

Off-site transport of triclopyr through runoff and sediment loss differs between the TEA and BEE formulations. For broadcast applications of triclopyr TEA, HQs for sensitive plants are 10 at the central bound and 39 at the upper bound at the maximum application rate. The corresponding values for backpack applications are 6.4 and 26. All other HQs for sensitive plants are substantially lower than the LOC. For triclopyr BEE, the HQ of 15 for sensitive plants at the upper bound at the maximum application rate is identical for both ground application methods. HQs for sensitive plants at the lower and central bounds and for tolerant plants are substantially lower than the LOC for BEE.

In many locations, runoff and sediment losses will be insubstantial. In other areas, sensitive species of plants could be damaged. If triclopyr is applied at a site that may be conducive to runoff or sediment loss, refined estimates of offsite transport should be considered.

For tolerant plant species contaminated by surface water used for irrigation, the HQs are far below a LOC, for both triclopyr formulations and application methods at the highest application rate. For triclopyr TEA, the HQs for sensitive plant species are above a LOC for broadcast application at the central (2) and upper (388) bounds at the highest application rate and for backpack applications at the central (1.6) and upper (256) bounds. For triclopyr BEE, the HQs for sensitive and tolerant plant species are identical; above a LOC for both broadcast and backpack application at the upper (32) bound at the highest application rate. The generic estimates of exposure on which these HQs are based may not represent all site-specific conditions. Site-specific HQs are influenced greatly by the extent of irrigation and concentrations of triclopyr in surface water.

HQs for the exposure of non-target plants to contaminated soil transported by wind are substantially below the LOC. Soil erosion by wind might pose a risk to sensitive plant species when triclopyr is applied to bare ground, but impacts could vary substantially with site-specific conditions.

Terrestrial and Aquatic Microorganisms – The potential for substantial effects on soil microorganisms appears to be low. As stated in SERA 2011a, p. 139: "*As summarized in Section 4.1.2.6, laboratory bioassays conducted in artificial growth media suggest a very high degree of variability in the response of soil bacteria and fungi to triclopyr with NOAELs of up to 1000 ppm in some species and growth inhibition at concentrations as low as 0.1 ppm in other species. For triclopyr BEE, concentrations of triclopyr in the top 12 to 36 inches of soil range from about 0.04 to 0.1 ppm (Appendix 4, Table A4-2 and A4-4). The corresponding values for triclopyr TEA are essentially identical. If the laboratory bioassays were used to characterize risks to terrestrial microorganisms, transient inhibition in the growth of some bacteria or fungi might be expected. This inhibition could result in a shift in the population structure of microbial soil communities, but substantial impacts on soil, including gross changes in capacity of soil to support vegetation, do not seem plausible. This assessment is consistent with the field experience involving the use of triclopyr to manage vegetation.*"

Aquatic Organisms

The SERA 2011d (p. 130) risk assessment concluded that: "*Neither terrestrial nor aquatic applications of triclopyr TEA pose substantial risks to aquatic animals across the range of labeled application rates. Triclopyr BEE, however, is much more toxic than triclopyr TEA to aquatic animals. At application rates in excess of about 3 lb a.e./acre, peak concentrations of triclopyr BEE in surface water could pose acute risks to sensitive species of fish and aquatic phase amphibians. Similarly, acute risks to sensitive species of aquatic invertebrates could*

occur if application rates exceed about 1.5 lb a.e./acre. The likelihood of acute risks to aquatic animals depends very much on site-specific conditions. In areas with low rates of rainfall, acute risks to aquatic animals would be negligible, so long as drift to surface water were minimal. In areas with high rates of rainfall, the surface water contamination is more likely. Because triclopyr BEE is not persistent in soil or surface water, longer-term risks to aquatic animals after terrestrial applications of triclopyr BEE appear to be negligible."

The application rates for triclopyr anticipated in the VTP and alternatives will be within the range of those analyzed in the SERA RA for Forest Service programs. The specimen labels for the two triclopyr products most commonly used in California, Garlon 3A and Garlon 4, prescribe application rates for forestry uses of up to 6 lbs a.e./acre/year (2 gallons), a smaller amount than used in the high application rate scenario in the SERA RA. For rangeland use, Garlon 3A and Garlon 4 can be applied at rates of up to 2 lbs a.e./acre/growing season (2/3 gal. for 3A, 1/2 gal. for 4), again a smaller amount than used in the high application rate scenario in the SERA RA. However, chemical applications in the VTP and alternatives will only be to terrestrial environments and will buffer waterbodies, so the likelihood of contamination of water will be minimal.

The risk characterization for TCP (an environmental metabolite of triclopyr) is considered quantitatively only for fish, because toxicity data are available only for fish. Except for accidental spills into small bodies of water, TCP is not likely to pose a risk to fish. Longer-term concentrations of TCP are far below the LOC.

Fish – For triclopyr TEA, the only HQ that exceeds the LOC for the accidental acute, non-accidental acute, and chronic exposure scenarios is for sensitive fish at the upper bound at the highest application rate. No risks to fish are identified, based on expected peak or longer-term concentrations of triclopyr acid in surface water.

Triclopyr BEE is much more toxic than triclopyr acid to fish. The HQs exceed the LOC for accidental acute exposures of fish even at the lowest application rate of 0.1 lb. a.e./acre (for sensitive species, 2 at the central bound and 20 at the upper bound and for tolerant species, 2.4 at the upper bound). At the upper bound at the highest application rate, the HQs are 1,331 for sensitive species and 161 for tolerant species. For non-accidental acute exposure, HQs are substantially below a LOC, except for a HQ of 2 for sensitive fish at the upper bound. For chronic exposures, all HQs are substantially below a LOC.

In the unlikely event of a large amount of triclopyr BEE being spilled into a small body of water, adverse effects on fish could be expected and would probably cause substantial fish kills. Because triclopyr BEE will not persist in surface water, no species of fish are likely to be at risk from longer-term exposure.

Terrestrial applications of both formulations of triclopyr will result in the contamination of surface water with TCP. The HQs exceed the LOC for accidental acute exposures of fish even

at the lowest application rate of 0.1 lb. a.e./acre (for sensitive species, 1 at the central bound and 10.1 at the upper bound and for tolerant species, 2.9 at the upper bound). At the upper bound at the highest application rate, the HQs are 673 for sensitive species and 192 for tolerant species. Most HQs for non-accidental acute exposure are substantially below a LOC. All HQs for chronic exposures are substantially below a LOC. Except for accidental spills into small bodies of water, TCP is not likely to pose a risk to fish. Longer-term concentrations of TCP are far below the LOC.

Amphibians – No toxicity data are available for TCP for reptiles or terrestrial phase amphibians. Consequently, risks to these groups of organisms are not characterized for TCP. As stated in SERA 2011a, p. 137: "*In the absence of data, the U.S. EPA/OPP will typically characterize risks to amphibians based on the risk characterization for birds. In the recent EPA risk assessment on the California red-legged frog, U.S. EPA/OPP (2009a, p. 75) uses toxicity studies on birds, identical to those used in the current risk assessment, to derive RQs ranging from 0.01 to about 5, based on acute exposures, and from about 1 to 134, based on chronic exposures.*"

For aquatic-phase amphibians, characterization of risk is essentially identical to that for fish. Triclopyr TEA is much less toxic than triclopyr BEE to amphibians, TEA having no HQs exceeding a LOC. For triclopyr BEE, HQs exceed the LOC for accidental acute exposures of sensitive (but not tolerant) species of fish at the central (1.8) and upper (18.2) bounds at the lowest application rate of 0.1 lb. a.e./acre. At the upper bound at the highest application rate of 6.6 lbs. a.e./acre, the HQs are 1,211 for sensitive species and 29 for tolerant species. For non-accidental acute exposure, HQs are substantially below a LOC, except for a HQ of 2 for sensitive amphibians at the upper bound.

Except for accidental spills into small bodies of water, triclopyr is not likely to pose a risk to aquatic-phase amphibians. There is a lack of adequate chronic exposure data for aquatic-phase amphibians, so a formal quantitative risk characterization is not developed. This is not a major limitation in characterizing long-term risk, as concentrations of triclopyr BEE in surface water are very low.

Aquatic Invertebrates – For aquatic invertebrates, characterization of risk is very similar to that for fish. Triclopyr TEA is much less toxic than triclopyr BEE to aquatic invertebrates, TEA having no HQs exceeding a LOC, except for a HQ of 5 for sensitive species at the upper bound at the highest application rate after an accidental spill into a small water body. For triclopyr BEE, HQs exceed the LOC for accidental acute exposures of sensitive (but not tolerant) aquatic invertebrates at the central (4) and upper (40) bounds at the lowest application rate of 0.1 lb. a.e./acre. At the upper bound at the highest application rate of 6.6 lbs. a.e./acre, the HQs are 2,692 for sensitive species and 34 for tolerant species. For non-accidental acute exposure, HQs are substantially below a LOC, except for a HQ of 4 for sensitive aquatic invertebrates at the upper bound. All HQs for chronic exposures are

substantially below a LOC. Except for accidental spills into small bodies of water, triclopyr is not likely to pose a risk to aquatic invertebrates.

Aquatic Plants (Algae and Macrophytes) – Triclopyr TEA is much less toxic than triclopyr BEE to algae. Triclopyr TEA HQs exceed a LOC for accidental acute exposures of sensitive algae at the upper bound (7.9) at the lowest application rate of 0.1 lb. a.e./acre. At the upper bound at the highest application rate, the HQs are 527 for sensitive species and 30 for tolerant species. For non-accidental acute exposure, HQs are substantially below a LOC, except for a HQ of 7 for sensitive algae at the upper bound. For chronic exposures, all HQs are substantially below a LOC, except for a HQ of 1.7 for sensitive algae at the upper bound. Except for accidental spills into small bodies of water, triclopyr TEA is not likely to pose a risk to aquatic algae.

Triclopyr BEE HQs exceed a LOC for accidental acute exposures of sensitive algae even at the lower bound (16.2) at the lowest application rate of 0.1 lb. a.e./acre. At the upper bound at the highest application rate, the HQs are 86,514 for sensitive species and 121 for tolerant species. For non-accidental acute exposure, most HQs are substantially below a LOC, except for a HQ of 141 for sensitive algae at the upper bound at the highest application rate (21 at the upper bound at the typical rate). Most HQs for chronic exposures, are substantially below a LOC. Accidental spills of triclopyr BEE into small bodies of water will likely kill sensitive species of aquatic algae and might damage tolerant species. Adverse effects are also likely in an area where substantial drift or offsite movement in runoff is likely. This is unlikely in arid regions, but as rainfall rates increase, so does the potential for substantial runoff and subsequent damage to aquatic algae.

For aquatic macrophytes, triclopyr TEA is much more toxic than triclopyr BEE. Triclopyr TEA HQs exceed a LOC for accidental acute exposures only for sensitive aquatic macrophytes at the lower bound (45) at the lowest application rate. At the upper bound at the highest application rate, the HQs are 242,240 for sensitive species and 22 for tolerant species. For non-accidental acute exposure, HQs are substantially below a LOC for tolerant species, but the HQ for sensitive species is 3,168 at the upper bound at the highest application rate. Most HQs for chronic exposures are substantially below a LOC, except for a HQ of 792 for sensitive species at the upper bound.

