APPENDIX C

ECOLOGICAL RISK ASSESSMENT

TABLE OF CONTENTS

Page

List of Tables

List of Figures

LIST OF ACRONYMS, ABBREVIATIONS, AND SYMBOLS

ac	-	Acres
a.i.	-	Active ingredient
ATV	-	All terrain vehicle
BA	-	Biological Assessment
BLM	-	Bureau of Land Management
BW	-	Body weight
CALPUFF	-	California Puff Model
CA OEHHA	-	California Office of Environmental Health Hazard Assessment
cm	-	Centimeter
cms	-	Cubic meters per second
CREAMS	-	Chemical Runoff Erosion Assessment Management System
C_{pond}	-	Concentrations from pond surface water
EC_{25}	-	Concentration causing 25% inhibition of a process (effect concentration)
EC_{50}	-	Concentration causing 50% inhibition of a process (median effective concentration)
Ed.	-	Edition
EEC	-	Estimated exposure concentration
EFED	-	Environmental Fate and Effects Division
EIS	-	Environmental Impact Statement
ERA	-	Ecological risk assessment
FCM	-	Food chain multiplier
FIFRA	-	Federal Insecticide, Fungicide, and Rodenticide Act
ft	-	Feet
g	-	Grams
gal	-	Gallon
GLEAMS	-	Groundwater Loading Effects of Agricultural Management Systems
HSDB	-	Hazardous Substances Data Bank
in	-	Inch
IRIS	-	Integrated Risk Information System
kg	-	Kilogram
km	-	Kilometers
L	-	Liters
lbs	-	Pounds
LC_{50}	-	Concentration causing 50% mortality (median lethal concentration)
LD_{50}	-	Dose causing 50% mortality (median lethal dose)
LOAEL	-	Lowest observed adverse effect level
LOC	-	Level of concern
m	-	Meters
mg	-	Milligrams
mg/kg	-	Milligrams per kilogram
mg/L	-	Milligrams per Liter
µg	-	Micrograms
µm	-	Micrometers
MRID	-	Master Record Identifier Number
MSDS	-	Material safety data sheet
NASA	-	National Aeronautics and Space Administration
NASQAN	-	National Stream Quality Accounting Network
NCDC	-	National Climatic Data Center
NOAEL	-	No observed adverse effect level
NMFS	-	National Marine Fisheries Service
NWS	-	National Weather Service

LIST OF ACRONYMS, ABBREVIATIONS, AND SYMBOLS (Cont.)

OPP	-	Office of Pesticide Programs
PEIS	-	Programmatic Environmental Impact Statement
PM	-	Particulate matter
PM_{10}	-	Particulate matter less than 10 micrometers in diameter or less
$PM_{2.5}$	-	Particulate matter less than 2.5 micrometers in diameter or less
ROW	-	Rights-of-way
RQ	-	Risk quotient
RTE	-	Rare, threatened, and endangered
RTEC	-	Registry of Toxic Effects of Chemical Substances
SAMSON	-	Solar and Meteorological Surface Observation Network
SDTF	-	Spray Drift Task Force
TRV	-	Toxicity reference value
TSP	-	Total suspended particulates
US	-	United States
USDA	-	United States Department of Agriculture
USDI	-	United States Department of the Interior
USEPA	-	United States Environmental Protection Agency
USFWS	-	United States Fish and Wildlife Service
USGS	-	United States Geological Survey
USLE	-	Universal Soil Loss Equation
>	-	Greater than
<	-	Less than
=	-	Equal to

APPENDIX C

ECOLOGICAL RISK ASSESSMENT

The purpose of this appendix is to summarize the ecological risks to plants and animals from 10 herbicides currently used, or proposed for use, by the United States Department of the Interior Bureau of Land Management (USDI BLM). More detailed assessments of these risks are given in Ecological Risk Assessments (ERA) prepared for each herbicide (see ENSR 2005a-j). These ERAs will be used by the BLM, in conjunction with analyses of other treatment effects on plants, animals, and other resources, to determine which of the proposed treatment alternatives evaluated in the Programmatic Environmental Impact Statement (PEIS) should be employed by the BLM. The BLM field offices will also utilize these ERAs for guidance on the proper application of herbicides to ensure that impacts to plants and animals are minimized to the extent practical when treating vegetation and do not pose unacceptable risks to non-target species, including rare, threatened, and endangered (RTE) species. The U.S. Fish and Wildlife Service (USFWS) and National Oceanic and Atmospheric Administration National Marine Fisheries Service (NMFS), will also use the information provided by the ERAs to assess the potential impact of vegetation treatment actions on fish and wildlife and their critical habitats.

The herbicide active ingredients (a.i.) evaluated in the ERAs are bromacil, chlorsulfuron, diflufenzopyr, diquat, diuron, fluridone, imazapic, Overdrive® (a mix of dicamba and diflufenzopyr), sulfometuron methyl, and tebuthiuron. Updated risk assessment methods were developed for the ERA process and are described in a separate document, *Vegetation Treatments Programmatic EIS Ecological Risk Assessment Methodology* (hereafter referred to as the "Methods Document" [ENSR 2004]). In addition, eight other herbicides are currently being used by the BLM and are proposed for continued use. These herbicides have been evaluated in a previous BLM EIS (USDI BLM 1991), as well as more recently in an invasive plant EIS prepared by the U.S. Department of Agriculture Forest Service (Forest Service; USDA Forest Service 2005).

Structure and Methodology of the Ecological Risk Assessment

Problem Formulation

Assessment endpoints represent "explicit expressions of the actual environmental value that is to be protected, operationally defined by an ecological entity and its attributes" (U.S. Environmental Protection Agency [USEPA] 1998). In the context of the screening-level, programmatic risk assessment, ecological entities include terrestrial invertebrates and vertebrates, non-target plants, and aquatic organisms (including RTE species). The essential biological requirements (i.e., survival, growth, and reproduction) for each of these groups of organisms are the attributes to be protected from herbicide exposure. Assessment endpoints, for the most part, reflect direct effects of an herbicide on these organisms, but indirect effects were also considered (particularly for threatened and endangered salmonids).

Measures of effect are measurable changes in an attribute of an assessment endpoint (or its surrogate, as discussed below) in response to a stressor to which it is exposed (USEPA 1998). For the screening-level ERA, the measures of effect associated with the assessment endpoints generally consisted of acute and chronic toxicity data (from pesticide registration documents and from the available scientific literature) for the most appropriate surrogate species. Rather than assess potential ecological risk to the large number of species found on public lands, surrogate species were used to represent classes of receptors (e.g., small mammalian herbivores, large avian piscivores [fish-eating birds]). In general, the surrogate species selected were those for which toxicity data were available from tests conducted in support of the USEPA pesticide registration process. Extrapolating chemical toxicity from a surrogate species to a particular species of concern can introduce extrapolation uncertainties (Fairbrother and Kapustka 1996, SERA 2000), but is often necessary in an ERA.

Assessment endpoints (and associated measures of effect) were generated in the problem formulation for each herbicide. Selection of specific assessment endpoints depends on the type of herbicide and its use pattern (e.g., terrestrial vs. aquatic application) and on the availability of appropriate toxicity data. Assessment endpoints include:

- Assessment Endpoint 1: *Acute mortality to mammals, birds, invertebrates, and non-target plants*. Measures of effect included median lethal effect doses (the dose lethal to 50% of organisms tested [LD_{50}]) from acute toxicity tests with these organisms or suitable surrogates.

- Assessment Endpoint 2: *Acute mortality to fish, aquatic invertebrates, and aquatic plants*. Measures of effect included median lethal effect concentrations (the concentration lethal to 50% of organisms tested [LC_{50}]) from acute toxicity tests with these organisms or suitable surrogates (e.g., other coldwater fish are used to represent threatened and endangered salmonids).

- Assessment Endpoint 3: *Adverse direct effects on growth, reproduction, or other ecologically important sublethal processes*. Measures of effect included standard chronic toxicity test endpoints such as the no observed adverse effect level ([NOAEL] the dose or concentration tested at which no adverse effects on test organisms were noted) for both terrestrial and aquatic organisms. Depending on data available for a given herbicide, chronic endpoints reflect either sublethal individual impacts (e.g., survival, growth, physiological impairment, behavior), or population-level impacts (e.g., reproduction [Barnthouse 1993]). For salmonids, careful attention was paid to smoltification (i.e., development of tolerance to seawater and other changes of parr [freshwater stage salmonids] to adulthood), thermoregulation (i.e., ability to maintain body temperature), migratory behavior, and other important life processes, if such data were available. With the exception of non-target plants, standard acute and chronic toxicity test endpoints were used for estimates of direct herbicide effects on RTE species. To add conservatism to the RTE assessment, levels of concern (LOCs) for RTE animals were lower than for typical species. Lowest available germination NOAELs were used to evaluate RTE plants.

- Assessment Endpoint 4: *Adverse indirect effects on the survival, growth, or reproduction of salmonids*. Measures of effect for this assessment endpoint depended on the availability of appropriate scientific data. Unless literature studies were found that explicitly evaluated the indirect effects of the target herbicides to salmonids and their habitat, estimates of indirect effects were qualitative. Such qualitative estimates of indirect effects include general evaluations of the potential risks to food (typically represented by acute and/or chronic toxicity to aquatic invertebrates) and cover (typically represented by potential for destruction of riparian vegetation). The USEPA Office of Pesticide Programs (OPP) is currently applying approaches similar to these qualitative evaluations for RTE species effects determinations and consultations.

Exposure Characterization

The BLM uses herbicides in a variety of programs (e.g., maintenance of rangeland and recreational sites) using several different application methods (e.g., application by aircraft, vehicle, backpack). In order to assess the potential ecological impacts of these herbicide uses, a variety of exposure scenarios were considered. These scenarios were selected based on actual BLM herbicide usage under a variety of conditions. There are differences among the individual herbicide risk assessment results based on the actual uses of a particular herbicide. Differences may include those attributable to application methodology (ground vs. aerial), area of application (forest vs. non-forest), or herbicide type (aquatic vs. terrestrial).

The exposure scenarios considered in the ERAs were organized by potential exposure pathways. In general, the exposure scenarios describe how a particular receptor group (e.g., terrestrial animals) may be exposed to the herbicide as a result of a particular exposure pathway. These exposure scenarios were designed to address herbicide exposure that may occur under a variety of conditions:

- Direct spray of the receptor or waterbody

- Indirect contact with dislodgeable foliar residue

- Ingestion of contaminated food items
- Off-site drift of spray to terrestrial areas and waterbodies
- Surface runoff from the application area to off-site soils or waterbodies
- Wind erosion resulting in deposition of contaminated dust
- Accidental spills to waterbodies

These scenarios were developed to address potential acute and chronic impacts to receptors under a variety of exposure conditions that may occur within public lands. These exposure conditions include normal application situations and associated off-site transport (via drift or wind erosion of dust), as well as accidental spills, and long-term overland flow to off-site soils and waterbodies (primarily via surface runoff and root-zone groundwater flow).

Additional details regarding specific receptors (e.g., receptor size, diet, RTE species status), application rates (e.g., typical vs. maximum applications rates, accidental spills), duration of herbicide exposure (e.g., one-time event, long-term exposure), and toxicity endpoints (e.g., acute, chronic) are discussed below. Additional information can be found in the individual ERAs and associated risk assessment spreadsheets compiled for each herbicide (ENSR 2005a-j).

Because of the differences in the application methods for terrestrial and aquatic herbicides, there were fewer exposure scenarios for aquatic herbicides. Off-site transport of the aquatic herbicides via surface runoff and wind erosion were not considered to be realistic scenarios for these applications and were therefore not considered for the aquatic herbicides. However, accidental direct spray of aquatic herbicides onto terrestrial receptors and off-site drift onto terrestrial plants were considered. The more conservative direct spray scenario was assumed to address any potential impacts from the other transport mechanisms. Details of the exposure scenarios considered in the risk assessments are presented in Section 3.0 of the Methods Document (ENSR 2004).

Herbicide levels resulting in potential risk to surrogate species were calculated using conservative assumptions. Exposure scenarios were included that are unlikely to occur (e.g., direct spray of receptor or waterbody, accidental spills to waterbodies). Furthermore, animals were assumed to have a home range equal to the application area, whereas many animals would range outside of application areas, reducing exposure. In addition, all applied herbicide was assumed to be biologically available, and no attempt was made to assess the tendency of herbicide degradation or water flow to decrease herbicide concentrations and exposure (see Appendix B of the ERAs [ENSR 2005a-j] for equations and calculations for each of the different exposure scenarios).

Exposure characterizations depend on the selection of appropriate fate and transport models that predict herbicide concentrations in various environmental media, such as tissues, soils, and water. Some of these models are fairly straightforward and only require simple algebraic calculations (e.g., water concentrations from direct aerial spray), but others require more complex computer models (e.g., aerial deposition rates, transport from soils).

The AgDRIFT® computer model (see page C-86) was used to estimate off-site herbicide transport due to spray drift. The Groundwater Loading Effects of Agricultural Management Systems (GLEAMS) computer model (see page C-86) was used to estimate off-site transport of herbicide in surface runoff and root-zone groundwater transport (Knisel and Davis 2000). The computer model California Puff (CALPUFF; see page C-86) was used to predict the transport and deposition of herbicides sorbed (i.e., reversibly or temporarily attached) to wind-blown dust. Each model simulation was approached with the intent of predicting the maximum potential herbicide concentration that could result from the given exposure scenario.

Effects Characterization

The ecological effects characterization phase of an ERA entails a compilation and analysis of the stressor-response relationships and any other evidence of adverse impacts from exposure to each herbicide. This evidence consisted mostly of toxicity studies conducted in support of USEPA pesticide registration, which generally include the following (additional studies may be required depending on herbicide use patterns and characteristics:

- Avian oral LD_{50}
- Avian dietary LC_{50}
- Freshwater fish acute LC_{50}

- Freshwater invertebrate acute LC_{50}

Additional tests, if required for a particular herbicide, may include honeybee acute toxicity, avian reproduction, non-target plant toxicity, and chronic fish life-cycle tests, among others. As data were not available for all receptors or for threatened/endangered species, extrapolation of risk based on surrogate species data was necessary. Species for which toxicity data were available were not necessarily the most sensitive species to a particular herbicide (these species are used as laboratory test organisms because they are generally sensitive to stressors and they can be maintained under laboratory conditions). The selected toxicity value for a receptor is based on a review of the available data for the most appropriate, sensitive surrogate species.

In the majority of cases, toxicological data do not exist for the specific ecological receptors of concern. Consequently, toxicological data for surrogate species were evaluated and used to establish quantitative benchmarks for the ecological receptors of concern. These benchmark values are referred to as toxicity reference values (TRVs). This section of text briefly describes the process used to derive TRVs. Once developed, TRVs were compared with predicted environmental concentrations of the herbicide to determine the likelihood of adverse effects to ecological receptors.

Literature Review

The literature review process for deriving TRVs consisted of assembling relevant literature, evaluating these information sources, and then establishing specific numeric values for each ecological receptor. Literature sources included published manuscripts, unpublished study reports, and electronic databases. Once data from these various sources were compiled, the information was reviewed to determine its acceptability for deriving ecological TRVs for each of the 10 herbicides. In order to be classified as an "acceptable" study, the research had to be suitable and of adequate quality (see following sections).

Data Suitability

For each chemical, the available literature was evaluated to determine if the data were suitable for use in deriving TRVs. Early in the ERA process, the BLM identified receptors that were representative of ecological guilds (i.e., general taxonomic groups comprised of animals or plants that perform particular roles in the ecosystem, including small and large mammals, small and large birds, piscivorous birds, fish, reptiles, insects, amphibians, terrestrial and aquatic plants, and algae) and their primary routes of exposure. Evaluation of suitability was based on these ecological receptors and routes of exposure. Specifically, a study was considered suitable if the following criteria were met:

- The material tested was one of the 10 herbicides;
- the test species was in the same guild as an ecological receptor;
- the route of exposure matched the primary routes of exposure for species in that guild; and
- the toxicity assessment endpoint (e.g., mortality, reproductive success, growth) was considered to be ecologically relevant.

For the majority of studies, the acute statistical measures of effect consisted of LD_{50}, LC_{50}, or EC_{50} (the concentration resulting in a defined effect in 50% of the receptors tested) values. Adverse effect levels in chronic studies were most frequently reported as lowest observed adverse effect levels (LOAELs). Levels at which no effects were noted were generally reported in chronic studies as NOAELs. Several additional statistical endpoints were evaluated for terrestrial plants, including EC_{25} (the concentration resulting in a defined effect in 25% of the receptors tested), NOAEL, and highest and lowest NOAEL (for germination and emergence endpoints only).

Data Adequacy

Once determined to be suitable, a study was then evaluated to determine whether the data were adequate. For peer-reviewed literature, two senior toxicologists independently determined data adequacy. Each paper was scored based on several selection criteria, including documentation of number of test organisms, statistical analysis, and proper use and performance of controls. Based on these reviews, the study was classified as either "adequate" or "not adequate."

Toxicity Reference Value Development

Study findings met both data adequacy and suitability criteria were used to develop ecological TRVs. From these studies, statistical endpoints were compiled into a matrix for each chemical and for each receptor. Data were further subdivided into acute adverse effect levels, chronic adverse effect levels, and no adverse effect levels.

Endpoints for a receptor and routes of exposure were converted to the same units (e.g., mg/kg body weight [BW]). Endpoints for aquatic organisms and terrestrial plants were reported based on exposure concentrations (mg/L and lbs/acre [ac], respectively). Dose-based endpoints (e.g., LD_{50}) were used for the remaining receptors. When possible, dose-based endpoints were obtained directly from the literature. When dosages were not reported, dietary concentration data (e.g., LC_{50}) were converted to dose-based values following the methodology recommended in USEPA risk assessment guidelines (Sample et al. 1996). See the ERA Methods document (ENSR 2004; Table 2-3) for a summary of animal body weights and feeding and drinking ingestion rates that were used to convert concentration endpoints to dose-based endpoints.

Toxicity Reference Value Derivation

Once the data were expressed in comparable units, the numeric values from studies classified as "acceptable" were compared to derive TRVs. For each chemical, receptor, and route of exposure, the lowest reported acute statistical endpoint was selected as the acute TRV. Acute TRVs were derived first to provide an upper boundary for the remaining TRVs; chronic TRVs and NOAELs were always equivalent to, or less than, the acute TRV.

The toxicity endpoint for most acute studies was mortality, immobilization, or failure to germinate, as assessed during a short-term exposure. In some cases, acute data were not always available. Consequently, chronic TRVs, based on longer exposure periods and associated endpoints such as growth and reproduction, were developed to provide supplementary data to the risk assessment. Conversely, when no valid statistical endpoints from chronic studies were available, the chronic TRV was set equal to the acute TRV. In the majority of cases, however, chronic data were available. Before the chronic NOAEL TRV was determined, a chronic LOAEL was identified, which was the lowest herbicide level that was found to cause significant adverse effects in a chronic study. Once a LOAEL was established, the chronic NOAEL TRV was established as the highest NOAEL value that was less than both the LOAEL and the acute TRV.

Use of the Uncertainty Factor

In some cases, a TRV for a particular assessment endpoint had to be extrapolated from available TRVs using an uncertainty factor. Based on a review of the application of uncertainty factors (Chapman et al. 1998), an uncertainty factor of 3 was considered to be appropriate for ecological TRV derivation in this document. For example, a chronic or an acute TRV (e.g., 100 mg/kg BW) could be divided by an uncertainty factor of 3 to obtain an extrapolated NOAEL (e.g., 33 mg/kg BW). Conversely, if a NOAEL value was available, but a chronic TRV was lacking, the NOAEL TRV could be multiplied by 3 to extrapolate the chronic TRV (but not the acute TRV).

Risk Characterization

The risk characterization phase of an ERA consists of a quantitative estimate of the ecological risks, a description of data used in support of these risk estimates (including data gaps where appropriate), and an overall interpretation of the potential ecological impacts of each herbicide (following consideration of uncertainties in the analyses).

In order to address potential risks to ecological receptors, risk quotients (RQs) were calculated by dividing the estimated exposure concentration (EEC) for each of the previously described scenarios by the appropriate toxicity endpoint, an herbicide-specific TRV. The TRV may be a surface water or surface soil effects concentration, or a species-specific toxicity value derived from the literature.

The RQs were then compared against LOCs established by the USEPA OPP to assess potential risk to non-target organisms. These LOCs are used by the USEPA's OPP to analyze potential risk to non-target organisms and to assess the need to consider regulatory action (Table C-1). Distinct® USEPA LOCs are currently defined for the following risk presumption categories:

- Acute high risk – the potential for acute risk is high.
- Acute restricted use – the potential for acute risk is high, but may be mitigated.
- Acute RTE species – RTE species may be adversely affected.

TABLE C-1
Levels of Concern

Risk Presumption		RQ	LOC
Terrestrial Animals [1]			
Birds	Acute high risk	EEC/LC_{50}	0.5
	Acute restricted use	EEC/LC_{50}	0.2
	Acute RTE species	EEC/LC_{50}	0.1
	Chronic risk	EEC/NOAEL	1
Wild mammals	Acute high risk	EEC/LC_{50}	0.5
	Acute restricted use	EEC/LC_{50}	0.2
	Acute RTE species	EEC/LC_{50}	0.1
	Chronic risk	EEC/NOAEL	1
Aquatic Animals [2]			
Fish and aquatic invertebrates	Acute high risk	EEC/LC_{50} or EC_{50}	0.5
	Acute restricted use	EEC/LC_{50} or EC_{50}	0.1
	Acute RTE species	EEC/LC_{50} or EC_{50}	0.05
	Chronic risk	EEC/NOAEL	1
	Chronic risk, RTE species	EEC/NOAEL	0.5
Plants [3]			
Terrestrial/semi-aquatic plants	Acute high risk	EEC/EC_{25}	1
	Acute RTE species	EEC/NOAEL	1
Aquatic plants	Acute high risk	EEC^2/EC_{50}	1
	Acute RTE species	EEC/NOAEL	1

[1] Estimated Environmental Concentration is in mg $_{\text{prey wet weight}}$/kg $_{\text{BW}}$ for acute scenarios and mg $_{\text{prey wet weight}}$/kg $_{\text{BW}}$/day for chronic scenarios.
[2] Estimated Environmental Concentration is in mg/L.
[3] Estimated Environmental Concentration is in lb/acre.

- Chronic risk - the potential for chronic risk is high.

A "chronic RTE species" risk presumption category for aquatic animals was added for this risk assessment. The LOC for this category was set to 0.5 to reflect a conservative 2-fold difference in contaminant sensitivity between RTE and surrogate test fishes (Sappington et al. 2001).

Risk quotients (RQs) and LOCs were tabulated and compared for all appropriate exposure scenarios and surrogate species described above. The ecological risk implications of various exposure estimates can be readily determined by noting which RQs exceed the corresponding LOCs. Over 1,000 RQs were generated in each ERA. While all RQs are presented in the supporting documentation of the risk assessment and available to BLM field offices, only selected values (e.g., those exceeding LOCs) are discussed within the text of each ERA report (ENSR 2005a-j).

Rare, Threatened, and Endangered Species

To specifically address potential impacts to RTE species, two types of RQ evaluations were conducted. For RTE terrestrial plant species, the RQ was calculated using different toxicity endpoints but keeping the same LOC (1) for all scenarios. The plant toxicity endpoints were selected to provide extra protection to RTE species. In the direct spray, spray drift, and wind erosion scenarios, the selected toxicity endpoints were an EC_{25} for "typical" species and a NOAEL for RTE species. In runoff scenarios, high and low germination NOAELs were selected to evaluate exposure for typical and RTE species, respectively.

The evaluation of RTE terrestrial animals and aquatic species was addressed using a second type of RQ evaluation. The same toxicity endpoint was used for both typical and RTE species in all scenarios, but the acute LOC was lowered for RTE species (see Table C-1).

Uncertainty Analysis

For any ERA, a thorough description of uncertainties is a key component that serves to identify possible weaknesses in the analysis and to elucidate what impact such weaknesses might have on the final risk conclusions. In general, an uncertainty analysis lists the uncertainties, followed by a logical discussion of what bias, if any, the uncertainty may introduce into the risk conclusions. This bias would be represented in qualitative terms that best describe whether the uncertainty might: 1) underestimate risk, 2) overestimate risk, 3) be neutral with regard to the risk estimates, or 4) be unable to be determined without additional study. Key categories of uncertainty for the herbicides ERAs include:

- *Limited toxicity data available for a given herbicide.* For some herbicides, the only toxicity data available may be those studies conducted as part of the USEPA pesticide registration process. In this case, chronic toxicity data may be limited or non-existent and may not include sublethal studies of importance relevant to assessment endpoint 4 (*Adverse indirect effects on the survival, growth, or reproduction of salmonids*). When relevant studies did not exist, the uncertainties were discussed as thoroughly as possible.

- *The potential indirect effects of herbicides on RTE salmonids.* Unless actual field studies were identified for a given herbicide, this discussion was limited to only qualitative estimates of potential indirect impacts on salmonid populations and communities. Such qualitative estimates were limited to a general evaluation of the potential risks to food (typically represented by acute and/or chronic toxicity to aquatic invertebrates) and cover (typically represented by potential for destruction of riparian vegetation or aquatic vegetation, if appropriate). The USEPA OPP is using similar approaches for RTE species effects determinations and consultations.[1]

- *Extrapolating from laboratory to field studies.* It is preferable to base any ecological risk analysis on reliable field studies that can clearly identify and quantify the amount of potential risk from particular exposure concentrations of the chemical of concern as field studies provide a more accurate representation of environmental conditions. When available, incident reports for the USEPA's Environmental Fate and Effects Division (EFED) were reviewed in an attempt to validate both exposure models and/or hazards to ecological receptors. For many of the new herbicides, however, such studies were not available. Most available incident reports present incomplete data, and explicit information linking herbicide exposure and resulting effects are difficult to interpret. In these cases, best professional judgment was used to evaluate the potential bias, if any, the lack of field studies had on risk conclusions. It should be noted, though, that in most cases laboratory studies actually overestimate risk relative to field studies, supporting the conservative nature of the risk assessment (Fairbrother and Kapustka 1996).

- *Ecological risks of inert ingredients, adjuvants, degradates, and tank mixtures.* From an ecological point of view, it is desirable to estimate risks not just from the a.i. of an herbicide, but also from the cumulative exposure to other ingredients. However, using currently available models (e.g., GLEAMS), deterministic risk calculations (i.e., exposure modeling, effects assessment, and RQ calculations) can only be conducted for a single a.i. However, qualitative estimates were made of the potential additional risks (if any) posed by chemicals added to the a.i. of an herbicide, such as inert ingredients (ingredients lacking active properties though still potentially toxic), adjuvants (chemicals used to enhance the pharmacological or toxic agent effect of the a.i.), and degradates (chemicals created during the natural breakdown or decomposition of another chemical).

Evaluating the potential additional/cumulative risks from mixtures of pesticides is substantially more difficult, particularly at the level of a PEIS. The composition of such mixtures is highly site-specific, and thus nearly impossible to address at the programmatic level. However, the label information from each of the 10 herbicides mentions that most can be "tank mixed" with other herbicides and insecticides. Therefore, for each herbicide, a qualitative evaluation

[1] http://www.epa.gov/oppfead1/endanger/effects.

was made of the potential additional risk that might result from applying each as part of a label-approved tank mix. It should be emphasized that this evaluation was only qualitative, based on risk conclusions from existing ERAs conducted for an earlier EIS (USDI BLM 1991), for the USDA Forest Service, or by the USEPA for registration and/or re-registration. Such an analysis can only be qualitative unless reliable scientific evidence exists to suggest whether the joint action of the herbicides is additive, synergistic, or antagonistic.

- *Estimates of herbicide exposure concentrations.* As in any screening or higher-tier ERA, a discussion of potential uncertainties from fate and exposure modeling is necessary to identify potential overestimates or underestimates of risk. In particular, the uncertainty analysis focused on which environmental characteristics (e.g., soil type, annual precipitation) exert the most significant numeric impact on model outputs. The results of the uncertainty analysis have important implications about the ability to apply risk calculations to different site characteristics from a risk management point of view.

Application Methods and Herbicide Usage

Table C-2 provides herbicide usage statistics, including application sites, application methods, and application rates.

Aerial Application

Aerial application is conducted from fixed-wing planes and/or rotary helicopters in the BLM Rangeland, Public Domain Forestland, Energy and Mineral Sites, Rights-of-way (ROW), Recreation, and Aquatics programs. ERA modeling assumed that herbicides were applied with buffers of 100, 300, and 900 feet (ft) from evaluated receptors, and application heights varied depending on whether the application area was forested or not.

Ground Application

Ground applications take place in the BLM Rangeland, Public Domain Forestland, Energy and Mineral Sites, ROW, Recreation, and Aquatics programs. Applications are conducted on foot or horseback using backpack sprayers or from vehicles (truck, all-terrain vehicle [ATV], boat) using spot or boom/broadcast (low or high boom) methods. For modeling purposes, herbicides were applied with buffers of 25, 100, and 900 ft from evaluated receptors.

Aquatic Application

There are four zones in a body of water that may be treated for the management of aquatic weeds: water surface, total water volume, bottom 1 to 3 ft of water, and the bottom soil surface.

When working in the water surface zone, generally only a fourth to a third of the surface area is treated at a time. Applications are made to floating or emergent weeds with the spray mixture being applied directly to the plants. When treating the total water volume, applications can be made through the metering or injecting of the herbicide into the water from booms trailing behind the boat or as a spray over the water surface. Applications of this type are made to submersed aquatic plants and algae. Treatments to the deepest 1 to 3 ft of water are generally made by attaching several flexible hoses at specific intervals on a rigid boom. Each hose is equipped with a nozzle and may be weighted to reach the depth desired. The length of hose and the speed of the boat carrying the application equipment also affect the depth of application. Such applications are beneficial because they apply the herbicide in a layer nearer the area where the herbicide can be taken up by the weedy species. The final zone, bottom soil surface, refers to applications made to the bottom soil of a drained pond, lake, or channel.

To treat small areas, a compressed-air sprayer with a hand-operated pump may be all that is needed. For larger areas, a boat-mounted pump-and-tank rig with one line may be used to treat emergent plants on a spot treat basis. A boom attached to the boat may be used when broadcast applications are made to the surface of the water, and booms with flexible hoses attached to the boom may be used to make the application below the water surface. Applications of granules and slow-release pellets can be made either using a cyclone spreader or by hand. The granules sink to the bottom, where the chemical is slowly released in the relatively small volume of water where the new shoots are beginning to grow.

Floating and emergent vegetation in static water (i.e., water in ponds, lakes, or reservoirs that have little or

TABLE C-2
Herbicide Application Methods and Usage Statistics

Herbicide	Programs/Treatment Areas	Application Method	Application Rate (lbs. a.i./acre)	
			Typical	Maximum
Bromacil	Energy and mineral sites Rights-of-way Recreation	Backpack, horseback, ATV, and truck (spot, boom/broadcast)	4.0	12.0
Chlorsulfuron	Rangeland Energy and mineral sites Rights-of-way Recreation	Plane, helicopter backpack, horseback, ATV, and truck (spot, boom/broadcast)	0.047	0.062
Diflufenzopyr	Rangeland Energy and mineral sites Rights-of-way Recreation	Backpack, horseback, ATV, and truck (spot, boom/broadcast)	0.075	0.1
Diquat	Aquatic	Plane, helicopter backpack, horseback, ATV, and truck (spot, boom/broadcast)	1.0	4.0
Diuron	Energy and mineral sites Rights-of-way Recreation	Backpack, horseback, ATV, and truck (spot, boom/broadcast)	6.0	20.0
Fluridone	Aquatic	Plane, helicopter backpack, horseback, ATV, and truck (spot, boom/broadcast)	0.15	1.3
Imazapic	Rangeland Public domain forestland Energy and mineral sites Rights-of-way Recreation	Plane, helicopter backpack, horseback, ATV, and truck (spot, boom/broadcast)	0.0313	0.1875
Overdrive®	Rangeland Oil and gas Rights-of-way Recreation	Backpack, horseback, ATV, and truck (spot, boom/broadcast)	0.2625	0.35
Sulfometuron methyl	Public domain forestland Energy and mineral sites Rights-of-way Recreation	Helicopter Backpack, horseback, ATV, and truck (spot, boom/broadcast)	0.14	0.38
Tebuthiuron	Rangeland Energy and mineral sites Rights-of-way Recreation	Plane, helicopter backpack, horseback, ATV, and truck (spot, boom/broadcast)	0.5	4.0

no inflow and outflow) is managed by direct foliage applications of the spray mixture by aircraft, with ground equipment operated from the bank if the pond is small or if the weeds occur only around the margins, or from a boat using various types of booms or hand applicators.

Aquatic vegetation in flowing water is more difficult to manage. Floating and emergent vegetation, when treated in flowing water, require the same treatment techniques as they do in the static water. Submersed vegetation and algae can be controlled effectively in flowing water only by continuously applying enough herbicide at a given spot to maintain the needed concentration and contact time.

Non-target Species Exposure Characterization

As described earlier, a number of exposure scenarios were developed to address potential acute and chronic impacts to receptors under a variety of exposure conditions that may occur within public lands. These exposure conditions include normal application situations, accidental spills, and associated off-site transport via spray drift, windblown dust, or surface runoff and root-zone groundwater. In general, the exposure scenarios describe how a particular receptor group (e.g., terrestrial animals, terrestrial plants, aquatic plants) may be exposed to an herbicide in a complete exposure pathway. The selected pathways and relevant dose calculations are described in more detail in the following sections.

This section discusses the exposure of terrestrial and aquatic receptors to the 10 herbicides proposed for new and continued use on public lands in 17 western states. The processes of surrogate species selection and the calculation of exposure data based on species biology and herbicide application rates are presented below.

Ecological Receptors

Surrogate Species

Use of surrogate species in a screening ERA is necessary to address the broad range of species likely to be encountered on public lands as well as to accommodate the fact that toxicity data may be restricted to a limited number of species. In this ERA, surrogates were selected to account for variation in the nature of potential herbicide exposure (e.g., direct contact, food chain) as well as to ensure that different taxa and their behaviors were considered. Generally, the surrogate species that were used in the ERAs are species commonly used as representative species in ecological risk assessments. Many of these species are common laboratory species, or are described in the USEPA (1993a, b) *Wildlife Exposure Factors Handbook*. Other species were included in the California Wildlife Biology, Exposure Factor, and Toxicity Database (California Office of Environmental Health Hazard Assessment [CA OEHHA] 2003), or are those recommended by USEPA OPP for tests to support pesticide registration.

Toxicity data from surrogate species were used in the development of TRVs. The surrogate species used for development of TRVs in each herbicide ERA are presented in Table C-3. For vertebrate terrestrial animals, in addition to these surrogate species, specific species were selected to represent populations of similar species (Table C-4). Interspecies extrapolation of toxicity data often produces unknown bias in risk calculations; therefore, higher trophic-level species were grouped according to shared life-history traits (e.g., herbivore vs. carnivore). Whenever possible, the species selected are found throughout the range of land included in the PEIS; all species selected are found in at least a portion of the range. The surrogate species are common species whose life histories are well documented (USEPA 1993a, b; CA OEHHA 2003). Because species-specific data, including body weight and food ingestion rates, can vary for a single species throughout its range, data from studies conducted in western states or with western populations were selected preferentially. This life-history procedure was not done for plants, invertebrates, and fish, as most exposure of these species to herbicides is via direct contact (e.g., foliar deposition, dermal deposition, dermal/gill uptake) rather than ingestion of contaminated prey items. Therefore, altering the life history of these species would not result in more or less exposure. In addition, potential impacts to non-target terrestrial plants were considered by evaluating two non-target plant receptors: the “typical” (i.e., non-RTE) species and the RTE species.