Triclopyr BEE HQs exceed a LOC for accidental acute exposures for aquatic macrophytes at the lower bound (4.2) for sensitive species and the upper bound (42.3) at the lowest application rate. At the upper bound at the highest application rate, the HQs are 2,817 for sensitive species and 391 for tolerant species. For non-accidental acute exposure, most HQs are substantially below a LOC, except for a HQ of 5 for sensitive species at the upper bound at the highest application rate. Most HQs for chronic exposures are substantially below a LOC.

Risks are characterized in SERA 2011a, p. 142 as follow: *"The HQs for aquatic macrophytes following terrestrial applications of triclopyr BEE are much lower than those for triclopyr TEA. The assessment of likely effects on aquatic macrophytes, however, is one example where the use of toxicity values and exposure estimates for triclopyr BEE to develop HQs is probably not justified. As discussed in Section 3.2.3.4.3, triclopyr BEE will rapidly degrade to triclopyr acid. Consequently, for the risk characterization of aquatic macrophytes, the HQs for triclopyr TEA applications should be applied to the assessment of triclopyr BEE applications, since triclopyr TEA is also rapidly hydrolyzed to triclopyr acid. Thus, for both triclopyr TEA and triclopyr BEE terrestrial applications, risks to aquatic macrophytes are substantial. As with algae, these risks will be much less in arid areas, so long as drift to surface water is avoided. If substantial drift occurs, damage to aquatic macrophytes following applications of either triclopyr TEA or triclopyr BEE could occur."* Depending on site-specific conditions, damage to aquatic macrophytes could be evident over a prolonged period of time. The longer-term HQs for sensitive species of aquatic macrophytes are based on estimates of average concentrations of triclopyr in water over a 1-year period.

1.4 INDIRECT EFFECTS FROM IMPLEMENTING THE VTP AND ALTERNATIVES

1.4.1 ENVIRONMENTAL EFFECTS

1.4.1.1 Wildlife

The indirect effects of herbicide treatments on wildlife are dependent on many factors, including the habitat type, specific subsequent activity design, climate, bioregion, and specific ecological requirements of individual species. Information on responses of wildlife to fuel reduction treatments, including herbicide treatments, is sparse to totally lacking. As a rule, negative effects will be greatest for species dependent on the fuels being removed, while positive effects will be greatest for species that have evolved in fire-dependent and other disturbance-prone ecosystems. Native species found in fire prone areas in California should generally be adapted to vegetation disturbances caused by herbicide treatments.

Some herbicide (but not borax) treatments, such as shrubland conversion to rangeland, are likely to significantly modify wildlife habitat. Others will only modify habitat slightly, such as noxious weed treatments on rangeland and understory shrub treatments following forest thinning. Herbicide treatments to control shrubs will normally increase the amount and diversity of grasses and forbs.

> *While herbaceous weed control results in a significant reduction in wildlife forage and cover species during the first growing season after application, research has shown that this effect is temporary, and many species begin to reappear in the first year. By*

the end of the second growing season, the diversity and quantity of herbaceous plants are comparable to untreated areas. (McNabb 1997)

Indirect effects on wildlife will vary over time and differ depending upon the species. Certain effects might be detrimental for some species, as by a reduction in the supply of preferred food or a degradation of habitat, yet beneficial to others, as by an increase in food or prey availability or an enhancement of habitat. This is especially true for species that have very small, localized populations, such as the endangered Lange's metalmark butterfly that exists only in the 55-acre Antioch Dunes National Wildlife Refuge. However, it is unlikely that the effects on large populations of wildlife of vegetation modification, at the spatial and temporal scale of these treatments, would be substantial.

1.4.1.2 Vegetation

The indirect effects of herbicide treatments on special-status plant species depends upon whether the microsite created is favorable or not to the establishment, spread, growth, and/or viability of a species. Rangeland improvement treatments that remove shrubs will open the ground to full sunlight and the drying effects of increased wind speeds, which will adversely affect shade-adapted plants. Conversely, plants that thrive in hot, dry environments will likely spread, if a local seed source is available. Salvage logging after a large fire, followed by herbicide treatments to control shrubs to enhance establishment and growth of conifers, have in some cases resulted in a proliferation of herbaceous species compared to untreated areas (DiTomaso 1995).

Fuelbreak treatments, especially those that remove most of the native vegetation and disturb the soil, create microsites that are conducive to the introduction, establishment, and spread of noxious weeds. If noxious weeds are growing near such treatments, and especially if they are species that propagate from windblown seeds that establish in open areas, which most do, it can be expected that these species will dominate the treatment areas to the detriment of native species. This is especially true if herbicide maintenance treatments follow within a few years of each other.

Treatments to control or eradicate noxious weeds, to the extent that they are effective, will likely open new microsites for the expansion of adapted special status plants that are already growing in the treatment area, can spread too it, or are seeded in or planted by humans. These plants will then have the benefit of a growing site that has less competition for resources from other plants.

1.4.1.3 Invasive Non-Native Plants

Many of the noxious weeds that are aggressively invasive are adapted to disturbed sites with little or no shade. Conversion of shrub fields to rangeland or even for wildlife habitat improvement will generally be done by mechanical, hand, or prescribed fire or herbivory

treatments. Herbicide treatments following the initial treatments will effectively prevent the regrowth of shrubs and perpetuate the microsite conditions that favor the establishment and spread of most species of noxious weeds.

Herbicide maintenance treatments in shaded fuelbreaks in forest environments are not common, but may become more so if vegetation treatment funding levels decrease. In many locations in California, shaded fuelbreaks are being established along road rights-of-way. Road openings provide abundant sunlight, which enhances the establishment and growth of new plants and the regrowth of sprouting species cut during fuelbreak establishment. To remain effective, these fuelbreaks will need to be maintained, which can be done cheaply and effectively using herbicides applied by backpack sprayers or from vehicles.

However, some studies indicate that repeated herbicide treatments, by controlling some species but not others and by creating favorable seedbeds, create microsites favorable to the invasion of noxious weeds. It is known that road openings are conducive to the spread of windborne seeds of such species as star thistle and pampas grass. Therefore, herbicide treatments of roadside shaded (or unshaded) fuelbreaks could result in invasion, reinvasion, or spread of noxious weeds found in the area.

Herbicide treatments to control or eradicate noxious weeds, to the extent that they are effective, will likely open new microsites for the expansion of adapted native plants, if they are already growing in the treatment area, can spread too it, or are planted or seeded by humans. To the extent that native plants can reoccupy and hold disturbed sites, there will likely be a reduction in the population of noxious weeds.

1.4.1.4 Air Quality

There is growing concern about pesticide pollution in California's air basins, especially in the Sacramento and San Joaquin Valley, Sierra Nevada, and Colorado Desert bioregions. There is evidence that current U.S. EPA and CDPR regulations, which define pesticide drift as the total amount of off-site drift that occurs during and immediately after a pesticide application, is inadequate to prevent 80-95% of the total drift of volatile pesticides (Kegley 2003). Detailed analysis of the California Air Resources Board (ARB) monitoring data shows that:

> *". . for about 45% of total pesticides applied in California, the bulk of off-site pesticide movement occurs as the pesticide volatilizes (evaporates) after application. ARB monitoring data show that concentrations of pesticides in air peak between eight and 24 hours after the start of application, with concentrations declining over several days to several weeks. Data presented in this report make it clear that while controls at the time of application are necessary to reduce application-related spray drift, such measures are not sufficient to control post-application drift of volatile pesticides. To adequately address the full range of adverse effects caused by drift, post-application drift must be regulated as well as drift that occurs during applications." (Kegley 2003)*

It is also thought that spray drift is not adequately controlled by regulatory language on pesticide labels. The U.S. EPA is in the process (since 2000) of making labels more consistent (ibid).

One of the highest priorities of the CDPR is reducing pesticide emissions that contribute to air pollution and health problems. Details of the Environmental Monitoring Branch "Air Protection Program" are available at http://www.cdpr.ca.gov/docs/emon/ehap.htm. As stated on the CDPR website (http://www.cdpr.ca.gov/docs/emon/pubs/tac/tac_prog.htm) (CDPR 2012):

> *"With the enactment of California's Toxic Air Contaminant Act the Legislature created the statutory framework for the evaluation and control of chemicals as toxic air contaminants (TACs). The statute defines TACs as air pollutants that may cause or contribute to increases in serious illness or death, or that may pose a present or potential hazard to human health. Included in the definition are substances listed as Hazardous Air Pollutants (HAPs) under section 7412 of Title 42 of the United States Code. The Department of Pesticide Regulation (DPR) is responsible for the evaluation of pesticides as TACs.*
>
> *In general, the law focuses on the evaluation and control of pesticides in ambient community air. In implementing the law, DPR must: 1) conduct a review of the physical properties, environmental fate and human health effects of the candidate pesticide; 2) determine the levels of human exposure in the environment; and 3) estimate the potential human health risk from those exposures. The law requires DPR to list in regulation those pesticides that meet the criteria to be TACs. DPR must then determine the appropriate degree of control measures for the pesticide. DPR may conduct compliance monitoring to assure that users adhere to the control measures as appropriate."*

As stated on CDPR's website (CDPR 2012), "DPR's TAC Program consists of two phases: risk assessment (evaluation and identification) and risk management (control)."

> *The law requires the preparation of a report: ... for each pesticide evaluated that includes: an assessment of exposure of the public to ambient concentrations of the pesticide; a risk assessment, which includes data on health effects, including potency, mode of action, and other biological factors; an overview of the environmental fate and use of the pesticide; and the results of air monitoring studies conducted in California to measure the levels of the candidate pesticide present in ambient air. The report is reviewed by the Office of Environmental Health Hazard Assessment, the ARB, and is made available for public review. Based on the results of these reviews, the draft report is revised as appropriate. The draft undergoes a rigorous peer review for scientific soundness by the Scientific Review Panel, a panel of experts representing a range of*

scientific disciplines. Based on the results of this comprehensive evaluation, the Director of the DPR determines whether the candidate is a TAC. If the Director determines the pesticide the criteria to be a TAC, DPR declares the pesticide a TAC in regulation, and adds it to the TAC list."

As per the California Code of Regulations Title 3. Food and Agriculture, Division 6. Pesticides and Pest Control Operations, Chapter 4. Environmental Protection, Subchapter 2. Air, Article 1. Toxic Air Contaminants, Section 6890, for a pesticide to be listed as a TAC its concentrations in ambient air must be:

". . greater than the following levels (for the purposes of this Section, a threshold is defined as the dose of a chemical below which no adverse effect occurs):

(a) For pesticides which have thresholds for adverse health effects, this level shall be ten-fold below the air concentration which has been determined by the director to be adequately protective of human health.

(b) For pesticides which do not have thresholds for adverse health effects, this level shall be equivalent to the air concentration which would result in a ten-fold lower risk than that which has been determined by the director to be a negligible risk."

As per a CDPR memorandum (CDPR 2007):

"Pesticide VOCs [volatile organic compounds] can contribute to the formation of ground-level ozone, which when present in high concentrations is harmful to human health and vegetation. The federal Clean Air Act requires each state to submit a state implementation plan (SIP) for achieving and maintaining federal ambient air quality standards, including the ozone standard. In 1994, California's Air Resources Board and CDPR developed a SIP element to track and reduce pesticidal sources of VOCs in five regions that do not meet the 1-hour ozone standard (ozone nonattainment areas): Sacramento Metro, San Joaquin Valley, Southeast Desert, Ventura, and South Coast. On February 21, 2006, the U.S. District Court (Eastern District of California) ordered CDPR to implement regulations by January 1, 2008, to achieve the VOC emission reduction goals."

Herbicides can enter the air and drift as droplets, particles, or vapors to affect offsite, non-target species, including humans. Storrie (2004) describes these three modes of transport:

*"**Droplet drift** is the easiest to control because under good spraying conditions, droplets are carried down by air turbulence and gravity, to collect on plant surfaces. Droplet drift is the most common cause of off-target damage caused by herbicide application. For example, spraying fallows with glyphosate under the wrong conditions often leads to severe damage to establishing crops.*

> ***Particle drift*** *occurs when water and other herbicide carriers evaporate quickly from the droplet leaving tiny particles of concentrated herbicide. This can occur with herbicide formulations other than esters. Instances of this form of drift have damaged susceptible crops up to 30 km [18.6 miles] from the source.*
>
> ***Vapour drift*** *is confined to volatile herbicides such as 2,4-D ester. Vapours may arise directly from the spray or evaporation of herbicide from sprayed surfaces. Use of 2,4-D ester in summer can lead to vapour drift damage of highly susceptible crops such as tomatoes, cotton, sunflowers, soybeans and grapes. This may occur hours after the herbicide has been applied.*
>
> *Vapours and minute particles float in the airstream and are poorly collected on catching surfaces. They may be carried for many kilometres in thermal updraughts before being deposited. Sensitive crops may be up to 10,000 times more sensitive than the crop being sprayed. Even small quantities of drifting herbicide can cause severe damage to highly sensitive plants."*

Herbicides can also move off site when sprayed vegetation is burned, although it is difficult to determine the exact amount due to the presence of large volumes of smoke, which is composed of many toxic compounds from combustion of vegetation.

Droplet and particle drift is largely dependent on droplet size, height above the ground of spray apparatus, herbicide formulation, tank mix, temperature, humidity, and wind velocity. Table D.4-1 shows the lateral distances that various sizes of droplets can drift in a 3 MPH wind and emphasizes why it is critical to manage herbicide spraying to reduce droplet size and drift.

Recommended droplet sizes for adequate herbicide coverage are related to the mode of action of the herbicide. Since pre-emergence herbicides (hexazinone and sulfometuron methyl) that are applied to the soil are generally dispersed by mechanical incorporation or precipitation, coarse droplets (greater than 450 microns) can reduce drift risk while ensuring uniform control. Spray droplet size has the greatest influence on the control effectiveness of post-emergence herbicides (clopyralid, glyphosate, imazapyr, triclopyr, and sometimes sulfometuron methyl). These herbicides are readily translocated within plants and may be applied with droplet sizes of around 350-450 microns. Generally, for herbicides, spray droplet size should be greater than 200 microns. (Colquhoun 2001)

It is not expected that herbicide drift under the VTP and alternatives would be excessive. Only ground spray methods would be used. Most sprays would likely be from low pressure, low volume equipment that produces relatively large droplets that are released close to the target. In addition, wind velocities near the ground tend to be lower than with increasing height. In combination, drift will be much less than that which would occur with aerial spraying.

Table D.4-1

Spray Droplet Size and Potential Drift Distance

Droplet Diameter (microns)	Type Of Droplet	Time Required to Fall 10 Feet	Lateral Distance Droplets Travel While Falling 10 Feet in a 3 MPH Wind
5	fog	66 minutes	3 miles
20	very fine spray	4.2 minutes	1,100 feet
100	fine spray	10 seconds	44 feet
240	medium spray	6 seconds	28 feet
400	coarse spray	2 seconds	8.5 feet
1,000	fine rain	1 second	4.7 feet

From Storrie 2004

Vapor drift is primarily affected by the volatility of the herbicide active ingredient formulation (esters are more volatile than salts or acids), climatic conditions (air temperature, humidity, and wind velocity), and soil conditions (texture and organic matter). Some herbicides are more volatile than others (see Table D.4-2). Ester formulations (i.e., triclopyr BEE) and the Velpar L® formulation of hexazinone are relatively volatile in comparison to the other herbicides analyzed in this PEIR.

A study conducted in Canada demonstrated that 3 to 4 percent of both 2,4-D amine and the highly volatile ester drifted out of the target area as spray droplets. An additional 25 to 30 percent of the ester, however, drifted as vapor in the first 30 minutes after spraying, while no additional movement of the amine was detected (Grover & Yoshida 1972).

In a study published by CDPR (CDPR 2002), monitoring was done off-site to determine the movement of three herbicides away from treatment areas following ground applications of glyphosate, triclopyr, and liquid hexazinone and aerial applications of granular hexazinone during 1997 to 2001. To summarize the results:

> *Glyphosate, triclopyr, and hexazinone were detected off-site following application. Triclopyr residues were detected up to 50-100 ft from the spray area in regions where*

it was co-applied with glyphosate. It is assumed that glyphosate also traveled distances equivalent to that of triclopyr, but remained undetected, likely due to its higher MDL [maximum detectable level]. Hexazinone is also suspected to have been transported off site in rain runoff/snowmelt from a liquid hexazinone treatment site and also transported off-site in dust residue from a granular hexazinone treatment site during aerial application.

Table D.4-2

Emission Potential of VTP Chemicals Used in 2010 in California

Chemical	Emission Potential (EPtog & EProg in %) [1/]			
	Forestland Chemicals		Rangeland Chemicals	
	Range	Most Used Formulations	Range	Most Used Formulations
Borax, Sodium Tetraborate Decahydrate	1.53	1.53	not used '00-'10	not used '00-'10
Clopyralid, Monoethanolamine Salt	2.76	2.76	2.76	2.76
Glyphosate, Diammonium Salt	not used '00-'10	not used '00-'10	only 5 lbs. used '10	only 5 lbs. used '10
Glyphosate, Dimethylamine Salt				
Glyphosate, Isopropylamine Salt	0-5.71	0-1.31	0-39.15	0 & 5.71
Glyphosate, Potassium Salt	4.80	4.80	4.80	4.80
Hexazinone	0-37.6	0.99 & 37.6 [3/]	0-37.6	unknown - used '07 & '08
Imazapyr, Isopropylamine Salt	0.01-0.04	0.01	0.01-0.04	0.01
Sulfometuron-Methyl	1.02-3.70	1.02	not used '00-'10	not used '00-'10
Triclopyr, Butoxyethyl Ester (BEE)	1.89-39.15	31.63 & 39.15	31.33-44.72	31.33 & 31.63
Triclopyr, Triethylamine Salt (TEA)	11.23-11.70	11.70	5.71-11.70	5.71 & 11.70

[1/] EPtog = % of product that contributes to VOC emissions of total organic gases, EProg = % of product that contributes to VOC emissions of reactive organic gases; [2/] Also formerly known as isooctyl ester (U.S. EPA 2005d); [3/] Velpar L® formulation

Soil textures influence the degree of herbicide volatilization from soil surfaces. Most of the herbicides analyzed in this PEIR do not adsorb tightly to soil particles (primarily clay and organic matter). Those that do not adsorb tightly (clopyralid, hexazinone, imazapyr, sulfometuron methyl, and triclopyr TEA) are more likely to volatilize, particularly if they are in a formulation that readily volatilizes. None of the herbicides with a low adsorption potential are more than moderately volatile in the formulations in which they are most commonly used, apart from the Velpar L® formulation of hexazinone, which has a high emission potential.

The length of time a chemical will remain on-site and will thus be able to volatilize will be determined to a large extent by its persistence. Persistence in soil is primarily affected by soil texture, climate, and microbial action. Persistence on plant surfaces is determined primarily by climate and exposure to sunlight. The herbicides with the longest potential persistence in soil (borax, clopyralid, hexazinone, imazapyr, sulfometuron methyl and triclopyr TEA), mostly have a low emission potential, except for the Velpar L® formulation of hexazinone, which has a high emission potential and triclopyr TEA, which has a moderately high emission potential.

Herbicide treatments are sometimes done to “brown vegetation” prior to applying prescribed fire to remove the dead vegetation (usually six months to a year following the treatment). Prescribed fire could volatilize herbicide residues found in the vegetation. Burning by itself produces toxic compounds that are respiratory irritants and some of which are carcinogens. Although the combustion products of most herbicides have not been examined in detail, it is not likely that they will add significantly to the hazard of burning alone. It is not possible in this PEIR to assess the extent to which the practice of brown and burn would occur or where it would occur on the landscape, as this practice is done on a voluntary basis.

It is possible that in some situations, such as in air quality non-attainment air basins or near residential areas, herbicide treatments will be used instead of prescribed burning as maintenance treatments. To the extent that this is done, additional smoke would be avoided, so air quality will be unaffected. It is not possible in this PEIR to assess the extent to which prescribed burning will be replaced by herbicide treatments or where these treatments would occur on the landscape, as herbicide treatments are done on a voluntary basis.

1.4.1.5 Water Quality

To the extent that herbicide treatments remove vegetation that protects the soil surface from erosion by rainfall, especially on coarse-textured, erosive soils, such as those derived from granitic rocks, water quality could decline, at least temporarily. On the other hand, except for “brown and burn” scenarios, herbicide treatments kill or inhibit vegetation

but do not remove it from the site, as does prescribed burning or mechanical treatments. Mechanical treatments that disturb the soil would likely result in greater surface erosion than herbicide treatments alone. In such cases herbicide treatments, would protect the soil surface, and water quality, more than the aforementioned treatments.

Some of the herbicides likely to be used under the VTP and alternatives have the potential to travel into waterbodies by spray drift, wind erosion of contaminated soil, surface runoff from treated areas, and/or by leaching into groundwater. Water quality impacts from herbicide treatments are addressed in Section 4.14 of this EIR.

1.4.1.6 Recreation

Herbicide treatments under the VTP may occur on public lands. Herbicide treatments on these lands have a greater possibility of affecting the public than those on private lands, where access to the public is by invitation only.

Public perception of the hazards of herbicide treatments are variable and run the gamut, from the belief that they are benign and beneficial to certainty that they are poisoning humans and the wild denizens of the natural world. Vegetation treated with herbicides tends to be highly visible and unsightly as it yellows, withers, and dies. Until treated areas have re-vegetated, the aesthetic sensibilities of many recreational visitors to public lands will likely be offended if treatments are highly visible or of great extent.

The ultimate effect of negative public perception would likely result in, as it has to date, increased public pressure on resource managers, regulators, and legislators to restrict herbicide applications, not only on public lands but also on private lands. Negative public perception could be alleviated by more robust toxicity testing, as stated in an article by Guynn et. al. in the Wildlife Society Bulletin (WSB 2004):

> *Future research efforts should address public concerns about forest herbicide use and contribute to a basis for defining socially acceptable applications. Information on the toxicity of surfactants, nonactive ingredients, and chemical mixtures (tank mixes) and increasing the number of sentinel species, especially amphibians, would address major public concerns.*

1.4.1.7 Geology & Soils

Killing vegetation that is buffering the soil surface from rainfall impact has the potential to increase surface erosion. This is particularly likely when vegetation is removed from coarse-textured, erosive soils, such as those derived from granitic rocks. If such erosion occurs, it is conceivable that it could remove the duff and top soil horizons, where the bulk of the soil organic matter is located. This would likely reduce soil productivity, at least in the short term.