Very few laboratory studies have been conducted using reptiles or amphibians. Therefore, data specific to the adverse effects of a chemical on species of these taxa are often unavailable. These animals, being cold-blooded, have very different rates of metabolism than mammals or birds (i.e., they require lower rates of

TABLE C-3
Surrogate Species Used in Quantitative ERA Evaluations

Surrogate Species	Scientific Name	Receptor	Herbicide
Honeybee	*Apis mellifera*	Pollinating insects	Bromacil, chlorsulfuron, diflufenzopyr, diquat, diuron, fluridone, imazapic, sulfometuron methyl, and tebuthiuron
Mouse	*Mus musculus*	Mammals	Bromacil, diquat, fluridone, sulfometuron methyl, and tebuthiuron
Rat	*Rattus norvegicus* spp.	Mammals	Bromacil, chlorsulfuron, diflufenzopyr, diquat, diuron, fluridone, imazapic, sulfometuron methyl, and tebuthiuron
Dog	*Canis familiaris*	Mammals	Bromacil, chlorsulfuron, diflufenzopyr, diquat, diuron, fluridone, imazapic, sulfometuron methyl, and tebuthiuron
Rabbit	*Leporidae* sp.	Mammals	Bromacil, chlorsulfuron, diflufenzopyr, diquat, diuron, fluridone, imazapic, sulfometuron methyl, and tebuthiuron
Guinea pig	*Cavia* sp.	Mammals	Chlorsulfuron
Mallard	*Anas platyrhynchos*	Birds	Bromacil, chlorsulfuron, diflufenzopyr, diquat, diuron, fluridone, imazapic, and sulfometuron methyl
Bobwhite quail	*Colinus virginianus*	Birds	Bromacil, diflufenzopyr, diquat, diuron, sulfometuron methyl, and tebuthiuron
Ring-necked pheasant	*Phasianus colchicus*	Birds	Diquat, fluridone, and imazapic
Japanese quail	*Coturnix coturnix*	Birds	Diquat
Chicken	*Gallus gallus*	Birds	Tebuthiuron
Rape	*Orobanche* sp.	Non-target terrestrial plants	Bromacil
Soybean	*Glycine max*	Non-target terrestrial plants	Bromacil, diuron, and imazapic
Canola	*Brassica napus*	Non-target terrestrial plants	Chlorsulfuron
Dyer's woad (weed)	*Isatis tinctoria*	Non-target terrestrial plants	Chlorsulfuron
Turnip	*Brassica rapa*	Non-target terrestrial plants	Diflufenzopyr
Tomato	*Lycopersicon esculentum*	Non-target terrestrial plants	Diflufenzopyr and diuron
Corn	*Zea mays*	Non-target terrestrial plants	Diquat and imazapic
Garden pea	*Pisum sativum*	Non-target terrestrial plants	Diuron
Vegetative crop	9 species, monocotyledons and dicotyledons	Non-target terrestrial plants	Imazapic
Onion	*Allium cepa*	Non-target terrestrial plants	Imazapic
White mustard	*Sinapis alba*	Non-target terrestrial plants	Sulfometuron methyl
Leafy spurge	*Euphorbia esula*	Non-target terrestrial plants	Sulfometuron methyl
Sorghum	*Sorghum bicolor*	Non-target terrestrial plants	Sulfometuron methyl
Sugarbeet	*Beta vulgaris*	Non-target terrestrial plants	Sulfometuron methyl
Cabbage	*Brassica* sp.	Non-target terrestrial plants	Tebuthiuron
Daphnid	*Daphnia magna*	Aquatic invertebrates	Bromacil, chlorsulfuron, diflufenzopyr, diquat, imazapic, and sulfometuron methyl
Daphnid	*Ceriodaphnia dubia*	Aquatic invertebrates	Sulfometuron methyl and tebuthiuron
Scud	*Gammarus fasciatus*	Aquatic invertebrates	Diuron
Amphipod	*Hyalella azteca*	Aquatic invertebrates	Diquat
Midge	*Chironomus tentans*	Aquatic invertebrates	Fluridone
Snail	*Helisoma* and *Physa* spp.	Aquatic invertebrates	Tebuthiuron

TABLE C-3 (Cont.)
Surrogate Species Used in Quantitative ERA Evaluations

Surrogate Species	Scientific Name	Receptor	Herbicide
Fathead minnow	*Pimephales promelas*	Fish	Bromacil, diuron, sulfometuron methyl, and tebuthiuron
Channel catfish	*Ictalurus punctatus*	Fish	Chlorsulfuron
Bluegill sunfish	*Lepomis macrochirus*	Fish	Diflufenzopyr and tebuthiuron
Walleye	*Stizostedion vitreum*	Fish	Diquat
Brown trout	*Salmo trutta*	Fish/salmonids	Chlorsulfuron
Rainbow trout	*Oncorhynchus mykiss*	Fish/salmonids	Bromacil, diflufenzopyr, diquat, fluridone, imazapic, and sulfometuron methyl
Cutthroat trout	*Oncorhynchus clarki*	Fish/salmonids	Diuron
Sago pondweed	*Potamogeton pectinatus*	Non-target aquatic plants	Chlorsulfuron
American pondweed	*Potamogeton nodosus*	Non-target aquatic plants	Fluridone
Green algae	*Selanastrum capricornutum*	Non-target aquatic plants	Bromacil, diflufenzopyr, diuron, and tebuthiuron
Algae	*Chlorella pyrenoidosa*	Non-target aquatic plants	Diuron
Duckweed	*Lemna* sp.	Non-target aquatic plants	Diquat and imazapic
Macrophyte	*Myriophyllum sibiricum*	Non-target aquatic plants	Sulfometuron methyl

TABLE C-4
Vertebrate Surrogate Species Evaluated by Life History

Species	Scientific Name	Trophic Level/Guild	Pathway Evaluated
American robin	*Turdus migratorius*	Avian invertivore/vermivore/insectivore	Ingestion
Canada goose	*Branta canadensis*	Avian granivore/herbivore	Ingestion
Deer mouse	*Peromyscus maniculatus*	Mammalian frugivore/herbivore	Direct contact and ingestion
Mule deer	*Odocoileus hemionus*	Mammalian herbivore/gramivore	Ingestion
Bald eagle (northern)	*Haliaeetus leucocephalus alascanus*	Avian carnivore/piscivore	Ingestion
Coyote	*Canis latrans*	Mammalian carnivore	Ingestion

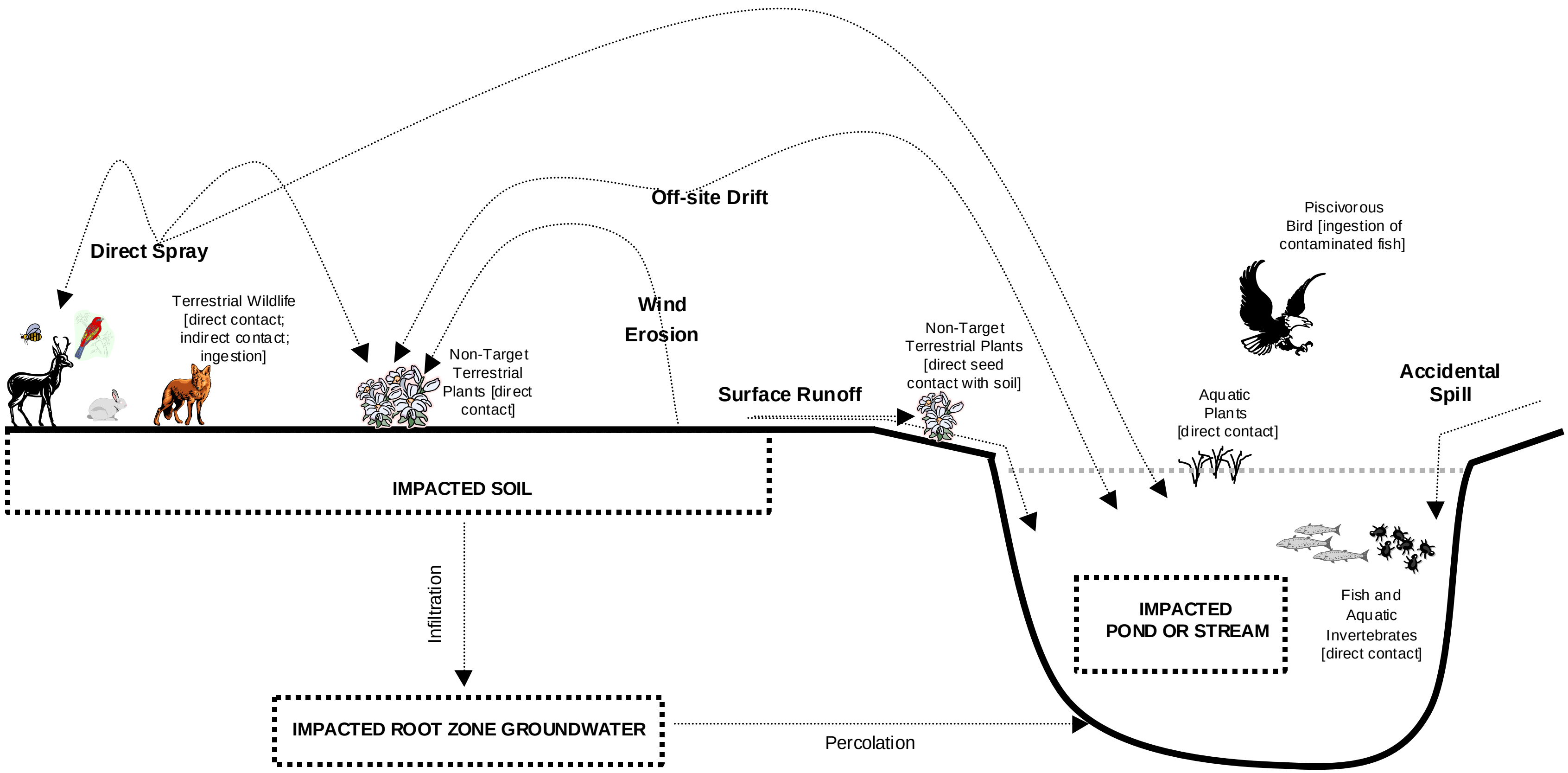
food consumption). Nonetheless, mammals and birds were used as the surrogate species for reptiles and adult amphibians because of the lack of data for these taxa (fish were used as surrogates for juvenile amphibians).

Aquatic exposure pathways were evaluated using fish, aquatic invertebrates, and non-target aquatic plants for two types of generic aquatic habitat: 1) a small pond ¼-acre in area, 1 meter (m) in depth, and 1,011,715 liters (L) in volume, and 2) a small stream representative of Pacific Northwest low-order streams that provide habitat for critical life-stages of anadromous salmonids (the stream is defined as 2 m-wide and 0.2-m deep, with a mean water velocity of approximately 0.3 m per second, resulting in a base flow discharge of 0.12 cubic meters per second [cms]).

Exposure Pathways

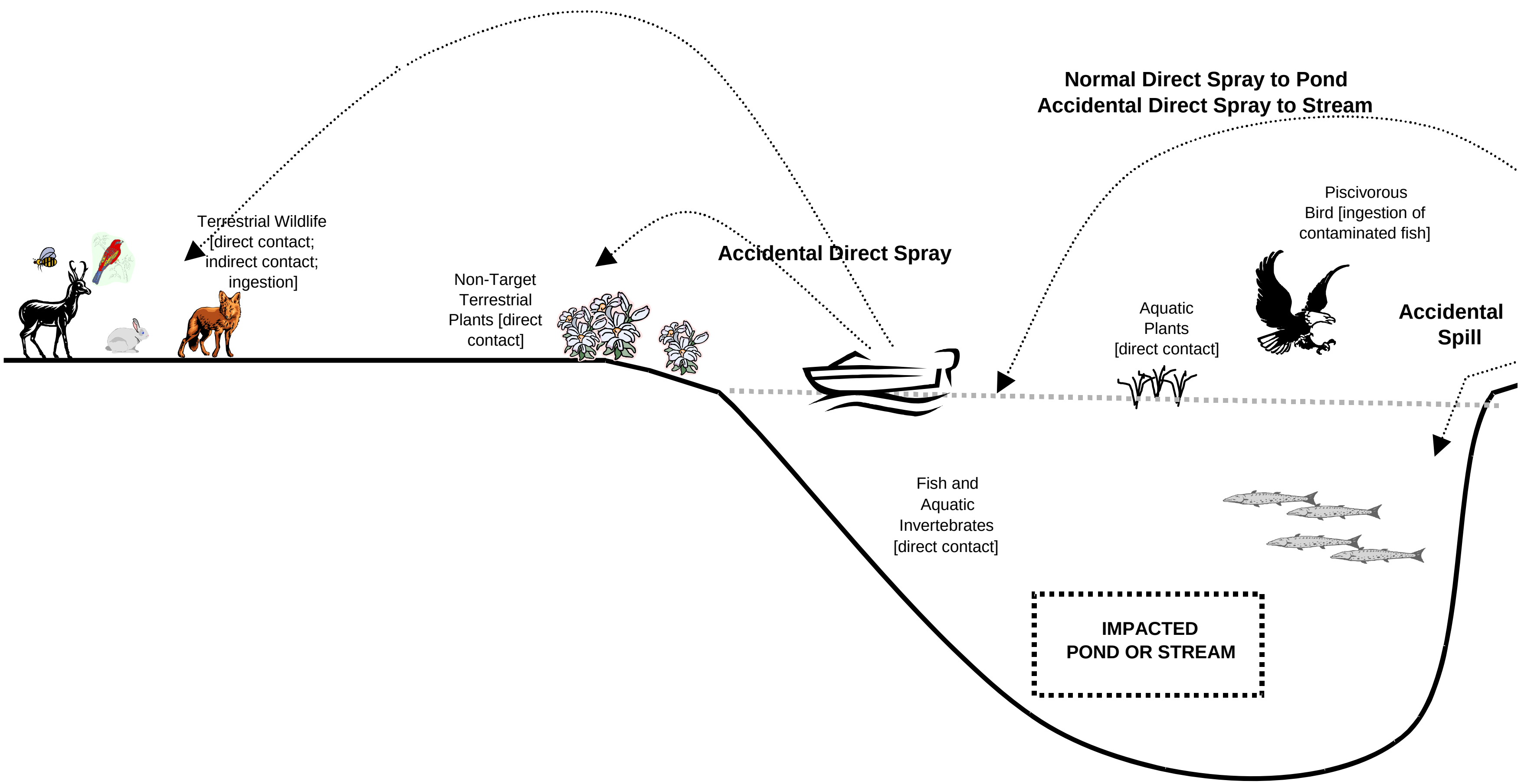
The following is a brief description of the scenarios used to address potential impacts to non-target organisms both within the area where the herbicide is being applied and outside the application area (accidental exposures not typical of BLM practices). Conceptual models were developed that provide working hypotheses about how terrestrial (Figure C-1) and aquatic (Figure C-2) herbicides might pose risk to the ecosystem and ecological receptors. The conceptual models indicate the possible exposure

FIGURE C-1. Conceptual Model for Terrestrial Herbicides.



Application of terrestrial herbicides may occur by aerial (i.e., plane, helicopter) or ground (I.e., truck, backpack) methods.

FIGURE C-2. Conceptual Model for Aquatic Herbicides.



Application of aquatic herbicides may occur from a boat or from the shoreline.

pathways for the herbicides, as well as the receptors evaluated for each exposure pathway.

Direct Spray

Plant and wildlife species may be unintentionally impacted during application of either a terrestrial or aquatic herbicide as a result of direct spray of the receptor or the waterbody inhabited by the receptor, indirect contact with dislodgeable foliar residue after herbicide application, or consumption of prey items sprayed during application. These exposures may occur within the application area (e.g., consumption of prey items) or outside of the application area (e.g., terrestrial plants accidentally sprayed during application of aquatic herbicide). Generally, impacts outside of the intended application area are accidental exposures that are not typical of BLM application practices.

Direct Spray of Terrestrial or Aquatic Herbicide on Terrestrial Wildlife

Scenarios involving direct spray of an herbicide consider acute exposures of vertebrate and invertebrate species considered most sensitive to herbicide exposure under laboratory conditions. It was assumed that small mammals are the most sensitive terrestrial vertebrate species, and that mobile pollinating invertebrates that spend time foraging among different plant species (e.g., honeybees) are the most sensitive terrestrial invertebrate receptor. If available literature data for a particular herbicide suggested that other terrestrial vertebrates (e.g., birds) or invertebrates (e.g., earthworms) were more sensitive, the more sensitive receptor was used for this scenario.

The extent of exposure from direct spray of a receptor is based on three variables: the herbicide application rate, the surface area of the receptor species, and the rate of dermal absorption. Both typical and maximum herbicide application rates were evaluated for each herbicide for representative sensitive species. For each receptor it was assumed that exposure occurred over one-half the body surface. The surface area calculation was obtained from the *Wildlife Exposure Factors Handbook* (USEPA 1993a).

Two scenarios were evaluated for the honeybee and small mammal to address the potential differences in absorption. The first case considered 100% absorption (intake through the skin) over 24 hours (i.e., all of the herbicide falling on the receptor was assumed to penetrate the skin). The second scenario considered the absorbed dose over 24 hours assuming first order dermal absorption (i.e., taking into consideration the potential for some herbicide to not be absorbed).

Indirect Contact with Foliage after Direct Spray of Terrestrial or Aquatic Herbicide

Scenarios involving direct spray of an herbicide consider only acute exposures. Foliage that has been sprayed with herbicide may transfer this herbicide to terrestrial animals through dermal contact with dislodgeable foliar residue. However, there is little information available on the potential magnitude of this transfer from plant to animal. Therefore, it was assumed that the amount of herbicide transferred to the animal was $^{1}/_{10}$ the amount the animal received during direct spray scenarios. This assumption was based on the work of Harris and Solomon (1992). It was also assumed that all herbicide transferred to the outside of the animal was completely adsorbed within 24 hours.

Ingestion of Food Items Contaminated by Direct Spray of Terrestrial or Aquatic Herbicide

Scenarios involving ingestion of food items consider both acute and chronic exposures. The terrestrial receptors considered for these scenarios included small and large mammals and small and large birds. Ingestion rates for the species consuming contaminated food items were obtained from field studies based on allometric equations presented in the *Wildlife Exposure Factors Handbook* (USEPA 1993a). It was conservatively assumed that the exposed receptors obtain 100% of their diet from the herbicide contaminated prey items and that 100% of the applied herbicide drifts onto the prey item. Concentrations of the herbicide on vegetation and insects were predicted using individual herbicide application rates and generic residue relationships for different types of vegetation derived by Hoerger and Kenaga (1972). The residue rate for forage crops was used as a surrogate for contaminated insects. Residue rates were not available for small mammals.

Two exposure scenarios were considered for the ingestion of contaminated prey items. The first scenario assumes that the prey item is consumed on the same day it is contaminated with herbicide (no degradation period). Ingested doses for this scenario were compared to acute toxicity endpoints. The second scenario assumes the prey is consumed up through 90 days after the application of the herbicide. Assuming first-order decay rates, the herbicide dose is predicted as a time-weighted average of the herbicide mass on the foliage over the 90-day period. This dose is compared to chronic toxicity endpoints.

Direct Spray of Terrestrial or Aquatic Herbicide on Non-target Terrestrial Plants

In the direct spray scenario, a non-target plant is sprayed during normal application of the terrestrial herbicide. Unintended direct spray of a non-target receptor is considered an accidental exposure scenario that is not typical of BLM application practices. The typical and maximum application rates were used to represent the amount accidentally sprayed on the non-target species. These application rates were directly compared to appropriate toxicity endpoints to determine potential impacts to non-target typical and RTE plants.

Direct Spray of Terrestrial or Aquatic Herbicide onto Pond

The normal application of aquatic herbicides to a pond was considered to evaluate potential impacts to aquatic receptors other than the target plant species. For this scenario, the typical and maximum application rates of the herbicides were applied directly to the pond, and the associated water concentration was calculated based on the pond area and volume. Neither degradation nor sorption of the herbicide to sediments, aquatic vegetation, or suspended solids were considered, resulting in a conservative estimate of the herbicide concentration in pond water. The pond water concentrations were compared against appropriate acute and chronic toxicity endpoints to evaluate potential impacts to fish, aquatic invertebrates, and non-target aquatic plants.

Accidental Direct Spray of Terrestrial or Aquatic Herbicide onto Stream

Aquatic and terrestrial herbicides may be accidentally directly sprayed onto the surface of a stream (stream plants are not targeted with herbicide applications). The typical and maximum application rates of the herbicides were applied directly to the stream, and the associated water concentrations were calculated. Degradation and sorption of the herbicide and transport from flow of the stream were not considered, and therefore, this represents a conservative calculation of the stream water concentration (essentially an instantaneous concentration). The stream concentrations were compared against appropriate acute and chronic toxicity endpoints to evaluate potential impacts to fish, aquatic invertebrates, and non-target aquatic plants.

Off-site Drift

During normal application of herbicides, it is possible for a portion of the herbicide to drift outside of the treatment area and deposit onto non-target receptors. Off-site spray drift and resulting terrestrial deposition rates and waterbody (pond and stream) concentrations were predicted using the computer model AgDRIFT® Version 2.0.05. AgDRIFT® is a product of the Cooperative Research and Development Agreement between the USEPA’s Office of Research and Development and the Spray Drift Task Force (SDTF; a coalition of pesticide registrants). It is based on, and represents an enhancement of, the computer program for agricultural dispersion (AGDISP). AGDISP was developed by the National Aeronautics and Space Administration (NASA), the USDA Forest Service, and the U.S. Army. AgDRIFT® was developed for use in regulatory assessments of off-site drift associated with agricultural use of pesticides through aerial, ground, or orchard/airblast applications. AgDRIFT® is based upon the idea that pesticide or herbicide drift is primarily a function of application technique (e.g., droplet size and release height), environmental conditions, and physical properties of the spray solution, and is not a function of the chemical properties of the a.i. itself. The computational approach employed by AgDRIFT® is based on a method that has evolved over a period of more than 20 years, and yields high correlation with field measurement data sets. The model was selected for use in this risk assessment because of its existing use in regulatory assessments of off-target drift and its suitability to this particular application.

AgDRIFT® enables the user to take a tiered approach to the modeling of drift by allowing the user to choose between three tiers (Tiers I, II, and III) of increasingly complex evaluations of off-target drift and deposition. The basic difference between the three tiers is the number of model input variables the users can change. Further, Tier I supports the evaluation of aerial and ground application scenarios, whereas Tiers II and III only support the evaluation of aerial application scenarios (agricultural and forestry applications). Tier I is based on a set of standard “Good Application Practices,” requires little knowledge of the actual application conditions or herbicide properties, and allows the user to modify a small number of model variables. Tiers II and III are based on the same set of “Good Application Practices;” however Tiers II and III allow the user to modify variables to make the scenario evaluated representative of the conditions under which herbicides will be applied. Tier I was used in the ERAs to evaluate off-site drift associated with ground application scenarios, and Tier II was used to evaluate off-site drift associated with agriculture-like (e.g., rangeland) and forestry application scenarios. The implementation of the Tier I ground and Tier II aerial application model and the model input variables (including the variables specific to the application method and environmental setting and specific to the herbicide being evaluated) are discussed and presented in Appendix A of the Methods document (ENSR 2004).

In accordance with actual BLM herbicide practices, ground application scenarios were modeled using a low- or high-placed boom. Aerial application scenarios were modeled from either a helicopter or a plane at two different heights representing forested or non-forested land types. Drift depositions were estimated at 25, 100, and 900 ft from the application area for ground applications and at 100, 300, and 900 ft for aerial applications. The AgDRIFT® model determined the fraction of the application rate that would be deposited on the off-site location without considering herbicide degradation.

Off-site Drift of Terrestrial Herbicide onto Plants

Surface soil concentrations calculated by AgDRIFT® were directly compared against appropriate toxicity endpoints to determine potential impacts to non-target typical and RTE plants.

Off-site Drift of Terrestrial Herbicide onto Pond or Stream

During normal application, is possible for a portion of the terrestrial herbicide to drift outside of the treatment area. This off-site drift may eventually reach a pond or stream and contaminate the waterbody. AgDRIFT® was used to calculate pond and stream water concentrations of the herbicide in the various application scenarios. The waterbody concentrations calculated by AgDRIFT® do not consider herbicide degradation, sorption, or dissipation, and likely overestimate actual concentrations. AgDRIFT® does consider the dilution of the herbicide in the volume of the pond, but the rate of deposition estimated by AgDRIFT® was diluted into the stream based on the assumed flow rate.

The predicted surface water concentrations in the pond and stream as a result of the various application scenarios were compared to the appropriate acute and chronic toxicity endpoint for each of the three aquatic receptors.

Consumption of Fish from Pond Contaminated by Off-site Drift of Terrestrial Herbicide

Off-site drift of herbicide may eventually reach off-site ponds and contaminate the resident fish population, which may be consumed by piscivorous bird species. In this scenario, impacted pond water is modeled using AgDRIFT® input variables described above. Exposure for the piscivorous bird was evaluated by modeling fish tissue concentrations from pond surface water (C_{pond}) employing bioconcentration factors and food chain multipliers (FCMs) for different trophic levels. Food chain multipliers assumed a trophic level 3 for fish and a trophic level 2 for the prey of fish (e.g., aquatic invertebrates). FCMs were obtained from USEPA (1995a).

The calculated dose to the piscivorous bird is a function of the concentration in the fish tissue, the food ingestion rate in wet weight, the proportion of the diet that is contaminated (assumed to be 100%), and the body weight of the bird. The dose estimate to the piscivorous bird was compared to appropriate chronic toxicity values.

Surface Runoff

Precipitation may result in the transport of herbicide applied to soils from the application area via surface runoff and root-zone groundwater flow. GLEAMS was used in this risk assessment to calculate soil concentrations at the site of application, transport of herbicides to adjacent soils, and the amount of herbicide that might runoff into aquatic habitats (e.g., ponds, streams). One benefit of GLEAMS is the ability to estimate a wide range of potential herbicide exposure concentrations as a function of site-specific parameters, such as soil characteristics and annual precipitation.

GLEAMS Overview

GLEAMS is a modified version of the CREAMS (Chemical Runoff Erosion Assessment Management System) model that was originally developed to evaluate non-point source pollution from field-size areas. Specifically, the hydrology, plant nutrient, and pesticide components of the CREAMS model were modified to consider movement of water and chemicals within and through the root zone. These modifications allow the GLEAMS model to simulate edge-of-field and bottom-of-root-zone loadings of water, sediment, pesticides, and plant nutrients from the complex climate-soil-management interactions. Agricultural pesticides are simulated by GLEAMS using three major components:

- *Hydrology* – considers the effects of precipitation, surface runoff, and percolation through the unsaturated zone of the soil and simulates the effects of vegetation on surface water runoff, infiltration, and evapotranspiration.

- *Erosion* – considers the movement of sediment over the land surface using the Universal Soil Loss Equation (USLE) and pesticide loss associated with particle erosion.

- *Pesticide* – considers chemical-specific characteristics (e.g., soil adsorption, decay) and application methods to determine the amount of herbicide that is available for extraction into surface runoff and/or movement into the soil profile and groundwater.

The GLEAMS model has evolved through several versions from its inception in 1984 to the present, and it has been evaluated in numerous climatic and soil regions around the world. The model was selected for use in this investigation because of its widespread acceptance, its suitability to this particular application, and the previous use of the model to support similar risk assessments for the USDA Forest Service (e.g., SERA 2003).

Data Requirements. The information required for a GLEAMS simulation includes a wide variety of site-specific data to describe the climate, topography, subsurface soils, vegetation type and growing potential, and herbicide-specific properties. The following briefly describes a subset of the data required to successfully simulate the effects of an herbicide on an agricultural site using GLEAMS:

- *Precipitation* – Daily rainfall records for the entire simulation period are required to provide input to the hydrologic simulation. The volume of precipitation strongly influences the amount of runoff and percolation of associated herbicides.

- *Climate* – Daily averages of standard meteorological data are necessary to define precipitation as either rain or snow and to calculate variations in monthly evapotranspiration. Because evapotranspiration is a large component of the hydrologic cycle, the climate (e.g., temperature, humidity) can affect the volume of water moving through the application area.

- *Soil characteristics* – Soil characteristics (as identified by soil type) are applied to the GLEAMS model to facilitate the calculation of runoff and percolation from the application area.

- *Vegetation/ground cover* – Plant growth controls the partitioning of pesticide to either the soil or foliar surfaces and controls the rate of evapotranspiration.

- *Herbicide properties* – The varying distribution of pesticide concentrations predicted by GLEAMS in an agricultural system is largely dependent on the chemical-specific properties used in the model, such as sorption coefficients and decay rates. As these values are herbicide-specific and can vary significantly, concentrations predicted by GLEAMS can be quite different among herbicides.

GLEAMS Model Scenarios

The GLEAMS model was run using a variety of model inputs designed to simulate a broad range of representative environmental conditions. The effect of changing environmental conditions on the export of herbicide from an application area was assessed in two distinct phases:

- *Variable soil type and annual precipitation* – The effects of soil type and cumulative annual precipitation were investigated by developing a single realistic GLEAMS scenario (base case) and then varying these two components. Soil type and precipitation were selected for the first phase of the modeling application because they are the factors most likely to affect the outcome of a simulation. The model was used to calculate herbicide export in environments with the three soil types (sand, loam, and clay) assuming annual precipitation rates of 5, 10, 25, 50, 100, 150, 200, and 250 inches (in). In total, there were 24 simulation combinations in this first phase of the modeling application.

- *Variable physical characteristics* – The effect of varying six physical parameters (soil type, soil erodibility factor, size of application area, hydraulic slope, surface roughness, and vegetation type) was investigated by changing each parameter individually. There were 3 variations for each of the 6 parameters, resulting in 18 simulations in this second phase of the modeling application.

The combination of scenarios included in each of the two phases of GLEAMS modeling produced results for 42 simulations. These simulations provide an indication of the effects of a variety of environmental conditions on the export of herbicide to off-site receptors. These scenarios were used to predict herbicide concentrations in soil and in the surface water of a stream and a pond.

The GLEAMS model predicts daily herbicide export rates. Because conservative assumptions were used in the model, it is likely that the export rates predicted by GLEAMS are high. This is substantiated by the comparison of the GLEAMS export rates modeled here to measured data presented by Lerch and Blanchard (2003), where GLEAMS export rates were higher than measured rates. Details of this comparison are presented in Appendix B of the ERA Methods document (ENSR 2004).

The daily export rates were used to calculate both surface soil and ambient water concentrations. The predicted runoff and percolation rates, and the mass of herbicide associated with each of these exports, were used to determine the amount of herbicide deposited at the edge of the application area.

The soil concentrations were calculated as 52, 7-day average concentrations from the final year of the GLEAMS run (when the model reaches a quasi-steady state). Ambient water concentrations were calculated using GLEAMS model daily predictions of herbicide export rates for acute and chronic exposure scenarios in a river and a pond immediately adjacent to the application area. Acute exposure scenario concentrations were calculated from the maximum 3-day average herbicide export rate from the last year of the simulation. Chronic exposure scenario concentrations were calculated from the daily average herbicide export rate from the last year of the simulation.

The following subsections present a general overview of the calculation of media concentrations in a variety of different terrestrial exposure scenarios (the surface runoff scenarios are not considered relevant for the aquatic herbicides). A more detailed discussion (including assumptions and equations) is presented in Appendix B of the ERA Methods document (ENSR 2004). Each herbicide risk assessment (ENSR 2005a-j) contains an herbicide-specific description of the model outputs.

Surface Runoff of Terrestrial Herbicide to Off-site Soils

The maximum of the 7-day average loadings calculated by GLEAMS was assumed to affect a soil area immediately downslope of the application area. The loading was expressed as a proportion of the total herbicide loading to the application area. For example, if 30% of the applied herbicide was found to run off, the soil concentration off-site was predicted to be 30% of that in the application area. These off-site soil concentrations were compared against appropriate toxicity endpoints to determine potential impacts to non-target typical and RTE plants. This particular exposure pathway may impact seed germination; therefore, toxicity data relevant to seed germination, a sensitive endpoint, were used for evaluation.

Overland Flow of Terrestrial Herbicide to Off-site Pond and Stream

As described previously, precipitation may result in the transport of terrestrial herbicides via surface runoff and root-zone groundwater flow. This overland flow of herbicide applied to soil (via runoff and groundwater) may eventually reach an off-site pond or stream resulting in the contamination of the waterbody. The daily predictions of herbicide export rates from the GLEAMS model were used to calculate ambient water concentrations of herbicide in the various watershed scenarios. GLEAMS considers the subsequent runoff and the natural decay processes that reduce the ambient pond water concentrations over time. Pond concentrations were calculated by assuming a fixed pond volume and a daily inflow of mass and water to the pond depending on recent precipitation, runoff, and percolation characteristics. The GLEAMS exports were used to calculate two pond and two stream water concentrations for comparison against acute and chronic toxicity endpoints for each of the three aquatic receptor groups.

Consumption of Fish from Pond Contaminated by Surface Runoff of Terrestrial Herbicide

Surface runoff containing herbicide bound to soil may eventually reach an off-site pond, and resident fish may uptake herbicide. The fish, in turn, may be consumed by piscivorous bird species. In this scenario, impacted pond water was modeled using the GLEAMS model described above. Since bioaccumulation is a long-term process, the chronic exposure concentration (i.e., the overall average concentration from the final year of the GLEAMS run) was used to predict fish tissue concentrations. A bioconcentration factor and food chain multipliers for different trophic levels were included in the estimate. Food chain multipliers assumed a trophic level 3 for fish and a trophic level 2 for the prey of fish (e.g., aquatic invertebrates). Food chain multipliers were obtained from USEPA (1995a). The calculated dose to the piscivorous bird is a function of the concentration in the fish tissue, the food ingestion rate in wet weight, the proportion of the diet that is contaminated (assumed to be 100%), and the body weight of the bird.

Wind Erosion and Off-site Transport of Terrestrial Herbicide

Dry conditions and wind may also allow transport of herbicide from the application area as wind-blown soil (fugitive dust) onto non-target plants some distance away. This transport due to wind erosion of the surface soil was modeled using the USEPA's guideline air quality CALPUFF air pollutant dispersion model (referenced in Appendix W of 40 CFR Part 51; see the *Air Quality Modeling for BLM Vegetation Treatment Methods* report ["Air Quality Modeling report;" ENSR 2005k] for CALPUFF details and assumptions). CALPUFF "lite" version 5.7 was selected because of its ability to screen potential air quality impacts within and beyond 50 kilometers (km) and its ability to simulate plume trajectory over several hours of transport based on limited meteorological data. Three distinct watersheds were modeled using CALPUFF to determine herbicide concentrations in particulate matter assumed to deposit on plants (i.e., total suspended particulates [TSPs] ranging between 0.1 and 50 micrometers [µm] in diameter, particulate matter [PM] 2.5 µm in diameter and smaller [$PM_{2.5}$], and PM_{10} µm in diameter and smaller). The concentrations were modeled after a wind event, with dust deposition estimates calculated at distances ranging from 1.5 to 100 km from the application area. At each radius considered, the maximum predicted rate of herbicide deposition in a given wind event was calculated. The dust estimates calculated within the model were then compared against the appropriate non-target plant toxicity values.

The dust exposure scenario was not considered for aquatic herbicides.

Source Characterization

A high wind event may cause the surface soil (with the applied herbicide) to migrate from the application area. In this modeled event, CALPUFF determines the rate of herbicide deposition as a function of the rate of dust deposition at the downstream receptor location. It was assumed that all of the applied herbicide was adsorbed by the top 1 mm of soil. The depth of 1 mm is believed to be conservative (thinner affected soil depths result in elevated herbicide emissions during fugitive dust events), and is less than that assumed by others (e.g., SERA [2003] assumed 1 cm). The modeling assumed a square, flat area of 1,000 acres was treated with herbicide applied from the air using a fixed-wing aircraft. For suspended particulate matter, modeled impacts were directly proportional to the modeled emission rate. Therefore, the modeling assumed a unit rate of chemical application/deposition (i.e., 1.0 gram per square meter). The model results can be scaled directly to accommodate varying application rates to bare, undisturbed soil. The modeling results were expressed as the fractional downwind deposition based on this initial application.

Determination of Wind Erosion Event

The CALPUFF model was used to estimate acute exposures. The maximum 1-hour and 24-hour, and annual average, deposition rates from a conservative impact migration event (i.e., an event modeled using very conservative properties such that the potential for dust migration is high) were computed for the distance ranges being modeled (1.5 to 100 km). Although a given area would be sprayed with herbicide only once per year, a full year was modeled to consider a large range of meteorological conditions that could influence the herbicide migration potential for a single event. The highest impact was considered to represent a "reasonable, but conservative" impact under the range of the meteorological conditions tested. For this modeling, there was no initial restriction on the timing of the herbicide application except as noted below.

The herbicide was assumed to adhere to undisturbed surface soil, which can be picked up and transported by sufficiently high winds. The threshold wind speed for such an event is linked to the "friction velocity," which is a measure of the mechanical turbulence at the soil-atmosphere interface, and thus is a good gauge of the ability of the wind to pick up surface particles. Friction velocity increases with increasing wind speed and increasing surface roughness. Threshold friction velocities for undisturbed soils were determined from Gillette (1988), as described in the *Air Quality Modeling for BLM Vegetation Treatment Methods* report (ENSR 2005k). The BLM (Ypsilantis 2003) identified appropriate soil types for each of the "example" modeling analysis locations, as discussed in more detail in the air quality modeling report.

The CALPUFF modeling procedures assumed that, for each modeled hour of the entire year, the friction velocity exceeded the threshold friction velocity for undisturbed soil. A portion of the herbicide spray mass from the 1,000-acre area therefore became airborne, subject to additional conditions listed below. This assumption is conservative because it also assumes that all of the chemical herbicide would be present in the soil at the commencement of a windy event, and that no reduction due to vegetation interception/up-take, leaching, solar or chemical half-life would have occurred since the time of aerial application. However, the use of a full year of meteorology provides a robust procedure to assess the maximum meteorological condition (i.e., the weather most likely to cause dust migration) for short-term impacts.

In addition to the threshold friction velocity requirement for hourly fugitive emissions of windblown soil, other triggering conditions were considered:

- Wet soil adjustment - assumed no hourly particulate matter emissions when there was measurable hourly precipitation (at least 0.01 in).
- Frozen soil adjustment - assumed no hourly particulate matter emissions when the hourly ambient temperature was at/below 28 degrees Fahrenheit.
- Snow cover adjustment - assumed no hourly particulate matter emissions when the hourly snow depth was at least 1 inch.
- Operational adjustment - assumed only one application of chemical herbicide per given 1,000 ac location in the same year.

For these conditions, the surface soil is resistant to movement because it is wet, frozen, and/or covered with an insulating layer of snow. It was assumed that there would be no spraying on a snow-covered surface, although a layer of snow could appear after a spraying event.

Determination of Herbicide Emission Rates

The initial incorporation depth of herbicide (1 mm) determines the concentration of herbicide on eroded dust and defines the depth of erosion at which the mass of herbicide would be exhausted. This mixed depth is based on fast-acting physical processes and does not include leaching of herbicide into the soil due to precipitation.

The mixed layer depth is estimated to account for three processes:

- Physical infiltration of the herbicide into the soil. This is likely a minor factor as little herbicide volume is available to drive infiltration.
- Settling of the herbicide at different depths relative to a given elevation due to uneven soil surface.
- Preferential erosion of fine-grained soils by the wind resulting in segregation of soil particles and mixing of the surface layer.

It was also assumed that there is an even distribution of the herbicide across the soils and that the mass of the herbicide is negligible compared to the mass of the soil. Given a typical soil density of 1 g/cm^3, the mass of a 1-mm depth of soil occupying a square meter is 1,000 grams (g). This represents the total mass of soil per square meter that has to be removed by the wind before all of the herbicide is re-suspended. Using the meteorological data for each site, the mass of soil removed by the wind was calculated for every hour that herbicide re-suspension is possible. The fraction of the herbicide applied to the area that could be released was determined by dividing the mass of soil removed per square meter by 1,000 g (per square meter) for each hour. This percentage was applied at each herbicide's maximum application rate. The resulting value represented the amount of herbicide potentially released each hour, which was assigned to each of the three particle sizes ($PM_{2.5}$, PM_{10}, and TSP) to estimate potential herbicide deposition.

Calculation of Herbicide Deposition

The deposition algorithm in CALPUFF simulated the effective mass distribution of the adsorbed herbicide, based on particulate matter size (small particles have a larger surface area relative to their mass and, therefore, will carry the majority of the herbicide mass). Dispersion modeling estimated the maximum 1-day and 30-day deposition values at each receptor distance. The results were scaled for typical and maximum application rates.

Watersheds Evaluated

Three watersheds were used in the simulation:

- Glasgow International Airport, Glasgow, Montana
- Medford/Jackson County Airport, Medford, Oregon
- Lander/Hunt Field, Lander, Wyoming

These locations were selected as representative of various regions of the western states addressed by the PEIS. For each location, 1 year of surface meteorological data from the Solar and Meteorological Surface Observation Network (SAMSON) data set that has been produced by National Climatic Data Center (NCDC) was used. After a review of available data capture, the most recent SAMSON year with complete surface and mixing height data was selected for each station. The SAMSON data set is particularly applicable for CALPUFF modeling because it contains hourly values of relative humidity and solar radiation, which are needed for chemical transformation calculations. Mixing height data for these sites were obtained from the USEPA's Technology Transfer Network Support Center for Regulatory Air Models.[2] The highest impact was considered to represent a reasonable, but conservative impact under the range of meteorological conditions tested.

Further details about the CALPUFF model inputs and assumptions can be found in the air quality modeling report (ENSR 2005k).

Accidental Spill

Two spill scenarios were modeled to represent worst-case potential impacts to the pond. The scenarios included a truck or a helicopter spilling entire loads (200 gallon [gal] spill and 140 gal spill, respectively) of herbicide mixed for the maximum application rate into the ¼ acre, 1-meter-deep pond. To represent an acute exposure event for the three types of aquatic receptors, the pond concentration was compared against the appropriate toxicity endpoint for the fish, aquatic invertebrates, and non-target aquatic plants.

The concentration of herbicide in the pond water is based on the concentration in the spilled solution, the volume spilled, and the volume of the pond, assuming instantaneous mixing.

Non-target Species Effects Characterization

This section summarizes the toxicity of the 10 herbicides to terrestrial and aquatic species found on public lands. There are eight terrestrial herbicide a.i./formulations (bromacil, chlorsulfuron, diflufenzopyr, diuron, imazapic, Overdrive®, sulfometuron methyl, and tebuthiuron) and two aquatic herbicide a.i. (diquat and fluridone). The effects these herbicide active ingredients have on non-target species were evaluated for different categories of plants and animals (via surrogate species). The categories of terrestrial species evaluated include terrestrial plants, mammals, birds, reptiles, amphibians, and insects; the categories of aquatic species evaluated include aquatic plants, fish, aquatic invertebrates, and amphibians (in aquatic life stages).

This effects characterization section identifies the TRVs selected from detailed laboratory and field toxicity studies evaluating the acute and chronic impacts of proposed herbicide usage on non-target plants and animals. TRVs were chosen as the lowest reported value for a given type of herbicide exposure (e.g., acute dermal exposure). For a given surrogate species, if an herbicide toxicity level exceeds the chosen TRV, that species and those it represents are at risk of experiencing adverse effects as a result of herbicide application.

[2] http://www.epa.gov/ttn/scram/

Terrestrial Species Effects Characterization

The TRVs representing the effects of each of the 10 herbicides on terrestrial species are presented below and are summarized by receptor type in Tables C-5 to C-15.

Bromacil

Bromacil poses a low toxicity hazard to terrestrial animals (mammals, birds, and honeybees [USEPA 1996]). However, terrestrial plants are sensitive to bromacil, with concentrations as low as 0.0023 pounds (lbs) a.i./ac affecting the growth of non-target plants (about 0.06% of the typical application rate).

Mammals

Based on USEPA re-registration documents (USEPA 1996), bromacil is considered to pose a low to moderate acute oral and dermal toxicity hazard to mammals. The oral LD_{50} (641 milligram [mg] a.i./kilogram [kg] BW) and chronic dietary NOAEL (13.3 mg a.i./kg BW-day) were selected as the dietary small mammal TRVs. The dermal small mammal TRV was established at >5,000 mg a.i./kg BW. Because no large mammal LD_{50}s were identified in the available literature, the small mammal LD_{50} was used as a surrogate value. The large mammal dietary NOAEL TRV was established at 4.65 mg a.i./kg BW-day.

Birds

Data from the available literature indicate that bromacil has low toxicity to birds. The bobwhite quail dietary LD_{50} (>30,195 mg/kg BW) and chronic NOAEL (936 mg/kg BW-day) were selected as the small bird dietary TRVs. The mallard dietary LD_{50} (>5,000 mg/kg BW) and NOAEL (155 mg/kg BW-day) were selected as the large bird dietary TRVs.

Terrestrial Invertebrates

Bromacil poses practically no toxicity risk to invertebrates. The honeybee dermal LD_{50} TRV was set at 193 microgram (μg)/bee. Based on a honeybee weight of 0.093 g, this TRV was expressed as 2,075 mg/kg BW.

Terrestrial Plants

Terrestrial plants are at high risk from exposure to bromacil. The lowest and highest germination-based NOAELs were selected to evaluate risk to terrestrial plants in surface runoff scenarios. Emergence endpoints were used when germination data were unavailable. These TRVs were 0.0117 and 0.188 lb a.i./ac. Two additional endpoints were used to evaluate other plant scenarios. These included an EC_{25} of 0.0023 lb a.i./ac and a NOAEL of 0.008 lb a.i./ac (extrapolated from the EC_{25} by dividing by an uncertainty factor of 3).

Chlorsulfuron

Chlorsulfuron poses little to no acute toxicity hazard to mammals via dermal and oral exposure. Adverse effects to small mammals have been documented from long-term dietary exposure to chlorsulfuron. Chlorsulfuron also has low toxicity to birds and slight toxicity to honeybees.

Adverse effects to non-target terrestrial plants occurred at concentrations as low as 0.047 lb a.i./ac.

Mammals

Based on USEPA re-registration documents (USEPA 2002), chlorsulfuron is characterized as not acutely toxic to mammals via dermal and oral exposure routes. The oral LD_{50} (1,363 mg a.i./kg BW) and chronic dietary NOAEL (5 mg a.i./kg BW-day) were selected as the dietary small mammal TRVs. The dermal small mammal TRV was established at >3,400 mg a.i./kg BW. Because no large mammal LD_{50}s were identified in the available literature, the small mammal LD_{50} was used as a surrogate value. The large mammal dietary NOAEL TRV was established at 65.6 mg a.i./kg BW-day.

Birds

Data from the available literature indicate that chlorsulfuron has low toxicity to birds. The bobwhite quail dietary LD_{50} (>16,970 mg/kg BW) and chronic NOAEL (100 mg/kg BW-day) were selected as the small bird dietary TRVs. The mallard dietary LD_{50} (>1,500 mg/kg BW) and NOAEL (99 mg/kg BW-day) were selected as the large bird dietary TRVs.

Terrestrial Invertebrates

Chlorsulfuron poses a slight toxicity risk to invertebrates. The honeybee dermal LD_{50} TRV was set at >25 μg/bee. Based on a honeybee weight of 0.093 g, this TRV was expressed as 269 mg/kg BW.

Terrestrial Plants

Chlorsulfuron is very highly toxic to plants. The lowest and highest germination-based NOAELs were selected to evaluate risk in surface runoff scenarios. These TRVs were 0.0157 and 0.0052 lb a.i./ac, based on the unverified dyer's woad germination study (EC_{100} divided by uncertainty factors of 3 and 9, respectively). Two additional endpoints were used to evaluate other plant scenarios. These included a life-cycle NOAEL of 0.000021 lb a.i./ac and an EC_{25} of 0.000063 lb a.i./ac (extrapolated from the NOAEL by multiplying by an uncertainty factor of 3).

Diflufenzopyr

As defined by the USEPA, diflufenzopyr alone poses little to no acute toxicity hazard to mammals via dermal and oral exposure. Adverse effects to small mammals have been documented from long-term dietary exposure to technical grade diflufenzopyr. Diflufenzopyr is practically non-toxic to birds and causes slight toxicity to honeybees. However, adverse effects to non-target terrestrial plant species have occurred at concentrations as low as 0.0008 lbs. a.i./ac, which is approximately 1/100 of the typical application rate.

Mammals

Based on USEPA conditional registration documents (USEPA 1999), diflufenzopyr is characterized as having low toxicity to small mammals. The oral LD_{50} (3,300 mg/kg BW) and chronic dietary NOAEL (42.2 mg/kg BW-day) were selected as the dietary small mammal TRVs. The dermal small mammal TRV was established at >5,000 mg/kg BW. Since no large mammal LD_{50}s were identified in the available literature, the small mammal LD_{50} was used as a surrogate value. The large mammal dietary NOAEL TRV was established at 59 mg/kg BW-day.

Birds

Data from the available literature indicate that diflufenzopyr has low toxicity to birds. The small bird chronic dietary NOAEL was set at 634 mg/kg BW-day. Since an acute adverse effect level was not established in the literature, the NOAEL was multiplied by an uncertainty factor of 3, resulting in a small bird dietary LD_{50} of 16,970 mg/kg BW. Similarly, the large bird dietary NOAEL TRV was set at 105 mg/kg BW-day, and using an uncertainty factor of 3, the LD_{50} was estimated to be >2,810 mg/kg BW.

Terrestrial Invertebrates

A honeybee dermal toxicity test suggests that diflufenzopyr has low toxicity to terrestrial invertebrates. Because a suitable LD_{50} could not be determined from the literature, the NOAEL was multiplied by an uncertainty factor of 3. The resulting honeybee dermal LD_{50} TRV was calculated to be 75 μg/bee. Based on a honeybee weight of 0.093 g, this TRV was expressed as 806 mg/kg BW.

Terrestrial Plants

Terrestrial plants appear to be at high risk of toxic effects from application of diflufenzopyr. The lowest and highest germination-based NOAELs were selected to evaluate risk in surface runoff scenarios of the risk assessment (TRVs = 0.028 and 0.0001 lb a.i./ac). Emergence endpoints were used when germination data were unavailable. Two additional endpoints were used to evaluate other plant scenarios: an EC_{25} of 0.0008 lb a.i./ac and a NOAEL of 0.0003 lb a.i./ac (extrapolated from the EC_{25} by dividing by an uncertainty factor of 3).

Diquat

Diquat is moderately toxic to mammals, particularly via dermal exposure. Diquat is also moderately toxic to birds and honeybees. In addition, adverse effects to non-target terrestrial plant species occurred with exposure to low concentrations of diquat (0.0046 lbs a.i./acre; 0.5% of the typical application rate).

Mammals

According to USEPA re-registration eligibility documents (USEPA 1995b), diquat is considered to be moderately toxic to mammals. Based on these findings, the oral LD_{50} (121 mg/kg BW) and chronic dietary NOAEL (0.8 mg a.i./kg BW-day) were selected as the dietary small mammal TRVs. The dermal small mammal TRV was established at 262 mg a.i./kg BW. Since no large mammal LD_{50}s were identified in the available literature, the small mammal LD_{50} was used as a surrogate value. The large mammal dietary NOAEL TRV was established at 0.5 mg a.i./kg BW-day.

Birds

Data from the literature indicate that diquat has moderate toxicity to birds. The bobwhite quail dietary LD_{50} (150 mg a.i./kg BW) and chronic NOAEL (12 mg a.i./kg BW-day) were selected as the small bird dietary TRVs. The pheasant dietary LD_{50} (215 mg a.i./kg BW) and the mallard dietary chronic NOAEL (0.6 mg a.i./kg BW-day) were selected as the large bird dietary TRVs.

Terrestrial Invertebrates

A dermal toxicity study in honeybees suggests that diquat has low toxicity to terrestrial invertebrates. The honeybee dermal TRV was set at 47 μg/bee, the 5-day LD_{50} value. Based on a honeybee weight of 0.093 g, this TRV was expressed as 505 mg/kg BW.

Terrestrial Plants

Terrestrial plants appear to be at high risk of toxic effects from exposure to diquat. Two endpoints were used to evaluate terrestrial plant scenarios for aquatic herbicides. These included an EC_{25} and a NOAEL. Since the lowest EC_{25} identified in the database (0.047 lb a.i./ac) was lower than the lowest reported NOAEL, the terrestrial plant NOAEL TRV (0.0016 lb a.i./ac) was calculated by dividing the EC_{25} value by an uncertainty factor of 3.

Diuron

Diuron is not considered to be highly toxic to most terrestrial species. In mammals, diuron is considered to have low acute oral and dermal toxicity. However, adverse effects have been demonstrated in mammals from long-term exposure to diuron in the diet. Diuron is slightly toxic to birds but essentially non-toxic to honeybees. Significant adverse effects were noted in non-target terrestrial plant species after 14 days exposure to concentrations as low as 0.08 lb a.i./ac.

Mammals

Because fairly large single doses of diuron are required before adverse effects are noted, diuron is considered to have low acute toxicity to mammals, but diuron does have moderate chronic toxicity to mammals. The oral LD_{50} (1,017 mg a.i./kg BW) and chronic dietary NOAEL (2.5 mg a.i./kg BW-day) were selected as the dietary small mammal TRVs. The dermal small mammal TRV was established at >2,500 mg a.i./kg BW. Since no large mammal LD_{50}s were identified in the available literature, the small mammal acute LD_{50} (1,017 mg a.i./kg BW-day) was used as a surrogate value. The large mammal dietary chronic NOAEL TRV was established at 0.6 mg a.i./kg BW-day.

Birds

Data from available literature indicate that diuron has low toxicity to birds (toxicity is higher for large birds than small birds). The bobwhite quail dietary LD_{50} (5,225 mg/kg BW-day) and the mallard dietary LD_{50} (865 mg/kg BW-day) were selected as the small and large bird dietary TRVs. Since NOAEL values for small and large birds were unavailable, the LD_{50}s were divided by an uncertainty factor of 3 to derive NOAEL TRVs of 348 and 58 mg/kg BW-day for small and large birds, respectively.

Terrestrial Invertebrates

A dermal toxicity study in honeybees suggests that diuron has low toxicity to terrestrial invertebrates. The honeybee dermal LD_{50} TRV was set at 145.03 μg/bee. Based on a honeybee weight of 0.093 g, this TRV was expressed as 1,560 mg/kg BW.

Terrestrial Plants

Terrestrial plants appear to be at high risk of toxic effects from diuron application. The lowest and highest germination-based NOAELs were selected to evaluate risk in surface runoff scenarios. However, because germination data were not available for diuron, the emergence TRVs of 0.047 and 12 lb a.i./ac were selected instead. Two additional endpoints were used to evaluate other plant scenarios. These included a seed emergence EC_{25} of 0.08 lb a.i./ac and a vegetative vigor NOAEL of 0.001 lb a.i./ac.

Fluridone

Fluridone has low toxicity to most terrestrial animals. Studies conducted with mammals found that acute exposure to fluridone usually does not cause adverse effects, even to mammals that were exposed to fluridone for longer periods of time or during pregnancy. Similarly, short-term exposure to fluridone did not result in adverse effects in birds, even at high exposure levels. Long-term exposure to fluridone did result in reduced growth in large and small birds. Fluridone was practically non-toxic to honeybees. While no quantitative data were found to evaluate fluridone's effects on terrestrial plants, the manufacturer's user guide (Eli Lilly and Company 2003) provided qualitative results indicating that the sensitivity of terrestrial plants is variable. Some species (e.g., grasses and sedges) were more sensitive than other plant species (e.g., willow).

Mammals

Fluridone poses little acute risk to small mammals. The oral LD_{50} (>10,000 mg a.i./kg BW) and chronic dietary NOAEL (8 mg a.i./kg BW-day) were selected as the dietary small mammal TRVs. The dermal small mammal TRV was established at >5,000 mg a.i./kg BW. Since no large mammal LD_{50}s were identified in the available literature, the small mammal LD_{50} (>10,000 mg a.i./kg BW) was used as a surrogate value. The large mammal dietary NOAEL TRV was established at 75 mg a.i./kg BW-day. Overall, acute exposure to fluridone causes few adverse effects to mammals, but adverse effects can occur if mammals are chronically exposed to fluridone. Small mammals may be slightly more susceptible to fluridone than large mammals.

Birds

Information related to avian exposure to fluridone suggests that acute oral exposure to fluridone is practically non-toxic to birds. The bobwhite quail dietary LD_{50} (>13,135 mg/kg BW) and chronic NOAEL (604 mg a.i./kg BW-day) were selected as the small bird dietary TRVs. The mallard dietary LD_{50} (>2,270 mg/kg BW) and NOAEL (100 mg a.i./kg BW-day) were selected as the large bird dietary TRVs.

Terrestrial Invertebrates

Honeybee dermal toxicity studies indicate that fluridone has low toxicity to terrestrial invertebrates. Because an LD_{50} was not established in the literature, the NOAEL was multiplied by an uncertainty factor of 3, resulting in an LD_{50} of 1,088 μg/bee. Based on a honeybee weight of 0.093 g, this TRV was expressed as 11,699 mg a.i./kg BW.

Terrestrial Plants

No quantitative toxicity studies were found in the reviewed literature that addressed toxicity of fluridone to terrestrial plants. In the manufacturer's user's guide (Eli Lilly and Company 2003), grasses and some sedges are considered to be "sensitive" or "intermediate" in their tolerance to the Sonar herbicide, while rushes tend to be "intermediate" to "tolerant." Shoreline plants, such as willow and cypress, were considered "tolerant," while the tolerance of members of the evening primrose and acanthus families was classified as "intermediate."

Imazapic

The information identified during the literature review indicates that imazapic is not highly toxic to most terrestrial animal species, although it is fairly toxic to non-target terrestrial plant species. Since the herbicide is rapidly metabolized and excreted in urine and feces, imazapic does not bioaccumulate in animals. In mammals, pesticide registration studies found that exposure to imazapic does not frequently cause adverse effects, even at relatively high dose levels. Nevertheless, mammals may be more susceptible during pregnancy, and large mammals may be slightly more sensitive to imazapic than small mammals. During short-term acute exposures, imazapic did not cause adverse effects in birds; however, long-term exposure to imazapic did result in reduced growth in large and small birds. For terrestrial plants, significant adverse effects were noted in non-target plant species after 14 days exposure to concentrations as low as 0.01 lb a.i./ac (approximately 1/3 of the typical application rate).

Mammals

Included in the registration reports were acute oral toxicity studies conducted in rats that demonstrated that exposure to imazapic typically does not cause adverse effects, even at relatively high dose levels (e.g., >5,000 mg a.i./kg BW; SERA 2001). Based on these findings, the oral LD_{50} (>5,000 mg a.i./kg BW) and chronic dietary NOAEL (1,728 mg/kg BW-day) were selected as the dietary small mammal TRVs. The dermal small mammal TRV (LD_{50}) was established at >2,000 mg/kg BW. Because a NOAEL was not identified in the available literature, it was calculated by dividing the LOAEL (137 mg/kg BW-day) by an uncertainty factor of 3, resulting in a large mammal dietary NOAEL TRV of 46 mg/kg BW-day. Overall, exposure to imazapic causes few adverse effects to mammals under most circumstances, even at high concentrations. However, large mammals may be more susceptible to imazapic than small mammals, and mammals may be more susceptible to imazapic during pregnancy.

Birds

Based on available data, imazapic appears to have low toxicity to birds. No adverse effects were observed in bobwhite quail administered imazapic at dose levels as high as 2,150 mg/kg BW for 21 days (USEPA 2003). The bobwhite quail dietary LD_{50} (>15,095 mg/kg BW) and chronic NOAEL (113 mg/kg BW-day) were selected as the small bird dietary TRVs. The mallard dietary LD_{50} (>2,500 mg/kg BW) and NOAEL (65 mg/kg BW-day) were selected as the large bird dietary TRVs.

Terrestrial Invertebrates

A dermal toxicity study in honeybees suggested that imazapic poses low toxicity hazard to terrestrial invertebrates. The honeybee dermal LD_{50} TRV was set at >100 μg/bee. Based on a honeybee weight of 0.093 g, this TRV was expressed as 1,075 mg/kg BW.

Terrestrial Plants

Imazapic appears to be highly toxic to terrestrial plants. The lowest and highest germination-based NOAELs were selected to evaluate risk in surface runoff scenarios. These TRVs were 0.064 and 0.032 lb a.i./acre. Two additional endpoints were used to evaluate other plant scenarios. These included an EC_{25} of 0.01 lb a.i./acre and an NOAEL of 0.008 lb a.i./acre.

Overdrive®

Overdrive® is a formulation containing dicamba and diflufenzopyr. Because Overdrive® is a recently approved herbicide, no Overdrive® toxicity data were identified. However, the herbicide Distinct® contains the same ratio of dicamba and diflufenzopyr, and several Distinct® studies were identified in the literature review. Therefore, Distinct®, dicamba, and diflufenzopyr toxicity data were examined to evaluate the toxicity of Overdrive® to receptor species.

Diflufenzopyr poses little to no acute toxicity hazard to mammals via dermal and oral exposure, Distinct® herbicide poses a slight toxicity hazard to mammals, and dicamba is considered to be slightly toxic to mammals via dermal and oral exposures. Adverse effects to small mammals have been documented from long-term dietary exposure to technical grade diflufenzopyr. Long term exposures to dicamba did not show significant mortality, reproductive, or teratogenic effects at the tested levels (up to 25 mg/kg/day).

Diflufenzopyr and dicamba are considered practically non-toxic to birds. Diflufenzopyr causes slight toxicity to honeybees, but dicamba is considered non-toxic to honeybees. For terrestrial plants, adverse effects to non-target species occurred at diflufenzopyr concentrations as low as 0.0008 lbs. a.i./acre, at dicamba concentrations as low as 0.00027 lb a.i./ac, and at Distinct® concentrations as low as 0.0043 lb a.i./ac.

Diflufenzopyr is moderately toxic to fish and aquatic invertebrates, while dicamba has low toxicity to aquatic organisms. Diflufenzopyr was toxic to aquatic macrophytes, specifically duckweed, with Distinct® being more toxic and dicamba being less toxic.

Mammals

Diflufenzopyr poses little to no acute toxicity hazard to mammals via dermal and oral exposure, but may pose chronic risk. Distinct® and dicamba pose slight toxicity hazards to mammals via dermal and oral exposures. The dermal small mammal TRVs were established at >5,000 mg/kg BW for diflufenzopyr and Distinct®, and >5,050 mg/kg BW for dicamba. The dietary small mammal diflufenzopyr TRV based on the oral LD_{50} was 3,300 mg/kg BW for diflufenzopyr. The dietary small mammal TRV based on the oral LD_{50} was 566 mg/kg BW for dicamba. The dietary small mammal TRV based on the oral LD_{50} was 1,600 mg/kg BW for Distinct®.

Based on the NOAEL, the chronic dietary small mammal TRV was established at 42.2 mg/kg BW-day for diflufenzopyr and at 3 mg/kg BW-day for dicamba. No small mammal chronic studies were reported for Distinct® or Overdrive®, and therefore, no TRV could be developed.

Because no large mammal LD_{50}s for diflufenzopyr, dicamba, or Distinct® were identified in the available literature, the small mammal LD_{50} was used as surrogate values. In addition, no large mammal chronic toxicity data were identified for Distinct® or Overdrive®, and consequently no TRV could be developed. Based on the available data, the large mammal dietary NOAEL TRV for diflufenzopyr was established at 59 mg/kg BW-day, and the chronic large mammal dietary TRV was established at 0.15 mg/kg BW-day for dicamba.

Birds

Data from the available literature indicate that diflufenzopyr has low toxicity to birds. The diflufenzopyr acute small bird dietary LD_{50} TRV was set at >16,970 mg/kg BW based on the bobwhite quail, and the acute large bird dietary LD_{50} TRV was set at >2,810 mg/kg BW. The diflufenzopyr chronic small bird dietary NOAEL was set at 634 mg/kg BW-day, based on the bobwhite quail, and the large bird

NOAEL was set at 105 mg/kg BW-day, based on the mallard.

Dicamba is classified as practically non-toxic to birds. The dicamba acute small bird dietary LD_{50} was set at >30,190 mg/kg BW, based on the bobwhite quail, and the large bird LD_{50} was set at >5,000 mg/kg BW, based on the mallard. The dicamba chronic small bird dietary NOAEL was set at 170 mg/kg BW-day, based on the bobwhite quail, and the large bird NOAEL was set at 92 mg/kg BW-day, based on the mallard.

Only one acute study was identified for Distinct®. The Distinct® acute small bird dietary LD_{50} was set at >18,360 mg/kg BW, based on the bobwhite quail. Because no chronic data were available, the 8-day NOAEL, 3,672 mg/kg BW-day, was used as the small bird NOAEL TRV. Due to a lack of additional data, no large bird TRVs were derived.

Terrestrial Invertebrates

Diflufenzopyr and dicamba appear to be practically non-toxic to terrestrial invertebrates, as represented by the honeybee. The honeybee dermal LD_{50} for diflufenzopyr was calculated to be 806 mg/kg BW. For dicamba, the LD_{50} value was calculated as 974 mg/kg BW. No honeybee data were identified for Distinct®.

Terrestrial Plants

Because germination data were unavailable, the lowest and highest emergence-based NOAELs were selected to evaluate risk in surface runoff scenarios of the risk assessment. The diflufenzopyr TRVs were 0.0001 and 0.028 lb a.i./acre. Two additional endpoints were used to evaluate other plant scenarios. These included an EC_{25} of 0.0008 lb a.i./acre and a NOAEL of 0.0003 lb a.i./acre.

The dicamba TRVs were <0.0022 and 0.53 lb a.i./ac. To evaluate other plant scenarios, two additional endpoints were used. These included the lowest dicamba EC_{25} of 0.00027 lb a.i./ac and the highest NOAEL that was still below the selected $EC_{2.5}$. The only NOAEL that met this criteria was the <0.0022 lb a.i./ac germination value.

The Distinct® TRVs were 0.0016 and 0.046 lb a.i./ac. To evaluate other plant scenarios, two additional endpoints were used. These included the lowest Distinct® EC_{25} of 0.0043 lb a.i./ac and the highest NOAEL that was still below the selected EC_{25} of 0.004 lb a.i./ac for vegetative vigor in tomatoes (USEPA 2003; Master Record Identifier (MRID) number 45047301).

Sulfometuron Methyl

Sulfometuron methyl has low toxicity to most terrestrial species. In mammals, sulfometuron methyl is considered to have low acute oral and dermal toxicity. However, adverse effects were demonstrated in mammals from long-term exposure to sulfometuron methyl in the diet or via oral gavage during pregnancy. Sulfometuron methyl is essentially non-toxic to birds and honeybees. There appears to be little difference in the high sensitivities of weeds and non-target plants to sulfometuron methyl. Pine species are less sensitive than broadleaves or grasses. Rare, threatened, and endangered species do appear to be particularly sensitive to sulfometuron methyl.

Mammals

Sulfometuron methyl is considered to have low acute toxicity to mammals, but moderate chronic toxicity. The oral LD_{50} (>5,000 mg a.i./kg BW) and the chronic dietary NOAEL (18 mg a.i./kg BW-day) were selected as the dietary small mammal TRVs. The dermal small mammal TRV was established at >8,000 mg a.i./kg BW. Because no large mammal LD_{50}s were identified in the available literature, the small mammal LD_{50} was used as a surrogate value, and the large mammal chronic dietary NOAEL TRV was established at 28 mg a.i./kg BW-day.

Birds

In the studies evaluated, no adverse effects have been demonstrated in birds exposed to sulfometuron methyl. The bobwhite quail dietary LD_{50} (>16,970 mg a.i./kg BW-day) and extrapolated NOAEL (1,131 mg a.i./kg BW-day) were selected as the small bird dietary TRVs. The mallard dietary LD_{50} (>2,300 mg a.i./kg BW-day) and extrapolated NOAEL (153 mg a.i./kg BW-day) were selected as the large bird dietary TRVs.

Terrestrial Invertebrates

A honeybee dermal toxicity study suggests that sulfometuron methyl is of low toxicity to terrestrial invertebrates. The honeybee dermal LD_{50} TRV was set at 300 μg/bee (extrapolated from the NOAEL). Based on a honeybee weight of 0.093 g, this TRV was expressed as 3,226 mg/kg BW.

Terrestrial Plants

Sulfometuron methyl appears to be highly toxic to terrestrial plants. The lowest and highest germination-based NOAELs were selected to evaluate risk in surface runoff scenarios of the risk assessment. These terrestrial plant TRVs were established as 0.000028 and 1.12 lb a.i./ac, based on emergence data. Two additional endpoints were used to evaluate other plant scenarios; these included an EC_{25} of 0.22 lb a.i./ac BW and a NOAEL of 0.000028 lb a.i./ac.

Tebuthiuron

Tebuthiuron has moderate toxicity to most terrestrial species. In mammals, tebuthiuron is considered to have low acute dermal toxicity, but adverse effects can occur when organisms are exposed for greater periods of time (e.g., via diet or oral gavage). Tebuthiuron is essentially non-toxic to birds and slightly toxic to honeybees. Tests conducted on crop plant species found adverse effects at concentrations as low as 0.03 lbs a.i./ac (6% of the typical application rate).

Mammals

Tebuthiuron is considered to have moderate toxicity to mammals. The oral LD_{50} (58 mg a.i./kg BW) and chronic dietary NOAEL (7 mg a.i./kg BW-day) were selected as the dietary small mammal TRVs. The dermal small mammal TRV was established at >5,000 mg a.i./kg BW. The large mammal dietary LD_{50} was established at >500 mg a.i./kg BW-day, and the NOAEL TRV was established at 12.5 mg a.i./kg BW-day.

Birds

In the studies evaluated, no adverse effects were reported in birds exposed to tebuthiuron. The small bird dietary LD_{50} was established at >15,440 mg a.i./kg BW, based on the bobwhite quail study. A small bird dietary NOAEL value was calculated by dividing the daily dose by an uncertainty factor of 3. The resulting NOAEL was 1,029 mg a.i./kg BW-day. The large bird dietary LD_{50} was established at >2,545 mg a.i./kg BW-day, based on the mallard duck. The large bird NOAEL was established at 1,000 mg a.i./kg BW-day, based on hens.

Terrestrial Invertebrates

Based on a honeybee dermal toxicity study, tebuthiuron appears to have low toxicity to terrestrial invertebrates. The honeybee dermal LD_{50} TRV was set at 30 μg a.i./bee. Based on a honeybee weight of 0.093 g, this TRV was expressed as 323 mg a.i./kg BW.

Terrestrial Plants

Available data suggest that tebuthiuron has high toxicity to terrestrial plants. The NOAEL TRVs were established at >6 and 0.01 lb a.i./ac (extrapolated from the EC_{25} of 0.03 lb a.i./ac), based on germination and emergence data, respectively. Two additional endpoints were used to evaluate other plant scenarios.

Aquatic Species Effects Characterization

This aquatic effects characterization section summarizes the acute and chronic toxicity study results demonstrating the impacts of proposed herbicide usage on non-target aquatic plants, aquatic invertebrates, fish, and aquatic life stages of amphibians.

The acute toxicity levels of the herbicides are classified according to the observed LC_{50} values:

- less than 0.1 mg/L – very highly toxic
- 0.1 to 1 mg /L – highly toxic
- 1 to 10 mg/L – moderately toxic
- 10 to 100 mg/L – slightly toxic
- greater than 100 mg/L – practically non-toxic

Bromacil

Bromacil is slightly toxic to effectively non-toxic to most aquatic animals. For fish, acute toxic effects of bromacil occurred at concentrations of 36 mg/L, and coldwater fish species appear to be slightly more sensitive to bromacil than warmwater species. Also, bromacil does not tend to bioconcentrate appreciably in fish tissue. Compared to fish, aquatic invertebrates are less sensitive to acute bromacil exposures, with acute adverse effect concentrations occurring at 65 mg a.i./L. In contrast, growth of the green algae, *Selenastrum capricornutum*, was adversely impacted by bromacil concentrations as low as 0.0068 mg/L. No acceptable toxicity studies were found for amphibians.

Fish

Bromacil is slightly toxic to fish. The coldwater 96-hour LC_{50} of 36 mg/L was selected as the acute TRV (the lower of the cold- and warmwater fish endpoints were selected as the TRVs for fish), and the warmwater fish NOAEL of 0.33 mg /L was used as the TRV for chronic effects.

Amphibians

Bromacil is slightly toxic to amphibians. The LC_{50} (230 mg/L) was selected as an amphibian acute TRV. The NOAEL was extrapolated from the LC_{50} using an uncertainty factor of 3. The resulting NOAEL TRV was 77 mg/L.

Aquatic Invertebrates

Bromacil is also slightly toxic to aquatic invertebrates. The LC_{50} (65 mg a.i./L) was selected as the invertebrate acute TRV. Since no NOAEL value in the reviewed literature was lower than the LC_{50}, the LC_{50} was divided by an uncertainty factor of 3 to estimate a NOAEL TRV of 22 mg a.i./L. It may be noted that the use of this NOAEL TRV to evaluate chronic scenarios is conservative, as it is based on a short-term study, but not a chronic study.

Aquatic Plants

Bromacil is very highly toxic to aquatic plants. The EC_{50} (0.0068 mg/L) was selected as the aquatic plant acute TRV. Because no NOAEL values in the reviewed literature were lower than the EC_{50}, the EC_{50} was divided by an uncertainty factor of 3 to estimate a NOAEL TRV of 0.0023 mg /L. It may be noted that the use of this NOAEL TRV to evaluate chronic scenarios is conservative, as it is based on a short-term study, but not a chronic study.

Chlorsulfuron

Chlorsulfuron is slightly toxic to fish and aquatic invertebrates. No toxicity studies conducted on amphibian species were found in the literature reviewed. Chlorsulfuron is very highly toxic to aquatic macrophytes. Aquatic macrophytes are adversely affected by concentrations as low as 0.00007 mg/L.

Fish

Chlorsulfuron is slightly toxic to fish. The coldwater 96-hour LC_{50} of 40 mg/L was selected as the acute TRV (the lower of the coldwater and warmwater fish endpoints). The warmwater fish NOAEL (17 mg /L; extrapolated from the LC_{50}) was used as the TRV for chronic effects (coldwater and warmwater fish species may have comparable sensitivity to chlorsulfuron). Chlorsulfuron is not likely to bioconcentrate in fish tissue. It may be noted that the use of this NOAEL TRV to evaluate chronic scenarios is conservative, as it is based on a short-term study, but not a chronic study.

Aquatic Invertebrates

Chlorsulfuron is practically non-toxic to aquatic invertebrates. The LC_{50} (368.9 mg/L) was selected as the invertebrate acute TRV and the 21-day NOAEL (20 mg/L) was selected as the chronic TRV.