Such a scenario is unlikely, however, as herbicide treatments alone do not remove vegetation. It is more plausible that as vegetation dies and sheds leaves and other plant parts, the organic litter layer that protects the soil surface from rainfall impact and overland water flow would increase in depth. This would have the effect of increasing the depth of the protective layer and as the litter decomposes, increasing the organic matter in the upper soil layer, thus enhancing soil productivity.

There is some concern that herbicides would have an adverse effect on soil productivity by damaging soil microorganisms. All the herbicides analyzed in this PEIR, however, are broken down by microbial action, except for borax, which is an inorganic compound. Studies reported in the SERA RAs indicate that adverse effects from herbicides to soil microorganisms are unlikely for most herbicides, using typical or worst-case exposure assumptions at the typical application rates. Field studies indicate that for most herbicides (especially glyphosate) there may either be no effect or an increase in microorganisms. However, field studies indicate that sulfometuron methyl “inhibited growth of several soil microorganisms and caused significant growth inhibition in *Salmonella typhimurium* after exposure periods of less than 3 hours” (SERA 2004c).

The risk of borax to insects and soil microorganisms was not characterized in SERA 2006i. Although borax is used to control fungi and insects, the atypical method of application of Sporax® (to individual tree stumps) combined with the likelihood that it would only be applied under the CFIP in the VTP and alternatives, makes it unlikely that there would be widespread exposure to insects and non-target microorganisms. Any effects to soil microorganisms, and thus soil productivity, would likely be localized and of limited extent.

The estimates of risk from soil contamination are general rather than site-specific, as the persistence and movement of chemicals in soil are complex and dependent upon variable, site-specific factors, primarily soil texture, organic matter content, microbial activity, and rainfall.

1.5 UNCERTAINTIES AND UNKNOWNS

There are a number of uncertainties and unknowns regarding the risks associated with using the herbicides analyzed in this PEIR. The following summarizes the uncertainties and unknowns, as discussed in more detail in the preceding risk analysis and in Wildlife Society Symposium publications from the 10th annual conference of the Wildlife Society in Burlington, VT. (WSB 2004)

- Some aspects of the toxicity and fate of herbicides, such as the role of some surfactants and other adjuvants, and possible synergistic effects of multiple chemicals applied simultaneously (i.e., tank mixes), remain unknown.

- FIFRA toxicity testing is not entirely adequate. Herbicides are only tested on a small number of sentinel species, generally under controlled conditions, and only on herbicide active ingredients. Testing of impacts to adult amphibians and to reptiles is largely absent. Tests on individual organisms cannot be used to predict how complex ecosystems would react to herbicides.
- Inert ingredients are not necessarily chemically inert and can be toxic themselves, or can potentially affect toxicity of the herbicide when applied.
- No comprehensive studies have evaluated the impacts of tank mixtures of herbicides. The fundamental types of interactions in these mixtures are additive (toxicity of the mixture is equivalent to the sum of the toxicities of the individual components), antagonistic (less than the sum), or synergistic (greater than additive). Synergistic toxicity is problematic in assessing risk and is complicated by the existence of multiple mechanisms by which it can occur. The toxicity of tank mixtures is generally considered to be the same as the most toxic herbicide, which may or may not be an accurate portrayal.
- No comprehensive field studies have evaluated the impacts of multiple herbicide treatments for site preparation and release, or the combined impacts of mechanical treatments followed by herbicide treatments. Effects of herbicides in combination with fire are not well understood.
- Previous research on herbicide effects has suffered from being conducted at small temporal and spatial scales.
- More scientific rigor needs to be incorporated into herbicide-forest biodiversity studies. Only 25% of researchers collected pre-treatment data, only 40% used control plots, only 56% used replication, and only 45% of study results were peer reviewed (WSB 2004, Summary).
- Interagency consultations between the U.S. EPA and the U.S. FWS on the effects of 64 pesticides on the endangered California red-legged frog, including five of the herbicides proposed for use in the VTP (2,4-D, glyphosate, hexazinone, imazapyr, and triclopyr) and one (atrazine) that might be used off-program, need to be completed to determine the effects on this species, as per CBD v. U.S. EPA & U.S. FWS, 2011.
- There is a need for studies on alternatives to herbicides, including prescribed fire, manual and mechanical cutting, mulches, grazing animals, cover cropping, and ground based and spot application systems.
- Herbicides are often perceived by the public to cause harm to the environment, and thus, many public land managers are reluctant to use them. A major problem in managing natural resources in today's sociopolitical environment is that there have been too few integrated comparisons of forest vegetation management alternatives, and too few syntheses of information to provide a scientific basis for decision-making.

Studies in California have shown what appears to be a strong association between upwind pesticide applications (but not with the herbicides analyzed in this PEIR) and amphibian declines downwind. The relationship seems to be consistent across a number of different species representing at least three independent ranges. Given that amphibian populations appear to be declining worldwide, there is an urgent need for additional research on the role of pesticides in this decline. As reported in (Davidson 2004):

> *Several recent studies in the Sierra Nevada (Datta et al., 1998, Sparling et al., 2001) have documented current-use pesticide residues in the non-declining Pacific treefrog (Hyla regilla). This work needs to be extended to current-use pesticide residues in declining species, and with better geographic coverage to allow for an analysis of the relationship between declines and pesticide residues in frogs. In addition, laboratory experiments are needed to assess possible causal mechanisms of pesticide impacts at field-relevant doses. Given the findings here, examination of the impact of cholinesterase-inhibiting pesticides on immune response, hibernation, and other life functions could be especially illuminating.*

During research for this risk assessment, an abundance of information from different sources was evaluated. Some of this information was contradictory, some was from regions with different ecosystems, and some was based on herbicide formulations not approved for use in California. It is recommended that a solid science foundation be established, using organizational frameworks whenever possible, to capture social and ecological concerns and knowledge regarding herbicides specifically and pesticides in general. This would likely result in more light and less heat being generated in planning for and using herbicides in resource management.

1.6 EFFECTS IN RELATION TO VTP GOALS

To the extent herbicide treatments modify the type, quantity, and continuity of existing live fuels and reduce their regrowth in areas previously treated by other methods, wildland fire behavior would be modified, the risk and severity of high intensity fires and associated suppression costs would be reduced, catastrophic loss of life and property from fires would be less, there would be less air pollution and greenhouse gases produced, and adverse impacts to water quality would be lower. These goals would be met under the VTP and alternatives, which all propose 6,000 acres of herbicide treatment per year.

Herbicide treatments may be used to reduce noxious weeds and non-native invasive plants or to increase the quantity or quality of plant species that would improve browse for wildlife and domestic stock. To the extent that herbicide treatments are used for these purposes, forestland and rangeland resources would be enhanced. These goals would

be met by the VTP and alternatives which all propose to treat 6,000 acres per year with herbicides.

Should funds be limited for the various CAL FIRE vegetation treatment programs, herbicide treatments, because they are generally less costly on a per acre basis than other vegetation treatment methods, would enable more acres to be treated than by other treatment methods. This will have the net effect of enhancing the VTP goals on more acres across California.

1.7 SUMMARY OF EFFECTS

1.7.1 HUMAN HEALTH EFFECTS

1.7.1.1 Overview

All chemicals potentially used under the VTP and alternatives have low acute oral, dermal and inhalation toxicity (Categories III - Caution or Category IV) (there is currently no inhalation study for NP9E). All the chemicals have low acute dermal irritation (Category IV), except for boric acid and NP9E. Boric acid (but not borax) is listed as a dermal irritant (Category III – Caution) and NP9E is listed as severely irritating (Category II – Warning). Given the low acute oral, dermal, and inhalation toxicity for most of the proposed chemicals, none are required to be labeled with the word POISON and a skull and crossbones. No chemicals are skin sensitizers, with the exceptions of triclopyr BEE and TEA.

Boron compounds are suspected of being absorbed more rapidly across damaged skin than intact skin. Thus, individuals with large areas of damaged skin should avoid using boron products, such as Sporax®. Undiluted NP9E may lead to skin sensitization, but such exposures are only likely to occur when it is mishandled. Some evidence suggests that dermal damage may also occur when in direct contact with high levels of clopyralid. Adverse effects can be largely avoided if workers use personal protective equipment and industrial hygiene procedures, as required by law.

Based on acute eye irritation studies, the Sporax® form of borax, clopyralid acid, hexazinone, and triclopyr TEA are all listed as primary eye irritants (Category I - Danger) that can cause severe, irreversible eye damage. Depending on the test study, imazapyr varies from a Category I to a Category III classification. NP9E is listed as severely irritating (Category II – Warning). Adverse effects to workers who do not wear eye protection, as legally required, are plausible. Although acute eye irritation is minimal (Category III - Caution) for clopyralid monoethanolamine salt, glyphosate, sulfometuron methyl, and triclopyr BEE, it is also advisable for workers to wear eye protection when handling these chemicals.

The WHO primarily uses only oral and dermal acute toxicity test results to determine classification. The WHO (2009) did not find any chemicals potentially used in the VTP and alternatives to be extremely or highly hazardous (Table D.3-6). Hexazinone, and triclopyr are categorized as moderately hazardous and borax, clopyralid and glyphosate as only slightly hazardous. Imazapyr and sulfometuron methyl were found to be unlikely to present acute hazard in normal use.

The WHO classifications are for the active ingredients only and are not for any specific formulation. The final classification of these chemicals might be different, depending upon their formulation. However, *evidence suggests that overall, whether assessed by the U.S. EPA or the WHO, chemicals potentially used in the VTP and alternatives do not pose a high acute toxicity hazard.*

Per U.S. EPA chemical assessments, reproductive and developmental toxicity symptoms only occurred at chemical dosages that were at or above the threshold of parental toxicity (ATPT) for chemicals potentially used in the VTP and alternatives, with the exception of borax (Table D.3-8). *None of the chemicals potentially used are listed on the California U.S. EPA's Safe Drinking Water and Toxic Enforcement Act of 1986 (Proposition 65) as chemicals known to cause reproductive toxicity* (OEHHA 2011).

Per the U.S. EPA, none of the active ingredients proposed for use in the VTP and alternatives are known carcinogens or mutagens (Table D.3-9). Similarly, *none of the chemicals proposed for use are on the California EPA's Safe Drinking Water and Toxic Enforcement Act of 1986 (Proposition 65) list of chemicals that are known to cause cancer* (Cal EPA 2011). While clopyralid is not thought to be a carcinogen, hexachlorobenzene, a manufacturing contaminant of clopyralid, is a carcinogenic impurity of concern. However, *hexachlorobenzene is found at average concentrations of less than 2.5 ppm in technical grade clopyralid, well below the cancer risk level used by the USDA/FS when assessing carcinogenicity*.

While neurotoxicity and immunotoxicity studies are now required as a part of new data requirements, these tests have not yet been completed for all chemicals proposed for use under this PEIR. Currently, most conclusions regarding neurotoxicity and immunotoxicity of chemicals are usually based on observations from toxicological studies not specific to evaluating the nervous and immune systems (see Table D.3-10). Of chemicals potentially used in the VTP and alternatives, direct effects to the nervous system were only found for boric acid/ borate salts at high dosages. Direct immunotoxicity effects were not observed for any chemicals potentially used in the VTP.