Aquatic Plants

Chlorsulfuron is very highly toxic to aquatic plants. The EC_{50} (0.00007 mg a.i./L) was selected as the aquatic plant acute TRV. The highest NOAEL below the acute TRV was the NOAEL from the same study (0.004 mg a.i./L).

Diflufenzopyr

Diflufenzopyr is moderately toxic to fish and aquatic invertebrates. No toxicity studies conducted on amphibian species were found in the literature reviewed. Diflufenzopyr is also toxic to aquatic macrophytes, which are adversely affected by diflufenzopyr and its various formulations at concentrations as low as 0.0078 mg/L. There do not appear to be appreciable differences in sensitivities among aquatic macrophytes, diatoms, and algae.

Fish

Results from coldwater and warmwater fish species suggest that diflufenzopyr has relatively low toxicity to fish species. The lower of the coldwater and warmwater fish endpoints were selected as the TRVs for fish. Therefore, the coldwater 96-hour LC_{50} of 106 mg/L was selected as the acute TRV, and the warmwater fish NOAEL of 16 mg/L was used as the TRV for chronic effects.

Aquatic Invertebrates

Based on toxicity studies in daphnids, diflufenzopyr appears to be slightly to moderately toxic to aquatic invertebrates. The EC_{50} (15 mg/L) and NOAEL (9.7 mg/L) were selected as the invertebrate TRVs.

Aquatic Plants

In studies with duckweed, diflufenzopyr has high toxicity to aquatic plants. In 14-day toxicity tests, 50% of the duckweed plants were adversely affected by concentrations as low as 0.11 mg a.i./L (the EC_{50}) of Distinct® herbicide (USEPA 2003). The green algae EC_{50} (0.1 mg/L) and NOAEL (0.0078 mg/L) were selected as the aquatic plant TRVs.

Diquat

Diquat has relatively high toxicity to fish and aquatic invertebrates. Diquat does not appear to appreciably bioconcentrate in fish tissue. No acute toxicity studies conducted on amphibian species were found in the literature. Aquatic macrophytes were adversely affected by diquat concentrations as low as 0.00075 mg/L. There did not appear to be appreciable differences in sensitivities among aquatic macrophytes, diatoms, and algae.

Fish

Results from coldwater and warmwater fish species suggest that diquat has high toxicity to fish species. The lower of the coldwater and warmwater fish endpoints were selected as the TRVs for fish; therefore, the warmwater 96-hour LC_{50} of 0.75 mg a.i./L was selected as the acute TRV. Because the NOAEL in a chronic study on rainbow trout was determined to be <0.5 mg a.i./L, the coldwater fish NOAEL was calculated by dividing this value by an uncertainty factor of 3. The resulting NOAEL TRV for coldwater fish species was 0.17 mg a.i./L; this was selected as the chronic fish TRV. In addition, the bioconcentration potential for diquat is low.

Amphibians

In a chronic toxicity study on northern leopard frogs, diquat was found to have moderate toxicity to amphibians. In a 16-day exposure, frogs were adversely affected by diquat concentrations as low as 5 mg/L, while no adverse effects were observed at 2 mg/L. The NOAEL (2 mg/L) was selected as an amphibian chronic TRV.

Aquatic Invertebrates

Toxicity studies on daphnids and amphipods suggest that diquat is highly toxic to aquatic invertebrates. The LC_{50} (0.14 mg/L) was selected as the invertebrate acute TRV, and the NOAEL of 0.044 mg/L was selected as the chronic TRV.

Aquatic Plants

Duckweed toxicity studies suggest that diquat is very highly toxic to aquatic plants. In 14-day studies, 50% of the duckweed plants were adversely affected by concentrations as low as 0.00075 mg/L of diquat (i.e., the EC_{50}; USEPA 2003; MRID 41883002). The EC_{50} (0.00075 mg/L) was selected as the aquatic plant acute TRV. Because no NOAEL values in the reviewed literature were lower than the EC_{50}, the EC_{50} was divided by an uncertainty factor of 3 to estimate a NOAEL TRV of 0.0003 mg/L.

Diuron

Diuron is moderately toxic to fish and highly toxic to aquatic plants and aquatic invertebrates. Toxicity tests indicate that diuron is toxic to fish species at concentrations as low as 0.71 mg/L. Diuron has a low to moderate potential to bioconcentrate in fish tissue. Amphibians were less sensitive to diuron than any other aquatic taxa. Aquatic invertebrates were affected by diuron concentrations of 0.16 mg a.i./L. Aquatic plants were affected at concentrations as low as 0.0013 mg a.i./L (about 0.02% of the typical application rate).

Fish

Diuron is considered moderately to highly toxic to fish. The lower of the coldwater and warmwater fish endpoints were selected as the TRVs for fish. Therefore the coldwater 96-hour LC_{50} of 0.71 mg/L was selected as the acute TRV, and the warmwater fish NOAEL of 0.033 mg a.i./L was used as the TRV for chronic effects.

Amphibians

Toxicity tests suggest that diuron is slightly toxic to amphibians. Acute toxicity was observed in amphibians exposed to diuron concentrations of 12.7 mg/L. In chronic toxicity tests, adverse effects on growth were observed at concentrations of 14.5 mg/L, with no effects observed at 7.6 mg a.i./L. The LC_{50} (12.7 mg/L) was selected as an amphibian acute TRV, and the NOAEL (7.6 mg/L) was selected as the chronic TRV.

Aquatic Invertebrates

Diuron is considered to have relatively high toxicity to aquatic invertebrates. The LC_{50} (0.16 mg/L) was selected as the invertebrate acute TRV. Since none of the observed chronic NOAEL values were below the selected acute TRV, the chronic LOAEL from a 28-day daphnid assay was divided by an uncertainty factor of 3 to estimate a chronic NOAEL TRV of 0.067 mg a.i./L.

Aquatic Plants

Toxicity tests on green algae (the most sensitive of aquatic plants tested) suggest that diuron is very highly toxic to aquatic plants. The EC_{50} (0.0013 mg/L) was selected as the aquatic plant acute TRV, and the NOAEL (0.00044mg/L) was selected as the chronic TRV.

Fluridone

In the available literature, aquatic plants were affected by concentrations less than 1 mg/L. Acute and chronic toxicity tests indicate that fluridone is toxic to fish species at concentrations less than 10 mg/L, and some adverse effect concentrations approach 1 mg/L. No data were found to evaluate the toxicity of fluridone to amphibians. Acute toxicity concentrations for aquatic invertebrates were as low as 1.3 mg/L, which is equal to the maximum application rate.

Fish

Fluridone is considered to be moderately toxic to fish species. The lower of the coldwater and warmwater fish endpoints were selected as the TRVs for fish. Therefore, the coldwater 96-hour LC_{50} of 4.2 mg a.i./L was selected as the acute TRV, and the warmwater fish NOAEL of 0.48 mg a.i./L was used as the TRV for chronic effects.

Amphibians

No toxicity studies for amphibians were found in the literature reviewed for fluridone.

Aquatic Invertebrates

Fluridone appears to be moderately toxic to aquatic invertebrates. Acute toxicity was observed in aquatic invertebrates exposure to fluridone concentrations as low as 1.3 mg/L. NOAELs for several species were derived from chronic or short-term chronic studies. The NOAEL for *D. magna* is 0.2 mg/L and the NOAELs for *Gammarus pseudolimnaeus* and *Chironomus plumosus* are 0.6 mg/L. The LC_{50} (1.3 mg/L) was selected as the invertebrate acute TRV, and the NOAEL of 0.6 mg/L was selected as the chronic TRV.

Aquatic Plants

Toxicity studies on American pondweed suggest that fluridone is moderately toxic to aquatic plants. No adverse effects to aquatic macrophytes were detected with fluridone concentrations of 1 mg/L, and the NOAEL was set at 1 mg/L. Because no EC_{50} values were identified in the literature, the NOAEL was multiplied by an uncertainty factor of 3 to estimate an EC_{50} of 3 mg/L.

Imazapic

Imazapic is relatively toxic to aquatic plants, but is much less toxic to aquatic animal species. Aquatic plants were affected at concentrations as low as 0.0042 mg/L. Toxicity tests indicate that imazapic has low toxicity to fish species and does not appreciably bioconcentrate in fish tissue. No data were found to evaluate the toxicity of imazapic to amphibians. Most studies reported that aquatic invertebrates were unaffected by imazapic concentrations of 100 mg/L; however, one unverifiable report suggested that chronic toxicity to aquatic invertebrates may occur at concentrations as low as 0.18 mg/L.

Fish

Imazapic is considered to have low toxicity to fish species. The coldwater 96-hour LC_{50} of >100 mg/L (the lower of the coldwater and warmwater fish endpoints) was selected as the acute TRV. The LC_{50} was divided by an uncertainty factor of 3, to produce a coldwater fish NOAEL of 33 mg/L used as the TRV for chronic effects. It may be noted that the selected chronic TRV, extrapolated from an LC_{50} indicating essentially no risk, is 3 times lower than the true chronic NOAEL observed for warmwater fish. This may overestimate chronic risk to fish.

Amphibians

No toxicity studies for amphibians were found in the published literature or in USEPA registration documents.

Aquatic Invertebrates

Imazapic is generally considered to have low toxicity to aquatic invertebrates. The LC_{50} (>100 mg/L) was selected as the invertebrate acute TRV. The 21-day NOAEL (96 mg a.i./L) was selected as the invertebrate chronic TRV.

Aquatic Plants

Toxicity studies on duckweed suggest that imazapic is very highly toxic to aquatic plants. In these studies, 25% of the duckweed plants were adversely affected by concentrations of 0.0042 mg/L after 14 days exposure. The no effect concentration in this study was 0.0026 mg/L. Compared to duckweed, freshwater algae and diatoms were at least 10 times more tolerant of imazapic. In 5-day acute toxicity tests, LC_{50} values for algae and diatoms were greater than the highest concentration tested (at least 0.04 mg/L). The aquatic plant TRVs were set at 0.0042 mg/L (EC_{25}) and 0.0026 mg/L (NOAEL).

Overdrive®

Based on toxicity data from dicamba, diflufenzopyr, and Distinct®, and using the conservative assumption that Overdrive® is slightly more toxic than dicamba and diflufenzopyr, Overdrive® may be considered slightly toxic to fish, moderately toxic to aquatic invertebrates (Distinct® was much less toxic to aquatic invertebrates than dicamba), and highly toxic to aquatic plants. Dicamba is practically non-toxic to amphibians, but no data were available for the other chemicals.

Fish

Toxicity tests suggest that diflufenzopyr is practically non-toxic to fish, and dicamba is slightly toxic to fish. No fish toxicity tests were identified for Distinct®. The selected fish TRVs for diflufenzopyr were established at 106 mg/L (warmwater LC_{50}) and 16 mg/L (coldwater NOAEL). The selected fish TRVs for dicamba were established at 28 mg/L (coldwater LC_{50}) and 9.3 mg/L (estimated coldwater NOAEL). No chronic toxicity studies on freshwater fish were found in the available literature, and therefore all TRVs are based on acute duration endpoints.

The bioconcentration factor for diflufenzopyr is 3.16, indicating that diflufenzopyr would not appreciably bioconcentrate in fish tissue (National Library of Medicine 2002). In contrast, the bioconcentration factor for dicamba range from 8 to 28, indicating that dicamba may bioconcentrate in fish tissue (HSDB 2002).

Amphibians

A single amphibian toxicity study was found during the literature review, and it suggested that dicamba is practically non-toxic to amphibians. The 96-hour toxicity test with dicamba (as the a.i. in Banvel) using tadpoles of two frog species resulted in LC_{50}s of 106 and 185 mg a.i./L (Johnson 1976). A NOAEL of 35.3 mg a.i./L was estimated by applying an uncertainty factor of 3 to the lowest LC_{50}.

Aquatic Invertebrates

Toxicity tests indicate that diflufenzopyr is slightly toxic, dicamba is moderately toxic, and Distinct® is practically non-toxic to aquatic invertebrates. One diflufenzopyr acute toxicity test using water fleas (e.g., *Daphnia magna*) was found in the literature. The selected invertebrate TRVs for diflufenzopyr were established at 15 mg/L (EC_{50}) and 9.7 mg/L (NOAEL), indicating that diflufenzopyr is slightly toxic to aquatic invertebrates.

Several dicamba aquatic invertebrate tests were identified, resulting in LC_{50}s ranging from 3.8 mg/L for the scud (Hurlbert 1975) to >1,000 mg/L for the water flea (Forbis et al. 1985). The selected dicamba LC_{50} (3.8 mg /L) was divided by an uncertainty factor of 3, to result in a dicamba NOAEL of 1.27 mg/L.

One 48-hour acute Distinct® water flea test was identified. No effects were observed at the highest tested concentration, 130 mg a.i./L (USEPA 2003; MRID 45310903). The NOAEL (130 mg/L) was multiplied by an uncertainty factor of 3, to result in a Distinct® EC_{50} of 390 mg/L.

No chronic toxicity studies on freshwater aquatic invertebrates were found in the available literature, and therefore, all TRVs are based on acute duration endpoints.

Aquatic Plants

Standard toxicity tests conducted on aquatic plants, including aquatic macrophytes, freshwater diatoms, and algae, suggest that diflufenzopyr, dicamba, and Distinct® are highly toxic to aquatic plants. The green algae EC_{50} (0.1 mg/L) and NOAEL (0.0078 mg/L) were selected as the aquatic plant TRVs for diflufenzopyr. The selected dicamba EC_{50} (0.1 mg a.i./L) was divided by an uncertainty factor of 3, to result in a dicamba NOAEL of 0.033 mg a.i./L. Based on the data above, the selected aquatic plant TRVs for Distinct® were established at 0.11 mg/L (EC_{50}) and 0.0023 mg/L (NOAEL).

Sulfometuron Methyl

Sulfometuron methyl is toxic to aquatic plants. Tests indicate that sulfometuron methyl has low acute toxicity to fish and aquatic invertebrates, though chronic toxicity can occur from long-term exposure. Sulfometuron methyl has a low potential to bioconcentrate in fish tissue. Overall, amphibians were more sensitive to sulfometuron methyl than most other aquatic biota.

Fish

Sulfometuron methyl is considered moderately toxic to fish. The coldwater 96-hour LC_{50} of >148 mg/L (the lower of the coldwater and warmwater fish endpoints) was selected as the acute TRV, and the warmwater fish NOAEL of 0.71 mg/L was used as the TRV for chronic effects.

Amphibians

Toxicity tests on frogs suggest that sulfometuron methyl is moderately toxic to amphibians. After 96-hours of exposure, 50% of the frogs exposed to sulfometuron methyl concentrations as low as 4.2 mg/L exhibited malformations. In chronic toxicity tests with this same species, malformations were observed in frogs exposed to concentrations as low as 1 mg/L, with no effects observed at 0.1 mg/L. The EC_{50} (4.2 mg/L) was selected as the amphibian acute TRV and the NOAEL (0.1 mg a.i./L) was selected as the chronic TRV.

Aquatic Invertebrates

Sulfometuron methyl is considered to have slight acute toxicity to aquatic invertebrates (chronic toxicity is higher). The LC_{50} (802 mg/L) was selected as the invertebrate acute TRV and the 21-day NOAEL (6.1 mg/L) was selected as the chronic TRV. However, one unverifiable report suggested that chronic toxicity to aquatic invertebrates may occur at concentrations as low as 0.18 mg/L.

Aquatic Plants

Sulfometuron methyl was most toxic to water milfoil, an aquatic macrophyte; tests on this plant suggest that this herbicide is very highly toxic to aquatic plants. Adverse effects to 50% of the milfoil plants (the EC_{50}) were observed in concentrations containing 0.00012 mg a.i./L, and the EC_{25} was 0.00006 mg a.i./L (Roshon et al. 1999). The EC_{50} (0.00012 mg a.i./L) was selected as the aquatic plant acute TRV. Because a NOAEL was not reported, it was extrapolated by dividing the EC_{50} by an uncertainty factor of 3; the resulting NOAEL was 0.00004 mg a.i./L.

Tebuthiuron

Tebuthiuron has low toxicity to coldwater and warmwater fish and amphibians and slight toxicity to aquatic invertebrates. Amphibians were more tolerant of tebuthiuron than fish. While tebuthiuron was not highly toxic to aquatic plants under acute exposure conditions, chronic exposure resulted in toxicity at relatively low concentrations. Tebuthiuron is not expected to bioconcentrate in aquatic organisms.

Fish

Tebuthiuron is considered to have low acute toxicity to fish (chronic toxicity is higher). The warmwater 96-hour LC_{50} of 112 mg/L (the lower of the coldwater and warmwater fish endpoints) was selected as the acute TRV and the warmwater fish NOAEL of 9.3 mg/L was used as the TRV for chronic effects.

Amphibians

Toxicity tests on bullfrogs suggest that tebuthiuron is practically non-toxic to amphibians. After 96-hours of exposure, the LC_{50} concentration was determined to be less than 398 mg/L, but greater than 306 mg/L. The LC_{50} (398 mg/L) was selected as an amphibian acute TRV. Because there was no suitable NOAEL reported in the literature, the NOAEL was extrapolated from the LC_{50} using an uncertainty factor of 3; the resulting NOAEL TRV was 133 mg/L.

Aquatic Invertebrates

Tebuthiuron has low acute toxicity to aquatic invertebrates (chronic toxicity is higher). In 48-hour aquatic toxicity tests, acute toxicity was observed in aquatic invertebrates exposed to concentrations of 297 mg/L of tebuthiuron. In chronic tests with chironomids, adverse effects were observed in the lowest concentration tested, 0.2 mg/L. No adverse effects were observed in snails exposed to 0.1 mg/L. The LC_{50} (297 mg/L) was selected as the invertebrate acute TRV. The snail NOAEL (0.1 mg/L) was selected as the invertebrate chronic TRV.

Aquatic Plants

Tebuthiuron appears to be highly toxic to aquatic plants. In acute toxicity tests, the EC_{50} was reported to be as low as 0.05 mg/L. NOAELs ranged from 0.013 mg/L for green algae to 0.18 for various algal species. The EC_{50} (0.05 mg/L) was selected as the aquatic plant acute TRV, and the NOAEL (0.013 mg/L) was selected as the aquatic plant chronic TRV.

Rare, Threatened, and Endangered Species Characterization

Rare, threatened, and endangered species have the potential to be impacted by herbicides applied for vegetation control. Rare, threatened, and endangered species are of potential increased concern to screening level ERAs, which utilize surrogate species and generic assessment endpoints to evaluate potential risk, rather than examining site- and species-specific effects to individual RTE species. Several factors complicate the evaluation of the effects of herbicide applications on RTE species:

- Toxicological data specific to the species (and sometimes even class) of organism are often absent from the literature.
- The other assumptions involved in the ERA (e.g., rate of food consumption, surface-to-volume ratio) may differ for RTE species relative to selected surrogates and/or data for RTE species may be unavailable.
- The high level of protection afforded RTE species by regulation and policy suggests that secondary effects (e.g., potential loss of prey or cover), as well as site-specific circumstances that might result in higher rates of exposure, should receive more attention.

A common response to these issues is to design screening level ERAs to be highly conservative. This includes assumptions such as 100% exposure to an herbicide by simulating scenarios where the organism lives year-round (not likely for larger and migratory animals) in the most affected area (i.e., area of highest concentration), or that the organism consumes only food items that have been impacted by the herbicide. The screening level ERA incorporates additional conservatism in the assumptions used in the herbicide concentration models such as GLEAMS. Even with highly conservative assumptions in the ERA, however, concern may still exist over the potential risk to specific RTE species.

Potential direct impacts to receptors, including RTE species, are the measures of effect typically used in screening level ERAs. Direct impacts, such as those resulting from direct or indirect contact or ingestion were assessed in the ERAs by comparing calculated RQs to receptor-specific LOCs. An RQ greater than the LOC indicates the potential for risk to that receptor group via that exposure pathway. As described below, the selection of TRVs and the use of LOCs were pursued in a conservative fashion in order to provide a greater level of protection for RTE species.

The LOCs used in the ERA (Table C-1) were developed by the USEPA for the assessment of pesticides (LOC information obtained from Michael Davy, USEPA OPP on June 13, 2002). In essence, the LOCs act as uncertainty factors often applied to TRVs. For example, using an LOC of 1.0 provides the same result as dividing the TRV by 10. The LOC for avian and mammalian RTE species is 0.1 for acute and chronic exposures. For RTE fish and aquatic invertebrates, acute and chronic LOCs were 0.05 and 0.5, respectively. Therefore, up to a 20-fold uncertainty factor has been included in the TRVs for animal species. As noted below, such uncertainty factors provide a greater level of protection to the RTE to account for the factors listed in the beginning of this section.

For RTE plants, the exposure concentration, TRVs, and LOCs provided a direct assessment of potential impacts. For all exposure scenarios, the maximum modeled concentrations were used as the exposure concentrations. The TRVs used for RTE plants were selected based on highly sensitive endpoints, such as germination, rather than direct mortality of seedlings or larger plants. Conservatism has been built into the TRVs during their development; the lowest suitable endpoint concentration available was used as the TRV for RTE plant species. Therefore, the RQ calculated for RTE plant exposure is intrinsically conservative. Given the conservative nature of the RQ, and to be consistent with USEPA policy, no additional levels of protection were required for the LOC (i.e., all plant LOCs are 1).

Protection levels for different species and individuals vary. Some organisms are protected on a community level; that is, slight risk to individual species may be acceptable if the community of organisms (e.g., wildflowers, terrestrial insects) is protected. Generally, community level organisms include plants and invertebrates. Other organisms are protected on a population level; that is, slight risk to individuals of a species may be acceptable if the population, as a whole, is not endangered. However, RTE species are protected as individuals; that is, risk to any single organism is considered unacceptable. This higher level of protection motivates much of the conservative approach taken in this ERA. Surrogate species were grouped by general life strategy: sessile (i.e., plants), water dwelling (i.e., fish), and mobile terrestrial vertebrates (i.e., birds, mammals, and reptiles). The approach to account for RTE species was divided along the same lines. Plants, fish, insects, and aquatic invertebrates were assessed using TRVs developed from surrogate species. All species from these taxa (identified in Appendix C of each ERA [ENSR 2005a-j]) were represented by the surrogate species presented in Tables C-3 and C-4.

Non-target Species Risk Characterization

The risk analysis evaluates the effects of the 10 herbicides on non-target terrestrial and aquatic flora and fauna. The risk that non-target flora and fauna face from herbicide applications depends on the toxicity of herbicides to individual species and the exposure of organisms to herbicides as a result of BLM applications. In order to address potential risks to ecological receptors, RQs were calculated by dividing the EEC for each of the previously described exposure pathways by the appropriate herbicide-specific TRV. Over 1,000 RQs were generated in each ERA. These RQs were then compared to LOC established by the USEPA OPP for all appropriate exposure scenarios and surrogate species. Ecological risk is implied when RQs exceeded the corresponding LOCs. LOCs are defined for the following risk presumption categories (see Table C-1):

- Acute high risk – the potential for acute risk is high.
- Acute restricted use – the potential for acute risk is high, but may be mitigated.
- Acute RTE species – RTE species may be adversely affected.
- Chronic risk – the potential for chronic risk is high.

Specific risks to applicable terrestrial and aquatic receptor groups from each individual herbicide are presented below according to the particular exposure pathway. See the tables and figures in Section 4 of the ERAs for each herbicide for risk information on ecological receptor groups according to herbicide application method. Indirect risks to salmonids are presented at the end of each herbicide section. In addition to direct effects of herbicides on salmonids and other fish species in stream habitats (i.e., mortality due to herbicide concentrations in water), reduction in vegetative cover or food supply may indirectly impact individuals or populations. No literature studies were identified that explicitly evaluated the direct or indirect effects of the herbicides on salmonids and their habitat; therefore, only qualitative estimates of indirect effects were possible. These estimates were accomplished by evaluating predicted impacts to prey items and vegetative cover in the stream scenarios of accidental direct spray, off-site drift, and surface runoff. An evaluation of impacts to non-target terrestrial plants was also included as part of the discussion of vegetative cover within the riparian zone. Food items for salmonids and other potential RTE species may include other fish species, aquatic invertebrates, and aquatic plants. It should be noted that the selected chronic fish TRV was based on a NOAEL for the fathead minnow (*Pimephales promelas*), which is much lower than any chronic values identified for salmonids. This indicates that chronic impacts to salmonids may be overestimated in this assessment.

Bromacil

Direct Spray

Terrestrial Wildlife

In general, acute RQs for terrestrial wildlife were below the most conservative LOC (0.1; acute risk RTE species at the typical application rate). However, direct spray of the pollinating insect resulted in elevated RQs at both the typical and maximum application rates. This is a conservative scenario that assumes the insect absorbs 100% of the herbicide with no degradation or limitations to uptake. Acute RQs above the most conservative LOC were also predicted at the maximum

TABLE C-5
Selected Toxicity Reference Values for Bromacil

Receptor	Selected TRV[1]	Units	Duration[2]	Endpoint[3]	Species	Notes
Receptors Included in the Food Web Model						
Terrestrial Animals						
Honeybee	193	ug/bee	NR	LD_{50}		Technical grade; no % a.i. listed
Large bird	> 5,000	mg/kg bw	8 d	LD_{50}	Mallard	Dietary; 80% a.i. product
Large bird	155	mg/kg bw-day	22 w	NOAEL	Mallard	98.1% a.i. product
Piscivorous bird	155	mg/kg bw-day	22 w	NOAEL	Mallard	98.1% a.i. product
Small bird	> 30,195	mg/kg bw	8 d	LD_{50}	Bobwhite quail	80% a.i. product
Small bird	936	mg/kg bw-day	21 w	NOAEL	Bobwhite quail	98.1% a.i. product
Small mammal	13.3	mg a.i./kg bw-day	2 y	NOAEL	Rat	
Small mammal - dermal	> 5,000	mg a.i./kg bw	NR	LD_{50}	Rabbit	
Small mammal - ingestion	641	mg a.i./kg bw	NR	LD50	Rat	Water exposure; no diet available
Large mammal	641	mg a.i./kg bw	> 14 d	LD_{50}	Rat	Small mammal value used
Large mammal	4.65	mg a.i./kg bw-day	2 y	NOAEL	Dog	
Terrestrial Plants						
Typical species - direct spray, drift, dust	0.0023	lb a.i./acre	NR	EC_{25}	Rape	Vigor
RTE species - direct spray, drift, dust	0.0008	lb a.i./acre	NR	NOAEL	Rape	Extrapolated from EC_{25}; vigor
Typical species - runoff	0.188	lb a.i./acre	14 d	NOAEL	Soybean	Emergence; no germination data
RTE species - runoff	0.0117	lb a.i./acre	NR	NOAEL	Rape	Emergence; no germination data
Aquatic Species						
Aquatic invertebrates	65	mg a.i./L	48 h	EC_{50}	Water flea	
Fish	36	mg/L	96 h	LC_{50}	Rainbow trout	96.6% a.i. product
Aquatic plants and algae	0.0068	mg/L	5 d	EC_{50}	Green algae	96.5% a.i. product
Aquatic invertebrates	22	mg a.i./L	48 h	NOAEL	Water flea	Extrapolated from EC_{50}
Fish	0.33	mg a.i./L	64 d	NOAEL	Fathead minnow	Extrapolated from chronic LOAEL
Aquatic plants and algae	0.0023	mg/L	5 d	NOAEL	Green algae	Extrapolated from EC_{50}

TABLE C-5 (Cont.)
Selected Toxicity Reference Values for Bromacil

Receptor	Selected TRV[1]	Units	Duration[2]	Endpoint[3]	Species	Notes
Additional Endpoints						
Amphibian	230	mg a.i./L	48 h	LC_{50}	Tadpole	"Unacceptable;" no other data; no % a.i. provided
Amphibian	77	mg a.i./L	48 h	NOAEL	Tadpole	Extrapolated from LC_{50}; "unacceptable;" no other data; no % a.i. provided
Warmwater fish	71	mg/L	48 h	LC_{50}	Bluegill	No % a.i. provided
Warmwater fish	0.33	mg a.i./L	64 d	NOAEL	Fathead minnow	Extrapolated from chronic LOAEL
Coldwater fish	36	mg/L	96 h	LC_{50}	Rainbow trout	96.6% a.i. product
Coldwater fish	16.9	mg/L	96 h	NOAEL	Rainbow trout	96.6% a.i. product

[1] Piscivorous bird TRV = Large bird chronic TRV; and fish TRV = Lower of coldwater and warmwater fish TRVs.
[2] Duration: h = hours; d = days; w = weeks; m = months; y = years; and NR = not reported.
[3] Toxicity endpoints for terrestrial animals: LD_{50} to address acute exposure, and NOAEL to address chronic exposure. Toxicity endpoints for terrestrial plants: EC_{25} to address direct spray, drift, and dust impacts on typical species; NOAEL to address direct spray, drift, and dust impacts on RTE species; highest germination NOAEL to address surface runoff impacts on typical species; and lowest germination NOAEL to address surface runoff impacts on RTE species. Toxicity endpoints for aquatic receptors: LC_{50} or EC_{50} to address acute exposure (appropriate toxicity endpoint for non-target aquatic plants will be an EC_{50}) and NOAEL to address chronic exposure.

TABLE C-6
Selected Toxicity Reference Values for Chlorsulfuron

Receptor	Selected TRV[1]	Units	Duration[2]	Endpoint[3]	Species	Notes
Receptors Included in the Food Web Model						
Terrestrial Animals						
Honeybee	> 25	μg/bee	48 h	LD_{50}		98.2% a.i. product
Large bird	> 1,500	mg/kg bw	8 d	LD_{50}	Mallard	Technical grade; no % a.i. listed
Large bird	99	mg/kg bw-day	1 generation	NOAEL	Mallard	98.2% a.i. product
Piscivorous bird	99	mg/kg bw-day	1 generation	NOAEL	Mallard	98.2% a.i. product
Small bird	> 16,970	mg/kg bw	8 d	LD_{50}	Bobwhite quail	Technical grade; no % a.i. listed
Small bird	100	mg/kg bw-day	27 w	NOAEL	Bobwhite quail	98.2% a.i. product
Small mammal	1,363	mg a.i./kg bw	NR	LD_{50}	Guinea pig	Small mammal value used
Small mammal – dermal	66	mg a.i./kg bw-day	1 y	NOAEL	Dog	
Small mammal – ingestion	5	mg a.i./kg bw-day	90 d	NOAEL	Rat	
Large mammal	> 3,400	mg a.i./kg bw	24 h	LD_{50}	Rabbit	
Large mammal	1,363	mg a.i./kg bw	NR	LD_{50}	Guinea pig	Water exposure; no diet available
Terrestrial Plants						
Typical species - direct spray, drift, dust	0.000063	lb a.i./acre	life cycle	EC_{25}	Canola	Extrapolated from NOAEL
RTE species - direct spray, drift, dust	0.000021	lb a.i./acre	life cycle	NOAEL	Canola	
Typical species - runoff	0.0157	lb a.i./acre	NR	NOAEL	Dyer's woad (weed)	Extrapolated from germination EC_{100}
RTE species - runoff	0.0052	lb a.i./acre	NR	NOAEL	Dyer's woad (weed)	Extrapolated from germination NOAEL
Aquatic Species						
Aquatic invertebrates	368.9	mg a.i./L	48 h	LC_{50}	Water flea (*D. magna*)	91% a.i. product
Fish	40	mg/L	96 h	LC_{50}	Brown trout	No % a.i. listed
Aquatic plants and algae	0.0007	mg a.i./L	96 h	EC_{50}	Duckweed	Technical grade; no % a.i. listed
Aquatic invertebrates	20	mg a.i./L	21 d	NOAEL	Water flea (*C. dubia*)	95.4% a.i. product
Fish	17	mg/L	96 h	NOAEL	Channel catfish	91% a.i. product; extrapolated from LC_{50}
Aquatic plants and algae	0.0004	mg a.i./L	96 h	NOAEL	Duckweed	Technical grade; no % a.i. listed

TABLE C-6 (Cont.)
Selected Toxicity Reference Values for Chlorsulfuron

Receptor	Selected TRV[1]	Units	Duration[2]	Endpoint[3]	Species	Notes
Additional Endpoints						
Amphibian	no data					
Amphibian	no data					
Warmwater fish	50	mg a.i./L	96 h	LC_{50}	Channel catfish	91% a.i. product
Warmwater fish	17	mg a.i./L	96 h	NOAEL	Channel catfish	91% a.i. product; extrapolated from LC_{50}
Coldwater fish	40	mg a.i./L	96 h	LC_{50}	Brown trout	No % a.i. listed
Coldwater fish	31	mg a.i./L	77 d	NOAEL	Rainbow trout	97.9% a.i. product

[1] Piscivorous bird TRV = Large bird chronic TRV; and fish TRV = Lower of coldwater and warmwater fish TRVs.
[2] Duration: h = hours; d = days; w = weeks; m = months; y = years; and NR = not reported.
[3] Toxicity endpoints for terrestrial animals: LD_{50} to address acute exposure, and NOAEL to address chronic exposure. Toxicity endpoints for terrestrial plants: EC_{25} to address direct spray, drift, and dust impacts on typical species; NOAEL to address direct spray, drift, and dust impacts on RTE species; highest germination NOAEL to address surface runoff impacts on typical species; and lowest germination NOAEL to address surface runoff impacts on RTE species. Toxicity endpoints for aquatic receptors: LC_{50} or EC_{50} to address acute exposure (appropriate toxicity endpoint for non-target aquatic plants will be an EC_{50}) and NOAEL to address chronic exposure.

TABLE C-7
Selected Toxicity Reference Values for Diflufenzopyr

Receptor	Selected TRV[1]	Units	Duration[2]	Endpoint[3]	Species	Notes
Receptors Included in the Food Web Model						
Terrestrial Animals						
Honeybee	75	µg/bee	48 h	LD_{50}		Extrapolated from NOAEL; 99.4% a.i. product
Large bird	> 2,810	mg/kg bw	8 d	LD_{50}	Mallard	94.7% a.i. product
Large bird	105	mg/kg bw-day	21 w	NOAEL	Mallard	94.3% a.i. product
Piscivorous bird	105	mg/kg bw-day	21 w	NOAEL	Mallard	94.3% a.i. product
Small bird	> 16,970	mg/kg bw	8 d	LD_{50}	Bobwhite quail	94.7% a.i. product
Small bird	634	mg/kg bw-day	20 w	NOAEL	Bobwhite quail	94.3% a.i. product
Small mammal	42.2	mg/kg bw-day	2 generation	NOAEL	Rat	93% a.i. product
Small mammal - dermal	> 5,000	mg/kg bw	NR	LD_{50}	Rabbit	96.4% a.i. product
Small mammal - ingestion	3,300	mg/kg bw	NR	LD_{50}	Rat	Water exposure; no diet available; 98.1% a.i. product
Large mammal	3,300	mg/kg bw	NR	LD_{50}	Rat	Small mammal value
Large mammal	59	mg/kg bw-day	1 y	NOAEL	Dog	No % a.i. listed
Terrestrial Plants						
Typical species - direct spray, drift, dust	0.0008	lb a.i./acre	14 d	EC_{25}	Turnip	Based on emergence
RTE species - direct spray, drift, dust	0.0003	lb a.i./acre	14 d	NOAEL	Turnip	Extrapolated from EC_{25}
Typical species – runoff	0.028	lb a.i./acre	14 d	NOAEL	Tomato	No germination data; based on emergence
RTE species - runoff	0.0001	lb a.i./acre	NR	NOAEL	Turnip	No germination data; based on emergence
Aquatic Species						
Aquatic invertebrates	15	mg/L	48 h	EC_{50}	Water flea	94.7% a.i. product
Fish	106	mg/L	96 h	LC_{50}	Rainbow trout	97.4% a.i. product
Aquatic plants and algae	0.1	mg/L	5 d	EC_{50}	Green algae	99.5% a.i. product
Aquatic invertebrates	9.7	mg/L	48 h	NOAEL	Water flea	94.7% a.i. product
Fish	16	mg/L	96 h	NOAEL	Bluegill sunfish	97.4% a.i. product
Aquatic plants and algae	0.0078	mg/L	5 d	NOAEL	Green algae	99.5% a.i. product

TABLE C-7 (Cont.)
Selected Toxicity Reference Values for Diflufenzopyr

Receptor	Selected TRV[1]	Units	Duration[2]	Endpoint[3]	Species	Notes
			Additional Endpoints			
Amphibian	no data					
Amphibian	no data					
Warmwater fish	> 135	mg/L	96 h	LC_{50}	Bluegill sunfish	97.4% a.i. product
Warmwater fish	16	mg/L	96 h	NOAEL	Bluegill sunfish	97.4% a.i. product
Coldwater fish	106	mg/L	96 h	LC_{50}	Rainbow trout	97.4% a.i. product
Coldwater fish	80	mg/L	96 h	NOAEL	Rainbow trout	97.4% a.i. product

[1] Piscivorous bird TRV = Large bird chronic TRV; and fish TRV = Lower of coldwater and warmwater fish TRVs.
[2] Duration: h = hours; d = days; w = weeks; m = months; y = years; and NR = not reported.
[3] Toxicity endpoints for terrestrial animals: LD_{50} to address acute exposure, and NOAEL to address chronic exposure. Toxicity endpoints for terrestrial plants: EC_{25} to address direct spray, drift, and dust impacts on typical species; NOAEL to address direct spray, drift, and dust impacts on RTE species; highest germination NOAEL to address surface runoff impacts on typical species; and lowest germination NOAEL to address surface runoff impacts on RTE species. Toxicity endpoints for aquatic receptors: LC_{50} or EC_{50} to address acute exposure (appropriate toxicity endpoint for non-target aquatic plants will be an EC_{50}) and NOAEL to address chronic exposure.

TABLE C-8
Selected Toxicity Reference Values for Diquat

Receptor	Selected TRV[1]	Units	Duration[2]	Endpoint[3]	Species	Notes
Receptors Included in the Food Web Model						
Terrestrial Animals						
Honeybee	47	µg/bee	5 d	LD_{50}		Extrapolated from NOAEL; no % a.i. listed
Large bird	215	mg a.i./kg bw	NR	LD_{50}	Ring neck pheasant	
Large bird	0.6	mg a.i./kg bw-day	1 generation	NOAEL	Mallard	
Piscivorous bird	0.6	mg a.i./kg bw-day	1 generation	NOAEL	Mallard	
Small bird	150	mg a.i./kg bw	NR	LD_{50}	Japanese quail	
Small bird	> 12	mg a.i./kg bw-day	NR	NOAEL	Bobwhite quail	
Small mammal	0.8	mg a.i./kg bw-day	104 w	NOAEL	Rat	
Small mammal - dermal	262	mg a.i./kg bw	NR	LD_{50}	Rabbit	
Small mammal - ingestion	121	mgkg bw	> 14 d	LD_{50}	Rat	Water exposure; no diet available; no % a.i. listed
Large mammal	121	mg/kg bw	> 14 d	LD_{50}	Rat	Small mammal value; no % a.i. listed
Large mammal	0.5	mg a.i./kg bw-day	1 y	NOAEL	Dog	
Terrestrial Plants						
Typical Species - direct spray, drift	0.0047	lb a.i./acre	NR	EC_{25}	Cotton	Vigor
RTE Species - direct spray, drift	0.0016	lb a.i./acre	NR	NOAEL	Cotton	Vigor; extrapolated from EC_{25}
Aquatic Species						
Aquatic invertebrates	0.14	mg/L	48 h	EC_{50}	Amphipod	No % a.i. listed
Fish	0.75	mg a.i./L	96 h	LC_{50}	Walleye	
Aquatic plants and algae	0.00075	mg a.i./L	14 d	EC_{50}	Giant duckweed	35.3% a.i. product
Aquatic invertebrates	0.044	mg a.i./L	life cycle	NOAEL	Water flea	41.4% a.i. product
Fish	0.17	mg a.i./L	NR	NOAEL	Rainbow trout	Extrapolated from LOAEL / swimming speed
Aquatic plants and algae	0.0003	mg a.i./L	14 d	NOAEL	Giant duckweed	Extrapolated from EC_{50}

TABLE C-8 (Cont.)
Selected Toxicity Reference Values for Diquat

Receptor	Selected TRV[1]	Units	Duration[2]	Endpoint[3]	Species	Notes
			Additional Endpoints			
Amphibian	no data					
Amphibian	2	mg/L	16 d	NOAEL	Northern leopard frog	No % a.i. listed
Warmwater fish	0.75	mg a.i./L	96 h	LC_{50}	Walleye	
Warmwater fish	0.58	mg a.i./L	34 d	NOAEL	Fathead minnow	41% a.i. product
Coldwater fish	14.83	mg/L	96 h	LC_{50}	Rainbow trout	19.8% a.i. product
Coldwater fish	0.17	mg a.i./L	NR	NOAEL	Rainbow trout	Extrapolated from LOAEL/swimming speed

[1] Piscivorous bird TRV = Large bird chronic TRV; and fish TRV = Lower of coldwater and warmwater fish TRVs.
[2] Duration: h = hours; d = days; w = weeks; m = months; y = years; and NR = not reported.
[3] Toxicity endpoints for terrestrial animals: LD_{50} to address acute exposure, and NOAEL to address chronic exposure. Toxicity endpoints for terrestrial plants: EC_{25} to address direct spray, drift, and dust impacts on typical species; NOAEL to address direct spray, drift, and dust impacts on RTE species; highest germination NOAEL to address surface runoff impacts on typical species; and lowest germination NOAEL to address surface runoff impacts on RTE species. Toxicity endpoints for aquatic receptors: LC_{50} or EC_{50} to address acute exposure (appropriate toxicity endpoint for non-target aquatic plants will be an EC_{50}) and NOAEL to address chronic exposure.

TABLE C-9
Selected Toxicity Reference Values for Diuron

Receptor	Selected TRV[1]	Units	Duration[2]	Endpoint[3]	Species	Notes
Receptors Included in the Food Web Model						
Terrestrial Animals						
Honeybee	145.03	μg/bee	48 h	LD_{50}		Technical grade; no % a.i. listed
Large bird	865	mg/kg bw	8 d	LD_{50}	Mallard	No % a.i. listed
Large bird	58	mg/kg bw-day	8 d	NOAEL	Mallard	No % a.i. listed; extrapolated from LD_{50}
Piscivorous bird	1,017	mg a.i./kg bw	NR	LD_{50}	Rat	Small mammal value
Small bird	0.6	mg a.i./kg bw-day	2 y	NOAEL	Dog	
Small bird	58	mg/kg bw-day	8 d	NOAEL	Mallard	No % a.i. listed; extrapolated from LD_{50}
Small mammal	5,225	mg/kg bw	8 d	LD_{50}	Bobwhite quail	No % a.i. listed
Small mammal – dermal	348	mg/kg bw-day	8 d	NOAEL	Bobwhite quail	No % a.i. listed; extrapolated from LD_{50}
Small mammal – ingestion	2.5	mg a.i./kg bw-day	3 m	NOAEL	Rat	
Large mammal	> 2,500	mg a.i./kg bw	> 14 d	LD_{50}	Unknown	
Large mammal	1017	mg a.i./kg bw	NR	LD_{50}	Rat	Water exposure; no diet available
Terrestrial Plants						
Typical species - direct spray, drift, dust	0.08	lb a.i./acre	NR	EC_{25}	Tomato	Based on seed emergence
RTE species - direct spray, drift, dust	0.001	lb a.i./acre	21 d	NOAEL		Vigor
Typical species - runoff	12	lb a.i./acre	14 d	NOAEL	Garden pea; soybean	Based on seed emergence
RTE species - runoff	0.047	lb a.i./acre	NR	NOAEL	Tomato	Based on seed emergence
Aquatic Species						
Aquatic invertebrates	0.16	mg/L	96 h	EC_{50}	Scud (*Gammarus*)	95% a.i. product
Fish	0.71	mg/L	96 h	LC_{50}	Cutthroat trout	95% a.i. product
Aquatic plants and algae	0.0013	mg/L	NR	EC_{50}	*Chlorella pyrenoidosa* (algae)	No % a.i. listed
Aquatic invertebrates	0.067	mg/L	28 d	NOAEL	Daphnid	98% a.i. product; extrapolated from chronic LOAEL
Fish	0.033	mg/L	chronic	NOAEL	Fathead minnow	98.6% a.i. product
Aquatic plants and algae	0.00044	mg/L	96 h	NOAEL	*Selenastrum* (algae)	98.6% a.i. product

TABLE C-9 (Cont.)
Selected Toxicity Reference Values for Diuron

Receptor	Selected TRV[1]	Units	Duration[2]	Endpoint[3]	Species	Notes
			Additional Endpoints			
Amphibian	12.7	mg/L	21 d	LC_{50}	Bullfrog	99.8% a.i. product
Amphibian	7.6	mg/L	21 d	NOAEL	Bullfrog	99.8% a.i. product
Warmwater fish	2.8	mg/L	96 h	LC_{50}	Bluegill sunfish	95% a.i. product
Warmwater fish	0.03	mg/L	chronic	NOAEL	Fathead minnow	98.6% a.i. product
Coldwater fish	0.71	mg/L	96 h	LC_{50}	Cutthroat trout	95% a.i. product
Coldwater fish	0.24	mg/L	96 h	NOAEL	Cutthroat trout	95% a.i. product; extrapolated from LC50

[1] Piscivorous bird TRV = Large bird chronic TRV; and fish TRV = Lower of coldwater and warmwater fish TRVs.
[2] Duration: h = hours; d = days; w = weeks; m = months; y = years; and NR = not reported.
[3] Toxicity endpoints for terrestrial animals: LD_{50} to address acute exposure, and NOAEL to address chronic exposure. Toxicity endpoints for terrestrial plants: EC_{25} to address direct spray, drift, and dust impacts on typical species; NOAEL to address direct spray, drift, and dust impacts on RTE species; highest germination NOAEL to address surface runoff impacts on typical species; and lowest germination NOAEL to address surface runoff impacts on RTE species. Toxicity endpoints for aquatic receptors: LC_{50} or EC_{50} to address acute exposure (appropriate toxicity endpoint for non-target aquatic plants will be an EC_{50}) and NOAEL to address chronic exposure.

TABLE C-10
Selected Toxicity Reference Values for Fluridone

Receptor	Selected TRV[1]	Units	Duration[2]	Endpoint[3]	Species	Notes
Receptors Included in the Food Web Model						
Terrestrial Animals						
Honeybee	1,088	µg/bee	48 h	LD_{50}		Extrapolated from NOAEL; 33.3% a.i. product
Large bird	> 2,270	mg/kg bw	8 d	LD_{50}	Mallard	Technical grade; assumed 95 - 97% a.i.
Large bird	100	mg a.i./kg bw-day	1 generation	NOAEL	Mallard	Reproduction
Piscivorous bird	100	mg a.i./kg bw-day	1 generation	NOAEL	Mallard	
Small bird	> 13,135	mg/kg bw	8 d	LD_{50}	Bobwhite quail	Technical grade; assumed 95 - 97% a.i.
Small bird	604	mg a.i./kg bw-day	1 generation	NOAEL	Bobwhite quail	Reproduction
Small mammal	8	mg a.i./kg bw-day	2 y	NOAEL	Rat	
Small mammal - dermal	> 5,000	mg a.i./kg bw	8 d	LD_{50}	Rabbit	
Small mammal - ingestion	> 10,000	mg a.i./kg bw	NR	LD_{50}	Mouse and rat	Water exposure; no diet available
Large mammal	> 10,000	mg a.i./kg bw	NR	LD_{50}	Mouse and rat	Small mammal value
Large mammal	75	mg a.i./kg bw-day	1 y	NOAEL	Beagle	
Terrestrial Plants						
Terrestrial plants - typical species	no data					
Terrestrial plants- RTE species	no data					
Aquatic Species						
Aquatic invertebrates	0.14	mg/L	48 h	EC_{50}	Amphipod	No % a.i. listed
Fish	0.75	mg a.i./L	96 h	LC_{50}	Walleye	
Aquatic plants and algae	0.00075	mg a.i./L	14 d	EC_{50}	Giant duckweed	35.3% a.i. product
Aquatic invertebrates	0.044	mg a.i./L	life cycle	NOAEL	Water flea	41.4% a.i. product
Fish	0.17	mg a.i./L	NR	NOAEL	Rainbow trout	Extrapolated from LOAEL / swimming speed
Aquatic plants and algae	0.0003	mg a.i./L	14 d	NOAEL	Giant duckweed	Extrapolated from EC_{50}

TABLE C-10 (Cont.)
Selected Toxicity Reference Values for Fluridone

Receptor	Selected TRV[1]	Units	Duration[2]	Endpoint[3]	Species	Notes
Additional Endpoints						
Amphibian	no data					
Amphibian	no data					
Warmwater fish	8.2	mg/L	96 h	LC_{50}	Channel catfish	98 – 99% a.i. product
Warmwater fish	0.5	mg/L	life cycle	NOAEL	Fathead minnow	98 – 99% a.i. product
Coldwater fish	4.2	mg/L	96 h	LC_{50}	Rainbow trout	98 – 99% a.i. product
Coldwater fish	1.4	mg/L	96 h	NOAEL	Rainbow trout	Extrapolated from LC_{50}

[1] Piscivorous bird TRV = Large bird chronic TRV; and fish TRV = Lower of coldwater and warmwater fish TRVs.
[2] Duration: h = hours; d = days; w = weeks; m = months; y = years; and NR = not reported.
[3] Toxicity endpoints for terrestrial animals: LD_{50} to address acute exposure, and NOAEL to address chronic exposure. Toxicity endpoints for terrestrial plants: EC_{25} to address direct spray, drift, and dust impacts on typical species; NOAEL to address direct spray, drift, and dust impacts on RTE species; highest germination NOAEL to address surface runoff impacts on typical species; and lowest germination NOAEL to address surface runoff impacts on RTE species. Toxicity endpoints for aquatic receptors: LC_{50} or EC_{50} to address acute exposure (appropriate toxicity endpoint for non-target aquatic plants will be an EC_{50}) and NOAEL to address chronic exposure.

TABLE C-11
Selected Toxicity Reference Values for Imazapic

Receptor	Selected TRV[1]	Units	Duration[2]	Endpoint[3]	Species	Notes
Receptors Included in the Food Web Model						
Terrestrial Animals						
Honeybee	> 100	μg/bee	48 h	LD_{50}		93.7% a.i. product
Large bird	> 2,500	mg a.i./kg bw	8 d	LD_{50}	Mallard	93.7% a.i. product
Large bird	65	mg a.i./kg bw-day	22 w	NOAEL	Mallard	96.9% a.i. product
Piscivorous bird	65	mg a.i./kg bw-day	22 w	NOAEL	Mallard	96.9% a.i. product
Small bird	> 15,095	mg a.i./kg bw	8 d	LD_{50}	Bobwhite quail	93.7% a.i. product
Small bird	113	mg a.i./kg bw-day	24 w	NOAEL	Bobwhite quail	96.9% a.i. product
Small mammal	1,728	mg/kg bw-day	3 m	NOAEL	Rat	Technical grade; no % a.i. listed; extrapolated from LOAEL
Small mammal - dermal	> 2,000	mg/kg bw	NR	LD_{50}	Rabbit	No % a.i. listed
Small mammal - ingestion	> 5,000	mg a.i./kg bw	NR	LD_{50}	Rat	Water exposure
Large mammal	> 5,000	mg a.i./kg bw	NR	LD_{50}	Rat	Same as small mammal value; water exposure
Large mammal	46	mg/kg bw-day	1 y	NOAEL	Dog	Technical grade; no % a.i. listed; extrapolated from LOAEL
Terrestrial Plants						
Typical species - direct spray, drift, dust	0.01	lb a.i./acre	14 d	EC_{25}	Corn	Based on seed emergence
RTE species - direct spray, drift, dust	0.008	lb a.i./acre	21 d	NOAEL	Soybean	Based on vegetative vigor
Typical species – runoff	0.064	lb a.i./acre	6 d	NOAEL	Vegetable crops	Based on seed germination
RTE species - runoff	0.032	lb a.i./acre	6 d	NOAEL	Onion	Based on seed germination
Aquatic Species						
Aquatic invertebrates	> 100	mg/L	48 h	LD_{50}	Water flea	93.7% a.i. product
Fish	> 100	mg/L	96 h	LD_{50}	Rainbow trout	93.7% a.i. product
Aquatic plants and algae	0.0042	mg/L	14 d	EC_{25}	Duckweed	96.9% a.i. product
Aquatic invertebrates	96	mg/L	21 d	NOAEL, ELEL	Water flea	97% a.i. product
Fish	33	mg/L	96 h	NOAEL	Rainbow trout	93.7% a.i. product; extrapolated from 96 h LC_{50}
Aquatic plants and algae	0.0026	mg/L	14 d	NOAEL	Duckweed	96.9% a.i. product

TABLE C-11 (Cont.)
Selected Toxicity Reference Values for Imazapic

Receptor	Selected TRV[1]	Units	Duration[2]	Endpoint[3]	Species	Notes
Additional Endpoints						
Amphibian	no data					
Amphibian	no data					
Warmwater fish	> 100	mg/L	96 h	LD_{50}	Bluegill	93.7% a.i. product
Warmwater fish	96	mg/L	32 d	NOAEL	Fathead minnow	97% a.i. product
Coldwater fish	> 100	mg/L	96 h	LD_{50}	Rainbow trout	93.7% a.i. product
Coldwater fish	33	mg/L	96 h	NOAEL	Rainbow trout	93.7% a.i. product; extrapolated from 96 h LC_{50}

[1] Piscivorous bird TRV = Large bird chronic TRV; and fish TRV = Lower of coldwater and warmwater fish TRVs.
[2] Duration: h = hours; d = days; w = weeks; m = months; y = years; and NR = not reported.
[3] Toxicity endpoints for terrestrial animals: LD_{50} to address acute exposure, and NOAEL to address chronic exposure. Toxicity endpoints for terrestrial plants: EC_{25} to address direct spray, drift, and dust impacts on typical species; NOAEL to address direct spray, drift, and dust impacts on RTE species; highest germination NOAEL to address surface runoff impacts on typical species; and lowest germination NOAEL to address surface runoff impacts on RTE species. Toxicity endpoints for aquatic receptors: LC_{50} or EC_{50} to address acute exposure (appropriate toxicity endpoint for non-target aquatic plants will be an EC_{50}) and NOAEL to address chronic exposure.

TABLE C-12
Selected Toxicity Reference Values for Overdrive®

Receptor	Selected TRV[1]	Units	Duration[2]	Endpoint[3]	Species	Notes
Receptors Included in the Food Web Model						
Terrestrial Animals						
Honeybee	75	ug/bee	48 h	LD_{50}		Extrapolated from NOAE; 9.4% a.i. product
Large bird	> 2,810	mg/kg bw	8 d	LD_{50}	Mallard	94.7% a.i. product
Large bird	105	mg/kg bw-day	21 w	NOAEL	Mallard	94.3% a.i. product
Piscivorous bird	3,300	mg/kg bw		LD_{50}	Rat	Small mammal value
Small bird	59	mg/kg bw-day	1 y	NOAEL	Dog	No % a.i. listed
Small bird	105	mg/kg bw-day	8 d	NOAEL	Mallard	94.7% a.i. product
Small mammal	> 16,970	mg/kg bw	8 d	LD_{50}	Bobwhite quail	94.7% a.i. product
Small mammal - dermal	634	mg/kg bw-day	20 w	NOAEL	Bobwhite quail	94.3% a.i. product
Small mammal - ingestion	42.2	mg/kg bw-day	2 generation	NOAEL	Rat	93% a.i. product
Large mammal	> 5,000	mg/kg bw		LD_{50}	Rabbit	96.4% a.i. product
Large mammal	3,300	mg/kg bw		LD_{50}	Rat	Water exposure; no diet available; 98.1% a.i. product
Terrestrial Plants						
Typical species - direct spray, drift, dust	0.0008	lb a.i./acre	14 d	EC_{25}	Turnip	Based on emergence
RTE species - direct spray, drift, dust	0.0003	lb a.i./acre	14 d	NOAEL	Turnip	Extrapolated from EC25
Typical species - runoff	0.028	lb a.i./acre	14 d	NOAEL	Tomato	No germination data; based on emergence
RTE species - runoff	0.0001	lb a.i./acre	NR	NOAEL	Turnip	No germination data; based on emergence
Aquatic Species						
Aquatic invertebrates	15	mg/L	48 h	EC_{50}	*D. magna*	94.7% a.i. product
Fish	106	mg/L	96 h	LC_{50}	Rainbow trout	97.4% a.i. product
Aquatic plants and algae	0.1	mg/L	5 d	EC_{50}	Green algae	99.5% a.i. product
Aquatic invertebrates	9.7	mg/L	48 h	NOAEL	*D. magna*	94.7% a.i. product
Fish	16	mg/L	96 h	NOAEL	Bluegill sunfish	97.4% a.i. product
Aquatic plants and algae	0.0078	mg/L	5 d	NOAEL	Green algae	99.5% a.i. product

TABLE C-12 (Cont.)
Selected Toxicity Reference Values for Overdrive®

Receptor	Selected TRV[1]	Units	Duration[2]	Endpoint[3]	Species	Notes
Additional Endpoints						
Amphibian	no data					
Amphibian	no data					
Warmwater fish	> 100	mg/L	96 h	LD_{50}	Bluegill	93.7% a.i. product
Warmwater fish	96	mg/L	32 d	NOAEL	Fathead minnow	97% a.i. product
Coldwater fish	> 100	mg/L	96 h	LD_{50}	Rainbow trout	93.7% a.i. product
Coldwater fish	33	mg/L	96 h	NOAEL	Rainbow trout	93.7% a.i. product; extrapolated from 96 h LC_{50}

[1] Piscivorous bird TRV = Large bird chronic TRV; and fish TRV = Lower of coldwater and warmwater fish TRVs.
[2] Duration: h = hours; d = days; w = weeks; m = months; y = years; and NR = not reported.
[3] Toxicity endpoints for terrestrial animals: LD_{50} to address acute exposure, and NOAEL to address chronic exposure. Toxicity endpoints for terrestrial plants: EC_{25} to address direct spray, drift, and dust impacts on typical species; NOAEL to address direct spray, drift, and dust impacts on RTE species; highest germination NOAEL to address surface runoff impacts on typical species; and lowest germination NOAEL to address surface runoff impacts on RTE species. Toxicity endpoints for aquatic receptors: LC_{50} or EC_{50} to address acute exposure (appropriate toxicity endpoint for non-target aquatic plants will be an EC_{50}) and NOAEL to address chronic exposure.

TABLE C-13
Selected Toxicity Reference Values for Sulfometuron Methyl

Receptor	Selected TRV[1]	Units	Duration[2]	Endpoint[3]	Species	Notes
Receptors Included in the Food Web Model						
Terrestrial Animals						
Honeybee	300	µg/bee	48 h	LD_{50}		Extrapolated from NOAEL; 99.8% a.i. product
Large bird	> 2,300	mg a.i./kg bw	8 d	LD_{50}	Mallard	
Large bird	153	mg a.i./kg bw-day	8 d	NOAEL	Mallard	Extrapolated from LD_{50}
Piscivorous bird	153	mg a.i./kg bw-day	8 d	NOAEL	Mallard	
Small bird	> 16970	mg a.i./kg bw	8 d	LD_{50}	Bobwhite quail	95.2% a.i. product
Small bird	1,131	mg a.i./kg bw-day	8 d	NOAEL	Bobwhite quail	Extrapolated from $LD_{50;}$ 95.2% a.i. product
Small mammal	18	mg a.i./kg bw-day	18 m	NOAEL	Mouse	
Small mammal - dermal	> 8,000	mg a.i./kg bw		LD_{50}	Rabbit	
Small mammal - ingestion	> 5,000	mg a.i./kg bw		LD_{50}	Rat	Water exposure; no diet available
Large mammal	> 5,000	mg a.i./kg bw		LD_{50}	Rat	Small mammal
Large mammal	28	mg a.i./kg bw-day	1 y	NOAEL	Dog	
Terrestrial Plants						
Typical species-direct spray, drift, dust	0.22	lb a.i./acre		EC_{25}	White mustard	Growth
RTE species-direct spray, drift, dust	0.000028	lb a.i./acre	14 d	NOAEL	Sorghum	Based on seed emergence
Typical species – runoff	1.12	lb a.i./acre		NOAEL	Leafy spurge	Based on seed emergence
RTE species – runoff	0.000028	lb a.i./acre		NOAEL	Sorghum, sugar beet	Based on seed emergence
Aquatic Species						
Aquatic invertebrates	802	mg/L		LC_{50}	Cladoceran	93% a.i. product
Fish	> 148	mg/L	96 h	LC_{50}	Rainbow trout	99.6% a.i. product
Aquatic plants and algae	0.00012	mg a.i./L		EC_{50}	Water milfoil	Based on root mass
Aquatic invertebrates	6.1	mg/L	21 d	NOAEL	Water flea (*D. magna*)	Extrapolated from EC_{50}; 99.1% a.i. product
Fish	0.71	mg/L	chronic	NOAEL	Fathead minnow	95% a.i. product
Aquatic plants and algae	0.00004	mg a.i./L		NOAEL	Water milfoil	Extrapolated from EC_{50}

TABLE C-13 (Cont.)
Selected Toxicity Reference Values for Sulfometuron Methyl

Receptor	Selected TRV[1]	Units	Duration[2]	Endpoint[3]	Species	Notes
Additional Endpoints						
Amphibian	4.2	mg/L		EC_{50}	African clawed frog	85% a.i. product
Amphibian	0.1	mg/L	chronic	NOAEL	African clawed frog	85% a.i. product
Warmwater fish	> 150	mg/L		LC_{50}	Bluegill sunfish	99.6% a.i. product
Warmwater fish	0.71	mg/L	chronic	NOAEL	Fathead minnow	95% a.i. product
Coldwater fish	> 148	mg/L	96 h	LC_{50}	Rainbow trout	99.6% a.i. product
Coldwater fish	49	mg/L	96 h	NOAEL	Rainbow trout	Extrapolated from LC_{50}; 99.6% a.i. product

[1] Piscivorous bird TRV = Large bird chronic TRV; and fish TRV = Lower of coldwater and warmwater fish TRVs.
[2] Duration: h = hours; d = days; w = weeks; m = months; y = years; and NR = not reported.
[3] Toxicity endpoints for terrestrial animals: LD_{50} to address acute exposure, and NOAEL to address chronic exposure. Toxicity endpoints for terrestrial plants: EC_{25} to address direct spray, drift, and dust impacts on typical species; NOAEL to address direct spray, drift, and dust impacts on RTE species; highest germination NOAEL to address surface runoff impacts on typical species; and lowest germination NOAEL to address surface runoff impacts on RTE species. Toxicity endpoints for aquatic receptors: LC_{50} or EC_{50} to address acute exposure (appropriate toxicity endpoint for non-target aquatic plants will be an EC_{50}) and NOAEL to address chronic exposure.

TABLE C-14
Selected Toxicity Reference Values for Tebuthiuron

Receptor	Selected TRV[1]	Units	Duration[2]	Endpoint[3]	Species	Notes
Receptors Included in the Food Web Model						
Terrestrial Animals						
Honeybee	30	μg/bee	NR	LD_{50}		
Large bird	> 2,545	mg a.i./kg bw	8 d	LD_{50}	Mallard	
Large bird	1,000	mg a.i./kg bw-day	30 d	NOAEL	Chicken	
Piscivorous bird	1,000	mg a.i./kg bw-day	30 d	NOAEL	Chicken	
Small bird	> 15,440	mg a.i./kg bw	8 d	LD_{50}	Bobwhite quail	
Small bird	1,029	mg a.i./kg bw-day	8 d	NOAEL	Bobwhite quail	Extrapolated from LD_{50}
Small mammal	7	mg a.i./kg bw-day	2 generations	NOAEL	Rat	
Small mammal - dermal	> 5,000	mg a.i./kg bw	NR	LD_{50}	Rabbit	
Small mammal - ingestion	58	mg a.i./kg bw	acute	LD_{50}	Mouse	Water exposure; no diet available
Large mammal	> 500	mg a.i./kg bw	acute	LD_{50}	Dog	
Large mammal	12.5	mg a.i./kg bw-day	90 d	NOAEL	Dog	Water exposure; no diet available
Terrestrial Plants						
Typical species - direct spray, drift, dust	0.03	lb a.i./acre	NR	EC_{25}	Cabbage	Based on seed emergence
RTE species-direct spray, drift, dust	0.01	lb a.i./acre	NR	NOAEL	Cabbage	Extrapolated from EC_{25}; based on seed emergence
Typical species – runoff	> 6	lb a.i./acre	5 d	NOAEL	10 species	Based on seed germination
RTE species – runoff	0.01	lb a.i./acre	5d	NOAEL	Cabbage	Extrapolated from EC_{25}; based on seed emergence
Aquatic Species						
Aquatic invertebrates	802	mg/L		LC_{50}	Cladoceran	93% a.i. product
Fish	> 148	mg/L	96 h	LC_{50}	Rainbow trout	99.6% a.i. product
Aquatic plants and algae	0.00012	mg a.i./L		EC_{50}	Water milfoil	Based on root mass
Aquatic invertebrates	6.1	mg/L	21 d	NOAEL	Water flea (*D. magna*)	Extrapolated from EC_{50}; 99.1% a.i. product
Fish	0.71	mg/L	chronic	NOAEL	Fathead minnow	95% a.i. product
Aquatic plants and algae	0.00004	mg a.i./L		NOAEL	Water milfoil	Extrapolated from EC_{50}

TABLE C-14 (Cont.)
Selected Toxicity Reference Values for Tebuthiuron

Receptor	Selected TRV[1]	Units	Duration[2]	Endpoint[3]	Species	Notes
Additional Endpoints						
Amphibian	< 398	mg/L	96 h	LC_{50}	Bullfrog	> 97% a.i. product
Amphibian	133	mg/L	96 h	NOAEL	Bullfrog	Extrapolated from LC_{50}
Warmwater fish	112	mg/L	96 h	LC_{50}	Bluegill sunfish	~ 100% a.i. product
Warmwater fish	> 9.3	mg a.i./L	33 d	NOAEL	Fathead minnow	Growth
Coldwater fish	115	mg/L	96 h	LC_{50}	Rainbow trout	> 97% a.i. product
Coldwater fish	26	mg/L	45 d	NOAEL	Rainbow trout	Growth; 98% a.i. product

[1] Piscivorous bird TRV = large bird chronic TRV; and fish TRV = Lower of coldwater and warmwater fish TRVs.
[2] Duration: h = hours; d = days; w = weeks; m = months; y = years; and NR = not reported.
[3] Toxicity endpoints for terrestrial animals: LD_{50} to address acute exposure, and NOAEL to address chronic exposure. Toxicity endpoints for terrestrial plants: EC_{25} to address direct spray, drift, and dust impacts on typical species; NOAEL to address direct spray, drift, and dust impacts on RTE species; highest germination NOAEL to address surface runoff impacts on typical species; and lowest germination NOAEL to address surface runoff impacts on RTE species. Toxicity endpoints for aquatic receptors: LC_{50} or EC_{50} to address acute exposure (appropriate toxicity endpoint for non-target aquatic plants will be an EC_{50}) and NOAEL to address chronic exposure.

application rate for ingestion of contaminated prey by the small mammalian herbivore, the large mammalian herbivore, the large avian herbivore, and the large carnivorous mammal, with the exception of the large mammalian herbivore with an RQ of 1.3, these RQs were below LOC for acute risk.

Risk quotients for chronic ingestion scenarios were below the associated LOC of 1, except the ingestion of contaminated prey by the small avian insectivore and the large mammalian carnivore. Chronic RQs for the small mammalian herbivore and the large avian herbivore were just above the LOC at the maximum application rate. The large mammalian herbivore scenario resulted in elevated chronic RQs at both the typical and maximum application rates.

Therefore, direct spray impacts may pose a risk to insects and large herbivores, primarily when the maximum application rate is used.

Non-target Plants – Terrestrial and Aquatic

As expected, because of the mode of action of herbicides, RQs for non-target terrestrial and aquatic plants impacted by direct spray were above the plant LOC of 1 for all modeled scenarios. RQs for direct spray of non-target terrestrial plants ranged from 1,740 to 15,000. RQs for non-target aquatic plants impacted by accidental direct spray of the pond or stream ranged from 66 to 2,924. Therefore, direct spray impacts are likely to pose a risk to plants in both aquatic and terrestrial environments.

Fish and Aquatic Invertebrates

Acute toxicity RQs for aquatic invertebrates in the pond were below the most conservative LOC of 0.05 (acute RTE species), indicating that direct spray impacts are not likely to pose a risk to these aquatic species. The predicted acute toxicity RQs for fish and aquatic invertebrates in the stream were above the most conservative LOC of 0.05 (acute risk RTE species). These results indicate the potential for acute risk to aquatic species, especially RTE species, in a stream accidentally sprayed with bromacil.

The chronic RQs for the accidental direct spray over the pond and stream scenarios were below the most conservative chronic LOC (0.5; chronic risk RTE species) for all aquatic invertebrate scenarios. These results indicate that impacts from direct spray are generally not likely to pose chronic risk to these aquatic species. However, chronic RQs for fish in the pond and stream impacted by accidental direct spray were above the chronic LOCs for RTE species and general chronic risk in most scenarios. This indicates the potential for chronic risk to fish due to accidental direct spray.

Off-site Drift

Non-target Plants – Terrestrial and Aquatic

All of the RQs for non-target terrestrial plants affected by off-site drift were above the plant LOC of 1. The RQs ranged from 1.17 (predicted 900 ft from application with a low boom at the typical application rate) to 312 (predicted 25 ft from application with a high boom at the maximum application rate). These results indicate that impacts from off-site drift pose a risk to non-target terrestrial plant species within 900 ft of the application area.

The majority of the RQs for non-target aquatic plants affected by off-site drift at the typical application rate were below the plant LOC of 1. However, RQs above the LOC were predicted for six chronic waterbody scenarios (25 ft from low boom applications, 25 and 100 ft from high boom applications in both the pond and the stream) and one acute stream scenario (25 ft from high boom application).

At the maximum application rate, off-site drift to the stream and pond resulted in elevated acute RQs 25 ft from low-boom applications and 25 and 100 ft from high-boom applications. Elevated chronic RQs were predicted in both waterbodies for these three scenarios as well as for the scenario of 100 ft from low boom applications. These results indicate that impacts from off-site drift may pose a risk to aquatic plants within 100 ft of the application area. In addition, slightly more elevated risks were predicted in the stream than in the pond.

Fish and Aquatic Invertebrates

Acute toxicity RQs for fish and aquatic invertebrates were all below the most conservative LOC of 0.05 (acute RTE species). All chronic RQs were well below the LOC for chronic risk to RTE species (0.5). These results indicate that impacts from off-site drift are not likely to pose acute or chronic risk to these aquatic species.

Piscivorous Birds

Risk quotients for piscivorous birds were all well below the most conservative terrestrial animal LOC (0.1), indicating that this scenario is not likely to pose a risk to piscivorous birds.

Surface Runoff

Non-target Plants – Terrestrial and Aquatic

Risk quotients for typical non-target terrestrial plant species affected by surface runoff were all below the plant LOC of 1, indicating that transport due to surface runoff is not likely to pose a risk to these receptors. Most RQs for RTE non-target terrestrial plant species were also below the plant LOC of 1; however, four scenarios did result in elevated RQs. These scenarios were for the base watershed with clay soils and greater than 100 in of rain per year at the maximum application rate (between 150 and 250 in of rain per at the typical application rate). Therefore, there is potential for risk to RTE plant species in this watershed type with high amounts of precipitation. This scenario is unlikely on most public lands because of arid and semi-arid conditions.

Risk quotients for non-target aquatic plants impacted by surface runoff exceeded the plant LOC for nearly all pond scenarios. Acute RQs for non-target aquatic plants in the stream were also above the plant LOC of 1 in 33 of the 42 scenarios at the typical application rate. At the maximum application rate, elevated RQs occurred in 36 of the 42 scenarios. These results indicate the likelihood for acute impacts to aquatic plants in the stream.

Chronic RQs in the stream were generally below the plant LOC at the typical application rate, except in the base watershed with sandy soils and precipitation of more than 50 in per year and in the larger application areas (100 and 1,000 ac). Most chronic stream RQs were above the plant LOC when the maximum application rate was considered. The only scenarios below this LOC were the base watershed with sand, clay, or loam soils and less than 25 in of rain per year, the base watershed with clay-loam soil and 50 in of rain per year, and the 1 acre application area. These results indicate the likelihood for chronic impacts to aquatic plants in the stream under most conditions.

Fish and Aquatic Invertebrates

Acute toxicity RQs for fish and aquatic invertebrates were below the most conservative LOC of 0.05 for all stream scenarios and nearly all pond scenarios. Three acute toxicity RQs for fish in the pond were just over the most conservative LOC, but below the remaining two acute LOCs, with values of 0.052, 0.056, and 0.051. These results indicate that impacts from surface runoff are not likely to pose a risk to most aquatic species, but may pose a slight risk to RTE fish.

Chronic risk RQs for aquatic invertebrates in the pond and stream and fish in the stream were well below the LOC for chronic risk to RTE species (0.5), indicating that these scenarios are not likely to result in long-term risk to these receptors. However, chronic risk RQs for fish in the pond were above the LOC for chronic risk to RTE species in several scenarios. At the typical application rate, elevated RQs ranged from 0.51 in the base watershed with sandy soil and 50 in of precipitation per year to 1.69 in the same watershed with 25 in of precipitation per year. Only two of these RQs were elevated above the chronic risk LOC of 1. At the maximum application rate, RQs over the LOC for chronic risk to RTE species (0.5) occurred in 35 of 42 modeled scenarios. These results indicate the potential for negative chronic impacts to fish in downgradient ponds due to surface runoff, especially at the maximum application rate.