Currently, information regarding endocrine disruption is vague, though per U.S. EPA and USDA/FS risk assessments, glyphosate, hexazinone, imazapyr and sulfometuron methyl are thought to have the potential to cause effects on the endocrine system with exposure,

though it remains unclear if the effects are direct or indirect (see Table D.3-10). Of the chemicals potentially used in the VTP and alternatives, currently only glyphosate are on the U.S. EPA Final List of Initial Pesticide Active Ingredients and Pesticide Inert Ingredients to be Screened (as part of Tier 1) for effects of endocrine disruption (FR 2009, p. 17579).

Of the chemicals potentially used in the VTP and alternatives, only triclopyr produces a metabolite - i.e., 3,5,6-trichloro-2-pyridinol (3,5,6-TCP) – that is toxic beyond the level of concern in some scenarios (see Table D.3-11). Clopyralid contains the impurities hexachlorobenzene and pentachlorobenzene, which are known carcinogens. Hexachlorobenzene is found at average concentrations of less than 2.5 ppm in technical grade clopyralid and pentachlorobenzene is found at average concentrations of less than 0.3 ppm. Hexachlorobenzene is ubiquitous and persistent in the environment and almost all people are exposed to it and have detectable concentrations in their bodies (SERA 2004a, p. 3-23). Some formulations of glyphosate that contain POEA surfactants contain the known carcinogenic contaminant 1,4-dioxane. These three carcinogens, however, are at concentrations well below the cancer risk level used by the USDA/FS when assessing carcinogenicity. Nicotinic acid, which is also known as Vitamin B3, is a metabolite of imazapyr and is a known neurotoxin; however, the minute amount in imazapyr poses no toxicity concern.

Forest Service risk assessments group chemical exposure to workers and members of the public into general exposure from normal use of chemicals and more severe accidental/incidental exposure resulting from misuse or unusual circumstances (SERA 2012). In Forest Service risk assessments, a number of specific scenarios are consistently used to characterize exposure of the general public (ibid and Table D.3-12). The assumptions made for these scenarios often make these scenarios implausible. When the standard scenarios were established for Forest Service public exposure assessments, the events were often designed to be intentionally extreme.

Extreme values, or upper and lower bounds of credible exposure levels, are typically used in Forest Service risk assessments. Consideration is also given to the estimated level of exposure most likely to occur, which is sometimes referred to as the central, or maximum likelihood estimate (ibid). The upper bound for each chemical is usually determined with the intent to encompass exposure of the most exposed individual. Moreover, when the lower bound exposure estimates are higher than the Level of Concern (LOC), this indicates that use of the pesticide will lead to an unacceptable risk (ibid).

In Forest Service risk assessments, the exposure and the dose-response assessments are used to quantitatively characterize risks. Hazard quotients (HQ) are values used to categorize risk for systemic toxicity effects (SERA 2012). All HQ values are directly proportional to the application rate (i.e., an HQ value of 2 at an application rate of 1 lb

a.e./acre would be 6 at an application rate of 3 lb a.e./acre). For acute exposures, HQs are in units of mg/kg bw/event whereas chronic exposures are in units of mg/kg bw/day. The HQ is usually calculated by dividing a projected level of exposure by an acceptable level of exposure, such as an RfD (*ibid*). Generally, an HQ greater than 1 indicates that risk is above the Level of Concern (LOC), or unacceptably high for the situation being considered, and that adverse health outcomes may be plausible. By contrast, an HQ less than or equal to 1 indicates that exposures are below the LOC and adverse effects are not expected. Still, when HQ values are 1 or greater, the plausibility of scenarios and assumptions made for each scenario should be considered before conclusions regarding risk levels are drawn.

It needs to be emphasized that for the risk characterizations that follow, regardless of studies and findings, "***[a]bsolute safety cannot be proven and the absence of risk can never be demonstrated***" (SERA 2012). There are always uncertainties, such as those associated with using data from surrogate mammals to represent human health risk. Thus, individuals should remain prudent and minimize chemical exposure when possible.

1.7.1.2 Chemical-Specific Effects to Workers and the Public

Borax

Workers - Since Sporax® is only applied in a granular form in a specialized way, scenarios inapplicable to general worker exposure, direct spray, oral exposure by ingestion of contaminated vegetation, fruit, or fish, and direct exposure from contaminated vegetation, were omitted from the Forest Service risk assessments. The only scenarios assessed were for exposure to workers from wearing contaminated gloves for 1 minute and for 1 hour, with HQs at the upper bound ranging from 0.00072 to 0.00576 mg/kg bw/event, *well under the LOC*.

Public

Scenarios: 1) direct spray of a child's whole body, 2) direct spray of a woman's feet and lower legs - Given that Sporax® is only applied in a granular form in a specialized way, the scenario involving a child being directly sprayed with a chemical was adapted to a child ingesting borax directly from a freshly treated stump. This scenario had a central HQ of 4.2 and values ranging from 2.1 to 16.2 for an ingestion of 50 to 400 mg of Sporax (5.67 to 45.36 mg B/day). Per the Forest Service risk assessment, such "estimated levels of exposure are below levels of exposure associated with nonlethal effects such as diarrhea and vomiting by factors of about 4 [184÷ 45.36] to 32 [184 ÷ 5.67]". Moreover, "lethal doses are in the range 505 mg B/kg/day and 765 mg B/kg/day, factors of about 11 to 135 below the estimated levels of exposure." *This indicates that if a child consumes borax from a stump, the child would likely experience vomiting and diarrhea as symptoms of toxicity. No other public exposure scenario was above the LOC.*

Scenarios: 1) consumption of contaminated fruit or vegetation by a woman, 2) long-term consumption of contaminated fruit or vegetation by a woman - None of these scenarios are applicable to borax.

Scenarios: 1) water consumption by a child after a spill, 2) consumption of contaminated fish by a man after a spill, 3) consumption of contaminated fish by subsistence populations following a spill, 4) water consumption by a child, 5) consumption of fish by subsistence populations, 6) water consumption by a man over a lifetime - The exposures for the accidental spill scenario are based on 6.25 to 25 pounds of borax spilling into a small pond. At these rates, the HQs for a small child consuming water contaminated by an accidental spill of Sporax® into a small pond range from 0.07 to 0.7, all below the LOC. Since risk is linearly related to the amount of Sporax® that is spilled into a pond, for spills of larger amounts, HQs could exceed the LOC.

For exposure by consumption of water contaminated by runoff, the range of Sporax® application rates considered is 0.1 lb/acre to 5 lbs/acre (0.01 to 0.57 lb B/acre), with a typical rate of 1 lb/acre (0.11 lb B/acre). HQs for acute exposure of a child and chronic exposure of an adult male to water contaminated by runoff are below the LOC for all application rates considered. The highest hazard quotient of 0.3 is associated with the upper bound for acute exposure of a child. Thus, *even at the highest application rate, there does not appear to be a risk associated with acute or chronic exposure to water contaminated by runoff.*

Clopyralid

Workers - At an application rate of 0.25 lb a.e./acre, all the general or incidental exposures to workers lead to HQ values substantially lower than the level of concern (LOC), so *no systemic effects are likely to occur among workers as a result of clopyralid exposure*.

Public

Scenarios: 1) direct spray of a child's whole body, 2) direct spray of a woman's feet and lower legs - At an application rate of 0.25 lb a.e./acre, *none of the short or long-term exposure scenarios approach a LOC based on central estimates.*

Scenarios: 1) consumption of contaminated fruit or vegetation by a woman, 2) long-term consumption of contaminated fruit or vegetation by a woman - At an application rate of 0.25 lb a.e./acre, *none of the short or long-term exposure scenarios approach a LOC based on central estimates*. Only for chronic effects at the upper bound for consumption of vegetation does the HQ (1.2) modestly exceed the LOC.

Scenarios: 1) water consumption by a child after a spill, 2) consumption of contaminated fish by a man after a spill, 3) consumption of contaminated fish by subsistence populations following a spill, 4) water consumption by a child, 5) consumption of fish by subsistence populations, 6) water consumption by a man over a lifetime - Only at the upper bound of the scenario of a child consuming water after a spill does the HQ (1.7) modestly exceed a LOC at the application rate of 0.25 lbs. a.e./acre. *This short-term exposure scenario is of no concern*. All other scenarios are substantially below a LOC.

Glyphosate

Workers - *Based on HQ values, the risk to workers from exposure to glyphosate is minimal.* The highest HQ for worker exposure is the upper bound for general broadcast spraying (HQ of 0.08 at normalized 1 lb a.e./acre). At the highest rate of application of 8 lbs a.e./acre used by the USDA/FS and potentially used under the VTP and alternatives, the highest HQ for occupational exposure is the upper bound associated with workers participating in broadcast foliar application.

Public

Scenarios: 1) direct spray of a child's whole body, 2) direct spray of a woman's feet and lower legs – Even at the upper bound at the highest application rate of 8 lbs. a.e./acre, none of these exposure scenarios leads to HQ values greater than 1, the LOC.

Scenarios: 1) consumption of contaminated fruit or vegetation by a woman, 2) long-term consumption of contaminated fruit or vegetation by a woman - *The only non-accidental exposure of potential concern involves contamination of vegetation shortly after application* (HQ of 0.7 at 1 lb a.e./acre). At the central (2 lb a.e./acre) and maximum (8 lb a.e./acre) application rates, the upper bound HQ values would be 1.35 and 5.4 respectively. Chronic exposure scenarios never resulted in LOCs, even when the maximum application of 8 lbs a.e./acre was used, as 0.9 was the highest HQ, which was for the chronic scenario involving contaminated vegetation. An HQ of 5 may raise concerns regarding adverse health effects to pregnant women and fetotoxicity. Formulas that contain surfactants and are used in South America have been associated with genotoxicity, though it is currently unclear if this finding is applicable to the U.S. formulations.

Scenarios: 1) water consumption by a child after a spill, 2) consumption of contaminated fish by a man after a spill, 3) consumption of contaminated fish by subsistence populations following a spill, 4) water consumption by a child, 5) consumption of fish by subsistence populations, 6) water consumption by a man over a lifetime - The accidental acute exposure involving a child consuming contaminated water after a spill has an HQ of 2.05 at the upper bound at the typical

application rate (HQ 8.2 at highest application rate). This scenario is quite arbitrary and thought to be inconsequential.

Hexazinone

Workers - Regardless of the formulation type, the upper bounds of general occupational exposure exceeded a LOC for broadcast and direct foliar application methods at a typical application rate of 2 lbs/acre (HQ of 6) and at the highest rate of 4 lbs/acre (HQ of 12). *Even at the lowest application rate (0.5 lbs/acre), the upper bound of hexazinone exposure exceeds the LOC (HQ of 1.5 lbs/acre) for broadcast application.* But *the highest upper bound HQ for any accidental exposure scenario was only 0.08, for wearing gloves contaminated with a liquid formulation for one hour. At central bounds, the LOC is exceeded (HQ 1.8) only at the highest application rate* while it only approaches the LOC (HQ 0.9) at the typical application rate. At the lower bounds, regardless of the application rate, HQs never reached a LOC. *The interpretation of these HQ values in the Forest Service risk assessment was that the level of acceptable risk for workers would be unacceptable unless all precautionary handling measures were followed (e.g. personal protection equipment is used) to minimize exposure.*

Public

Scenarios: 1) direct spray of a child's whole body, 2) direct spray of a woman's feet and lower legs – For these accidental acute exposure scenarios, all HQs are substantially lower than a LOC at the upper bound at the highest application rate.