Piscivorous Birds

Risk quotients for the piscivorous bird were all well below the most conservative terrestrial animal LOC (0.1), indicating that this scenario is not likely to pose a risk to piscivorous birds.

Wind Erosion and Transport Off-site

Risk quotients for typical and RTE terrestrial plants in this scenario were all well below the plant LOC (1), indicating that wind erosion is not likely to pose a risk to non-target terrestrial plants.

Accidental Spill to Pond

Risk quotients for the spill scenario were 1.2 for fish, 0.66 for aquatic invertebrates, and 6,330 for non-target aquatic plants. Potential risk to fish, aquatic invertebrates, and non-target aquatic plants were indicated for the truck spills mixed for the maximum application rate.

Potential Risk to Salmonids from Indirect Effects

Qualitative Evaluation of Impacts to Prey

Risk quotients in excess of the acute LOCs for aquatic invertebrates were only observed for the accidental direct spray scenario. All other acute and chronic RQs from accidental spray, off-site drift, and surface runoff scenarios were below the associated LOCs. Because aquatic invertebrates are not predicted to be directly impacted by herbicide concentrations in the stream as a result of normal applications, salmonids are not likely to be indirectly affected by a reduction in prey.

Qualitative Evaluation of Impacts to Vegetative Cover

Aquatic plant RQs for accidental direct spray scenarios were above the plant LOC at both the typical and maximum application rates. Elevated acute and chronic aquatic plant RQs (ranging from 1.01 to 10.7) were also observed as a result of off-site drift within 100 ft of the application area. Acute risk was observed for nearly all surface runoff scenarios. Chronic risk due to surface runoff was also predicted in most scenarios at the maximum application rate. At the typical application rate, minimal chronic risk was observed in the base watershed with sandy soils, and more significant risk was predicted when the application area was increased from 10 acres to 100 and 1,000 acres. These results indicate the potential for a reduction in cover and indirect impacts to salmonids as a consequence of multiple exposure pathways.

Risk quotients for terrestrial plants were elevated above the LOC for accidental direct spray scenarios at both the typical and maximum application rates, indicating the potential for a reduction in this plant community. Risk quotients above the plant LOC for typical terrestrial plants were also observed for all off-site drift scenarios modeled for bromacil. However, non-target terrestrial plant RQs in excess of the LOC, as a result of surface runoff, were only observed for the base watershed with clay soil and at least 100 in of rain per year. All other runoff scenarios predicted RQs less than 1. Therefore, in addition to the potential loss of aquatic vegetative cover, under most scenarios a reduction in riparian vegetation and loss of terrestrial vegetative cover to salmonids are likely results of accidental direct spray and off-site drift of bromacil.

Chlorsulfuron

Direct Spray

Terrestrial Wildlife

Risk quotients for terrestrial wildlife were all below the most conservative LOC of 0.1 (acute RTE species), indicating that direct spray impacts are not likely to pose a risk to terrestrial animals.

Non-target Plants – Terrestrial and Aquatic

As expected, because of the mode of action of herbicides, RQs for non-target terrestrial plants ranged from 746 to 6,667, and RQs for non-target aquatic plants ranged from 7.5 to 196. The lowest RQs were calculated for typical species at the typical application rate, and the highest RQs were calculated for RTE species at the maximum application rate. All of the RQs were above the plant LOC of 1, indicating that direct spray impacts pose a risk to plants in both aquatic and terrestrial environments.

Fish and Aquatic Invertebrates

RQs for fish and aquatic invertebrates were all below the most conservative LOC of 0.05 (acute RTE species), indicating that direct spray impacts are not likely to pose a risk to these aquatic receptors.

Off-site Drift

Non-target Plants – Terrestrial and Aquatic

The majority of the RQs for non-target terrestrial plants affected by off-site drift to soil were above the plant LOC of 1. Only RQs based on off-site drift 900 ft from ground application with a low or a high boom were below the plant LOC. These results indicate the potential for risk to off-site non-target terrestrial plants due to drift.

The majority of the RQs for non-target aquatic plants affected by off-site drift were below the plant LOC of 1. However, chronic toxicity RQs above the LOC occurred with some aerial applications. Chronic toxicity RQs in the stream were elevated for off-site drift 100 ft from applications by plane and helicopter at the maximum application rate.

Fish and Aquatic Invertebrates

Acute toxicity RQs for fish and aquatic invertebrates were all below the most conservative LOC of 0.05

(acute RTE species). All chronic RQs were well below the LOC for chronic risk to RTE species (0.5). These results indicate that off-site drift of chlorsulfuron is not likely to pose acute or chronic risk to these aquatic species.

Piscivorous Birds

Risk quotients for the piscivorous bird were all well below the most conservative terrestrial animal LOC (0.1), indicating that this scenario is not likely to pose a risk to piscivorous birds.

Surface Runoff

Non-target Plants – Terrestrial and Aquatic

Risk quotients for non-target terrestrial plants affected by surface runoff to off-site soil were all below the plant LOC of 1, indicating that transport due to surface runoff is not likely to pose a risk to typical or RTE terrestrial plant species.

Acute RQs for non-target aquatic plants in streams impacted by surface runoff of herbicide were generally below the plant LOC of 1. However, there were some scenarios where values exceeded the plant LOC at the typical or maximum application rate. At both the typical and maximum application rates, elevated RQs were predicted in sandy watersheds with annual precipitation above 100 in. In addition, at the maximum application rate, elevated acute RQs were predicted in the clay watershed with at least 100 in of annual precipitation. These scenarios are unlikely to occur on public lands because of arid and semi-arid conditions. Chronic RQs for non-target aquatic plants in the stream impacted by runoff or overland flow of herbicide were all below the plant LOC of 1. Therefore, it is possible that in some locations aquatic plants in the stream would be at acute risk from surface runoff of chlorsulfuron, but this transport mechanism is not likely to pose a chronic risk to aquatic plant species in the stream.

Risk quotients exceeded the LOC for several pond scenarios at both typical and maximum application rates. Elevated acute RQs based on the typical application rate ranged from 1.11 to 11.8 in the following scenarios: surface runoff through sandy soil in the base watershed with annual precipitation above 50 inches; through clay watersheds with annual precipitation above 25 inches; through loam watersheds with annual precipitation above 200 inches; and through three variations of the base watershed with 50 in of rain per year (silt loam, silt, and clay loam soils). Elevated acute RQs ranging from 1.54 to 35.3 were predicted at the maximum application rate resulting from surface runoff through the same scenarios that generated elevated RQs at the typical application rate, as well as sandy watersheds with at least 25 inches of precipitation per year and loam watersheds with at least 100 inches of precipitation per year. Of the 42 scenarios modeled for the pond, acute RQs were elevated above the LOC for 16 scenarios at the typical application rate and 19 scenarios at the maximum application rate. Chronic RQs ranging from 1.1 to 4.4 were predicted due to surface runoff to the pond at the typical application rate, and chronic RQs ranging from 1.2 to 13.1 were predicted due to surface runoff at the maximum application rate. Of the 42 scenarios modeled, chronic RQs were elevated above the LOC for 4 scenarios with the typical application rate and 12 scenarios with the maximum application rate. This suggests that aquatic plants in the pond are at acute and chronic risk from surface runoff of chlorsulfuron resulting from most application scenarios.

Fish and Aquatic Invertebrates

Acute toxicity RQs for fish and aquatic invertebrates were all below the most conservative LOC of 0.05 (acute RTE species) for all pond and stream scenarios, and chronic toxicity RQs were well below the LOC for chronic risk to RTE species (0.5), indicating that these scenarios are not likely to result in long-term risk to aquatic animals in streams or ponds.

Piscivorous Birds

Risk quotients for the piscivorous bird were all well below the most conservative terrestrial animal LOC (0.1), indicating that this scenario is not likely to pose a risk to piscivorous birds.

Wind Erosion and Transport Off-site

Risk quotients for typical and RTE terrestrial plants were all well below the plant LOC (1), indicating that wind erosion is not likely to pose a risk to non-target terrestrial plants.

Accidental Spill to Pond

Risk quotients for the spill scenarios resulted in elevated RQs only for non-target aquatic plants, with fish and aquatic invertebrates generating values below the identified LOC. Potential risk to non-target aquatic plants was indicated for both the truck and helicopter spills mixed for the maximum application rate.

Potential Risk to Salmonids from Indirect Effects

Qualitative Evaluation of Impacts to Prey

No RQs in excess of the appropriate acute or chronic LOCs were observed for aquatic invertebrates in any of the stream scenarios. Because aquatic invertebrates are not predicted to be directly impacted by herbicide concentrations in the stream, salmonids are not likely to be indirectly affected by a reduction in prey.

Qualitative Evaluation of Impacts to Vegetative Cover

Aquatic plant RQs for accidental direct spray scenarios were above the plant LOC at both the typical and maximum application rates, indicating the potential for a reduction in the aquatic plant community. In the unlikely event that a stream was accidentally sprayed, there would be the potential for indirect impacts to salmonids caused by a reduction in available cover.

Minimal elevated aquatic plant chronic RQs (RQs of 1.07 and 1.23) were also observed as a result of off-site drift from selected aerial applications of chlorsulfuron, indicating the potential for a reduction in cover overtime. No elevated aquatic plant acute RQs were predicted due to drift. No RQs in excess of the LOC were observed for aquatic plant species in the stream for any of the surface runoff scenarios.

Risk quotients for terrestrial plants were elevated above the LOC for accidental direct spray scenarios at both the typical and maximum application rates, indicating the potential for a reduction in this plant community. In addition, RQs for typical terrestrial plants were observed above the plant LOC (ranging from 1.52 to 21.4) for nearly all scenarios as a result of off-site drift. No RQs in excess of the LOC were observed for terrestrial plant species for any of the surface runoff scenarios. These results indicate the potential for a reduction in riparian cover under selected conditions.

Diflufenzopyr

Direct Spray

Terrestrial Wildlife

Risk quotients for terrestrial wildlife were all below the most conservative LOC of 0.1 (acute RTE species), indicating that direct spray impacts are not likely to pose a risk to terrestrial animals.

Non-target Plants – Terrestrial and Aquatic

Risk quotients for non-target terrestrial plants ranged from 93.8 to 333, and RQs for non-target aquatic plants ranged from 0.084 to 7.19. As expected because of the mode of action of herbicides, all of the terrestrial plant RQs were above the plant LOC of 1, indicating that direct spray impacts may pose a risk to these receptors. Aquatic plant RQs were below the plant LOC in all acute scenarios and above the plant LOC in all chronic scenarios, indicating the potential for long-term harm to these receptors.

Fish and Aquatic Invertebrates

All acute and chronic toxicity RQs for fish and aquatic invertebrates were below the most conservative LOC (0.05 for acute risk RTE species; 0.5 for chronic risk RTE species). These results indicate that impacts from direct spray are generally not likely to pose acute or chronic risk to these aquatic species.

Off-site Drift

Non-target Plants – Terrestrial and Aquatic

The majority of the RQs for non-target terrestrial plants affected by off-site drift to soils were below the plant LOC of 1. However, RQs did exceed the LOC (ranging from 1.13 to 7.0) for several application scenarios. Off-site drift 25 ft from ground application with a low or high boom at the typical and maximum application rates resulted in RQs above the LOC for both typical and RTE species. Additional risk was also predicted for RTE species within 100 ft of a low-boom application at the typical application rate, and within 100 ft of a high-boom application at the typical and maximum application rates. Therefore, there is potential risk to typical terrestrial plant species from off-site drift of diflufenzopyr within 25 ft of the application, and there is risk to RTE terrestrial plant species from herbicide drift within 100 ft of the application area.

All RQs for non-target aquatic plants affected by off-site drift were below the plant LOC of 1, indicating this transport mechanism is not likely to impact these receptors.

Fish and Aquatic Invertebrates

Acute toxicity RQs for fish and aquatic invertebrates were all below the most conservative LOC of 0.05 (acute RTE species), and all chronic RQs were well below the LOC for chronic risk to RTE species (0.5). These results indicate that impacts from off-site drift are not likely to pose acute or chronic risk to these aquatic species.

Piscivorous Birds

Risk quotients for piscivorous birds were all well below the most conservative terrestrial animal LOC (0.1), indicating that this scenario is not likely to pose a risk to these species.

Surface Runoff

Non-target Plants – Terrestrial and Aquatic

Risk quotients for typical non-target terrestrial plant species affected by surface runoff to off-site soil were all below the plant LOC of 1, indicating that transport due to surface runoff is not likely to pose a risk to these receptors. Most RQs for RTE non-target terrestrial plant species were also below the plant LOC of 1; however, a couple scenarios did result in elevated RQs at the typical or maximum application rate. These scenarios were surface runoff in the base watershed with clay soils and more than 25 inches of precipitation per year and runoff in the base watershed with silt-loam, silt, or clay-loam soils and 50 inches of precipitation per year. This indicates the potential for risk to RTE plant species in selected watersheds at the typical and maximum application rates with greater than 25 inches of precipitation per year.

Acute and chronic RQs for non-target aquatic plants in the pond and streams impacted by overland flow of diflufenzopyr were all below the plant LOC of 1. In addition, acute RQs for non-target aquatic plants in the pond were also below the plant LOC. These results indicate that this transport mechanism is not likely to pose a risk to aquatic plant species under these conditions.

Fish and Aquatic Invertebrates

Acute and chronic toxicity RQs for fish and aquatic invertebrates in ponds and streams were all below the most conservative LOCs (0.05 and 0.5, respectively), indicating that impacts from surface runoff are not likely to pose a risk to these aquatic species.

Piscivorous Birds

Risk quotients for piscivorous birds were all well below the most conservative terrestrial animal LOC (0.1), indicating that this scenario is not likely to pose a risk to piscivorous birds.

Wind Erosion and Transport Off-site

Risk quotients for typical and RTE terrestrial plants were all well below the plant LOC (1), indicating that wind erosion is not likely to pose a risk to non-target terrestrial plants.

Accidental Spill to Pond

Risk quotients for the spill scenario ranged from 0.00338 for fish and 0.0239 for aquatic invertebrates to 3.59 for non-target aquatic plants. Potential risk to non-target aquatic plants was indicated for the truck spill with diflufenzopyr mixed for the maximum application rate.

Potential Risk to Salmonids from Indirect Effects

Qualitative Evaluation of Impacts to Prey

No RQs in excess of the appropriate acute or chronic LOCs were observed for aquatic invertebrates in any of the stream scenarios. Because aquatic invertebrates are not predicted to be directly impacted by herbicide concentrations in the stream, salmonids are not likely to be indirectly affected by a reduction in prey.

Qualitative Evaluation of Impacts to Vegetative Cover

Chronic aquatic plant RQs for accidental direct spray scenarios were above the plant LOC at both the typical and maximum application rates, indicating the potential for a reduction in the aquatic plant community over time. Therefore, in the unlikely event that a stream is accidentally sprayed, there would be the potential for indirect impacts to salmonids caused by a reduction in available cover.

No RQs in excess of the LOC were observed for stream aquatic plant species for any of the off-site drift or surface runoff scenarios.

Risk quotients for terrestrial plants were elevated above the LOC for accidental direct spray scenarios at both the typical and maximum application rates, indicating the potential for a reduction in this plant community.

Risk quotients for typical terrestrial plants were also observed above the plant LOC (ranging from 1.13 to 7.00) as a result of off-site drift. Off-site drift 25 ft from ground application with a low or high boom resulted in RQs above the LOC at the typical and maximum application rates for both typical and RTE species. Additional risk was also predicted for RTE species within 100 ft of a low-boom application at the typical application rate and within 100 ft of a high-boom application at the typical and maximum application rates. These results indicate the potential for a reduction in riparian cover under selected conditions.

No RQs in excess of the LOC were observed for terrestrial plant species for any of the surface runoff scenarios.

Diquat

Direct Spray

Terrestrial Wildlife

Acute RQs for terrestrial wildlife were above the most conservative LOC of 0.1 (acute RTE species) for several scenarios. Accidental direct spray of the pollinating insect resulted in elevated RQs at both the typical and maximum application rates. Risk was also predicted for the pollinating insect as a result of indirect contact with foliage accidentally sprayed at the maximum application rate. No risks to the small mammal were predicted due to direct spray or indirect contact with foliage.

Acute exposure RQs were elevated above the associated LOC (0.1; acute risk RTE species) for two scenarios using the typical application rate (large mammalian herbivore and small avian insectivore) and for five scenarios at the maximum application rate (large and small mammalian herbivores, large avian herbivore, large mammalian carnivore, and small avian insectivore).

Chronic exposure RQs were elevated above the associated LOC (1.0) for three scenarios using the typical application rate (large and small mammalian herbivore and large avian herbivore) and for four scenarios at the maximum application rate (large mammalian and avian herbivores, small mammalian herbivore, and small avian insectivore).

This evaluation indicates that accidental direct spray impacts may pose a risk to insects, birds, and wildlife, primarily when the maximum application rate is used.

Non-target Plants – Terrestrial and Aquatic

As expected, because of the mode of action of herbicides, RQs for non-target terrestrial and aquatic plants impacted by direct spray were above the plant LOC of 1 for all modeled scenarios. Risk quotients for direct spray of non-target terrestrial plants ranged from 213 to 2,500. Risk quotients for non-target aquatic plants impacted by routine application to the pond or accidental direct spray of the stream ranged from 149 to 7,472. Therefore, direct spray impacts pose a risk to plants in both aquatic and terrestrial environments.

Fish and Aquatic Invertebrates

Acute and chronic toxicity RQs for fish and aquatic invertebrates in the pond and stream were all above the most conservative associated LOCs (0.05 for acute risk RTE species; 0.5 for chronic risk RTE species).

These results indicate there is potential risk to aquatic species, especially RTE species, in a pond or stream sprayed with diquat.

Piscivorous Birds

Risk quotients for piscivorous birds were all well below the most conservative terrestrial animal LOC (0.1), indicating that diquat application is not likely to pose a risk to piscivorous birds.

Off-site Drift to Terrestrial Plants

Risk quotients for both typical and RTE terrestrial plant species were elevated over the plant LOC of 1 for several scenarios. At the typical application rate, RQs were elevated for typical and RTE plant species within 900 ft of the aerial application of the herbicide (helicopter and fixed-wing plane) and within 100 ft of ground applications (high boom). At the maximum application rate, RQs for typical plant species were elevated for all aerial applications and within 100 ft of ground applications (low and high booms). Risk quotients for RTE plant species were elevated for all evaluated herbicide applications using the maximum rate, and for all but two scenarios at the typical application rate (low- and high-boom ground applications with 900 foot buffers). These results indicate that potential risk to non-target terrestrial plants exists due to off-site drift during application of this aquatic herbicide.

Accidental Spill to Pond

Potential risks to fish, aquatic invertebrates, and non-target aquatic plants were indicated for the truck and helicopter spills mixed for the maximum application rate. Risk quotients for the truck spill scenario were 19.1 for fish, 102 for aquatic invertebrates, and 19,129 for non-target aquatic plants. Risk quotients for the helicopter spill scenario were higher at 67 for fish, 359 for aquatic invertebrates, and 66,952 for non-target aquatic plants.

Potential Risk to Salmonids from Indirect Effects

Qualitative Evaluation of Impacts to Prey

Risk quotients in excess of the acute LOCs for aquatic invertebrates were observed for the accidental direct spray scenario at both the typical and maximum application rates. This conservative evaluation predicts that fish and aquatic invertebrates would be directly impacted by herbicide concentrations in the stream. Accordingly, their availability as prey item populations may be impacted, and this may result in an indirect effect on salmonids.

Qualitative Evaluation of Impacts to Vegetative Cover

Aquatic plant RQs for accidental direct spray scenarios were above the plant LOC at both the typical and maximum application rates. Therefore, there is the potential for indirect impacts to salmonids due to a reduction in available cover in the unlikely event that a stream is accidentally sprayed.

In addition, RQs for terrestrial plants were elevated above the LOC for accidental direct spray scenarios at both the typical and maximum application rates, indicating the potential for a reduction in this plant community and potential indirect impacts to salmonids due to a loss of riparian cover.

Diuron

Direct Spray

Terrestrial Wildlife

In general, most acute RQs for terrestrial wildlife were below the most conservative LOC of 0.1 (acute RTE species). However, direct spray of the pollinating insect resulted in elevated RQs at both the typical and maximum application rates. In addition, at the maximum application rate, risk was also predicted for the pollinating insect from indirect contact with foliage impacted by direct spray.

Risk quotients for acute ingestion scenarios were below the most conservative LOC (0.1; acute risk RTE species) when herbicide is applied at the typical rate, but above the LOC in all cases at the maximum application rate.

Risk quotients for chronic ingestion scenarios were above the associated LOC of 1.0 for three receptors (the small and large mammalian herbivores and the large mammalian carnivore) when herbicide is applied at the typical or maximum application rate. At the maximum application rate, elevated RQs were also predicted for the small mammalian herbivore and the large avian herbivore.

This evaluation indicates that direct spray impacts may pose a risk to insects, birds, and mammals, primarily when the maximum application rate is used.

Non-target Plants – Terrestrial and Aquatic

Risk quotients for non-target terrestrial plants ranged from 75 to 20,000, and RQs for non-target aquatic plants ranged from 517 to 25,474. All of the RQs were above the plant LOC of 1.0, indicating that, as expected, direct spray impacts pose a risk to plants in both aquatic and terrestrial environments.

Fish and Aquatic Invertebrates

Acute and chronic toxicity RQs for fish and aquatic invertebrates in the pond and stream were above the most conservative LOCs (0.05 for acute risk RTE species; 1.0 for chronic risk), indicating that direct spray impacts may pose a risk to these aquatic species.

Off-site Drift

Non-target Plants – Terrestrial and Aquatic

Many of the RQs for non-target terrestrial plants affected by off-site drift to off-site soils were above the plant LOC of 1.0. For typical terrestrial plant species, elevated RQs were predicted at the typical application rate 25 ft from application with a high boom and at the maximum application rate within 100 ft from application with a low or high boom. Elevated RQs were predicted for RTE terrestrial plant species under all off-site drift scenarios. These results indicate that terrestrial plants, particularly RTE species, located near applications areas may be impacted by herbicide drift.

The majority of the RQs for non-target aquatic plants affected by off-site drift were above the plant LOC of 1.0. The only scenario that did not consistently predict elevated RQs was off-site drift 900 ft from the application area. More elevated RQs were predicted with application using the high boom. These results indicate that off-site drift may impact aquatic plants in waterbodies adjacent to application areas.

Fish and Aquatic Invertebrates

Acute toxicity RQs for fish were below the most conservative LOC of 0.05 (acute RTE species) for all scenarios except one (off-site drift to the stream 25 ft from the maximum application with a high boom). Acute toxicity RQs for aquatic invertebrates were generally also below the most conservative LOC of 0.05 (acute RTE species). However, off-site drift to the pond and stream within 25 ft of a low-boom application or within 100 ft of a high-boom application at the maximum application rate predicted elevated RQs for aquatic invertebrates. Off-site drift within 25 ft of a high-boom application at the typical application rate also predicted a slightly elevated RQ (0.077) in the stream. These results indicate the potential for acute risk to fish and invertebrates due to off-site drift under selected application conditions.

Most chronic RQs were well below the LOC for chronic risk to RTE species (0.5). However, application at the maximum application rate resulted in elevated RQs for one aquatic invertebrate scenario (in the stream 25 ft from the high-boom application) and three fish scenarios (in the pond 25 ft from the high-boom application and in the stream 25 ft from the low- and high-boom applications). For fish, the only the scenario with an RQ above the chronic LOC of 1.0 was for the stream 25 ft from application at the maximum rate with a high boom. These results indicate minimal potential for chronic risk, except within 25 ft of the application area at the maximum application rate.

Piscivorous Birds

Risk quotients for piscivorous birds were all well below the most conservative terrestrial animal LOC (0.1), indicating that this scenario is not likely to pose a risk to piscivorous birds.

Surface Runoff

Non-target Plants – Terrestrial and Aquatic

Risk quotients for typical non-target terrestrial plant species affected by surface runoff to off-site soil were all below the plant LOC of 1.0, indicating that transport due to surface runoff is not likely to pose a risk to these receptors. Most RQs for RTE non-target terrestrial plant species were also below the plant LOC of 1.0; however, several scenarios did result in elevated RQs. At the typical application rate, RQs for the base watershed with clay soils and between 100 and 250 inches of rain per year (250 inches per year was the maximum rainfall modeled) ranged from 1.0 to 2.85. At the maximum application rate, RQs were elevated above 1.0 for the base watershed with clay soils and at least 50 inches of rain per year, for the base watershed with loam soils and at least 200 inches of rain per year, and for the base watershed with clay-loam soil and 50 inches of rain per year (no other rainfall amounts were modeled for this scenario). This indicates the potential for risk to RTE plant species in certain watersheds (with precipitation greater than 50 inches) at the typical or maximum application rates (these scenarios are unlikely on many public lands because of arid and semi-arid conditions).

Acute and chronic RQs for non-target aquatic plants impacted by herbicide runoff exceeded the plant LOC for nearly all pond scenarios modeled at both the typical and maximum application rates.

Acute RQs for non-target aquatic plants in the stream were also generally above the plant LOC of 1.0. At the typical application rate, elevated RQs occurred in 35 of the 42 scenarios. At the maximum application rate, elevated RQs occurred in 37 of the 42 scenarios. These results indicate the high potential for acute impacts to aquatic plants in the stream.

Chronic RQs in the stream at the typical application rate were above the plant LOC for several scenarios (base watershed with sandy soils and precipitation of more than 25 inches per year; base watershed with clay or loam soils and precipitation of more than 100 inches per year; and 100 and 1,000 ac application areas).

Most chronic stream RQs were above the plant LOC when the maximum application rate was considered. The only scenarios below this LOC were the base application watershed with sandy soils and less than 10 inches of rain per year; the base application watershed with clay or loam soil and less than 25 inches of rain per year; the 1 acre application area; and the base watershed with silt soil and 50 inches of rain per year. These results indicate the potential for chronic impacts to aquatic plants in the stream under most conditions at the maximum application rate.

Fish and Aquatic Invertebrates

Acute toxicity RQs for fish and aquatic invertebrates were above the most conservative LOC of 0.05 (acute RTE species) for nearly all pond scenarios. At the typical application rate, RQs were elevated above 0.05 for fish in 35 of 42 scenarios, and for aquatic invertebrates in 36 of 42 scenarios. At the maximum application rate, this increased to 36 and 38 of 42 scenarios for fish and aquatic invertebrates, respectively. Acute RQs for aquatic invertebrates in the stream were greater than the LOC of 0.05 for most scenarios at the maximum application rate (35 of 42 scenarios) and for high precipitation scenarios at the typical application rate (18 of 42 scenarios). Acute RQs for fish in the stream were greater than the LOC for high precipitation scenarios at the maximum application rate (16 of 42 scenarios) and for high precipitation scenarios in clay soils at the typical application rate (3 of 42 scenarios). This suggests that diuron poses substantial acute risks to aquatic animals in ponds and limited acute risks to aquatic stream animals (i.e., at the maximum application rate and in wet watersheds).

Chronic toxicity RQs in the stream were well below the LOC for chronic risk to RTE species (0.5), indicating that these scenarios are not likely to result in long-term risk to fish or aquatic invertebrates. However, chronic RQs for fish and aquatic invertebrates were elevated above this LOC in several pond scenarios. At the maximum application rate, RQs were elevated above 0.5 for fish in 36 of 42 scenarios, and for aquatic invertebrates in 33 of 42 scenarios. At the typical application rate, RQs were elevated above 0.5 for fish in 30 of 42 scenarios, and for aquatic invertebrates in 10 of 42 scenarios. At the typical application rate, only 10 of the fish RQs and 3 of the aquatic invertebrate RQs were above the chronic LOC of 1 for typical species, indicating significantly less risk to non-RTE species. These results indicate the potential for risk to fish and aquatic invertebrates in the pond, especially RTE species, as a result of surface runoff.

Piscivorous Birds

Risk quotients for the piscivorous bird exposed to surface runoff of diuron were all well below the most conservative terrestrial animal LOC (0.1), indicating that this scenario is not likely to pose a risk to piscivorous birds.

Wind Erosion and Transport Off-site

Risk quotients for typical and RTE terrestrial plants were all well below the plant LOC (1), indicating that wind erosion is not likely to pose a risk to non-target terrestrial plants.

Accidental Spill to Pond

Risk quotients for the spill scenario were 101 for fish, 448 for aquatic invertebrates, and 55,180 for non-target aquatic plants. Therefore, there is the potential for risk to fish, aquatic invertebrates, and non-target aquatic plants under the scenario of a truck spill with diuron mixed for the maximum application rate.

Potential Risk to Salmonids from Indirect Effects

Qualitative Evaluation of Impacts to Prey

Risk quotients in excess of the acute and chronic LOCs for aquatic invertebrates were observed for the accidental direct spray scenario.

The off-site drift scenarios predicted elevated RQs for aquatic invertebrates (mostly RTE species) under selected conditions, primarily within 100 ft of the application area at the maximum application rate (two scenarios predicted acute risk to aquatic invertebrates in the stream at the typical application rate and for a buffer of more than 100 and less than 900 ft). All chronic RQs for aquatic invertebrates in the stream impacted by surface runoff were below the associated LOCs. Acute RQs for these surface runoff scenarios were elevated above the most conservative LOC for several scenarios; most significantly for aquatic invertebrates at the maximum application rate.

Because fish may be directly impacted by herbicide concentrations in the stream as a result of normal applications, their availability as prey item populations may be impacted, and there may be an indirect effect on salmonids.

Qualitative Evaluation of Impacts to Vegetative Cover

Aquatic plant RQs for accidental direct spray scenarios were above the plant LOC at both the typical and maximum application rates, indicating the potential for a reduction in the aquatic plant community. Therefore there is the potential for indirect impacts to salmonids due to a reduction in available cover in the unlikely event that a stream is accidentally sprayed.

Elevated aquatic plant RQs were also observed in the stream scenario as a result of off-site drift of the ground application of the herbicide more than 100 and less than 900 ft from the stream, indicating the potential for a reduction in cover, most significantly at the maximum application rate (chronic risk to aquatic plants are also predicted with greater than a 900-foot buffer at the maximum application rate). Elevated RQs were also predicted for many of the surface runoff scenarios. These results indicate there is the potential for indirect impacts to salmonids due to reduction in available cover due to off-site drift and surface runoff of the applied herbicide.

Risk quotients for typical and RTE terrestrial plants were elevated above the LOC for accidental direct spray scenarios at both the typical and maximum application rates, indicating the potential for a reduction in this plant community. However, as discussed above, this scenario is unlikely to occur as a result of BLM practices and represents a worst-case scenario.

Risk quotients for typical terrestrial plants were also observed above the plant LOC (ranging from 1.1 to 5.19) as a result of off-site drift from the ground application of the herbicide. At the typical application rate, risk was predicted at least 25 ft and less than 100 ft from the application area, and at the maximum application rate, risk was predicted at least 100 ft and less than 900 ft from the application area. Elevated RQs for RTE species were also observed for all modeled application scenarios. These results indicate the potential for a reduction in riparian cover and indirect effects to salmonids due to off-site drift under selected conditions.

No RQs in excess of the LOC were observed for typical terrestrial plant species for any of the surface runoff scenarios. Elevated RQs were predicted for RTE terrestrial plant species under selected surface runoff conditions, primarily in clay or loam soils at high precipitation levels. These results indicate the limited potential for a reduction in riparian cover due to surface runoff, primarily when RTE plant species are present.

Fluridone

Direct Spray

Terrestrial Wildlife

Acute RQs for terrestrial animals were below the most conservative LOC of 0.1 (acute RTE species) for all scenarios. At the maximum application rate, the small mammalian herbivore had an RQ of 2.22, all other RQs were well below the LOC of 1. These results indicate that accidental direct spray impacts are not likely to pose a risk to insects, birds, or mammals.

Non-target Plants – Terrestrial and Aquatic

No toxicity data were identified for non-target terrestrial plant species; therefore, a quantitative evaluation is not possible. However, the ecological incident report described earlier suggests that impacts to terrestrial plants are possible due to unintended contact with fluridone. In the manufacturer's user's guide for the Sonar® aquatic herbicide (Eli Lilly and Company 2003), grasses and some sedges are considered to be "sensitive" or "intermediate" in their tolerance to the herbicide, while rushes tend to be "intermediate" to "tolerant." Shoreline plants, such as willow and cypress, were considered "tolerant," while the tolerance of members of the evening primrose and acanthus families was classified as "intermediate." No concentrations were associated with these qualitative statements. It is the more tolerant shoreline plants that are more likely to come in contact with fluridone during normal pond applications.

For aquatic plants, all of the RQs were below the plant LOC of 1, indicating that direct spray impacts are not predicted to pose a risk to aquatic plants in the stream or the pond.

Fish and Aquatic Invertebrates

Normal application of fluridone within a pond resulted in one RQ elevated over the associated LOC. The acute RQ for aquatic invertebrates in the pond impacted by the maximum application rate of fluridone was 0.11, just above the LOC for acute risk to RTE species (0.05).

Accidental direct spray of fluridone over the stream resulted in elevated acute and chronic RQs. Elevated acute RQs were 0.17 for fish at the maximum application rate, and 0.065 and 0.56 for invertebrates at the typical and maximum application rates, respectively. These RQs were all above the acute risk to RTE species LOC, but below or nearly consistent with the acute high risk LOC. Elevated chronic RQs were 1.5 for fish and 1.8 for invertebrates at the maximum application rate. These RQs were above the LOC for chronic risk to RTE species (0.5) and the LOC for chronic risk (1).

These results indicate there is potential for risk to aquatic species, especially RTE species, in a stream sprayed with fluridone. However, this scenario is not likely to occur as fluridone is reserved for use in ponds.

Piscivorous Birds

Risk quotients for the piscivorous bird were all well below the most conservative terrestrial animal LOC (0.1), indicating that the direct spray scenario is not likely to pose a risk to piscivorous birds.

Off-site Drift to Non-target Terrestrial Plants

As described previously, no toxicity data were identified for non-target terrestrial plant species; therefore, a quantitative evaluation of this scenario is not possible. However, impacts to terrestrial plants are possible due to unintended contact with fluridone.

It may be noted that the concentrations of fluridone predicted due to off-site drift are significantly lower than those modeled for accidental direct spray of fluridone on near-shore terrestrial plants. This comparison indicates that the maximum deposition (100 ft from aerial applications) was only 23.8% of the typical application rate and only 0.87% of the maximum application rate. On average, off-site drift modeled using the typical application rate was less than 10% of the typical application rate used in the direct spray scenario. Off-site drift modeled using the maximum application rate was less than 1% of the maximum application rate used in the direct spray scenario. This indicates the reduction in deposition and associated risks that occur with off-site drift relative to direct accidental spray.

Accidental Spill to Pond

Risk quotients for the truck spill scenario were 1.10 for fish, 3.58 for aquatic invertebrates, and 1.56 for non-target aquatic plants. Risk quotients for the helicopter spill scenario were slightly higher at 3.83, 12.6, and 5.44 for fish, aquatic invertebrates, and non-target aquatic plants, respectively. Therefore, potential risks to fish, aquatic invertebrates, and non-target aquatic plants are indicated for the unlikely events of truck and helicopter spills mixed for the maximum application rate.

Potential Risk to Salmonids from Indirect Effects

Qualitative Evaluation of Impacts to Prey

Risk quotients in excess of the acute LOCs for aquatic invertebrates were observed for the accidental direct spray scenario. This conservative evaluation predicts that aquatic invertebrates may be directly impacted by herbicide concentrations in the stream. Accordingly, their availability as prey item populations may be impacted, and there may be an indirect effect on salmonids.

Qualitative Evaluation of Impacts to Vegetative Cover

Aquatic plant RQs for accidental direct spray scenarios were below the plant LOC at both the typical and maximum application rates, indicating that impacts to the aquatic plant community are not predicted. This evaluation indicates that indirect impacts to salmonids due to a reduction in available cover are unlikely.

It is uncertain whether or not a reduction in riparian cover is likely, but a review of incident reports and the manufacturer's user's guide indicate that shoreline plant species are generally tolerant of fluridone exposures.

Imazapic

Direct Spray

Terrestrial Wildlife

Risk quotients for terrestrial wildlife were all below the most conservative LOC of 0.1 (acute RTE species), indicating that direct spray impacts are not likely to pose a risk to terrestrial animals.

Non-target Plants – Terrestrial and Aquatic

As expected, because of the mode of action of herbicides, RQs for non-target terrestrial plants ranged from 3.1 to 23.8, and RQs for non-target aquatic plants ranged from 0.82 to 41.0. The lowest RQs were calculated for typical species at the typical application rate, and the highest RQs were calculated for RTE species impacted at the maximum application rate. All of the RQs were above the plant LOC of 1, indicating that direct spray impacts pose a risk to plants in both aquatic and terrestrial environments. The only possible exception is the accidental direct spray of the pond at the typical application rate.

Fish and Aquatic Invertebrates

Acute toxicity RQs for fish and aquatic invertebrates were below the most conservative LOC of 0.05 (acute RTE species), indicating that direct spray impacts are not likely to pose a risk to these aquatic species. All chronic RQs were well below the LOC for chronic risk to RTE species (0.5). These results indicate that impacts from direct spray are not likely to pose acute or chronic risk to fish and aquatic invertebrates.

Off-site Drift

Non-target Plants – Terrestrial and Aquatic

The majority of the RQs for non-target terrestrial plants affected by off-site drift to soil were below the plant LOC of 1. However, RQs for several aerial application scenarios did exceed the LOC, with RQs between 1.06 and 6.98. Off-site drift 100 ft from aerial application by plane or helicopter over forested or non-forested lands consistently resulted in an RQ above the LOC at the maximum application rate. In addition, off-site drift 300 ft from the aerial application by a plane over forested land also predicted an elevated RQ at the maximum application rate. Risk at the typical application rate was only predicted for RTE species as a result of drift 100 ft from the aerial application by a plane over forested lands. The predicted RQ of 1.06 was only slightly over the LOC, indicating that use of the typical application rate is not likely to result in risk to most non-target terrestrial species.

The majority of the RQs for non-target aquatic plants affected by off-site drift were below the plant LOC of 1. However, as with impacts to terrestrial plants, RQs above the LOC occurred with some aerial applications resulting in RQs between 1.07 and 2.36. Off-site drift 100 ft from the aerial application by a plane over forested lands consistently resulted in acute and chronic RQs above the LOC at the maximum application rate in the pond and the stream. Off-site drift 100 ft from the aerial application by a helicopter over forested land also predicted an elevated chronic RQ in the stream when applied at the maximum application rate. No elevated RQs were predicted at the typical application rate. Slightly more elevated risks were predicted in the stream than the pond.

Fish and Aquatic Invertebrates

Acute toxicity RQs for fish and aquatic invertebrates were all below the most conservative LOC of 0.05 (acute RTE species). All chronic RQs were well below the LOC for chronic risk to RTE species (0.5). These results indicate that impacts from off-site drift are not likely to pose acute or chronic risk to these aquatic species.

Piscivorous Birds

Risk quotients for the piscivorous bird were all well below the most conservative terrestrial animal LOC (0.1), indicating that off-site drift is not likely to pose a risk to piscivorous birds.

Surface Runoff

Minimal acute risk to non-target aquatic plants in the pond may occur when herbicides are applied at the maximum rate in watersheds with sandy soils and precipitation between 50 and 150 inches per year (RQs were just above 1); chronic risks to non-target aquatic plants in the pond may occur in watersheds with sandy soil and annual precipitation of 25 inches or greater. No risks were predicted for non-target terrestrial plants, non-target aquatic plants in the stream, fish, aquatic invertebrates, or piscivorous birds.

Non-target Plants – Terrestrial and Aquatic

Risk quotients for non-target terrestrial plants affected by surface runoff to off-site soil were all below the plant LOC of 1, indicating that transport due to surface runoff is not likely to pose a risk to typical or RTE terrestrial plant species.

Most RQs for non-target aquatic plants in streams impacted by surface runoff of herbicide were below the plant LOC of 1. The one exception was an acute RQ of 1.03 (just above the LOC), when imazapic is applied at the maximum rate in a watershed with clay soils and at least 250 inches of precipitation per year. However, this is a minimal exceedance; transport due to surface runoff is not likely to pose a risk to aquatic plants species in streams.

Risk quotients exceeded the LOC for several pond scenarios at the maximum application rate. Acute RQs greater than the LOC were predicted at the maximum application rate in the base watershed with sandy soils and at least 25 inches of precipitation per year (RQs ranged up to 4.34), in clay and clay/loam watersheds with at least 50 inches of precipitation per year (RQs ranged up to 7.51), and in loam watersheds with at least 100 inches of precipitation per year (RQs ranged up to 1.97). Acute RQs greater than the LOC were predicted at the typical application rate in watersheds with clay soils and at least 150 inches of precipitation per year (RQs ranged up to 1.72). Chronic RQs were predicted in the base watershed with sandy soil and annual precipitation greater than 25 inches.

Fish and Aquatic Invertebrates

Acute toxicity RQs for fish and aquatic invertebrates were all below the most conservative LOC of 0.05 (acute RTE species) for all pond and stream scenarios, indicating that impacts from surface runoff are not likely to pose a risk to these aquatic species.

Chronic toxicity RQs were well below the LOC for chronic risk to RTE species (0.5), indicating that these scenarios are not likely to result in long-term risk to aquatic animals in streams or ponds.

Piscivorous Birds

Risk quotients for the piscivorous bird were all well below the most conservative terrestrial animal LOC (0.1), indicating that surface runoff is not likely to pose a risk to piscivorous birds.

Wind Erosion and Transport Off-site

Risk quotients for typical and RTE terrestrial plants were all well below the plant LOC (1), indicating that wind erosion is not likely to pose a risk to non-target terrestrial plants.

Accidental Spill to Pond

Risk quotients for the spill scenarios ranged from 0.00681 for fish and aquatic invertebrates to 564 for non-target aquatic plants. Potential risk to non-target aquatic plants was indicated for both the truck and helicopter spills mixed for the maximum application rate.

Potential Risk to Salmonids from Indirect Effects

Qualitative Evaluation of Impacts to Prey

No RQs in excess of the appropriate acute or chronic LOCs were observed for aquatic invertebrates in any of the stream scenarios. Because aquatic invertebrates are not predicted to be directly impacted by herbicide concentrations in the stream, salmonids are not likely to be indirectly affected by a reduction in prey.

Qualitative Evaluation of Impacts to Vegetative Cover

Aquatic plant RQs for accidental direct spray scenarios were above the plant LOC at both the typical and maximum application rates, indicating the potential for a reduction in the aquatic plant community. Therefore, in the unlikely event that a stream is accidentally sprayed, there would be the potential for indirect impacts to salmonids caused by a reduction in available cover.

Slightly elevated aquatic plant RQs (ranging from 1.45 to 2.6) were also observed as a result of off-site drift 100 ft from the aerial application of imazapic, indicating the potential for a reduction in cover. One slightly elevated acute RQ (1.03) was predicted for aquatic plant species in streams impacted from surface runoff in the base watershed with clay soil and annual precipitation of 250 inches. No other elevated acute or chronic RQs were observed for any other surface runoff scenarios.

Risk quotients for terrestrial plants were elevated above the LOC for accidental direct spray scenarios at both the typical and maximum application rates, indicating the potential for a reduction in this plant community.

Risk quotients for typical terrestrial plants were also observed above the plant LOC (ranging from 1.29 to 5.58) as a result of off-site drift from aerial application at the maximum rate. Off-site drift 100 ft from the application area resulted in risk when imazapic was applied from a helicopter or plane over forested and non-forested lands. Potential risk was also indicated 300 ft from the application area when applied by a plane over a forest. Elevated RQs for RTE species were also observed for the same application scenarios. These results also indicate the potential for a reduction in riparian cover under selected conditions.

Overdrive®

Direct Spray

Terrestrial Wildlife

Risk quotients for terrestrial wildlife were all below the most conservative LOC of 0.1 (acute RTE species), indicating that direct spray impacts are not likely to pose a risk to terrestrial animals. Risk quotients for chronic ingestion scenarios were below the associated LOC of 1 for all scenarios, except the ingestion of contaminated food items by the large mammalian herbivore. The scenario predicted elevated RQs of 1.4 and 12.8 at the typical and maximum application rates, respectively. This evaluation indicates that direct spray impacts may pose a risk to large herbivorous mammals, primarily when the maximum application rate is used.

Non-target Plants – Terrestrial and Aquatic

Risk quotients for non-target terrestrial plants ranged from 61.0 to 273, and RQs for non-target aquatic plants ranged from 0.267 to 107. All of the terrestrial plant RQs were above the plant LOC of 1, indicating that direct spray impacts may pose a risk to these receptors. Aquatic plant RQs were below the plant LOC in the acute pond scenarios and above the plant LOC in all other pond and stream scenarios, indicating the potential for acute risk in the stream and long-term risk of harm in the pond and stream.

Fish and Aquatic Invertebrates

Acute toxicity RQs for fish and aquatic invertebrates were below the most conservative LOC of 0.05 (acute RTE species), indicating that direct spray impacts are not likely to pose a risk to these aquatic species. In addition, all chronic toxicity RQs for fish and aquatic invertebrates were well below the LOC for chronic risk to RTE species (0.5). These results indicate that impacts from direct spray are generally not likely to pose acute or chronic risk to these aquatic species.

Off-site Drift

Non-target Plants – Terrestrial and Aquatic

Most of the RQs for typical species of non-target terrestrial plants affected by off-site drift to off-site soils were below the plant LOC of 1. RQs for typical non-target terrestrial plants were elevated (ranging from 1.30 to 2.14) when located 25 ft from ground application with a low boom at the maximum application rate, and with a high boom at the typical or maximum application rate. RQs for several application scenarios with RTE plant species did exceed the LOC, with RQs between 1.09 and 5.74. At the typical application rate, elevated RQs for RTE species were predicted 25 ft from ground application with a low boom and 100 ft from ground application with a high boom. At the maximum application rate, elevated RQs for RTE species were predicted 100 ft from ground application with a low or high boom. These results indicate the potential for risk to typical and RTE species located at least 25 to 100 ft from the application area, depending on the boom height and application rate.

The majority of the RQs typical species of non-target terrestrial plants affected by off-site drift to off-site soils were below the plant LOC of 1. RQs for typical non-target terrestrial plants were elevated (ranging from 1.30 to 2.14, depending on the testing scenario) when located 25 feet from ground application with a low boom at the maximum application rate and with a high boom at the typical or maximum application rate. RQs for several application scenarios with RTE plant species did exceed the LOC, with RQs between 1.09 and 5.74. At the typical application rate, elevated RQs for RTE species were predicted 25 feet from ground application with a low boom and 100 feet from ground application with a high boom. At the maximum application rate, elevated RQs for RTE species were predicted 100 ft from ground application with a low or high boom. These results indicate the potential for risk to typical and RTE species located at least 25 to 100 feet from the application area, depending on the boom height and application rate.

All RQs for non-target aquatic plants affected by off-site drift were below the plant LOC of 1, indicating this transport mechanism is not likely to impact these receptors.

Fish and Aquatic Invertebrates

Acute toxicity RQs for fish and aquatic invertebrates were all below the most conservative LOC of 0.05 (acute RTE species). All chronic RQs were well below the LOC for chronic risk to RTE species (0.5). These results indicate that off-site drift is not likely to pose acute or chronic risk to these aquatic species.

Piscivorous Birds

Risk quotients for the piscivorous bird were all well below the most conservative terrestrial animal LOC (0.1), indicating that this scenario is not likely to pose a risk to piscivorous birds.

Surface Runoff

Non-target Plants – Terrestrial and Aquatic

RQs for typical non-target terrestrial plant species affected by surface runoff to off-site soil were all below the plant LOC of 1, indicating that transport due to surface runoff is not likely to pose a risk to these receptors. Most RQs for RTE non-target terrestrial plant species were also below the plant LOC of 1. However, several scenarios did result in elevated RQs at the typical and maximum application rates. These scenarios included the base watershed with clay soils and more than 25 inches of precipitation per year and three variations on the base watershed with 50 inches of precipitation per year (silt loam, silt, and clay-loam soil). This indicates the potential for risk to RTE plant species in selected watersheds dominated by clay soils at the typical and maximum application rates with greater than 25 inches annual precipitation, with additional risk associated with soils dominated by silt and clay under situations exceeding 50 inches annual precipitation.

Acute and chronic RQs for non-target aquatic plants in the stream impacted by overland flow of herbicide were all below the plant LOC of 1. Acute RQs for non-target aquatic plants in the pond were also below the plant LOC, with one exception. An RQ of 1.04 was predicted at the maximum application rate in the base watershed with sandy soil and 150 inches of precipitation per year. However, this LOC exceedance was minimal and in general these results indicate that this transport mechanism is not likely to pose a risk to aquatic plant species under these conditions.

Chronic RQs exceeded the LOC for several pond scenarios. Elevated RQs ranged from 1.02 to 3.74 at the typical application rate and from 1.15 to 4.06 at the maximum application rate. Risk quotients above the plant LOC of 1 were predicted in 14 scenarios at the typical application rate and 16 scenarios at the maximum application rate. The maximum RQ was predicted in the base watershed with clay soils and 50 inches of precipitation per year.

Fish and Aquatic Invertebrates

Acute toxicity RQs for fish and aquatic invertebrates were all below the most conservative LOC of 0.05 (acute RTE species) for all pond and stream scenarios, indicating that impacts from surface runoff are not likely to pose a risk to these aquatic species.

Chronic risk RQs were well below the LOC for chronic risk to RTE species (0.5), indicating that these scenarios are not likely to result in long-term risk to aquatic animals in the stream or pond.

Piscivorous Birds

RQs for the piscivorous bird were all well below the most conservative terrestrial animal LOC (0.1), indicating that this scenario is not likely to pose a risk to piscivorous birds.

Wind Erosion and Transport Off-site

RQs for typical and RTE terrestrial plants were all well below the plant LOC (1), indicating that wind erosion is not likely to pose a risk to non-target terrestrial plants.

Accidental Spill to Pond

Risk quotients for the spill scenario were 0.0040 for aquatic invertebrates, 0.043 for fish and 14.3 for non-target aquatic plants. These scenarios are highly conservative and represent unlikely and worst case conditions (limited waterbody volume, tank mixed for maximum application). Spills of this magnitude are possible, but are not likely to occur. However, potential risk to non-target aquatic plants was indicated for the truck spill mixed for the maximum application rate.

Potential Risk to Salmonids from Indirect Effects

Qualitative Evaluation of Impacts to Prey

No RQs in excess of the appropriate acute or chronic LOCs were observed for fish or aquatic invertebrates in any of the stream scenarios. Because fish and aquatic invertebrates are not predicted to be directly impacted by herbicide concentrations in the stream, their availability as prey item populations is not likely to be impacted, and there is not likely to be an indirect effect on salmonids due to a lack of prey.

Qualitative Evaluation of Impacts to Vegetative Cover

No RQs in excess of the LOC were observed for aquatic plant species in the stream for any of the off-site drift or surface runoff scenarios. RQs for terrestrial plants were elevated above the LOC for accidental direct spray scenarios at both the typical and maximum application rates, indicating the potential for a reduction in this plant community. RQs for non-target typical and RTE terrestrial plants were also observed above the plant LOC as a result of off-site drift. At the typical application rate, elevated RQs were predicted 25 ft from ground application with a low boom and 100 ft from ground application with a high boom. At the maximum application rate, elevated RQs were predicted 100 ft from ground application with a low or high boom. Risk quotients in excess of the LOC were also predicted for RTE terrestrial plants due to surface runoff in clay watersheds with at least 25 inches of precipitation per year and in clay-loam, silt-loam, and silt watersheds with at least 50 inches of precipitation per year. These results indicate the potential for a reduction in riparian cover under selected conditions as a result of off-site drift and/or surface runoff.

Sulfometuron Methyl

Direct Spray

Terrestrial Wildlife

Risk quotients for terrestrial wildlife were all below the most conservative LOC of 0.1 (acute RTE species), indicating that direct spray impacts are not likely to pose a risk to terrestrial animals.

Non-target Plants – Terrestrial and Aquatic

The majority of RQs for terrestrial and aquatic plants were above the LOC of 1.0. Risk quotients for non-target terrestrial plants ranged from 0.636 to 13,571. Risk quotients for non-target aquatic plants under the accidental direct spray over pond scenario ranged from 131 to 1,060, and the RQs for the accidental direct spray over stream scenario ranged from 650 to 5,320. Therefore, direct spray impacts pose a risk to plants in both aquatic and terrestrial environments. The only exception was the accidental direct spray of the non-target terrestrial plants at the typical application rate.

Fish and Aquatic Invertebrates

Acute and chronic toxicity RQs for fish and aquatic invertebrates were below the most conservative LOCs (0.05 for acute risk RTE species; 0.5 for chronic risk RTE species). These results indicate that direct spray impacts are not likely to pose a risk to fish and aquatic invertebrates.

Off-site Drift

Non-target Plants – Terrestrial and Aquatic

The RQs for typical non-target terrestrial plants affected by off-site drift to soils were all below the plant LOC of 1. However, RQs for all of the RTE non-target terrestrial plants did exceed the LOC, with RQs between 3.43 and 2,536. Risks were more significant for helicopter forested applications than for any other scenario. These results indicate that off-site drift is not likely to result in significant risk to typical non-target terrestrial species, but risks to RTE species may occur.

The majority of the RQs for non-target aquatic plants affected by off-site drift were above the plant LOC of 1, indicating the potential for negative impacts as a result of off-site drift to waterbodies. Acute and chronic RQs in the pond and stream were elevated above the LOC for all aerial application scenarios, suggesting that sulfometuron methyl should not be sprayed aerially or that a buffer zone of more than 900 ft (maximum modeled distance) is needed. Elevated RQs were also predicted 100 ft from ground application areas. These results indicate that off-site drift has the potential to negatively impact aquatic plants, but that impact may be reduced through the use of wider buffer zones or the use of ground rather than aerial applications.

Fish and Aquatic Invertebrates

Acute and chronic toxicity RQs for fish and aquatic invertebrates were all below the most conservative LOCs. These results indicate that impacts from off-site drift are not likely to pose acute or chronic risk to these aquatic species.

Piscivorous Birds

Risk quotients for piscivorous birds were all well below the most conservative terrestrial animal LOC (0.1), indicating that this scenario is not likely to pose a risk to piscivorous birds.

Surface Runoff

Non-target Plants – Terrestrial and Aquatic

Risk quotients for typical non-target terrestrial plant species affected by surface runoff to off-site soil were all below the plant LOC of 1, indicating that transport due to surface runoff is not likely to pose a risk to these species. Risk quotients for RTE species were elevated over the plant LOC for several scenarios. At the typical application rate, elevated RQs for RTE species were predicted in the base watershed with clay soils and more than 10 inches of precipitation per year, in the base watershed with loam soil and 150 inches of precipitation per year, and in the base watershed with two soil variations (silt-loam and clay-loam) at 50 inches of precipitation per year. Chronic RQs were elevated in these same scenarios as well as in the base watershed with loam soil and more than 100 inches of precipitation per year and in the base watershed with silt soil and 50 inches of precipitation per year.

Acute and chronic RQs for non-target aquatic plants in the pond impacted by herbicide runoff were above the plant LOC of 1 for most scenarios. Only watersheds with relatively minimal annual precipitation (5 to 25 inches, depending on soil type) did not predict elevated RQs.

Acute toxicity RQs for aquatic plants in the stream were elevated above the plant LOC in 13 scenarios at the typical application rate and 17 scenarios at the maximum application rate. At the typical rate the following scenarios predicted potential risk: base watershed with sandy soil and greater than 25 inches of annual precipitation, base watershed with clay soil and greater than 100 inches of annual precipitation, and base watershed with loam soil and greater than 150 inches of annual precipitation. The four additional scenarios predicting risk at the maximum application rate were: base watershed with clay soil and greater than 50 inches of annual precipitation; base watershed with loam soil and greater than 100 inches of annual precipitation; base watershed with clay-loam soil and greater than 5 inches of annual precipitation; and base watershed with loam soil, greater than 5 inches of annual precipitation, and a 1,000 acre application area.

Minimal chronic risk to aquatic plants was predicted in the stream, with RQs of 1.02 predicted in two scenarios: base watershed with sand soil, and 200 or 250 inches of annual precipitation. At the maximum application rate, elevated RQs were also predicted in the base watershed with sand soil and greater than 50 inches of annual precipitation.

These results indicate the potential for risk to terrestrial and aquatic plants due to surface runoff under most conditions.

Fish and Aquatic Invertebrates

Acute toxicity RQs for fish and aquatic invertebrates were all below the most conservative LOC of 0.05 (acute RTE species) for all pond and stream scenarios, indicating that impacts from surface runoff are not likely to pose a risk to these aquatic species. In addition, chronic risk RQs were well below the LOC for chronic risk to RTE species (0.5); therefore, surface runoff scenarios are not likely to result in long-term risk to aquatic animals in the stream or pond.

Piscivorous Birds

Risk quotients for the piscivorous bird were all well below the most conservative terrestrial animal LOC (0.1), indicating that surface runoff is not likely to pose a risk to piscivorous birds.

Wind Erosion and Transport Off-site

Risk quotients for typical and RTE terrestrial plants were all well below the plant LOC (1), indicating that wind erosion is not likely to pose a risk to non-target terrestrial plants.

Accidental Spill to Pond

Risk quotients for the spill scenarios were below the associated acute LOC for fish and aquatic invertebrates. However, potential risk to non-target aquatic plants was indicated for both the truck and helicopter spills mixed for the maximum application rate.

Potential Risk to Salmonids from Indirect Effects

Qualitative Evaluation of Impacts to Prey

No RQs in excess of the appropriate acute or chronic LOCs were observed for aquatic invertebrates in any of the stream scenarios; therefore, salmonids are not likely to be indirectly affected by a reduction in prey.

Qualitative Evaluation of Impacts to Vegetative Cover

Aquatic plant RQs for accidental direct spray scenarios were above the plant LOC at both the typical and maximum application rates, indicating the potential for a reduction in the aquatic plant community. Therefore, in the unlikely case that a stream is accidentally sprayed, there would be the potential for indirect impacts to salmonids caused by a reduction in available cover.

Elevated aquatic plant RQs were also observed as a result of off-site drift more than 900 ft from aerial application and within 100 ft of the ground application of the herbicide, indicating the potential for a reduction in cover. Elevated RQs were also predicted for several surface runoff scenarios: primarily in sand watersheds with more than 25 inches of annual precipitation; clay watersheds with more than 50 inches of annual precipitation; and loam watersheds with more than 100 inches of annual precipitation.

Risk quotients for terrestrial plants were elevated above the LOC for accidental direct spray scenarios at both the typical and maximum application rates, indicating the potential for a reduction in riparian cover.

Risk quotients for RTE terrestrial plants were observed above the plant LOC under all of the modeled conditions. No risks were predicted for typical plant species. A similar pattern was also predicted for risks due to surface runoff. These results further indicate the potential for a reduction in riparian cover and possible indirect impacts to salmonids.

Tebuthiuron

Direct Spray

Terrestrial Wildlife

Risk quotients for the pollinating insect were above the most conservative LOC of 0.1 (acute RTE species) for impacts from direct spray of the insect (typical and maximum application rates) and indirect contact with foliage after direct spray (maximum application rate). These results suggest there may be potential for risk to pollinating insects due to direct spray and indirect contact with foliage.

Acute and chronic RQs for terrestrial wildlife impacted by the typical application rate were all below the most conservative LOC of 0.1 (acute RTE species). At the maximum application rate, three acute exposure scenarios (ingestion of contaminated food by small and large mammalian herbivores and the large avian herbivore) predicted RQs above the most conservative LOC (0.1; acute RTE species). The small mammalian herbivore acute RQ of 1.86 was also above the “acute high risk” LOC of 0.5. Two chronic exposure scenarios (ingestion of contaminated food by the smaller large mammalian herbivores) were above the chronic LOC of 1, with an RQ of 3.58 and 3.79, respectively at the maximum application rate. These results indicate that direct spray impacts are not likely to pose a risk to terrestrial animals at the typical application rate. Acute and chronic risk to avian and mammalian herbivores is predicted at the maximum application rate.

Non-target Plants – Terrestrial and Aquatic

Risk quotients for non-target terrestrial plants ranged from 16.7 to 400. Risk quotients for non-target aquatic plants in the pond ranged from 1.12 to 39.5, and RQs for non-target aquatic plants in the stream ranged from 5.6 to 172. Therefore, direct spray impacts pose a risk to plants in both aquatic and terrestrial environments.

Fish and Aquatic Invertebrates

Acute and chronic toxicity RQs for fish, and acute toxicity RQs for aquatic invertebrates were below the most conservative LOC of 0.05 (acute RTE species), indicating that direct spray impacts are not likely to pose acute or chronic risk to fish or acute risk to aquatic invertebrates in the pond or stream.

The chronic RQs for the aquatic invertebrates for the accidental direct spray ranged from 0.56 to 4.48 for the pond scenario and from 2.8 to 22.4 for the stream scenario. These values were greater than the LOC for chronic risk to RTE species (0.5), indicating the potential for risk to these receptors. These results suggest that impacts from direct spray may pose a chronic risk to RTE aquatic invertebrates in the pond or stream.

Off-site Drift

Non-target Plants – Terrestrial and Aquatic

The majority of the RQs for non-target terrestrial plants affected by off-site drift to soil were below the plant LOC of 1. However, RQs for 6 of the 24 application scenarios did exceed the LOC, with RQs between 1.04 and 6.59. Elevated RQs were predicted for RTE species impacted by off-site drift in the following situations: 25 ft from the ground application using a high boom at the typical application rate, 25 ft from the ground application using a low or a high boom at the maximum application rate, and 100 ft from the ground application using a high boom at the maximum application rate.

All of the acute and chronic toxicity RQs for non-target aquatic plants affected by off-site drift in the pond and stream were below the plant LOC of 1. These results indicate that impacts from off-site drift are not likely to pose acute or chronic risk to aquatic plants.

Fish and Aquatic Invertebrates

Acute and chronic toxicity RQs for fish and aquatic invertebrates were all below the most conservative LOCs. These results indicate that impacts from off-site drift are not likely to pose acute or chronic risk to these aquatic species.

Piscivorous Birds

Risk quotients for the piscivorous bird were all well below the most conservative terrestrial animal LOC (0.1), indicating that off-site drift of tebuthiuron is not likely to pose a risk to piscivorous birds.

Surface Runoff

Non-target Plants – Terrestrial and Aquatic

Risk quotients for typical, non-target terrestrial plant species affected by surface runoff to off-site soil were all below the plant LOC of 1, indicating that transport due to surface runoff is not likely to pose a risk to these species. Risk quotients for RTE species were elevated for four scenarios at the typical application rate: runoff from the base watershed with clay soil and annual precipitation of 100, 150, 200, and 250 inches (RQs ranged from 1.24 to 1.43). Risk quotients for RTE species were elevated for eight scenarios at the maximum application rate: runoff from the base watershed with clay soil and annual precipitation of 50, 100, 150, 200, and 250 inches and runoff from the base watershed with annual precipitation of 50 inches and three different soil types—silt-loam, silt, and clay-loam (RQs ranged from 1.34 to 11.5). These risk scenarios involve high levels of precipitation (50 inches and greater), and therefore, are not likely on most public lands, which experience arid and semi-arid conditions.

Acute RQs for non-target aquatic plants in the pond impacted by runoff of herbicide were generally below the plant LOC of 1 at the typical application rate. However, elevated acute RQs were predicted at the typical application rate in the base watershed with sandy soil and precipitation greater than 10 inches per year, in the base watershed with clay soil and precipitation of 50 to 100 inches per year, and in the clay loam variation of the base watershed with 50 inches of precipitation per year (no other precipitation levels modeled for this watershed). At the maximum application rate, elevated RQs were predicted in all but five modeled scenarios. These results indicate there is potential for acute impacts to aquatic plants in the pond under selected conditions at the typical application rate and under most conditions at the maximum application rate.

Chronic RQs for non-target aquatic plants in the pond were elevated above the plant LOC in several scenarios. At the typical application rate, 10 of the 42 RQs were above the plant LOC (ranging from 1.29 to 10.2). The majority of these exceedances occurred in sandy watersheds. At the maximum application rate, 37 of the 42 RQs were above the plant LOC. These results suggest the potential for chronic impacts to aquatic plants in the pond under selected conditions at the typical application rate and under most conditions at the maximum application rate.

Acute RQs for non-target aquatic plants in the stream impacted by surface runoff of herbicide were all below the plant LOC of 1 at the typical application rate. At the maximum application rate, elevated acute RQs were predicted in the base watershed with sandy soil and more than 50 inches of precipitation per year, in the base watershed with clay soil and more than 100 inches of precipitation per year, and in the loam watershed at 50 inches of precipitation per year when the application area was increased to 100 and 1,000 acres. Chronic RQs for non-target aquatic plants in the stream were below the LOC in all scenarios. These results indicate the potential for acute, but not chronic, impacts to aquatic plants in the stream under selected conditions at the maximum application rate.

Fish and Aquatic Invertebrates

Acute toxicity RQs for fish and aquatic invertebrates were all below the most conservative LOC of 0.05 (acute RTE species) for all pond and stream scenarios, indicating that surface runoff of tebuthiuron is not likely to pose acute risks to these aquatic species.

Chronic risk RQs for fish were well below the LOC for chronic risk to RTE species (0.5) in both pond and stream scenarios. At the typical application rate, chronic risk RQs for aquatic invertebrates were below the LOC for chronic risk to RTE species (0.5) in all but one modeled scenario (RQ of 1.28 for base watershed with sand soil and 10 inches of precipitation per year). At the maximum application rate in the pond scenario, chronic RQs for aquatic invertebrates were elevated above the LOC for chronic risk to RTE species (0.5) in 31 of 42 scenarios. In addition, 11 of the 42 pond RQs were above the chronic risk LOC (1). The majority of these elevated RQs occurred in watersheds with sandy soil. No RQs for aquatic invertebrates in the stream scenario were above their LOCs.

These results indicate that these scenarios are not likely to result in long-term risk to fish in the stream or pond or to aquatic invertebrates in the stream at the typical application rate. Long-term impacts to aquatic invertebrates in the pond, especially RTE species, may occur at the maximum application rate.

Piscivorous Birds

Risk quotients for the piscivorous bird were all well below the most conservative terrestrial animal LOC (0.1), indicating that surface runoff is not likely to pose a risk to piscivorous birds.

Wind Erosion and Transport Off-site

Risk quotients for typical and RTE terrestrial plants were all well below the plant LOC (1), indicating that wind erosion is not likely to pose a risk to non-target terrestrial plants.

Accidental Spill to Pond

Risk quotients for the truck spill scenario ranged from 0.048 for aquatic invertebrates to 287 for non-target aquatic plants. Risk quotients for the helicopter spill scenario were higher, ranging from 0.196 for aquatic invertebrates to 1,170 for non-target aquatic plants. Potential risk to fish and non-target aquatic plants was indicated for the truck spill, and risk to fish, aquatic invertebrates, and non-target aquatic plants was indicated for the helicopter spill.

Potential Risk to Salmonids from Indirect Effects

Qualitative Evaluation of Impacts to Prey

No RQs in excess of the appropriate acute or chronic LOCs were observed for aquatic invertebrates in any of the stream scenarios associated with off-site drift or surface runoff. However, chronic RQs for invertebrates were greater than the associated chronic LOC for the accidental direct spray scenario. Because aquatic invertebrates are not predicted to be directly impacted by herbicide concentrations in the stream during *normal* application of tebuthiuron, salmonids are not likely to be indirectly affected by a reduction in prey.

Qualitative Evaluation of Impacts to Vegetative Cover

Aquatic plant RQs for accidental direct spray scenarios were above the plant LOC at both the typical and maximum application rates. Therefore, there is the potential for indirect impacts to salmonids due to a reduction in available cover in the unlikely event that a stream is accidentally sprayed.

No elevated aquatic plant RQs were observed resulting from off-site drift to the stream. Acute RQs in excess of the LOC were observed for aquatic plant species in the stream for selected surface runoff scenarios at the maximum application rate, most strongly within sandy watersheds. No chronic RQs were elevated in the surface runoff scenarios. These results indicate the potential for a reduction in cover in some locations as a result of surface runoff when the herbicide is applied at the maximum rate.

Risk quotients for terrestrial plants were elevated above the LOC for accidental direct spray scenarios at both the typical and maximum application rates, indicating the potential for a reduction in this plant community in response to this unlikely event.

Risk quotients for typical terrestrial plants were also observed above the plant LOC as a result of off-site drift 25 ft from the ground application of the herbicide at the maximum rate. In addition, elevated RQs at the typical application rate were observed for RTE species at 25 ft from the ground application using a high boom. Elevated RQs at the maximum application rate were observed for RTE species at 25 ft from the ground application and 100 ft from the ground application using a high boom. These results indicate the potential for a reduction in riparian cover under selected conditions as a result of off-site drift from a nearby tebuthiuron application.

No RQs in excess of the LOC were observed for typical terrestrial plant species for any of the surface runoff scenarios. Elevated RQs as a result of surface runoff were observed for RTE terrestrial plant species in 4 of 42 scenarios at the typical application rate and 8 of 42 scenarios at the maximum application rate. Therefore, a reduction in plant cover is likely as a result of accidental direct spray and under selected scenarios as a result of off-site drift and surface runoff. In these circumstances, salmonids could be indirectly affected by the reduction in cover.

Uncertainty Analysis

For any ERA, a thorough description of uncertainties is a key component that serves to identify possible weaknesses in the analysis and to elucidate what impact such weaknesses might have on the final risk conclusions. The uncertainties of this risk assessment are discussed below (also see Table 7-1 in the herbicide ERAs [ENSR 2005a-j]).

Toxicity Data Availability

The majority of the available toxicity data was obtained from studies conducted as part of the USEPA pesticide registration process. There are a number of uncertainties related to the use of this limited data set in the risk assessment. In general, it would be preferable to base any ecological risk analysis on reliable field studies that clearly identify and quantify the amount of potential risk from particular exposure concentrations of the chemical of concern. However, in most risk assessments it is more common to extrapolate the results obtained in the laboratory to the receptors found in the field. It should be noted, however, that laboratory studies often overestimate risk relative to field studies (Fairbrother and Kapustka 1996).

Species for which toxicity data are available may not necessarily be the most sensitive species to a particular herbicide. These species have been selected as laboratory test organisms because they are generally sensitive to stressors and can also be maintained under laboratory conditions. Toxicity values for the most appropriate sensitive surrogate species for each receptor were selected by qualified toxicologists based on a thorough review of the available toxicity data; however, there is a possibility that some non-tested receptors in a given receptor group would be more sensitive.

Furthermore, the surrogate species used in the registration testing are not an exact match to the wildlife receptors included in the ERA. For example, avian data are only available for two primarily herbivorous birds: the mallard duck and the bobwhite quail. However, TRVs based on these receptors were also used to evaluate risk to insectivorous and piscivorous birds. Species with alternative feeding habits may be more or less sensitive to the herbicide than those species tested in the laboratory (see Tables C-3 and C-4 for a list of surrogate species and their receptor groups).

In general, the most sensitive available endpoint for the appropriate surrogate test species was used to derive TRVs. This approach is conservative as there may be a wide range of data and effects for different species. For example, the EC_{50}s available for aquatic invertebrates exposed to bromacil ranged from 65 mg a.i./L to >1,000 mg a.i./L. Accordingly, 65 mg a.i./L was selected as the aquatic invertebrate TRV, even though the majority of results were well above this value. In general, this selection criterion for TRVs has the potential to overestimate risk within the ERA.

In addition, several of the toxicity tests conducted during the registration process did not use herbicide formulations with 100% a.i. The assumption has been made that any toxicity observed in the tests is due to the herbicide a.i.; however, it is possible that the additional ingredients in the different formulations also had an effect. For purposes of TRV derivation and the ERA, it was assumed that all toxicity data applies to the a.i. itself and not the particular product formulation tested. This may result in an overestimate of risk to certain receptors and species guilds.