Scenarios: 1) consumption of contaminated fruit or vegetation by a woman, 2) long-term consumption of contaminated fruit or vegetation by a woman - The only non-accidental exposure scenario, long-term consumption of contaminated vegetation, that results in HQs that substantially exceed LOCs are at the highest application rate (4 lbs a.e./acre) of Velpar L (a liquid formulation) at low, central, and upper bounds (HQs of 0.4.1, 6, and 46 respectively). Even at the lowest application rate (0.5 lb a.e./acre), the LOC is exceeded at the upper range of exposure (HQ of 5.75) for broadleaf vegetation. The risk of exposure is much lower for granular formulations of hexazinone, with HQs of 0.2 for fruit and 1.8 for broadleaf vegetation at the upper bound at the highest application rate. *Given that granular application methods result in less residue on plants, particularly on the leaves of broadleaf vegetation and other plant parts that might collect similar levels of residue, this method should be favored over liquid hexazinone applications where public consumption of contaminated vegetation is probable.*

Scenarios: 1) water consumption by a child after a spill, 2) consumption of contaminated fish by a man after a spill, 3) consumption of contaminated fish by subsistence populations following a spill, 4) water consumption by a child, 5) consumption of fish by subsistence populations, 6) water consumption by a man

over a lifetime - The only acute exposure that leads to a HQ above the LOC is the accidental exposure involving consumption of contaminated water by a child after a spill into a small pond, which results in a HQ of 2 at the upper bound of the highest application rate (4 lbs a.e./acre) for Velpar L. However, *this scenario is highly arbitrary and implausible*. For chronic exposures, other than the consumption of contaminated vegetation, the highest HQ is 0.2, the upper range for the consumption of contaminated water at the maximum application rate. *This is below the LOC by a factor of 5*.

Imazapyr

Workers - Risks are characterized only for workers applying imazapyr by ground broadcast methods. The highest HQ for general exposures is 0.02, the upper bound at the typical application rate of 0.30 lbs a.e./acre of the HQ for workers involved in ground broadcast applications of imazapyr. *This is below the LOC by a factor of about 50*. The highest accidental HQ is 0.004, at the upper bound for a worker wearing contaminated gloves for 1 hour. No exposure assessment was done for cut surface or basal bark applications, as adequate worker exposure studies were not available. However, since cut surface applications would require the use of concentrated imazapyr solutions, exposures could reach a LOC in five hours of wearing contaminated gloves. Workers who use highly concentrated solutions of imazapyr should be especially careful to prevent prolonged skin contact with the chemical. Eye irritation is the only clear risk to humans and is most pertinent to workers. Injury to the eye is most likely to occur with occupational mishandling of imazapyr, and thus workers should be prudent to follow personal protection measures, such as wearing goggles. Currently, *no evidence suggests that systemic effects are likely to occur among workers as a result of exposure to imazapyr.*

Public

Scenarios: 1) direct spray of a child's whole body, 2) direct spray of a woman's feet and lower legs – Both scenarios resulted in accidental acute exposure HQs that were substantially below a LOC at the upper bound at the highest application rate.

Scenarios: 1) consumption of contaminated fruit or vegetation by a woman, 2) long-term consumption of contaminated fruit or vegetation by a woman - *The public is not likely to be at risk due to applications of imazapyr.* None of these scenarios resulted in an HQ that exceeded 1, the LOC, when calculated at an application rate of 1 lb a.e./acre. When using the upper bound at the maximum application rate of 1.5 lbs a.e./acre, the non-accidental acute scenario of an adult woman consuming contaminated vegetation resulted in a HQs of 1. Given the lack of adverse effects detected, HQ values that do exceed 1 are difficult to interpret. *Currently, no evidence suggests that systemic effects are likely to occur in the public as a result of imazapyr exposure.*

Scenarios: 1) water consumption by a child after a spill, 2) consumption of contaminated fish by a man after a spill, 3) consumption of contaminated fish by subsistence populations following a spill, 4) water consumption by a child, 5) consumption of fish by subsistence populations, 6) water consumption by a man over a lifetime - The public is not likely to be at risk due to applications of imazapyr. No dose has been identified that might pose a risk to humans. Based on the RfD of 2.5 mg/kg bw/day, the highest HQ is associated with an accidental spill of imazapyr into a small pond and the subsequent consumption of contaminated water by a small child. For this exposure scenario, the HQ is 1 at the upper bound at the highest application rate of 1.5 lbs a.e./acre. The risk assessment suggests that only very severe accidental spills would approach a LOC. HQs for all other scenarios are substantially below a LOC. *Currently, no evidence suggests that systemic effects are likely to occur in the public as a result of imazapyr exposure.*

NP9E

Workers - *No evidence indicates that typical acute and chronic exposures would lead to doses that exceed the LOC for workers, though some of the upper bounds did exceed it.* The upper bounds of general worker exposure resulted in levels above concern, with the LOC being double for broadcast application (HQ of 10.1) than directed (backpack) ground spray (HQ of 5.34). Despite the high LOCs at the upper bounds, there is not a high likelihood that workers will use such high levels of surfactants containing NP9E on a long-term basis. Additionally, workers are expected to use industrial hygiene practices while handling chemicals, which are not accounted for in worker exposures.

Public

Scenarios: 1) direct spray of a child's whole body, 2) direct spray of a woman's feet and lower legs – Neither of these scenarios resulted in HQs that exceeded the LOC.

Scenarios: 1) consumption of contaminated fruit or vegetation by a woman, 2) long-term consumption of contaminated fruit or vegetation by a woman - *No evidence indicates that typical acute and chronic exposures would lead to doses that exceed the LOC for the public, though some of the upper bounds did exceed it.* Chronic exposure leads to levels below concern. The scenario for consumption of contaminated fruit leads to acute or accidental exposures with unacceptable risk, but only the upper bounds were above the LOC (HQ 12). These findings indicate that *oral, rather than dermal, exposures are of the greatest concern for NP9E*, and help determine where the greatest mitigations may be necessary to minimize exposures to the public. Per the USDA/FS risk assessment, there should not be any substantial risk of long-term exposure to NP9E-based surfactants to the public.

Scenarios: 1) water consumption by a child after a spill, 2) consumption of contaminated fish by a man after a spill, 3) consumption of contaminated fish by subsistence populations following a spill, 4) water consumption by a child, 5) consumption of fish by subsistence populations, 6) water consumption by a man over a lifetime - No evidence indicates that typical acute and chronic exposures would lead to doses that exceed the LOC for the general public, though some of the upper bounds did exceed it. Oral rather than dermal exposures are of the greatest concern for NP9E. Chronic exposure leads to levels substantially below the LOC, though some accidental exposure scenarios lead to exposures of concern. At the upper bound, the HQ is 1.7 for consumption of contaminated fish by subsistence populations following a spill. The scenario relating to consumption of water by a child after a spill leads to the highest risk at lower, typical, and upper exposures levels (HQ values of 1.4, 4.6, and 17 respectively), but this scenario is highly arbitrary, which means that LOCs are not indicative of realistic risk to the public. Per the USDA/FS risk assessment, there should not be any substantial risk of long-term exposure to NP9E-based surfactants to the public.

Sulfometuron methyl

Workers - At the typical application rate used by the Forest Service and potentially used under the VTP and alternatives (0.045 lb a.e./acre), none of the upper limit HQ values for workers are at or above LOCs and most are substantially below a LOC. The highest general worker HQ is 0.34 at the typical application rate for broadcast application. At the maximum application rate (0.38 lb a.e./acre) the HQ for broadcast application is 2.9 and for direct foliar application it is 1.5, both of which are above the LOC.

The interpretation in Forest Service RAs is that an unacceptable level of risk could be expected for workers if the maximum application rates are used, the maximum acreage is treated per day, and the workers are not prudent in using sound hygiene practices and personal protection equipment. Given the low likelihood that all these factors would occur, and the conservative provisionary RfDs used by the Forest Service, *it is unlikely that workers would experience observable adverse effects.* The risk of adverse effects would be reduced or eliminated if lower application rates and fewer acres were treated.

Public

Scenarios: 1) direct spray of a child's whole body, 2) direct spray of a woman's feet and lower legs - At the typical application rate used by the Forest Service and potentially used under the VTP and alternatives (0.045 lb a.e./acre), all the upper bound HQ values for these scenarios are substantially below a LOC. For the public, *all acute exposures, both accidental and non-accidental, remained below the levels of concern at the maximum application rate of 0.38 lb a.e./acre. The risk of adverse effects to the public would be reduced or eliminated if lower application rates and fewer acres were treated.*

Scenarios: 1) consumption of contaminated fruit or vegetation by a woman, 2) long-term consumption of contaminated fruit or vegetation by a woman - At the typical application rate, the upper bound HQ values are substantially below a LOC. All acute exposures, both accidental and non-accidental, remained below the levels of concern at the maximum application rate of 0.38 lb a.e./acre. For, chronic exposures, only the upper bound relating to the consumption of contaminated vegetation was above the level of concern, with an HQ of 4.1. The risk of adverse effects to the public would be reduced or eliminated if lower application rates and fewer acres were treated.

Scenarios: 1) water consumption by a child after a spill, 2) consumption of contaminated fish by a man after a spill, 3) consumption of contaminated fish by subsistence populations following a spill, 4) water consumption by a child, 5) consumption of fish by subsistence populations, 6) water consumption by a man over a lifetime - At the typical and highest application rates, none of the upper bound HQ values for these scenarios are at or above LOCs and most are substantially below a LOC. It is unlikely that the public would experience observable adverse effects.

Triclopyr

Workers - *The LOC for occupational exposure is highly dependent on whether the acute or chronic RfD is used.* Based on the acute RfD, at an application rate of 1 lb a.e./acre none of the HQs were substantially above the LOC, but the acute RfD is only appropriate for male workers. Based on the chronic RfDs, HQs are below the LOC for triclopyr TEA. The central estimates for triclopyr BEE range from 0.7 to 1.2 at the typical application rate of 1 lb a.e./acre. All upper bound HQ values were above the LOC for both TEA and BEE forms of triclopyr when based on the chronic RfD for all application methods. In this case, BEE had higher HQ values than the TEA form of triclopyr (TEA 1.6 to 3, BEE 6 to 12). *One of the most likely exposures and risks for workers is from triclopyr being splashed into eyes, as it is moderately to severely damaging.* This as an avoidable hazard, as long as workers wear eye protection while handling triclopyr.

Public

Scenarios: 1) direct spray of a child's whole body, 2) direct spray of a woman's feet and lower legs – The HQ values for these two scenarios vary considerably between triclopyr TEA, BEE, and the metabolite TCP. The HQs for triclopyr TEA are both below a LOC at the upper bound at the typical application rate of 1 lb a.e./acre, but would exceed a LOC (HQs of 1.3 to 3.3) at the upper bound at the maximum application rate of 6.6 lbs a.e./acre. For triclopyr BEE, the HQ (1.4) exceeds the LOC for the direct spray of a woman's feet and lower legs at the upper bound at the typical application rate and exceeds a LOC (HQs of 4.6 to 9.2) at the maximum application rate. For TCP, the HQ for direct spray of a child exceeds a LOC at the central (8) and upper (123) bounds at the

typical application rate and exceeds a LOC for the direct spray of a woman's feet and lower legs at the upper (HQ 12) bound at the typical application rate. These HQs would be 6.6 times higher at the upper bound at the highest application rate.