Degradates, Inert Ingredients, Adjuvants, and Tank Mixtures

In a detailed herbicide risk assessment, it is preferable to estimate risks not just from the a.i. of an herbicide, but also from the cumulative risks of degradates, inert

TABLE C-15
Risk Levels Used to Describe Typical Herbicide Effects According to Exposure Scenario and Ecological Receptor Group

	BROMACIL		CHLOR		DIFLU		DIQUAT		DIURON		FLURIDONE		IMAZAPIC		OVERDRIVE		SULFM		TEBU	
	Typ	**Max**	**Typ**	**Max**	**Typ**	**Max**	**Typ**	**Max**	**Typ**	**Max**	**Typ**	**Max**	**Typ**	**Max**	**Typ**	**Max**	**Typ**	**Max**	**Typ**	**Max**
Direct Spray																				
Terrestrial animals	0 [14:16]	0 [8:16]	0 [16:16]	0 [16:16]	0 [16:16]	0 [16:16]	0 [10:16]	M [7:16]	0 [12:16]	0 [6:16]	0 [16:16]	0 [15:16]	0 [16:16]	0 [16:16]	0 [15:16]	0 [15:16]	0 [16:16]	0 [16:16]	0 [15:16]	0 [9:16]
Terrestrial plants	H [1:1]	H [1:1]	H [1:1]	H [1:1]	M [1:1]	H [1:1]	H [1:1]	H [1:1]	M [1:1]	H [1:1]	NE	NE	L [1:1]	M [1:1]	M [1:1]	H [1:1]	0 [1:1]	L [1:1]	M [1:1]	H [1:1]
RTE terrestrial plants	H [1:1]	H [1:1]	H [1:1]	H [1:1]	H [1:1]	H [1:1]	H [1:1]	H [1:1]	H [1:1]	H [1:1]	NE	NE	L [1:1]	M [1:1]	H [1:1]	H [1:1]	H [1:1]	H [1:1]	M [1:1]	H [1:1]
Fish pond	L [1:2]	L [1:2]	0 [2:2]	0 [2:2]	0 [2:2]	0 [2:2]	L [2:2]	M [1:2]	M [2:2]	H [1:2]	0 [2:2]	0 [2:2]	0 [2:2]	0 [2:2]	0 [2:2]	0 [2:2]	0 [2:2]	0 [2:2]	0 [2:2]	0 [2:2]
Fish stream	M [1:2]	M [1:2]	0 [2:2]	0 [2:2]	0 [2:2]	0 [2:2]	M [1:2]	M [2:2]	H [1:1]	H [1:1]	0 [2:2]	L [2:2]	0 [2:2]	0 [2:2]	0 [2:2]	0 [2:2]	0 [2:2]	0 [2:2]	0 [2:2]	0 [2:2]
Aquatic invertebrates pond	0 [2:2]	0 [2:2]	0 [2:2]	0 [2:2]	0 [2:2]	0 [2:2]	M [2:2]	H [1:2]	M [2:2]	H [1:2]	0 [2:2]	L [1:2]	0 [2:2]	0 [2:2]	0 [2:2]	0 [2:2]	0 [2:2]	0 [2:2]	L [1:2]	L [1:2]
Aquatic invertebrates stream	0 [2:2]	L [1:2]	0 [2:2]	0 [2:2]	0 [2:2]	0 [2:2]	H [1:2]	H [2:2]	H [1:1]	H [1:1]	L [1:2]	M [1:2]	0 [2:2]	L [1:2]	0 [2:2]	0 [2:2]	0 [2:2]	0 [2:2]	L [1:2]	M [1:2]
Aquatic plants pond	H [1:2]	H [2:2]	M [1:2]	H [2:2]	L [1:2]	L [1:2]	H [2:2]	H [2:2]	H [2:2]	H [2:2]	0 [2:2]	0 [2:2]	L [1:2]	L [2:2]	M [1:2]	M [1:2]	H [2:2]	H [2:2]	L [2:2]	M [1:2]
Aquatic plants stream	H [2:2]	H [2:2]	M [2:2]	H [2:2]	L [1:2]	L [1:2]	H [2:2]	H [2:2]	H [1:1]	H [1:1]	0 [2:2]	0 [2:2]	L [2:2]	M [2:2]	M [1:2]	H [1:2]	H [2:2]	H [2:2]	M [1:2]	H [1:2]
Piscivorous bird	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	0 [1:1]	0 [1:1]	NA	NA	NA	NA	NA	NA	NA	NA
Accidental Spill																				
Fish pond	NE	M [1:1]	NE	0 [2:2]	NE	0 [1:1]	NE	H [2:2]	NE	H [1:1]	NE	M [2:2]	NE	0 [2:2]	NE	0 [1:1]	NE	0 [2:2]	NE	M [1:2]
Aquatic invertebrates pond	NE	L [1:1]	NE	0 [2:2]	NE	0 [1:1]	NE	H [2:2]	NE	H [1:1]	NE	H [1:2]	NE	0 [2:2]	NE	0 [1:1]	NE	0 [2:2]	NE	L [1:2]
Aquatic plants pond	NE	H [1:1]	NE	H [2:2]	NE	H [1:1]	NE	H [2:2]	NE	H [1:1]	NE	L [2:2]	NE	H [2:2]	NE	M [1:1]	NE	H [2:2]	NE	H [2:2]

TABLE C-15 (Cont.)

Risk Levels Used to Describe Typical Herbicide Effects According to Exposure Scenario and Ecological Receptor Group

	BROMACIL		**CHLOR**		**DIFLU**		**DIQUAT**		**DIURON**		**FLURIDONE**		**IMAZAPIC**		**OVERDRIVE**		**SULFM**		**TEBU**	
	Typ	**Max**	**Typ**	**Max**	**Typ**	**Max**	**Typ**	**Max**	**Typ**	**Max**	**Typ**	**Max**	**Typ**	**Max**	**Typ**	**Max**	**Typ**	**Max**	**Typ**	**Max**
Off-Site Drift																				
Terrestrial animals	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NE	NE	NA	NA	NA	NA	NA	NA	NA	NA
Terrestrial plants	M [3:6]	M [3:6]	M [5:12]	M [8:12]	0 [4:6]	0 [4:6]	L [7:12]	M [7:12]	0 [5:6]	L [4:6]	NE	NE	0 [18:18]	0 [13:18]	0 [5:6]	0 [4:6]	0 [12:12]	0 [12:12]	0 [6:6]	0 [4:6]
RTE terrestrial plants	M [3:6]	H [3:6]	M [7:12]	M [7:12]	L [4:6]	L [4:6]	M [7:12]	M [7:12]	M [3:6]	H [3:6]	NE	NE	0 [17:18]	0 [13:18]	L [3:6]	L [4:6]	H [5:12]	H [8:12]	0 [5:6]	L [3:6]
Fish pond	0 [12:12]	0 [12:12]	0 [24:24]	0 [24:24]	0 [12:12]	0 [12:12]	NA	NA	0 [12:12]	0 [11:12]	NA	NA	0 [36:36]	0 [36:36]	0 [12:12]	0 [12:12]	0 [24:24]	0 [24:24]	0 [12:12]	0 [12:12]
Fish stream	0 [12:12]	0 [12:12]	0 [24:24]	0 [24:24]	0 [12:12]	0 [12:12]	NA	NA	0 [12:12]	0 [9:12]	NA	NA	0 [36:36]	0 [36:36]	0 [12:12]	0 [12:12]	0 [24:24]	0 [24:24]	0 [12:12]	0 [12:12]
Aquatic invertebrates pond	0 [12:12]	0 [12:12]	0 [24:24]	0 [24:24]	0 [12:12]	0 [12:12]	NA	NA	0 [12:12]	0 [9:12]	NA	NA	0 [36:36]	0 [36:36]	0 [12:12]	0 [12:12]	0 [24:24]	0 [24:24]	0 [12:12]	0 [12:12]
Aquatic invertebrates stream	0 [12:12]	0 [12:12]	0 [24:24]	0 [24:24]	0 [12:12]	0 [12:12]	NA	NA	0 [11:12]	0 [8:12]	NA	NA	0 [36:36]	0 [36:36]	0 [12:12]	0 [12:12]	0 [24:24]	0 [24:24]	0 [12:12]	0 [12:12]
Aquatic plants pond	0 [9:12]	L [7:12]	0 [24:24]	0 [24:24]	0 [9:12]	0 [8:12]	NA	NA	L [8:12]	M [6:12]	NA	NA	0 [36:36]	0 [34:36]	0 [9:12]	0 [8:12]	L [13:24]	L [12:24]	0 [12:12]	0 [12:12]
Aquatic plants stream	0 [8:12]	L [6:12]	0 [24:24]	0 [22:24]	0 [8:12]	L [6:12]	NA	NA	L [6:12]	M [6:12]	NA	NA	0 [36:36]	0 [33:36]	0 [8:12]	L [6:12]	L [14:24]	L [10:24]	0 [12:12]	0 [12:12]
Piscivorous bird	0 [6:6]	0 [6:6]	0 [12:12]	0 [12:12]	0 [6:6]	0 [6:6]	NA	NA	0 [6:6]	0 [6:6]	NA	NA	0 [18:18]	0 [18:18]	0 [6:6]	0 [6:6]	0 [12:12]	0 [12:12]	0 [6:6]	0 [6:6]
Surface Runoff																				
Terrestrial animals	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Terrestrial plants	0 [42:42]	0 [42:42]	0 [42:42]	0 [42:42]	0 [42:42]	0 [42:42]	NA	NA	0 [42:42]	0 [42:42]	NA	NA	0 [42:42]	0 [42:42]	0 [42:42]	0 [42:42]	0 [42:42]	0 [42:42]	0 [42:42]	0 [42:42]
RTE terrestrial plants	0 [39:42]	0 [38:42]	0 [42:42]	0 [42:42]	0 [34:42]	0 [33:42]	NA	NA	0 [38:42]	0 [34:42]	NA	NA	0 [42:42]	0 [42:42]	0 [34:42]	0 [33:42]	0 [32:42]	0 [28:42]	0 [38:42]	0 [34:42]
Fish pond	0 [65:84]	0 [46:84]	0 [84:84]	0 [84:84]	0 [84:84]	0 [84:84]	NA	NA	L [60:84]	L [48:84]	NA	NA	0 [84:84]	0 [84:84]	0 [84:84]	0 [47:84]	0 [84:84]	0 [84:84]	0 [84:84]	0 [84:84]
Fish stream	0 [84:84]	0 [84:84]	0 [84:84]	0 [84:84]	0 [84:84]	0 [84:84]	NA	NA	0 [81:84]	0 [68:84]	NA	NA	0 [84:84]	0 [84:84]	0 [84:84]	0 [83:84]	0 [84:84]	0 [84:84]	0 [84:84]	0 [84:84]

TABLE C-15 (Cont.)
Risk Levels Used to Describe Typical Herbicide Effects According to Exposure Scenario and Ecological Receptor Group

	BROMACIL		CHLOR		DIFLU		DIQUAT		DIURON		FLURIDONE		IMAZAPIC		OVERDRIVE		SULFM		TEBU	
	Typ	Max	Typ	Max	Typ	Max	Typ	Max	Typ	Max	Typ	Max	Typ	Max	Typ	Max	Typ	Max	Typ	Max
Surface Runoff (Cont.)																				
Aquatic invertebrates pond	0 [84:84]	0 [84:84]	0 [84:84]	0 [84:84]	0 [84:84]	0 [84:84]	NA	NA	0 [38:84]	L [34:84]	NA	NA	0 [84:84]	0 [84:84]	0 [84:84]	0 [84:84]	0 [84:84]	0 [84:84]	0 [83:84]	0 [53:84]
Aquatic invertebrates stream	0 [84:84]	0 [84:84]	0 [84:84]	0 [84:84]	0 [84:84]	0 [84:84]	NA	NA	0 [66:84]	0 [49:84]	NA	NA	0 [84:84]	0 [84:84]	0 [84:84]	0 [84:84]	0 [84:84]	0 [84:84]	0 [84:84]	0 [84:84]
Aquatic plants pond	M [70:84]	H [45:84]	0 [64:84]	0 [53:84]	0 [84:84]	0 [84:84]	NA	NA	M [50:84]	H [64:84]	NA	NA	0 [80:84]	0 [62:84]	0 [70:84]	0 [67:84]	L [42:84]	L [38:84]	0 [65:84]	L [55:84]
Aquatic plants stream	0 [45:84]	L [55:84]	0 [80:84]	0 [77:84]	0 [84:84]	0 [84:84]	NA	NA	L [35:84]	L [39:84]	NA	NA	0 [84:84]	0 [83:84]	0 [84:84]	0 [84:84]	0 [69:84]	0 [60:84]	0 [84:84]	0 [74:84]
Piscivorous bird	0 [42:42]	0 [42:42]	0 [42:42]	0 [42:42]	0 [42:42]	0 [42:42]	NA	NA	0 [42:42]	0 [42:42]	NA	NA	0 [42:42]	0 [42:42]	0 [42:42]	0 [42:42]	0 [42:42]	0 [42:42]	0 [42:42]	0 [42:42]
Wind Erosion																				
Terrestrial animals	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Terrestrial plants	0 [9:9]	0 [9:9]	0 [9:9]	0 [9:9]	0 [9:9]	0 [9:9]	NA	NA	0 [9:9]	0 [9:9]	NA	NA	0 [9:9]	0 [9:9]	0 [9:9]	0 [9:9]	0 [9:9]	0 [9:9]	0 [9:9]	0 [9:9]
RTE terrestrial plants	0 [9:9]	0 [9:9]	0 [9:9]	0 [9:9]	0 [9:9]	0 [9:9]	NA	NA	0 [9:9]	0 [9:9]	NA	NA	0 [9:9]	0 [9:9]	0 [9:9]	0 [9:9]	0 [9:9]	0 [9:9]	0 [9:9]	0 [9:9]
Fish pond	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Fish stream	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Aquatic invertebrates pond	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Aquatic invertebrates stream	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Aquatic plants pond	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Aquatic plants stream	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Piscivorous bird	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA

TABLE C-15 (Cont.)

Risk Levels Used to Describe Typical Herbicide Effects According to Exposure Scenario and Ecological Receptor Group

Notes:

The reported risk level is based on the risk level of the majority of the RQs for each exposure scenario within each of the above receptor groups and exposure categories (i.e., direct spray/spill, off-site drift, surface runoff, and wind erosion). As a result, risk may be higher than the reported risk category for some scenarios within each category. The reader should consult the risk tables in Section 4 of the ERAs to determine the specific scenarios that result in the displayed level of risk for a given receptor group.

Abbreviations

CHLOR = Chlorsulfuron
DIFLU = Diflufenzopyr
SULFM = Sulfometuron methyl
TEBU = Tebuthiuron

NA = Not Applicable
NE = Not Evaluated
Typ = Typical application rate
Max = Maximum application rate

Risk Categories

0 = No Risk (RQ<LOC)
L = Low Risk (RQ 1-10x LOC)
M = Moderate Risk (RQ 10-100x LOC)
H = High Risk (RQ >100 LOC)

Number in brackets represents Number of RQs in the Indicated Risk Level: Number of Scenarios Evaluated ingredients (inerts), and adjuvants. Other pesticides may also factor into the risk estimates, as herbicides can be tank mixed to expand the level of control and to accomplish multiple identified tasks (the BLM usually only tank mixes herbicides with other herbicides). However, using currently available models (e.g., GLEAMS), it is only practical to make deterministic risk calculations (i.e., exposure modeling, effects assessment, and RQ derivations) for a single a.i.

In addition, information on inerts, adjuvants, and degradates is often limited by the availability of, and access to, reliable toxicity data for these constituents. The sections below present a qualitative evaluation of the potential risks from degradates, inert ingredients, adjuvants, and tank mixtures.

Degradates

The potential toxicity of degradates should be considered when selecting an herbicide. However, it is beyond the scope of this risk assessment to evaluate all of the possible degradates of the various herbicide formulations of the 10 herbicides. Degradates may be more or less mobile and more or less toxic in the environment than their source herbicides (Battaglin et al. 2003). Differences in environmental behavior (e.g., mobility) and toxicity between parent herbicides and degradates makes prediction of potential impacts challenging. For example, a less toxic, but more mobile bioaccumulative, or persistent degradate may have a greater adverse impact due to residual concentrations in the environment. A recent study indicated that 70% of degradates had either similar or reduced toxicity to fish, daphnids, and algae than the parent pesticide. However, 4.2% of the degradates were more than an order of magnitude more toxic than the parent pesticide, with a few instances of acute toxicity values below 1 mg/L (Sinclair and Boxall 2003). No evaluations of impacts to terrestrial species were conducted in the study. The lack of data on the toxicity of degradates of the specific herbicides represents a source of uncertainty in the risk assessment.

Inerts

Pesticide products contain both active and inert ingredients. The terms “active ingredient” (a.i.) and “inert ingredient” have been defined by federal law—the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA)—since 1947. An a.i. is one that prevents, destroys, repels, or mitigates the effects of a pest, or is a plant regulator, defoliant, desiccant, or nitrogen stabilizer. By law, the a.i. must be identified by name on the label, together with its percentage by weight. An inert ingredient is simply any ingredient in the product that is not intended to affect a target pest. For example, isopropyl alcohol may be an a.i. and antimicrobial pesticide in some products; however, in other products, it is used as a solvent and may be considered an inert ingredient. The law does not require inert ingredients to be identified by name and percentage on the label, but the total percentage of such ingredients must be declared. Because neither the federal law nor the regulations define the term “inert” on the basis of toxicity, hazard or risk to humans, non-target species, or the environment, it should not be assumed that all inert ingredients are non-toxic.

The USEPA has a listing of regulated inert ingredients at http://www.epa.gov/opprd001/inerts/index.html. This listing divides inert ingredients into four lists. The number of inert ingredients found in the nine herbicides evaluated in the ERAs for each category is shown below (nine inerts were not found on the USEPA lists):

List 1 - Inert Ingredients of Toxicological Concern: None.

List 2 - Potentially Toxic Inert Ingredients: None.

List 3 - Inerts of Unknown Toxicity: 12.

List 4 - Inerts of Minimal Toxicity. List 4 is subdivided into List 4A (minimal risk inert ingredients) and List 4B (inerts that have sufficient data to substantiate that they can be used safely in pesticide products): Over 50.

Toxicity information was also searched in the following sources:

- TOMES (a proprietary toxicological database including USEPA’s IRIS, the Hazardous Substance Data Bank, and the Registry of Toxic Effects of Chemical Substances (RTEC).
- USEPA’s ECOTOX database which includes AQUIRE (a database containing scientific papers published on the toxic effects of chemicals to aquatic organisms).
- TOXLINE (a literature searching tool).

- Material Safety Data Sheets from suppliers.
- Other sources, such as the Farm Chemicals Handbook.
- Other cited literature sources.

Relatively little toxicity information was found. A few acute studies on aquatic or terrestrial species were reported. No chronic data, no cumulative effects data, and almost no indirect effects data (food chain species) were found for the inerts in the 10 herbicides.

A number of the List 4 compounds (Inerts of Minimal Toxicity) are naturally-occurring earthen materials (e.g., clay materials or simple salts) that would produce no toxicity at applied concentrations. However, some of the inerts, particularly the List 3 compounds and unlisted compounds, may have moderate to high potential toxicity to aquatic species based on information in Material Safety Data Sheets or on published data.

As a tool to evaluate List 3 and unlisted inerts in the ecological risk assessment, the exposure concentration of the inert compound was calculated and compared to toxicity information. As described in more detail in Appendix D of the ERAs, the GLEAMS model was set up to simulate the effects of a generalized inert compound in the base-case watershed (annual precipitation rate of 50 inches per year, application area of 10 acres, slope of 0.05, surface roughness of 0.015, erodibility of 0.401 tons per acre, and vegetation type of weeds) with a sand soil type. The chemical characteristics of the generalized inert compound were set at either extremely high or low values to describe it as either a very mobile or stable compound. The application rate of the inert/adjuvant compound was fixed at 1 lb a.i./acre. Under these conditions, the maximum predicted ratio of inert concentration to herbicide application rate was 0.69 mg/L per lb a.i./acre (3 day maximum in the pond), and in every case (acute and chronic, pond and stream scenarios) the inert concentrations exceeded herbicide a.i. concentrations.

In general, higher application rates resulted in higher exposure concentrations of surfactant inerts, exceeding 1 mg/L for the maximum pond scenario. This suggests that inerts associated with the application of herbicides may contribute to acute toxicity to aquatic organisms if they reach the aquatic environment. However, due to the lack of specific inert toxicity data, this may be an overestimate of the potential toxicity. It is assumed that toxic inerts would not represent a substantial percentage of the herbicide and that minimal impacts to the environment would result from these inert ingredients.

Adjuvants and Tank Mixtures

Evaluating the potential additional/cumulative risks from mixtures and adjuvants of pesticides is substantially more difficult than evaluating the inerts in the herbicide composition. While many herbicides are present in the natural environment along with other pesticides and toxic chemicals, it is extremely difficult to estimate the potential cumulative risks of such mixtures. The composition of such mixtures is highly site-specific, and thus nearly impossible to address at the programmatic level of the EIS.

Herbicide label information indicates whether a particular herbicide can be tank mixed with other pesticides. Adjuvants (e.g., surfactants, crop oil concentrates, fertilizers) may also be added to the spray mixture to improve the herbicide efficacy when mixed and applied to according to the label. Without product specific toxicity data, it is impossible to quantify the potential impacts of these mixtures. In addition, a quantitative analysis could only be conducted if reliable scientific evidence allowed a determination of whether the joint action of the mixture was additive, synergistic, or antagonistic. Such evidence is not likely to exist unless the mode of action is common among the chemicals and receptors.

Adjuvants

Adjuvants generally function to enhance or prolong the activity of an a.i. For terrestrial herbicides, adjuvants aid in proper wetting of foliage and absorption of the a.i. into plant tissue. Adjuvant is a broad term that includes surfactants, selected oils, anti-foaming agents, buffering compounds, drift control agents, compatibility agents, stickers, and spreaders. Adjuvants are not under the same registration guidelines as pesticides; the USEPA does not register or approve the labeling of spray adjuvants. Individual herbicide labels contain lists with "label-approved" adjuvants for use with a particular herbicide under specific conditions.

Following the same procedure used to address inerts in Appendix D of the ERAs, the GLEAMS model was used to estimate the potential portion of an adjuvant that might reach an adjacent waterbody via surface runoff. In addition, sources (Muller 1980; Lewis 1991;

Dorn et al. 1997; Wong et al. 1997) generally suggest that the acute toxicity of surfactants and anti-foam agents to aquatic life ranges from 1 to 10 mg/L, and that chronic toxicity ranges as low as 0.1 mg/L. This evaluation indicates that, for herbicides with high application rates, adjuvants have the potential to cause acute, and potentially chronic, risk to aquatic species. However, more specific modeling and toxicity data would be necessary to define the level of uncertainty. Selection of adjuvants is under the control of BLM land managers, and it is recommended that land managers follow all label instructions and abide by any warnings. In general, adjuvants compose a relatively small portion of the volume of herbicide applied; however, selection of adjuvants with limited toxicity and low volumes is recommended to reduce the potential for the adjuvant to influence the toxicity of the herbicide.

Tank Mixtures

The use of tank mixtures of labeled herbicides, along with the addition of an adjuvant (when stated on the label), may be an efficient use of equipment and personnel; however, knowledge of both products and their interactions is necessary to avoid unintended negative effects. In general, herbicide interactions can be classified as additive, synergistic, or antagonistic:

- Additive effects occur when mixing two herbicides produces the same response as the combined effects of each herbicide applied alone. The products neither hurt nor enhance each other.
- Synergistic responses occur when two herbicides provide a greater response than the added effects of each herbicide applied separately.
- Antagonistic responses occur when two herbicides applied together produce less control than if you applied each herbicide separately.

While a quantitative evaluation of all of these mixtures is beyond the scope of this ERA, a qualitative evaluation may be made if the assumption is made that the products in the tank mix will act in an additive manner. The predicted RQs for two active ingredients can be summed for each individual exposure scenario to see if the combined impacts result in additional RQs elevated above the corresponding LOCs.

The RQs for any two herbicides in a tank mix were combined to simulate a tank mix in Appendix E of each ERA (diquat, fluridone, and tebuthiuron are not generally tank mixed by the BLM and were not included in this analysis). The application rates within the tank mix are not necessarily the same as each individual a.i. applied alone. See Table 7-2 in each ERA (ENSR 2005a-j) for a comparison of the percent of RQs exceeding LOCs for each of the 10 herbicide active ingredients applied alone and in a tank mix.

These comparisons indicate that tank mixes for bromacil (with sulfometuron methyl) and imazapic with diflufenzopyr do not result in more RQs above the associated LOCs for birds, mammals, fish, and invertebrates (and aquatic plants for imazapic), than were predicted for bromacil, imazapic, or diflufenzopyr alone. Additional elevated RQs are predicted for both aquatic and RTE terrestrial plants when tank mixes of bromacil with sulfometuron methyl, and imazapic with diflufenzopyr, are applied (aquatic plant risk is not elevated versus imazapic applied alone). This suggests that in some cases plant species may be particularly sensitive to the tank mix. However, when chlorsulfuron and diuron are tank mixed, all receptors are at higher risk than with application of chlorsulfuron alone (risks are not higher than with the application of diuron alone), and most receptors are also at higher risk when sulfometuron methyl is applied with bromacil versus sulfometuron methyl alone.

The comparison of the RQs from herbicide a.i. and tank mixes of these herbicides indicate that results are specific to each tank mix. Aquatic plants and RTE terrestrial plants may be at greater risk from the tank mixed application than from the a.i. alone. However, in some cases all receptors are at greater risk and precautions (e.g., increased buffer zones, decreased application rates) should be taken to reduce risk. There is some uncertainty in this evaluation because herbicides in tank mixes may not interact in an additive manner; this may overestimate risk if the interaction is antagonistic, or it may underestimate risk if the interaction is synergistic. In addition, other products may also be included in tank mixes and may contribute to the potential risk.

Selection of tank mixes, like adjuvants, is under the control of BLM land managers. To reduce uncertainties and potential negative impacts, it is required that land managers follow all label instructions and abide by any warnings. Labels for both tank mixed products should be thoroughly reviewed and mixtures with the least potential for negative effects should be selected. This is especially relevant when a mixture is applied in a manner that may have increased potential for risk. Use of a tank mix under these conditions increases the level of uncertainty in risk to the environment.

Concentration Models

The ecological risk assessment relies on different models (i.e., AgDRIFT, GLEAMS, CALPUFF) to predict the off-site impacts of herbicide use. These models have been developed and applied in order to produce a conservative estimate of herbicide loss from the application area to off-site locations (via off-site drift, surface runoff, and wind erosion). The uncertainty analysis focused on which environmental characteristics (e.g., soil type, annual precipitation) exert the biggest numeric impact on model outputs. The results of this uncertainty analysis also have important implications for the ability to apply risk calculations to different site characteristics from a risk management perspective.

AgDRIFT®

Off-site spray drift and resulting terrestrial deposition rates and waterbody concentrations (hypothetical pond or stream) were predicted using the computer model, AgDRIFT® Version 2.0.05 (SDTF 2002). As with any complex ecological risk assessment model, a number of simplifying assumptions were made to ensure that the risk assessment results would be protective of most environmental settings encountered in the public land management program.

Predicted off-site spray drift and downwind deposition can be substantially altered by a number of variables intended to simulate the herbicide application process (e.g., nozzle type used in the spray application of an herbicide mixture, ambient wind speed, release height [application boom height], and evaporation). Hypothetically, any variable in the model that is intended to represent some part of the physical process of spray drift and deposition can substantially alter predicted downwind drift and deposition patterns. Recognizing the incomplete knowledge of all the scenarios likely to be encountered in BLM programs, these assumptions were developed to be conservative and likely result in overestimation of actual off-site spray drift and resulting environmental impacts.

GLEAMS

The GLEAMS model was used to predict the loading of herbicide to nearby soils, ponds, and streams from overland runoff, erosion, and root-zone groundwater runoff.

Herbicide Loss Rates

The trends in herbicide loss rates (herbicide loss computed as a percent of the herbicide applied within the watershed) and water concentrations predicted by the GLEAMS model echo trends that have been documented in a wide range of streams located in the Midwestern U.S. A recently published study (Lerch and Blanchard 2003) recognized three primary factors affecting herbicide transport to streams, and they can be organized into four general categories:

- Intrinsic factors – soil and hydrologic properties and geomorphologic characteristics of the watershed;
- Anthropogenic factors – land use and herbicide management;
- Climate factors – precipitation and temperature; and
- Herbicide factors – chemical and physical properties and formulation of herbicide product.

These findings were based on the conclusions of several prior investigations, data collected as part of the U.S. Geological Survey's (USGS) National Stream Quality Accounting Network (NASQAN) program, and the results of runoff and baseflow water samples collected in 20 streams in northern Missouri and southern Iowa. The investigation concluded that the median runoff loss rates for atrazine, cyanazine, acetochlor, alachlor, metolachlor, and metribuzin ranged from 0.33 to 3.9% of the mass applied—loss rates that were considerably higher than other areas of the U.S. The study indicated that the runoff potential was a critical factor affecting herbicide transport. The median total loss rates range from 0.27 to 36%, and the median runoff loss rates range from 0.02 to 0.27%.

The results of the GLEAMS simulations indicate trends similar to those identified in the Lerch and Blanchard (2003) study. First, the GLEAMS simulations demonstrated that the most dominant factors controlling herbicide loss rates are soil type and precipitation (increased precipitation and less porous soil types result in increased herbicide runoff). This was demonstrated in each of the GLEAMS simulations that considered the effect of highly variable annual precipitation rates and soil type on herbicide transport. In all cases, the GLEAMS model predicted that runoff loss rate was positively correlated with both precipitation rate and soil type. Second, estimating the groundwater discharge concentrations by using the predicted root-zone concentrations as a surrogate is extremely conservative. For example, while the median runoff loss rates predicted using GLEAMS range from 0.02 to 0.27%, the median total loss rates are substantially higher. This discrepancy may be due to the differences between the watershed characteristics in the field investigation and those used to describe the GLEAMS simulations, as well as to the conservative nature of the baseflow predictions.

Based on the results and conclusions of prior investigations, the runoff loss rates predicted by the GLEAMS model are approximately equivalent to loss rates determined within the Mississippi River watershed and elsewhere in the U.S., but the percolation loss rates are probably conservatively high. This confirms that the GLEAMS modeling approach used in the ERAs either approximates or overestimates the rate of loadings observed in the field.

CALPUFF

The USEPA's CALPUFF air pollutant dispersion model was used to predict impacts of the potential migration of herbicide between 1.5 and 100 km from the application area by windblown soil (fugitive dust). Several assumptions were made that could overestimate or underestimate the deposition rates obtained from this model.

The modeling conservatively assumed that all of the herbicide would be present in the soil at the commencement of a windy event, and that no reduction due to vegetation interception/up-take, leaching, or solar or chemical half-life would occur following aerial application. Thus, the model likely overestimates the deposition rates unless the herbicide is taken by the wind as soon as it is applied. It is more likely that a portion of the applied herbicide would be sorbed to plants or degraded over time.

The model assumes a 1-mm penetration depth, which is less than the depth used in previous herbicide risk assessments (SERA 2003) and the depth assumed in the GLEAMS model (1 cm surface soil). This penetration depth is conservative and likely overestimates impacts.

The use of flat terrain could underestimate deposition for mountainous areas. In these areas, hills and mountains would likely focus wind and deposition into certain areas, resulting in pockets of increased risk. The use of bare, undisturbed soil results in less uptake and transport than disturbed (e.g., tilled) soil. However, the BLM does not apply herbicides to agricultural areas so this assumption may be appropriate for public lands.

The surface roughness in the vicinity of the application site directly affects the deposition rates predicted by CALPUFF. The surface roughness length used in the CALPUFF model is a measure of the height of obstacles to the wind flow and varies by land-use types. Forested areas and urban areas have the highest surface roughness lengths (0.5 m to 1.3 m) while grasslands have the lowest (0.001 m to 0.10 m). Predicted deposition rates are likely to be higher near the application area and lower at greater distances if the surface roughness in the area is relatively high (above 1 meter, such as in forested areas). Therefore, overestimation of the surface roughness could overpredict deposition within about 50 km of the application area and underpredict deposition beyond 50 km. Overestimation of the surface roughness could occur if, for example, prescribed burning was used to treat a typically forested area prior to planned herbicide treatment.

The surface roughness also affects the calculated "friction velocity" used to determine deposition velocities, which in turn are used by CALPUFF to calculate the deposition rate. Friction velocity increases with increasing wind speed and increasing surface roughness. Higher friction velocities result in higher deposition velocities and higher deposition rates, particularly within about 50 km of the emission source.

The CALPUFF modeling assumes that the data from the selected National Weather Service stations are representative of meteorological conditions in the vicinity of the application sites. Site-specific meteorological data (e.g., from an on-site meteorological tower) could provide slightly different wind patterns, possibly due to local terrain, which could impact the deposition rates as well as locations of maximum deposition.

Overall, conservative assumptions employed in exposure characterization will tend to overestimate risk, and therefore, may provide a buffer for the effects of uncertainties.

Potential Indirect Effects on Salmonids

No actual field studies or ecological incident reports on the effects of the 10 herbicides on salmonids were identified during the ERA. Therefore, any discussion of direct or indirect impacts on salmonids was limited to qualitative estimates of potential impacts on salmonid populations and communities. In some herbicide evaluations TRVs are based on warmwater species because these have the highest risk, and this may result in an overestimate of risks to salmonids, which are coldwater species. These evaluations indicated that for most herbicides (except diuron, diquat, and fluridone), salmonids are not likely to be indirectly impacted by a reduction in food supply (i.e., fish and aquatic invertebrates). However, they could be affected by a reduction in vegetative cover, which may occur under some conditions.

It is anticipated that these qualitative evaluations overestimate the potential risk to salmonids because of the conservative selection of TRVs for salmonid prey and vegetative cover, the application of additional LOCs (with uncertainty/safety factors applied) to assess risk to RTE species, and the use of conservative stream characteristics in the exposure scenarios (i.e., low order stream, relatively small instantaneous volume, limited consideration of herbicide degradation or absorption in models).

Herbicide Application Recommendations

The following general recommendations are designed to reduce potential unintended impacts to the environment from the application of herbicides in the BLM vegetation management program (see the individual ERAs for recommendations specific to each herbicide):

- Select herbicide products carefully to minimize additional impacts from degradates, adjuvants, inert ingredients, and tank mixtures. This is especially important for application scenarios that already predict potential risk from the a.i. itself.

- Review, understand, and conform to the "Environmental Hazards" section on the herbicide label. This section warns of known pesticide risks to wildlife receptors or to the environment and provides practical ways to avoid harm to organisms or to the environment.

- Avoid accidental direct spray and spill conditions to reduce the largest potential impacts. Use the typical application rate, rather than the maximum application rate, to reduce risk to most species for most herbicides.

- Limit the use of terrestrial herbicides (especially bromacil, diuron, and sulfometuron methyl) in watersheds with downgradient ponds and streams if potential impacts to aquatic plants are of concern.

- Establish appropriate (herbicide specific) buffer zones to downstream waterbodies, habitats, or species/populations of interest (see Table C-16).

- Consider the proximity of application areas to salmonid habitat and the possible effects of herbicides on riparian and aquatic vegetation. Maintain appropriate buffer zones around salmonid-bearing streams (see Table C-16 and recommendations in individual ERAs).

- The results from these ERAs contribute to the evaluation of proposed alternatives in the PEIS and to the development of the Biological Assessment (BA), specifically addressing the potential impacts of vegetation treatments to proposed and listed RTE species on public lands. Furthermore, the ERAs will assist BLM field offices on the proper application of herbicides to ensure that impacts to plants and animals and their habitats are minimized to the extent practical.

TABLE C-16
Buffer Distances (feet) to Minimize Risk from Off-site Drift of Herbicides

Application Scenario	BROM	CHLR	DIFLU	DIQT	DIUR	FLUR	IMAZ	OVER	SULF	TEBU
Buffer Distance from Non-target Aquatic Plants										
Typical Application Rate										
Aerial	NA	0	NA	NE	NA	NE	0	NA	1,300	NE
Low Boom	100	0	100	NE	900	NE	0	100	900	0
High Boom	900	0	900	NE	1,000	NE	0	900	900	0
Maximum Application Rate										
Aerial	NA	300	NA	NE	NA	NE	300	NA	1,500	NE
Low Boom	900	0	900	NE	1,000	NE	0	900	900	0
High Boom	900	0	900	NE	1,000	NE	0	900	900	0
Buffer Distance from Non-target Terrestrial Plants										
Typical Application Rate										
Aerial	NA	1,350	NA	1,200	NA	NE	0	NA	0	NE
Low Boom	950	900	100	100	0	NE	0	0	0	0
High Boom	950	900	100	900	100	NE	0	100	0	0
Maximum Application Rate										
Aerial	NA	1,350	NA	1,200	NA	NE	900	NA	0	NE
Low Boom	1,000	1,000	100	900	200	NE	0	100	0	50
High Boom	1,000	1,000	100	900	500	NE	0	100	0	50
Buffer Distance from Rare, Threatened, and Endangered Terrestrial Plants										
Typical Application Rate										
Aerial	NA	1,400	NA	1,200	NA	NE	300	NA	1,500	NE
Low Boom	1,200	1,000	100	900	1,000	NE	0	100	1,100	0
High Boom	1,200	1,000	900	900	1,000	NE	0	900	1,100	50
Maximum Application Rate										
Aerial	NA	1,400	NA	1,200	NA	NE	900	NA	1,500	NE
Low Boom	1,200	1,050	900	1,000	1,000	NE	0	900	1,100	100
High Boom	1,200	1,000	900	1,000	1,000	NE	0	900	1,100	500

Buffer distances are the smallest modeled distance at which no risk was predicted. In some cases, buffer distances were extrapolated (if the largest distance modeled still resulted in risk) or interpolated (if greater precision was required).
NA = Not applicable; NE = Not evaluated; BROM = Bromacil; CHRL = Chlorsulfuron; DIFLU = Diflufenzopyr; DIQT = Diquat; DIUR = Diuron; FLUR = Fluridone; IMAZ = Imazapic; OVER = Overdrive®; SULF = Sulfometuron methyl; and TEBU = Tebuthiuron.

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