Because the upper bounds are above the LOC, caution is particularly warranted to avoid accidental spraying of the public. However, these scenarios are highly unlikely and are designed to be indicators of the most serious exposures that could result from accidental spraying of members of the public.

Scenarios: 1) consumption of contaminated fruit or vegetation by a woman, 2) long-term consumption of contaminated fruit or vegetation by a woman - The only triclopyr or TCP exposure scenarios of substantial concern involve the consumption of contaminated vegetation and fruit. These risks do not differ between the TEA and BEE formulations. For acute non-accidental and chronic (chronic values in parentheses) exposures to a young woman consuming contaminated vegetation the HQs at the upper bound at the typical application rate of 1 lb a.e./acre are 27 (4). At the typical application rate, the central bounds for the consumption of contaminated vegetation exceed or reach the LOC for acute exposures to triclopyr (HQ of 0.3) and to TCP (HQ of 6) and for chronic exposures to TCP (HQ of 1.3). Lower bounds of exposures are used as *best case* estimates and are generally intended to represent the feasibility of risk mitigation. The lower bound HQ for the exposure scenario involving a young woman consuming vegetation contaminated with triclopyr is 0.2 at an application rate of 1 lb a.e./acre and would reach a LOC (HQ 1) at an application rate of 5 lbs a.e./acre, and exceed (HQ 1.3) the LOC at the maximum application rate of 6.6 lbs a.e./acre.

Potential exposures to triclopyr TEA, BEE, and TCP also exceed the LOC at the upper bound of the HQs for both the non-accidental acute and longer-term consumption of contaminated fruit. For TEA and BEE, the HQs are 4 for acute and 3 for chronic exposures and for TCP the HQs are 2 for acute and 10 for chronic exposures. These HQs would be 6.6 times higher at the upper bound at the highest application rate. *The upper bound HQs are intentionally based on very conservative exposure assumptions that lead to assessments that may unrealistically magnify risks.*

Scenarios: 1) water consumption by a child after a spill, 2) consumption of contaminated fish by a man after a spill, 3) consumption of contaminated fish by subsistence populations following a spill, 4) water consumption by a child, 5) consumption of fish by subsistence populations, 6) water consumption by a man over a lifetime – The scenarios of greatest concern are for a child consuming contaminated water after a spill. For triclopyr TEA and BEE, the HQ at the upper bound at the typical application rate is 2 and for TCP is 82 (5 at the central bound). The risk assessment suggests that only very severe accidental spills would exceed a LOC and only for the metabolite TCP. However, this scenario is highly arbitrary, which means that

the LOCs are not indicative of realistic risk to the public. For all the other scenarios, the HQs are substantially below a LOC.

1.7.1.3 Chemical-Specific Effects to Sensitive Subgroups, Connected Actions, and Cumulative Effects

Sensitive Subgroups - Potential adverse effects to sensitive subpopulations of humans from chemical treatments are highly dependent on the toxicity of a specific chemical, the exposure to that chemical, the dose and length of time to which an individual is exposed, and the sensitivity of that individual to a specific chemical.

Connected Actions - Connected actions are typically activities other than those associated with the agent of concern that might impact an individual's response to that agent. Potentially significant connected actions associated with the risk assessments done by SERA and the USDA/FS include exposures to other agents that might alter an individual's response to the agent of concern (SERA 2005, p. 3-42). The Food Quality Protection Act requires that chemicals that are mixed with other chemicals that have the same mode of action relating to toxicity be assessed for synergistic, additive, or antagonistic effects.

Cumulative Effects - Cumulative effects refers to the consequences of repeated exposure to the chemicals potentially used in the VTP and alternatives as well as exposures to other chemicals that have the same mode of action as the chemical of concern. As stated in SERA 2005 (p. 3-41), "*It is beyond the scope of the current risk assessment to identify and consider all agents that might have the same mode of action. To do so quantitatively would require a complete set of risk assessments on each of the other agents that would be considered.*"

Borax

Sensitive Subgroups - Developing fetuses are a primary target of boron exposure. Since the RfD is based on the adverse fetal effect of weight loss, the reproduction related subgroups are accounted for throughout the entire Forest Service risk assessment. Testes are also targeted in male mammals and thus, while data is currently lacking, males with underlying testicular dysfunction may be at an increased risk of testicular issues induced by boron exposure.

Connected Actions - Connected actions are not of concern since borax is not mixed with other chemicals.

Cumulative Effects - Multiple exposures are not concerns given that the chronic RfD was used to calculate risk through the entire boron assessment. The concern is also lessened by the fact that boron is ubiquitous in nature. Exposures occur naturally at rates

of 0.14 to 0.36 mg/kg/day and potential application rates under the VTP and alternatives will not substantially contribute to the already existent background levels.

Clopyralid

Sensitive Subgroups - In toxicity studies clopyralid has been implicated in causing decreased body weight, increased kidney and liver weight, deceased red blood cell counts, as well as hyperplasia in gastric epithelial tissue. The likely critical effect in humans cannot be identified and effects are not consistent among test species or even between different studies on the same species. *It is unclear if individuals with pre-existing kidney, liver, or blood diseases would be particularly sensitive to clopyralid exposures. There are no data or case reports on idiosyncratic responses to clopyralid by individuals who suffer from multiple chemical sensitivity.*

Connected Actions - Although clopyralid may be applied in combination with other herbicides, *no data in the literature suggests that it will interact, either synergistically or antagonistically, with them.*

Cumulative Effects - Repeated exposure to levels of clopyralid below the toxic threshold should not be associated with cumulative toxic effects. All longer-term exposures are substantially below the LOC.

Glyphosate

Sensitive Subgroups - Sensitive subgroups include women and fetuses, but these are accounted for since a developmental study was used to establish the NOAEL used for the RfD. While not well understood, MCS may be a potential concern for glyphosate, as with other chemicals.

Connected Actions - The U.S. EPA has not determined if glyphosate shares toxicity mechanisms with other chemicals. Potentially the most important connected action is associated with surfactants. Given that glyphosate functions to inhibit some mixed-function oxidases, this is a plausible mechanism of interaction for other chemicals that function similarly. There has been no evidence of such effects, however, and this is only likely to be a potential when glyphosate is applied at much higher rates than done by the Forest Service or likely under the VTP and alternatives.

Cumulative Effects - The daily dose of glyphosate rather than the duration of exposure determines the toxicological response. Repeated exposure to levels of glyphosate below the toxic threshold should not be associated with cumulative toxic effects. All longer-term exposures are substantially below the LOC.

Hexazinone

Sensitive Subgroups - Hexazinone can induce fetal resorptions and other adverse developmental effects, so pregnant women and developing offspring may be sensitive subgroups particularly vulnerable to adverse effects of hexazinone. This potential has been explicitly accounted for given that the developmental endpoint was used in the risk assessment. The literature does not report any other subgroups that may be sensitive to hexazinone and there is no indication that it causes allergic responses or sensitization.

Connected Actions - There is almost no information available on the interaction of hexazinone with other compounds. There is no indication that the inerts and adjuvants in its formulations will increase the toxicity of hexazinone in humans or mammals. However, it is not unreasonable to suspect hexazinone would interact additively, synergistically or antagonistically with chemicals that share similar metabolic pathways. Such potential connected actions are beyond the scope of the risk assessment in this PEIR and are not evaluated by the Forest Service or the U.S. EPA.

Cumulative Effects - Cumulative effects may result from repeated exposures, multiple routes of exposure (i.e., oral and dermal), or exposures to chemicals that have connected modes of action. Forest Service risk assessments consider the effects of multiple, long-term exposures, evaluating risk in terms of both acute and chronic exposures to workers and the public.

Imazapyr

Sensitive Subgroups - Given the low toxicity of imazapyr, effects on sensitive subpopulations are thought to be minimal. Because imazapyr is a weak acid it would most likely be affected by other weak acids that are similarly excreted by the kidneys, though only at unrealistically high doses that nearly saturate kidneys.

Connected Actions - Given the low toxicity of imazapyr, the occurrence of connected actions is thought to be minimal. Both the low HQ values and conservative assumptions support that impacts of inerts, impurities and metabolites are minimal to imazapyr risk characterization. However, adjuvant interactions are a potential, but were beyond the scope of the USDA/FS risk assessment for imazapyr.

Cumulative Effects - Given the low toxicity of imazapyr to humans, cumulative effects are thought to be minimal. When characterizing risk of chemical use, cumulative effects may result if humans experience multiple exposures to imazapyr via multiple routes and/or events, or if humans are exposed to additional chemicals with the same toxicity mechanisms at the same time as exposure to imazapyr. At present, common mechanisms of toxicity have not been found between imazapyr and any other chemicals (similar or otherwise).

NP9E

Sensitive Subgroups - There are several groups of people that have the potential to be part of sensitive subgroups. There is some indication that some sensitive individuals are prone to develop contact allergies related to NP9E exposures. In addition, there is evidence that NP9E targets the kidneys and liver in mammals, so sensitive subgroups may consist of those individuals that have pre-existing impairment of the liver or kidneys. Per the Forest Service risk assessment, the likelihood of NP9E inducing reproductive effects should be low, though acute exposures may occur at the application rates that are within the range of fetal effects being a potential. Therefore, it is relevant to consider pregnant women an additional potential sensitive subgroup.

Connected Actions - NP9E has not been connected to any antagonistic or synergistic interactions relating to human health effects when mixed with other chemicals. This group of surfactants is not known to increase dermal absorption of herbicides and synergistic effects are not expected with repeated exposures of NP related compounds. Toxicological response appears to be dependent on daily doses rather than the duration of exposures. Additionally, any repeated-exposure effects should have been counted for through use of the chronic RfD. There is the potential for additive estrogenic effects to arise if NP related compounds or chemicals that act via similar estrogen-like (xenoestrogen) pathways cumulatively reach a high enough concentration. NP9E is abundant in a number of non-forestry related sources (e.g. personal care products, industrial and institutional detergents and cleaners, and the environment), and the amount of human exposure to NP9E as a result of forestry use is thought to be negligible.

Cumulative Effects - Repeated exposure to levels below the toxic threshold should not be associated with cumulative effects. However, estrogenic effects can be caused by additive amounts of NP, NPE, and their breakdown products. In other words, an effect could arise from the additive dose of a number of different xenoestrogens and phytoestrogens (hormone mimicking substances naturally present in plants), none of which individually have high enough concentrations to cause effects. Additive doses could come from sources removed from the herbicide application site, such as personal care products, detergents and soaps, foods, paints, and from the environment. Various studies have estimated the daily exposure of humans to NP and NPE from food and the environment. As presented in USDA/FS 2003b (p. 38), In terms of this risk assessment, the contribution of NP9E (workers exposure ranged from 0.000075 to 1.01 mg/kg/day) would contribute from 0.00075 up to 10 to any hazard quotient. This may be negligible depending upon the background exposures, lifestyles, absorption rates, and other potential natural or man-made chemical exposures that are used to determine overall risk to environmental xenoestrogens.

Sulfometuron methyl

Sensitive Subgroups - No adverse effects for sensitive subgroups was identified with evidence in the 2004 risk assessment for sulfometuron methyl conducted for the Forest Service. Given hematology and thyroid effects observed in mammalian studies, it was suggested that individuals with pre-existing anemia or thyroid function issues may be more susceptible to adverse effects.

Connected Actions - Per the Forest Service risk assessment, sulfometuron methyl formulations have not been connected to synergistic or antagonistic effects related to the mixing of sulfometuron methyl with other active ingredients and surfactants.

Cumulative Effects - Cumulative effects are not anticipated given that repeated exposures were explicitly considered through using a chronic RfD to evaluate the level of concern with repeated exposure.

Triclopyr

Sensitive Subgroups - Women of child bearing age are thought to be of concern due to reproductive and developmental effects found in exposure studies using mammals. Despite the lack of epidemiological evidence, there is a certain level of uncertainty, regarding the possibility of triclopyr causing adverse reproductive effects. Current evidence suggests, however, that toxicity to a fetus would only occur at doses that also caused frank signs of maternal toxicity. Despite the years triclopyr has been used, this chemical has never been implicated in causing frank signals of toxicity in male or female humans. Individuals with kidney disease may also be at greater susceptibility to adverse effects, since the kidneys are the target organ for triclopyr.

Connected Actions - Connected actions of triclopyr are associated with exposure to the triclopyr metabolite 3,5,6-trichloro-2-pyridinol (TCP). The Forest Service and U.S. EPA risk assessments consider all exposures to this compound as below the level of concern, although the Agency does not consider all oral exposures assessed in the Forest Service risk assessments. Like many herbicides, adjuvants are commonly used with triclopyr and some may be hazardous.

Cumulative Effects - The cumulative effects associated with triclopyr may include those associated with any additive effects that could potentially result from mixing of triclopyr with other chemicals, as well as effects resulting from repeated exposures. The additive effects associated with mixing adjuvants with triclopyr are beyond the scope of the USDA/FS risk assessments. It should be noted, however, that triclopyr is a weak-acid auxin herbicide, and thus, when mixed with other similar weak acids that function by the same mechanisms, such as clopyralid, additive risks would result. Repeated exposure is a cumulative effect accounted for using chronic exposure information in each Forest Service risk assessment.

1.7.2 ECOLOGICAL EFFECTS

Implementation of chemical treatments could in some cases result in adverse effects to non-target biological resources, particularly under marginally plausible, worst-case scenarios at chemical application rates higher than are likely to be used in the VTP and alternatives. Potential adverse effects are highly dependent on the lifeform, the toxicity of a specific chemical to that lifeform, the exposure of individuals to that chemical, the dose to which individuals are exposed, and the interaction of environmental factors that are not always fully understood.

Herbicides that would potentially be used in the VTP and alternatives would most likely be applied either by backpack or boom spray. It is likely that during application some portion of the herbicides would enter the air and drift off-site. The amount and distance of spray drift is dependent on a number of factors, including droplet size, wind speed, air temperature, humidity, inversion layer, the chemical formulation and tank mix, type of spray equipment and application method, height of spray equipment above the ground, and the area treated. It is also possible that a portion of the herbicides would volatilize from the surfaces on which they land and would adversely affect air quality, although this was not identified as a risk for the herbicides analyzed in this PEIR. The amount of volatilization is dependent primarily on the chemical formulation and tank mix, air temperature, humidity, and wind velocity. *Borax is unlikely to affect air quality* as it is not volatile, would be applied directly to fresh stumps that are moist (the chemical will likely adhere to the stump), will be applied in a manner (“salt shaker”) that would minimize off-site movement of powder in the air, and would be applied in forested areas where wind speeds tend to be minimal.

Because site-specific factors at the subsequent activity level cannot be predicted, the amount of drift and/or volatilization of herbicides and the absolute effect on air quality cannot be quantified. However, an attempt was made to model spray drift in the 2012 Worksheets that accompany the risk assessments for each chemical, for both backpack and broadcast applications. Adverse effects to off-site, non-target plants were specifically related to the chemical and method of application and were by far most likely to occur in sensitive terrestrial plant species. The limits of modeled adverse effects for broadcast sprays varied from 100 feet away from the treatment site for 2,4-D to >900 feet for sulfometuron methyl. Distances for backpack applications were substantially less. Adverse effects to tolerant plant species were rarely shown off-site, and then only within 25 feet of the treatment site. The most appropriate use of this information is to assess the relative toxicity of chemicals and the effect of the application method, as the amount of chemical drift is largely a function of wind speed, spray droplet size, and height of the spray from the vegetation being sprayed.

Chemicals would potentially be used in the VTP and alternatives only to treat terrestrial vegetation, so direct contamination of water resulting from normal use is unlikely. However, it is possible that chemicals would at times be used near Class I or II waterbodies and probable that they would be used near Class III watercourses. Inadvertent contamination of waterbodies or watercourses could occur. Direct spill, drift of spray, or runoff are the most likely routes for levels of chemical contamination of water that might cause adverse effects in aquatic organisms.

Other than for off-site drift of spray, the possibility of chemicals moving off-site into waterbodies is variable and dependent upon chemical properties (persistence, solubility in water, volatility, adsorption potential to soil) and environmental factors (soil texture, rainfall amount and timing, wind speed and topography, depth to water table, distance to waterbodies). As soil texture and rainfall amount and timing are highly variable across and within bioregions, both the primary routes of chemical transport (runoff, leaching, wind drift of soil, volatilization) and the mobility of chemicals would vary. Transport by runoff would be most likely on fine-textured soils (clay) and leaching most likely on coarse textured soils (sand) in bioregions with heavy rainfall events occurring shortly after chemical treatments. Wind erosion and volatilization will be most likely in drier, hotter bioregions with strong winds and topography that channels winds.

Although it is possible that chemical treatments would result in some portion of the herbicides, surfactants, or borax entering waterbodies, dilution and photolysis would generally rapidly minimize the chance of an organism receiving a high enough dose to cause adverse effects. Possible exceptions to this would be in shallow ponds, vernal pools, or narrow, shallow, and/or slow-moving streams, where dilution would either be less or at a slower rate. Sensitive aquatic macrophytes are likely to experience adverse effects, especially from spills of relatively large quantities of chemical. Borax is unlikely to move offsite into water and is nontoxic to humans and practically non-toxic to aquatic lifeforms, so would not affect water quality under normal use conditions.

Direct adverse effects are probable within treatment areas to **non-target terrestrial plants** that are sensitive to the specific chemicals applied. All the herbicides are effective toward sensitive plants. Sulfometuron methyl is known to be highly toxic to a wide variety of plants. In general, tolerant species would be unaffected or only slightly affected by herbicide treatments. Off-site effects are possible if chemicals move from treatment areas in sufficient quantities to adversely affect non-target plants. Off-site drift from broadcast spray can transport sufficient quantities of herbicides (especially glyphosate, imazapyr, and sulfometuron methyl) to adversely affect sensitive species over 900 feet from the application site. Backpack spray, however, would result in substantially lower concentrations of herbicides and for most herbicides would not result in off-site effects, even in sensitive species.

Direct adverse effects are plausible within treatment areas to **non-target terrestrial lifeforms** that are susceptible to the specific chemicals applied. However, *except for 2,4-D, which is slightly to moderately toxic to mammals and practically non-toxic to moderately toxic to birds, the chemicals analyzed and likely to be applied under the VTP and alternatives are only slightly toxic to practically non-toxic to terrestrial organisms*. Toxicity ranges are due to variable toxicities to different species in the same class. For example, dogs have an impaired ability to excrete weak acids so are more susceptible to toxic effects from herbicides and large mammals may be at greater risk from triclopyr than small mammals. Effects to reptiles are largely unknown, as no toxicity testing was available on this class of animal.

Direct adverse effects are also plausible within treatment areas to **non-target aquatic lifeforms** that are susceptible to the specific chemicals applied. However, *except for glyphosate formulations containing POEA, and triclopyr BEE, which are likely to adversely affect sensitive aquatic species, the chemicals analyzed and likely to be applied under the VTP and alternatives are only slightly toxic to practically nontoxic to aquatic organisms*. Although amphibians appear to be particularly at risk, little to no toxicity data exists for this class of animal, especially for adult amphibians, for most of the chemicals analyzed.

*Chemical treatments under the VTP and alternatives have the potential to adversely affect individuals or populations of **special status species**.* Direct adverse effects are probable within treatment areas to plants that are susceptible to the specific chemicals applied. *Sulfometuron methyl is known to be highly toxic to a wide variety of plants.* In general, tolerant species would be unaffected or only slightly affected by herbicide treatments. Off-site effects are possible if chemicals move from treatment areas in sufficient quantities to adversely affect non-target, sensitive plants.

Direct adverse effects are possible from specific chemicals to **special status terrestrial lifeforms** that are susceptible to the specific chemicals applied. However, *the chemicals analyzed and likely to be applied under the VTP and alternatives are only slightly toxic to practically non-toxic to terrestrial organisms*. Toxicity ranges are due to variable toxicities to different species in the same class. Effects to reptiles are largely unknown as no toxicity testing was available on this class of animal.

Direct adverse effects are also probable within treatment areas to **special-status aquatic lifeforms** that are susceptible to the specific chemicals applied. However, *except for glyphosate formulations containing POEA, and triclopyr BEE, which are likely to adversely affect sensitive aquatic species, the chemicals analyzed and likely to be applied under the VTP and alternatives are only slightly toxic to practically nontoxic to aquatic organisms.* Although amphibians appear to be particularly at risk, little or no toxicity data exists for this class of animal for most of the chemicals analyzed.

Indirect effects from changes in plant species composition, cover, and/or population size, are likely to affect habitat for both plant and non-plant special status species, either adversely or beneficially, depending upon the species and site-specific conditions that cannot be determined at the PEIR scale.

Because site-specific factors at the subsequent activity level cannot be predicted, the amount of drift and/or volatilization of herbicides and the absolute effect on air quality cannot be quantified. What can be predicted is that *the more volatile herbicide formulations, the esters (triclopyr BEE) and the Velpar L® formulation of hexazinone, will be more likely to volatilize, move off-site in the air, and temporarily affect air quality.* This will be more likely in bioregions where volatile formulations are most used (North Coast, Modoc, and Sierra), in the vegetation lifeforms in which they are most used (Conifer Forest and Conifer Woodland), and where air temperatures and wind velocities are higher and humidities are lower during typical herbicide spray seasons.

Historically the main forestland applications of the most volatile herbicides (triclopyr BEE, and the Velpar L® formulation of hexazinone) has been for site preparation for planting and for release of tree seedlings from vegetative competition. These uses would be limited to practices funded through CFIP, so the acreage treated would be relatively small. In 2010, triclopyr BEE comprised a little over 4% of the total forestland acreage treated by the chemicals analyzed in this PEIR, and hexazinone (formulations unspecified) comprised a little over 14%. *If herbicide treatments in the VTP and alternatives follow historical patterns, herbicides suited for forest management would potentially be used primarily in conifer forests in the North Coast, Modoc, and Sierra bioregions*.

For rangeland applications of the most volatile herbicides, 2,4-D EHE has been among the top ten chemicals used between 2000 and 2010, although in 2010 it was applied to less than 1% of the total rangeland acreage treated by the chemicals analyzed in this PEIR. Triclopyr BEE has been in the top ten for all years and is used more than 2,4-D EHE. It was applied to about 19% of the rangeland acreage treated in 2010. *If herbicide treatments on rangelands in the VTP follow historical patterns, these herbicides would potentially be used primarily in grasslands in the Sacramento Valley bioregion, grasslands and shrublands in Sierra and Central Coast bioregions, and shrublands in the South Coast bioregion.*